Atlantic Climate Adaptation Solutions Association

Solutions d'adaptation aux changements climatiques pour l'Atlantique

Managing Groundwater Resources

Assessing the impact of climate change on salt-water intrusion of coastal aquifers in Atlantic Canada

By

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ACASA is a non-profit organisation formed to coordinate project management and planning for climate change adaptation initiatives in Nova Scotia, New Brunswick, Prince Edward Island and Newfoundland and Labrador and supported through the Regional Adaptation Collaborative, a joint undertaking between the Atlantic Provinces, Natural Resources Canada and regional municipalities and other partners.

Preface

Atlantic Canada already faces the need to adapt to the impacts of climate change and these needs will increase in future. Governments, communities, businesses, and other stakeholders all have diverse but linked roles to play in the development of adaptation measures and strategies to reduce vulnerabilities and increase resilience to these climate change impacts. The Atlantic Climate Adaptation Solutions Association (ACASA) is being funded to facilitate collaborative activities in Atlantic Canada that will be needed to develop adaptations to climate change in the key area of coastal climate hazards including salt-water intrusion due to sea-level rise. The activities will include assessment of risks and vulnerabilities in selected communities, review of selected relevant regulations, policies, and guidelines, and evaluation of different approaches to address identified risks and opportunities. This project builds on the Climate Change Adaptation Strategy for Atlantic Canada, endorsed by the Atlantic Environment Ministers in June 2008.

Executive Summary

Many coastal communities in Atlantic Canada depend on groundwater resources as a principal source of potable water supply, either via municipal water supply systems or private wells. Because of the proximity of these water supplies to the coast, a phenomenon referred to as salt-water intrusion, can in cases, be a serious threat to the viability of these supplies with saline, marine groundwater “intruding” inland from the coast, raising the salinity of groundwater tapped by potable fresh water wells.

The effect of climate change on the region has been investigated from a variety of perspectives, with projected rises in sea level being one of the key concerns, particularly in relation to such threats as coastal erosion and storm surge damage to coastal infrastructure. Various sources have also speculated about the potential effect of climate change on the integrity of coastal aquifers, however to date these concerns have not been addressed in a rigorous fashion. The current work described here specifically addresses potential climate change impacts on saltwater intrusion of coastal aquifers, and the implications for coastal groundwater supplies in the Atlantic Provinces.

The current investigation, through a series of case studies, utilized a variety of approaches to shed light on the occurrence and controlling factors behind salt water intrusion in Atlantic Canada, and further assessed the extent to which climate change may alter the current situation. In some cases, the principal focus is devoted to characterizing occurrences of saline groundwater with a view to documenting the key factors determining the vulnerability of coastal aquifers to saltwater intrusion. This work has been conducted both on a local, detailed scale, and on a more regional basis. In other cases characterization of the existing interface between saline, marine groundwaters and fresh inland groundwater has been extended, to make predictions regarding the future extent of salt water intrusion under the influence of climate change and anthropogenic pressures.

The occurrence and severity of salt-water intrusion depends on the collective influence of several factors, including local geological, hydrogeological and climatic conditions, as well as anthropogenic factors such as the degree of exploitation of groundwater resources in coastal environments. Some of these factors can be considered as fixed (e.g. geological framework) whereas other factors can be dependent on climatic conditions or anthropogenic influences. In relation to climate change, the three key factors considered here are sea level rise, changes in groundwater recharge conditions and the rate of groundwater extraction.

The individual case studies presented here demonstrate how variable the occurrence and extent of salt-water intrusion can be, even on a local scale, and underline some important gaps in our knowledge of the distribution of salt-water intrusion in the Atlantic Region. Nonetheless, some significant conclusions can be drawn, with respect to both expected climate change impacts and the management of coastal groundwater resources in general.

The most significant finding from this work is the fact that sea level rise itself is likely to be of little significance to the integrity of coastal aquifers in the Atlantic Region. Even rises of sea level in the order of 1 metre are projected to result in only slight changes (roughly comparable in magnitude to

the change in sea level rise itself) to the boundary between fresh and saline groundwater in coastal aquifers.

By comparison, the results of numerical modelling suggest that the position of the salt-water fresh water interface is likely to be more sensitive to changes in groundwater recharge rates, as the rate of discharge of fresh groundwater toward the coast is a key factor determining the inland extent of intrusion of marine, saline groundwater. The factors controlling groundwater recharge rates are complex however, making firm predictions on future recharge rates more difficult to formulate.

Nonetheless, these effects, while not insignificant, are greatly overshadowed by sensitivity to groundwater extraction rates, especially in areas of significant levels of groundwater exploitation such as represented by municipal water supplies. Noteworthy also is the fact that the full extent of the effects of groundwater withdrawal on increased salinities in production wells can be expected in some cases to take decades to manifest itself, thus current groundwater quality may not be evidence of “sustainable” groundwater extraction rates with respect to salt-water intrusion. Furthermore, groundwater demand, while potentially linked to climate change, is controlled by more immediate factors (e.g. population growth), and is the one factor considered here to be within human control.

Based on the findings of the studies presented here, the most critical component in the sustainable management of coastal groundwater resources is clearly the management of groundwater extraction rates. As the factors determining the extent of salt-water intrusion are highly variable, this requires an assessment of the local vulnerability of groundwater resources in question, coupled with long term monitoring groundwater quality. Because the response of groundwater salinities to pumping may only be manifested after extended periods of time, ideally the establishment of targets for sustainable extraction rates would include numerical modelling under a variety of water demand and climate scenarios. Where trends toward increased salinity are detected, confirmation of the source of salinity should be sought, as the occurrence of saline groundwater may be attributable to a number of sources, individually or in combination.

The effects of climate change on future groundwater recharge rates is an issue of concern to water managers in general, as it has important implications for overall water budgets, including the management of sustainable groundwater extraction rates and base-flow contributions to surface water. In the context of salt-water intrusion, these effects may be of principal interest for long range development planning in coastal areas with relatively low water demand, as in areas of greater levels of groundwater exploitation, management of water demand will be the key consideration. The current state of the science in regard to climate change impacts on groundwater recharge rates is still in its infancy; however, this area clearly deserves continuing research because of its widespread implications for water resource management in general, and in this case, with respect to salt-water intrusion. As advances in this area progress, the findings should be integrated into the existing knowledge base on salt-water intrusion, and projections updated or revised as appropriate.

Finally, much of the information on salt water intrusion stems from investigations of existing occurrences or problems related to saline groundwater. As such, this information base is likely to be skewed toward conditions found in regions of greatest dependence on coastal groundwater resources, and may not be entirely representative of conditions elsewhere in the region. Continuation of regional reconnaissance work, identifying areas of potential vulnerability to salt-water intrusion, can help fill these gaps, and can provide planners, water managers and policy makers with valuable information on the feasibility and relative merits of future water supply options, especially in less developed areas of the region. As noted previously, the development or maintenance of individual water supply projects or facilities will continue to require the support of site specific investigations, focusing on such key factors as likely sources of salinity, characterization of the local salt water fresh water interface, and the climatic and anthropogenic factors that will determine the long term sustainability of local groundwater resources.

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I. Introduction

Coastal aquifers are an important water resource for the nearly one billion people living in the world's coastal regions (Small and Nicholls, 2003). Rising sea levels pose a threat to coastal aquifers worldwide and this has been noted by several synthesis studies on climate change (CC), (see for example Stern, 2006; IPCC, 2007; EPA, 2008). In coastal regions, there is a boundary between "inland" fresh groundwater (GW) and "saline GW of marine origin often referred to as the salt-water fresh water interface (SFI). Marine waters are denser than fresh water and may extend inland at depth below fresh GW (see Figure 1), a phenomenon referred to as seawater intrusion or salt-water intrusion (SWI), which results in the degradation of fresh GW resources. In an idealized, highly theoretical sense the position of the boundary is determined by the relative densities of salt and fresh water, and the elevation of the "inland" water table. In practice this boundary is not sharp, but is represented by a transition zone or zone of diffusion, and the position of the SFI is controlled by the amount of fresh GW flowing toward the coast, which in turn is dependent on the hydraulic characteristics of the aquifer and the rate of recharge (and thus discharge) of fresh GW. Because the geological and climatic factors that determine the extent of SWI vary by region, the extent of SWI can be highly variable.

In the absence of CC considerations, inland movement of the SFI has most frequently been attributed to over-pumping of coastal aquifers, and the effects of CC, including sea level rise, can be expected to exacerbate existing issues with SWI. A number of studies using analytical and numerical models have demonstrated that under some conditions, the position of the SFI in coastal aquifers could move inland significant distances (Masterson and Garabedian, 2007; Werner and Simmons, 2009; Webb and Howard, 2011). Climate change also has the potential to cause significant changes in groundwater recharge on a local scale, with decreases of more than 10% expected for 20% of the Earth's landmass as a whole (Döll, 2009). Areas experiencing significant decreases in recharge will see an even greater amount of SWI. As noted earlier, these effects will occur on top of the effects of groundwater extraction, which has been increasing globally (Wada et al., 2010

Within North America, much of the knowledge of SWI has been developed from studies conducted in the United States. The relative scarcity of work in Canada (Barlow and Reichard, 2009) is probably indicative of the smaller population located in coastal communities when compared to the United States. Only a few published studies have dealt with SWI in Atlantic Canada (e.g. Carr, 1969; Brown, 1971; Tremblay et al. 1973; van der Kamp, 1981; Rivard et al. 2003; Comte and Banton, 2007).

As shown by the electrical resistivity surveys presented by Rivard et al. (2003), the shallow (less than ~ 40 m depth) salt water-fresh GW mixing zone that exists naturally in coastal Carboniferous rock aquifers in Prince Edward Island and New Brunswick can be located relatively close to the shoreline, to distances as great as 200 m inland. Although they indicated that natural GW discharge to coastal areas may be of sufficient magnitude to prevent significant SWI, Rivard et al. (2003) also noted that "the situation could be different when pumping is involved (since pumping modifies significantly the flow pattern by reversing the hydraulic gradient, creating an encroachment of the zone of seawater)". In addition, Rivard notes that local factors such as site specific geological factors are likely to play a role in the extent of SWI.

Near the Town of Summerside, Prince Edward Island, Tremblay et al. (1973) identified two types of intrusion: 1) a landward migration of salt-water from the coast into the upper 24 m of the aquifer due to large GW withdrawals, and 2) up-coning of salt-water at a depth of approximately 115 m due to intermittent pumping. Salt-water intrusion on the island is facilitated by fractures within the Carboniferous rocks that increase the overall hydraulic conductivity of the aquifer. Brown (1971) reported that chloride concentrations in municipal supply wells (Town of Shippagan, northeast New Brunswick) located 550 m inland in a highly fractured Pennsylvanian-age sandstone aquifer rose to approximately 600 to 800 mg/L with continued use. In the nearby Magdalen Islands, Comte and Banton (2006) have used a combined geophysical-hydrogeological modelling approach to first determine the position of the SFI and, secondly, to simulate the response of the interface to several pumping scenarios. Their simulations under the assumption of increased pumping rates (e.g. summer pumping rates maintained throughout the year) show significant degradation of GW quality.

The previous research in Atlantic Canada suggests that SWI in coastal regions may be a concern, especially in locations with significant GW extraction. However, the additional longer term impacts associated with CC (e.g. sea level rise, changing temperature, precipitation and GW regimes) have not been investigated to date.

Globally it has been predicted that by 2100 sea level will rise from between 18 cm and 59 cm (IPCC 2007), while more recent studies have calculated that sea level rise is likely to be more than twice that amount (Rahmstorf 2010) On a more local scale, sea level rise will also depend on crustal subsidence. For example, R.J. Daigle Enviro (2011) has provided predictions of sea level rise for Richibucto based on global sea level rise trends and estimates of local vertical motion (crustal subsidence) with the anticipated change in relative sea level 0.93 m ± 0.38 m by the year 2100.

In addition to the direct effects of sea level rise, potential changes in recharge rates for GW could be expected to influence the extent or severity of SWI. However predictions for future recharge conditions hold greater uncertainty, as they depend on many factors beyond changes in net temperature or precipitation, including precipitation intensity and the form and seasonal distribution of precipitation.

With these factors in mind, the principle purpose of this study is to evaluate the potential consequences of CC with respect to SWI affecting coastal aquifers in Atlantic Canada. The intent of this work is to provide advice to various levels of government, planners and the general public on what degree of risk exists in light of projected rates of sea level rise or other changes to the regional hydrologic regime.

II. Approach to the Investigations

The prevalence and impacts of SWI on GW supplies, as well as the current state of knowledge of these factors, varies considerably between the four Atlantic Provinces and indeed for the most part, on a local scale. Because of these geographical differences, the current work involves selected case studies intended to represent some of the typical hydrogeological environments deemed to be most at risk, and to provide examples of strategies that may be useful in assessing the potential vulnerability of coastal GW resources to SWI. Accordingly, the specific studies described in this report are tailored to address the priorities of each region. These priorities are in large part shaped by such factors as settlement patterns in coastal regions, historical demand on coastal GW supplies, previous problems with SWI and availability of alternate sources of water.

Many factors contribute to the occurrence, extent and severity of SWI, and although the primary focus of this study is to evaluate the risk CC poses in relation to coastal GW supplies, as a fundamental first step it is necessary to understand the existing local extent and characteristics of SWI.

Studies conducted in New Brunswick (Mott and Butler, Richibucto area), Nova Scotia (Beebe, Communities of Pugwash and Wolfville) and PEI (Hansen and Ferguson, Summerside area), provide detailed characterization of local occurrences of saline GW and their relationship to the process determining the occurence of SWI, while work in Nova Scotia (Kennedy) and Newfoundland and Labrador (Adams) focuses on assessing the occurrence or risk of SWI on a more regional scale. Finally the works of Green & MacQuarrie (Richibucto, NB) and Hansen & Ferguson (Summerside and Lennox Island, PE) investigate the potential impact of CC on well documented cases of SWI. In these latter studies, key factors considered include the effects of sea level rise, potential changes in recharge, and the additional factor of changes in water demand via GW extraction on the position of the SFI. Collectively, these works provide important insight into the vulnerability of coastal aquifers, and more importantly, evaluate the relative importance of the key factors in the development and management of coastal aquifers under changing climatic conditions.

Because of the differing priorities, approaches and available information on SWI for specific regions within Atlantic Canada, a detailed description of the methodology is provided within the body of individual case studies.

III. Case Studies

The scientific research supporting this report is assembled from a series of case studies distributed throughout the Atlantic Provinces, and includes site specific investigations in Richibucto, New Brunswick, Summerside and Lennox Island in Prince Edward Island and Pugwash and Wolfville in Nova Scotia. In addition, a more regional approach to evaluating the risk of saltwater intrusion is developed for the Province of Nova Scotia in general and for a portion of southwestern Newfoundland. Much of this work has been presented at scientific conferences, and in some of these cases, are presented here as published in the relevant conference proceedings. Other works have been produced specifically for inclusion in this report.

The selection of locations for site specific investigations has generally been based on a combination of the known occurrence, or history of problems with saline groundwater, and existing dependence on groundwater to support developed areas of the respective provinces.

The scientific methodology used in each site specific study share some common elements, such as an evaluation of geological and hydrogeological conditions derived from well drilling logs and geochemical sampling programs. However, the specific details for each program vary according to the availability of information from previous works and techniques best suited to individual locations, and are thus described in each individual study. To preserve the scientific insight and interpretation of the individual researchers, their work is presented as originally supplied by the authors, with the exception of minor editorial/grammatical changes made for the purposes of clarity, and as a consequence there may be some variation in terminology used in individual case studies. Numbering of sections, figures and tables is maintained as originally prepared, although references are compiled collectively at the end of the overall report.

New Brunswick Case Studies

The phenomenon of SWI in New Brunswick has received the greatest amount of attention in south eastern portions of the province, where coastal communities are more reliant on GW than other areas. Previous work, in such communities as Shippagan and Richibucto, both of which rely on central GW supplies, has documented problems with increasing salinities in water supply wells.

In the current study, work in the community of Richibucto is documented in two closely related case studies. The first, *Salt water intrusion in a coastal sandstone aquifer at Richibucto, New Brunswick as revealed by geophysical and borehole surveys*, has been published in the proceedings for GeoHydro 2011 conference in Quebec City in September 2011, and characterizes the current nature of SWI through a combination of geophysical, hydrogeological and geochemical techniques. The second study, *An evaluation of the effects of climate change on seawater intrusion in coastal aquifers: a case study from Richibucto, New Brunswick,* documents the results of numerical modeling of climatic and hydrogeological elements to provide predictions for future conditions under factors relating to CC.

Case Study:
Salt water intrusion in a coastal sandstone aquifer at Richibucto, New Brunswick as revealed by geophysical and borehole surveys

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1. INTRODUCTION

The town of Richibucto, situated on the coast of the Northumberland Strait 65 km north of the city of Moncton (Figure 1), draws water from a municipal wellfield consisting of three production wells completed at depths between 23 and 51 m in a fractured sandstone aquifer. This aquifer is subdivided into upper and lower sandstone units separated by one to two closely spaced discontinuous layers of shale, with drilled thicknesses varying from 1.2 to 5 m. The production wells are located approximately 1 km from the coast and within 500 m of a tidally influenced brook (Figure 2). In recent years two production wells, PW1 and PW2, have experienced elevated levels of groundwater salinity. The issue has been addressed by decommissioning PW2, restricting the pumping rate in PW1, and adding two new wells: PW3 was completed at a shallower depth, and most recently, in 2010, PW4 was commissioned farther inland (MGI 1991, Stantec 2009). However, the possibility of recurrence remains a concern.

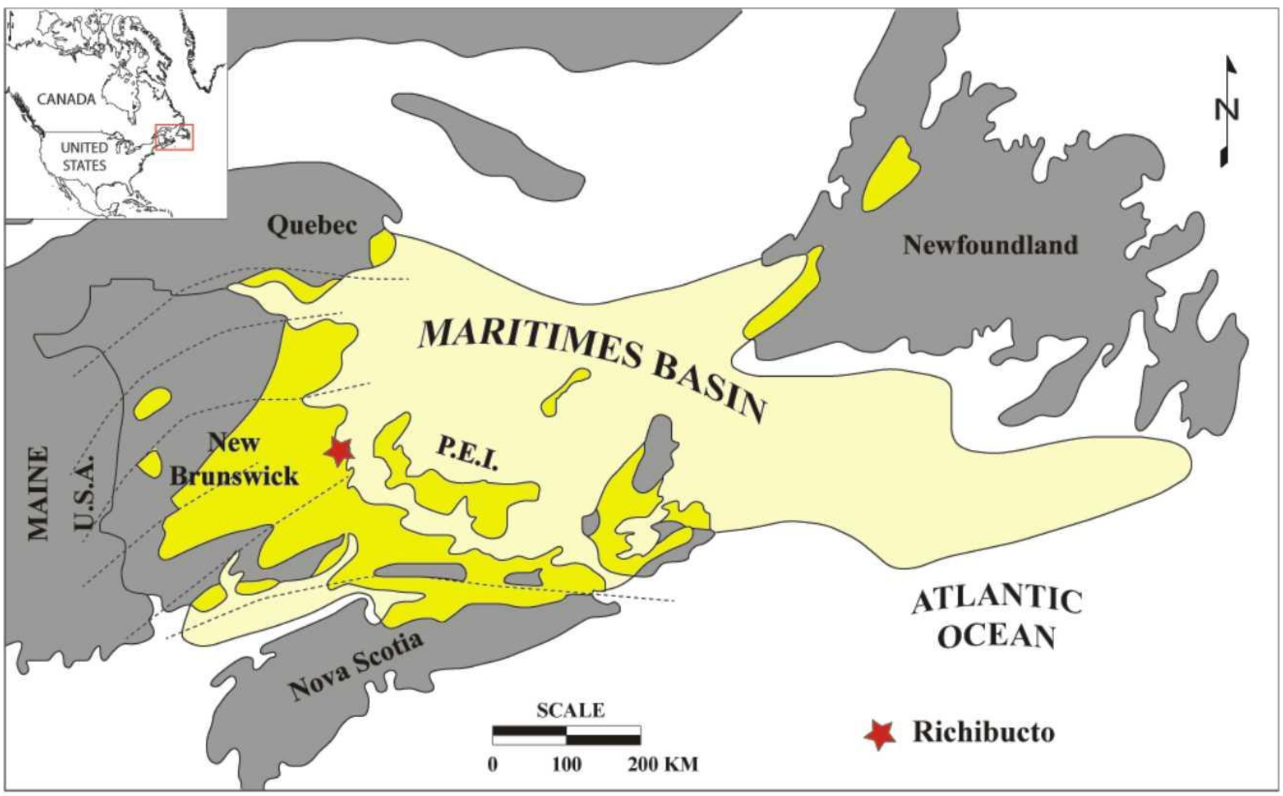


Figure 1. Regional map showing the extent of the Maritimes Basin in yellow and the location of Richibucto, NB.

In late July 2010 electrical resistivity tomography (ERT) surveys were conducted along several lines in and around the town of Richibucto in an effort to assess the current extent of seawater intrusion into the bedrock aquifer. The objective of this ERT survey was to determine if the spatial distribution of saline groundwater could be determined from resistivity variations in the sandstone aquifer units and used to infer any preferential pathways and mechanisms of intrusion in the Richibucto area. This information would aid a companion project (Green and MacQuarrie, 2012) in the development of a conceptual hydrogeological model for purposes of inferring the sensitivity of the area to increased salt water intrusion as a consequence of climate change.

In August 2011, two boreholes, extending to depths of 56 m near the coast and 89 m within the wellfield, were drilled and cored to investigate signs of salt water intrusion evident in the ERT sections. These boreholes were logged in September 2011 with various geophysical tools. In this paper, we use the geological and geophysical logs, along with

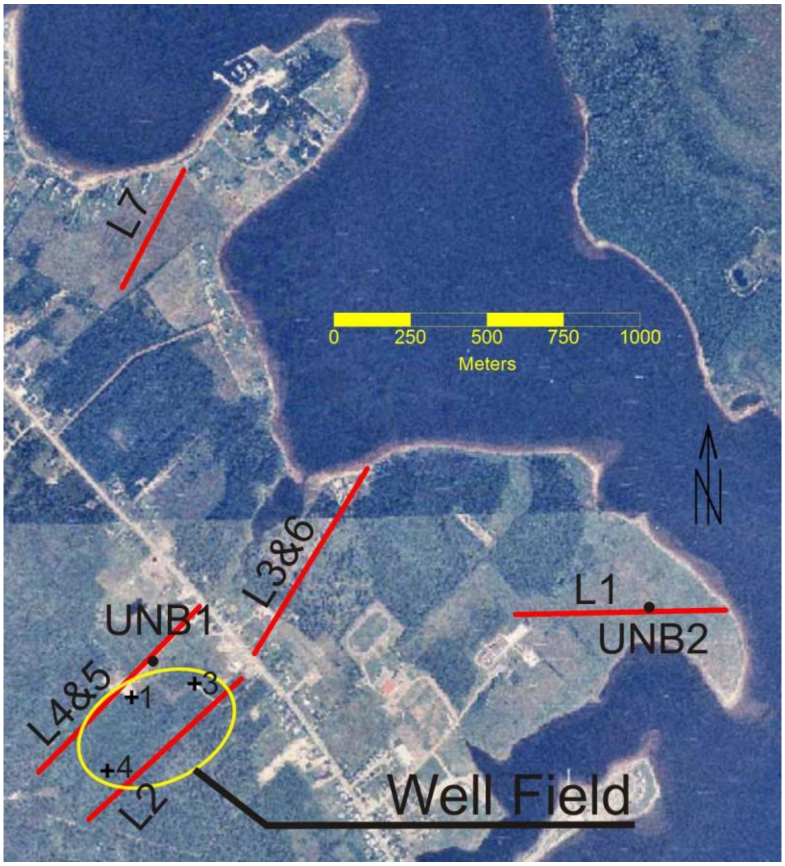


Figure 2. Aerial photo of the northern part of Richibucto, NB, showing the location of municipal pumping wells (black crosses with numbers indicating the production well ID) and ERT survey lines. Boreholes UNB1 and UNB2 drilled as part of this study are shown as black circles.

measurements of borehole water salinity to support interpretations of salt water intrusion evident in the two most informative ERT sections – one near the coast and one within the wellfield.

1.1 Hydrogeological Setting

The surficial geology around Richibucto well field consists of poorly sorted glacial till up to 3 m thick. The area has very little relief, and slopes gently towards the Northumberland Strait. Bedrock geology in the region is represented by the Richibucto Formation of the Pictou group, late Carboniferous in age and part of the post-accretionary intermontane Maritimes Basin (St. Peter, 1993) (see Figure 1). The Richibucto Formation is comprised of multilayered grey and lesser reddish brown sandstone, pebbly sandstones, and pebble conglomerates separated by reddish brown, very fine to fine grained sandstones and mudstones. The units are interpreted as fining up sequences deposited in a fluvial environment, possibly braided rivers (St. Peter and Johnson, 2009).

The town of Richibucto has intermittently experienced elevated chloride levels within its wellfield with water samples from PW2 measuring as high as 322 mg/L (S. Sullivan, personal communication, 2011), in excess of the Canadian drinking water limit of 250 mg/L. These elevated chloride readings were associated with increased pumping rates to accommodate flushing of municipal water mains. However, naturally elevated levels of chloride in aquifers of eastern New Brunswick are not uncommon, as illustrated by water sampling that was carried out as part of the Carboniferous Drilling Project in late 1970's (Ball et al., 1981). Figure 3 shows that chloride concentrations in excess of the drinking water guidelines (triangular symbols) were encountered at depths of 61 m (200 ft) and/or 121 m (400 ft) at four points along the Richibucto River extending southwest of Richibucto as well as to the east of Moncton and in several other localities. Many of these locations are far from the coast, which suggests that the saline waters may have been emplaced following the retreat of glaciers from this area when relative sea level was significantly higher. These observations seem to have been the basis for a recommendation (MGI 1991) that the town of Richibucto avoid drilling deeper than about 91 m (300 ft) to avoid intersecting saline water. Figure 3 shows that the validity of that recommendation is very site-dependent, as two wells drilled a few kilometres to the northwest of Richibucto exhibited low salinities at both 61 and 121 m depth. Nonetheless, the data exhibit a general increase in salinity with depth which is a trend one would expect to be particularly evident in coastal areas, such as Richibucto, where modern sea water intrusion processes are occurring.

2. METHODS

The two main hydrogeophysical methods used in the Richibucto area were surface ERT surveys and borehole geophysical logging. Figure 2 shows a map of the seven ERT survey lines, ranging from 400 to 800 m in length and oriented sub-perpendicular to the coast. The measurements presented in this paper were made using a 72-electrode Iris Syscal Pro resistivity imaging system using a Wenner array with unit electrode spacings ("a-spacings") ranging from 6 to 10 m and up to 23 depth levels (i.e. "n-values" up to 23). The two-dimensional resistivity inversion program RES2DINV (Loke and Barker, 1995) was used to generate models or 'sections' of inferred resistivity variations in the subsurface beneath each line to depths of up to 100m. The inverse modelling parameters including smoothing constraints designed to favour the recovery of models having lateral continuity are in agreement with the predominantly flat-lying geology. More details on the data acquisition, processing and inversion are presented by Mott et al. (2011).

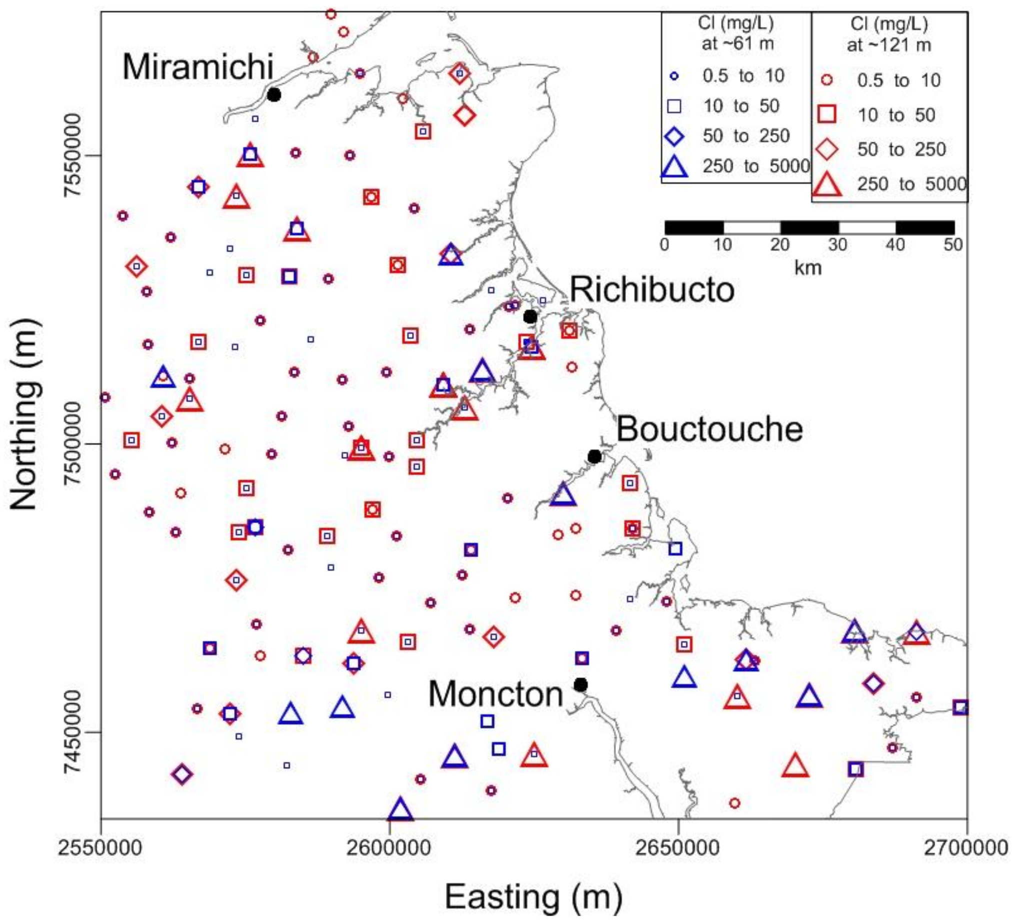


Figure 3. Map showing the locations of boreholes drilled and sampled for water chemistry in central-eastern New Brunswick as part of the Carboniferous Drilling Project (Ball et al., 1981). The blue and red symbols represent chloride concentrations measured in groundwaters extracted at depths of 61 and 121 m (200 and 400 ft), respectively. The larger, triangular symbols represent chloride concentrations in excess of the Canadian Drinking Water Guideline of 250 mg/L.

Areas of salt water intrusion were expected to be apparent as zones of anomalously low electrical resistivity (high electrical conductivity). However, it was recognized that shale layers would also be electrically conductive as a consequence of the high surface conductivity associated with clay particles. Two boreholes, UNB1 and UNB2 (Figure 2), were therefore drilled and cored with a geotechnical diamond drilling rig in order to investigate the geology and groundwater in two areas of particular interest identified by the ERT surveys. Borehole resistivity logging measurements were carried out in the new holes and in pre-existing holes for purposes of determining typical resistivities of shale and sandstone layers. Additional geophysical logs were used to confirm lithologies and identify fractures as discussed in the examples presented below.

Relating geophysical measurements of sandstone resistivity to the salinity of its pore water requires knowledge of (i) how salinity affects pore fluid resistivity, and (ii) how pore fluid resistivity in turn affects rock resistivity. Relationship (i) was investigated by taking samples of water from boreholes in Richibucto and plotting their electrical conductivities against their salinities as expressed in mg/L chloride (sodium chloride being the dominant solute in seawater). These water samples were acquired using a down hole sampling pump or depth-specific sampling devices known as Hydrasleeves™ after the wells had been purged. The purging process involved removing three well volumes of water from the borehole using a Honda “trash pump” connected to approximately 10 m of hose lowered down the open borehole. These samples give conductivities of the water within the well bore and are subject to tidal effects near the coast (Leblanc et al., 2012) and vertical flow between fractures. They do not necessarily give salinities and conductivities of the pore water in the rock matrix at the depths from which they were taken but represent the best estimates available at this time.

Relating pore fluid resistivity to bulk rock resistivity is more complex, because the latter is also dependent on the porosity of the rock, the tortuosity of the pore space, and the possible presence of surface or grain conductivity. Visual inspection of sandstone cores from boreholes UNB1 and UNB2 suggests little clay content so surface conductivity may not be important. In that case, the relationship between fluid resistivity and bulk resistivity may be expected to take the form of Archie's Law:

$$\rho_r = a\phi^{-m}\rho_f \quad , \qquad [1]$$

where ρ_r is the bulk resistivity of the rock,
ρ_f is the resistivity of the pore fluid,
ϕ is the porosity
m and *a* are empirically derived constants.

Drill core samples collected from UNB1 and UNB2 are currently being analysed in the lab at UNB to determine the empirical constants *a*, and *m* appropriate for the Richibucto formation sandstones making up the aquifer. However, in the absence of such measurements, it is common to assume the Humble formula for clay-free, water-saturated sandstones, for which a = 0.62 and m = 2.15. The porosities of several sandstone cores determined to date by hydrostatic weighing range from 5% to 15%. Applying the Humble formula with these porosities, the bulk resistivity is expected to be 355 to 37 times more resistive than that of the pore fluid. We anticipate, therefore, that the variations in the bulk resistivities observed in the ERT and well logging data may be a consequence of variations in either pore water salinity, sandstone vs shale lithology, or sandstone porosity. The roles played by each of those factors are considered in interpretation of the ERT sections below.

3. RESULTS AND DISCUSSION

We begin with a brief discussion of the subsurface geology and groundwater salinities encountered in boreholes UNB1 and UNB2, before moving on to a discussion of how spatial variations in groundwater salinity can be inferred non-invasively from the ERT surveys.

3.1 Subsurface Geology in boreholes UNB1, 2

The recovered core (see example in Figure 4) revealed a subset of the lithologies previously reported for the Richibucto Formation. It was dominated by grey sandstone sequences generally fining upward into red shaley sands and grey shale. However, the two boreholes did not encounter any of the red sandstones (indicative of more oxidizing conditions) or pebble conglomerates that St. Peter and Johnson (2009) reported in other exposures of the Richibucto Formation. Using the core, it was difficult to determine where fractures occurred because core breaks occurred frequently during recovery and there was a lack of oxidation surrounding any of the breaks. To accurately identify the fractures an optical televiewer was used. The fractures are predominantly sub-horizontal and appear to lie mainly along bedding planes exhibiting a variety of low angle dips and dip directions as would be expected in a fluvial depositional environment. Just over 80% of the fractures in the two bore holes have dips below 20º. Some of the larger and more visible fractures appear at the top of fining up sequences where coarse grained sandstone overlies shale. Although not directly related to the issue of salt water intrusion, fracture characterization is very relevant to the understanding of the aquifer and water supplies as flow through fractures is expected to dominate over flow through the porous matrix. More details on fracture apertures and distributions can be found in the geophysical logs presented later in this section.

Figure 4. Photograph of one end of a box of core spanning the contact between shale and overlying sandstone at 18.23 m depth in borehole UNB1. Lithologies from bottom (lower right) to top (upper left) include brown shale, grey shale, fine-grained grey sandstone, and coarse-grained grey sandstone. The fracture in bottom right within the brown shale had a dip of approximately 8.3º. Core is type NW, having a diameter of 47.6 mm (1.87 inches).

3.2 Groundwater Salinities

Figure 5 shows two conductivity versus chloride concentration plots. The plot of the left shows the results of sampling the water column at various depths in monitoring wells within the well field, as well as in monitoring well UNB2 near the coast and at one surface site along the St. Charles River. The dashed blue line represents the Canadian Drinking Water

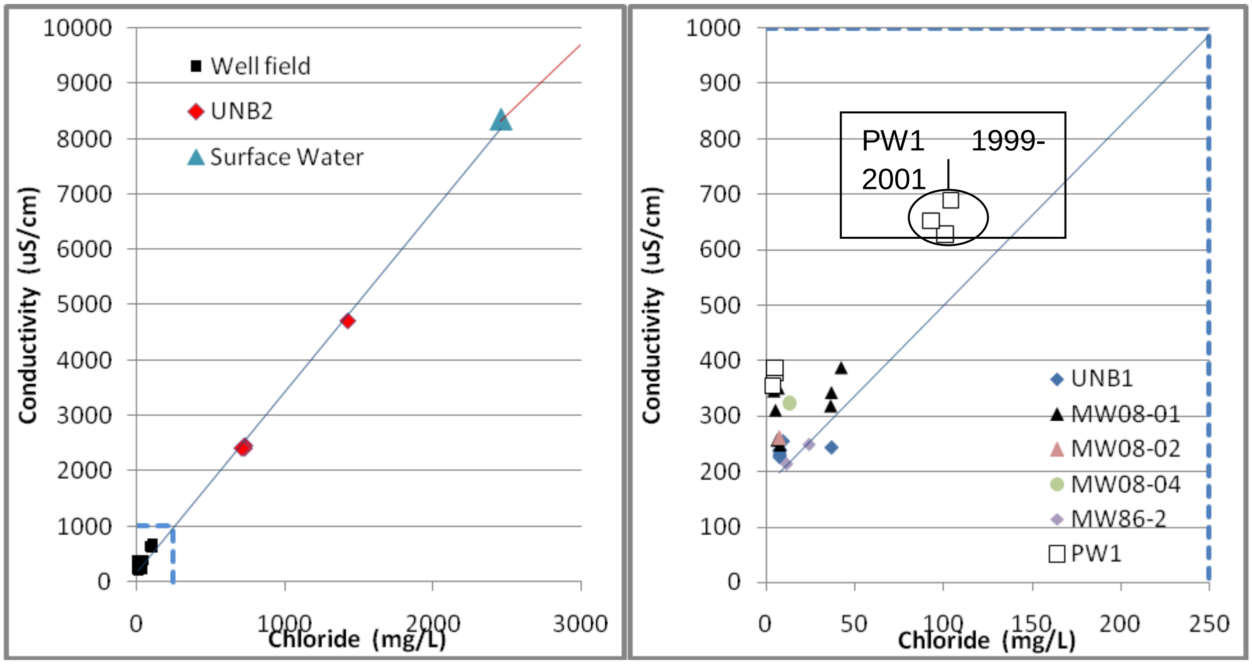


Figure 5. Graph showing the relationship between chloride concentration and conductivity in water samples taken from wells within the wellfield (including UNB1), monitoring well UNB2 near the coast, and from the St Charles River (just northwest of the ERT surveys). The drinking water limit for chloride is shown with the dashed blue line. The graph on the right shows the lower salinities and conductivities, sampled from the wellfield, at a larger scale.

Quality standard for chloride and its corresponding conductivity value (250 mg/L, ~1000 µS/cm). The second plot shows, at an expanded scale, results for only those waters sampled from monitoring wells within the well field. At the higher concentrations of chloride, a linear relationship can be established with conductivity. As chloride decreases well below the Canadian drinking water guidelines of 250 mg/L into the region of the samples from within the well field this linear relationship no longer applies as other ionic species (besides Cl- and Na+) become significant.

3.3 Electrical Resistivity Tomography Surveys

Of the seven ERT lines acquired, only the two most interesting and informative will be presented here. Line 1, acquired in a low-lying area on the north side of Richibucto Harbour, shows clear evidence of seawater intruding laterally from the coast. Line 4, located within the well field, suggests that the elevated salinities in PW1 are a consequence of salt water upconing from depth.

3.3.1 Coastal ERT Survey

Figure 6 shows the ERT section for Line 1 which passes through borehole UNB2, drilled to a depth of 56.87 m, near the coast (see map in Figure 2). A generalized geological log for UNB2, overlain on the ERT section, shows that the geology is dominated by sandstone with three thin layers of shale/shaley sandstone ranging from 0.5 to 5.5 m thick. A more detailed geological log for UNB2, and the corresponding geophysical logs are shown in the lower part of the figure.

Apart from a low resistivity zone at surface below the eastern third of the line, which might represent infiltration of tidal surges, the ERT section is dominated by a wedge-like zone of low resistivity extending inland from the coast, underlying more resistive formations above. The resistivity structure does not correlate with the lithological variations observed in borehole UNB2 particularly well, but is instead suggestive of a salt water wedge intruding laterally into the lower sandstones penetrated by the borehole. The rock resistivity measured within UNB2 by geophysical well logging is comparable to the ERT resistivity adjacent to the hole with the exception of the shale layer from 22 to 27.5 m which is not resolved in the ERT inversion. The ERT method is unable to resolve the low resistivity shale bed at this depth because of its small thickness to depth ratio. However, the transition from high to low resistivity at about 35 m depth in the resistivity log is reproduced in the ERT section.

Shale layers evident in core are also easily identified based on the geophysical logs. Higher gamma ray emission counts matched with low resistivities of 25 to 55 Ohm·m distinguish the shales from the sandstones. The sandstone resistivity varies widely, likely due to a combination of variations in porosity and pore water conductivity. Based on visual examination of the drill cores, it seems likely that short wavelength (metre-scale) variations in sandstone grain size and sorting may be accompanied by changes in porosity and/or permeability that in turn give rise to short wavelength changes in bulk resistivity. However, the gradual (long wavelength) decrease in resistivity with depth below the upper shale layer is more likely to be a consequence of increasing pore water salinity related to an increase in the chloride concentration as established earlier in the paper (see Figure 5) .

Note that the normal resistivity log contains four tracks with different electrode spacings (8, 16, 32, and 64 inches). The shorter spacings, which penetrate less deeply into the formation, tend to measure higher resistivities; this reflects the presence of fresh water introduced to the rock during drilling, forming a resistive 'flushed zone' near the borehole wall. Such flushed zones are likely to be better developed in areas of higher sandstone porosity and permeability, thereby contributing to the short wavelength variations in resistivity that are most evident in the short electrode spacing data. The longer electrode spacings provide measurements closer to the true resistivity of the formation but due to the large sampling volume they are unable to resolve thin beds and become distorted when approaching shale layers.

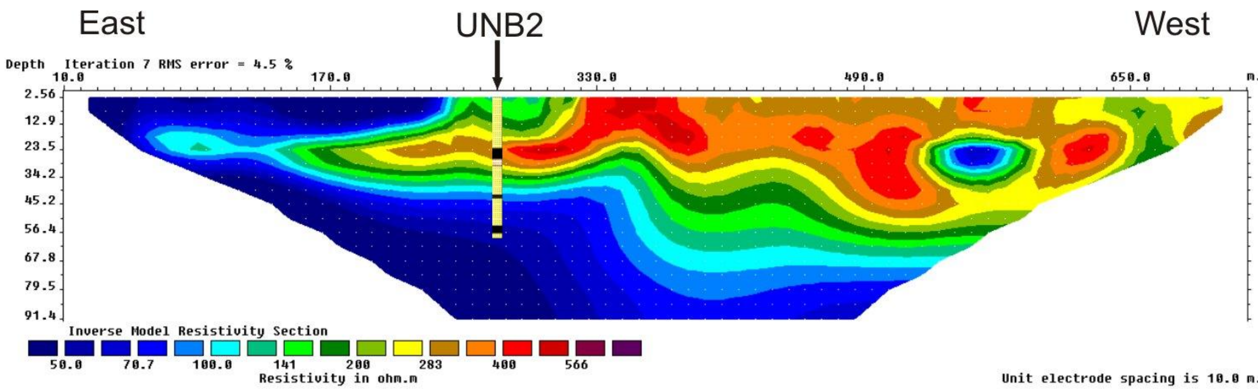
Two fluid conductivity logs, acquired on different dates, are shown side-by-side in Figure 6. The first, FCond 25°C (in red), was logged continuously with the Mount Sopris logging system at the same time as the other geophysical logs. The second, TLC-Cond25°C (in green), was logged about three weeks later with a Solinst Model 107 TLC meter at 1 m intervals. The logs have the same general shape with conductivity being nearly constant through the upper part of the borehole and increasing dramatically at 35 to 38 m depth. However, the conductivities were higher and the abrupt increase occurred at a shallower depth in the second log. Evidence from two water temperature-level-conductivity loggers emplaced in the borehole at depths of 8 and 28 m (LeBlanc et al., 2012) suggests that this variability is likely a consequence of tidal effects; it appears likely that saline water repetitively entered the borehole through fractures at depth during times of high tide and displaced some of the fresher water above until the tide receded again.

The geophysical logs in Figure 6 also include the locations of fractures, based on analyses of the optical televiewer logs. The vast majority (85%) of fractures, such as those corresponding to bedding plane partings, are subhorizontal. The remainder of the fractures have dips as high as 67° with the steepest at 37.17 m depth. The fractures appear to have no preferred orientation. The geophysical log on the far right in Figure 6 includes estimates of the apertures of fractures that appeared to be 2 mm or greater in the optical televiewer data. Three of the 96 fractures in borehole UNB2 were found to have significant apertures, ranging from 2 to 9 mm. We note that fractures occur at or near the top of each of the three shale layers encountered in borehole UNB2. Given the abrupt increase in borehole water conductivity observed between 35 and 40 m depth, we suspect that the fractures occurring at the tops of the shale layers at 41 and 52 m depth could be the dominant sources of saline water in this borehole.

Combining the above observations from ERT surveying, geological and geophysical logging, and water conductivity logging, we can interpret the highly conductive area in the deeper part of the ERT section as a seawater wedge extending inland beneath the coast.

3.3.2 Wellfield ERT Survey

The ERT section for Line 4, located within the well field, is shown in Figure 7 with geological logs for four wells, including PW1 and UNB1, overlain on it. The geophysical logs for borehole UNB1, drilled to a depth of 88.54 m, are also shown. From these data a reasonable interpretation of the subsurface can be rendered. The ERT section reveals the upper and lower sandstone aquifer as two resistive layers with a more conductive unit between the depths of 20 and 35 m. Comparing the ERT section to the UNB1 geological and resistivity logs it appears that this conductive layer is an artifact of the ERT inversion code's inability to resolve the two shale layers separately. Another point of interest on the ERT section is the drop in resistivity observed at a depth of approximately 70 m. This drop can be explained, in part, by the presence of another shale to shaley sandstone layer at depth of 67 – 78 m, characterized by low resistivities of 25 – 55 Ohm m in the geophysical well logs. However, it is also a consequence of the fact that sandstones below the shale have an average resistivity that is at least a factor of two lower than the resistivity of the sandstones above. Comparing gamma emissions in the sandstones immediately above and below the lower shale shows that there is no reason to expect that they differ in clay content. Furthermore, lab measurements (not shown here) reveal that sandstone porosities above and below the shale are very comparable. As a result, it would appear that pore waters beneath the lower shale may be twice as saline as those above that layer.



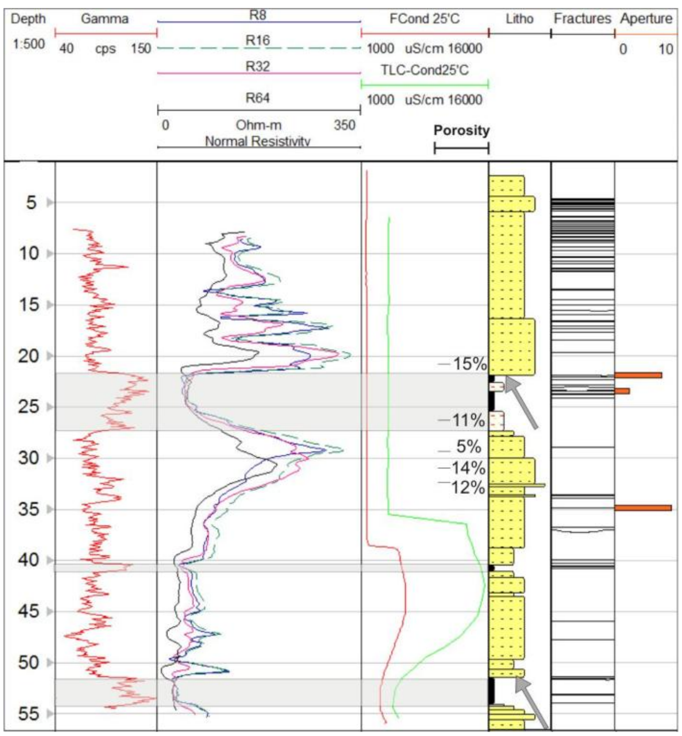
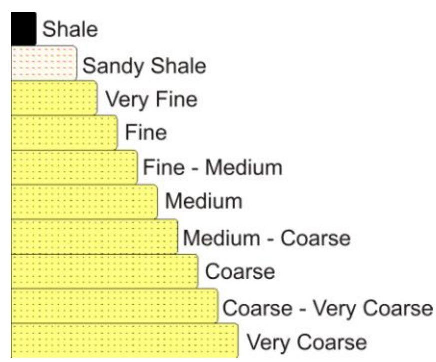


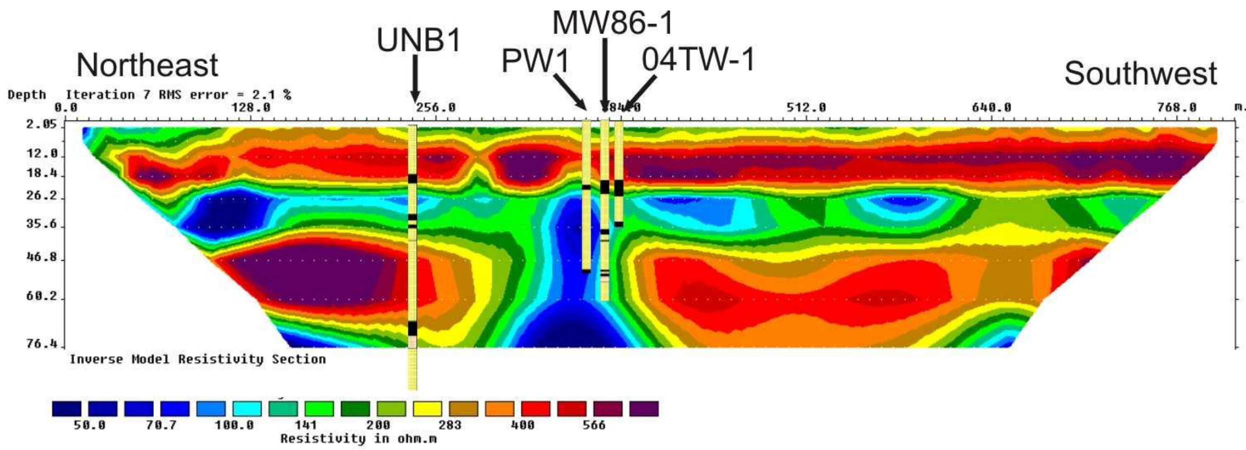
Figure 6. *Above*: ERT section 1 - an estimate of the true resistivity structure beneath Line 1 suggesting the presence of a saltwater wedge coming in from the coast. *Left*: Geophysical and geological logs for borehole UNB2 intersected by the ERT survey. Grey shading is used to highlight the shale layers and grey arrows indicate two fining upward sequences of grain size. *Below*: A legend for the geological log showing the three different types of lithologies and the different grain sizes for the sandstone.



The most prominent anomaly on the ERT section in Figure 7 is the low resistivity zone below PW1. This anomaly does not conform to the layered strata known to exist in the area. Forward modelling exercises (Mott et al., 2011) have demonstrated that the anomaly cannot be explained by variations in the shale thickness observed in the area or by the 26 m long steel well casing for PW1 located 17 m off-line. Instead, the most likely explanation is considered to be upconing of saline water from depth. The presence of more saline water at depth below the lower shale layer supported by the low resistivity layer present at the base of the ERT section and by resistivity logs from borehole UNB1 as outlined above. While the shale layer would be expected to impede upward water movement, its vertical permeability may be increased by moderately to steeply dipping fractures like the one found at 23.05 m depth (dipping 53°) in borehole UNB2 (Figure 6) and the fracture found at 69.05 m depth in borehole UNB1 (Figure 4).

4. CONCLUSIONS

ERT surveys results suggest that elevated groundwater salinities in the Richibucto wellfield are likely a consequence of salt water upconing beneath pumping wells. The presence of increased salinities at depth is confirmed by the observation that sandstones underlying a shale layer, approximately 80 m deep in borehole UNB1, have relatively low resistivities that cannot be readily explained by an increase in clay content or porosity. ERT surveys and borehole



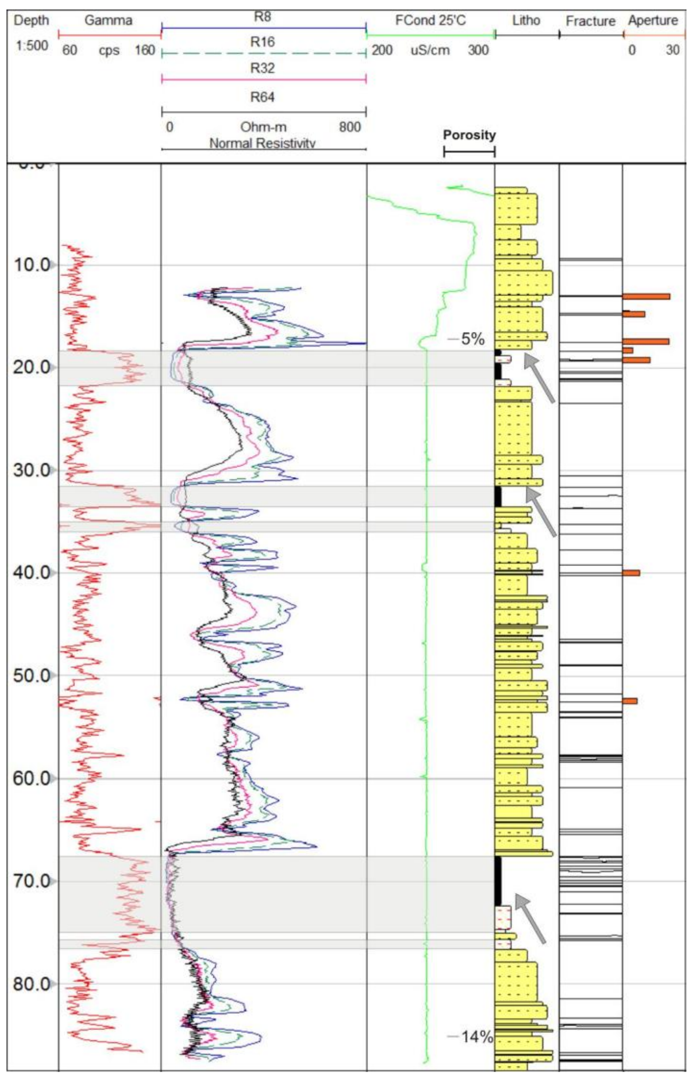


Figure 7. *Above*: ERT section 4 - an estimate of the true resistivity structure beneath Line 4 revealing a predominantly layered subsurface modified by the possible upconing of saline water beneath PW1. The locations and geological logs for boreholes UNB1, PW1, MW86-1, and 04TW-1 are also shown. *Left*: Geophysical and geological logs for borehole UNB1, annotated with measured sandstone porosities at two depths. Grey shading highlights the shale layers and three grey arrows indicating fining up sequences of grain size.

geophysics also reveal a salt water wedge extending inland from the coast in a very low-lying area. These results show that ERT surveys can be an effective tool in identifying the extent and modes of salt water intrusion into the similar Carboniferous sandstone aquifers that underlie many coastal communities in eastern New Brunswick. Repeat surveys may be used to assess changes in the extent of salt water intrusion that may accompany climate change and sea level rise or other stressors such as increased pumping. Ongoing work, including (i) an examination of the depth of investigation of ERT surveys in this environment, (ii) forward modelling to determine the method's relative sensitivity to variations in groundwater salinity versus variations in shale layer thickness, and (iii) lab measurements on drill core samples to determine the relationship between pore water salinity and bulk sandstone resistivity will serve to refine and strengthen these assertions.

Case Study:
An evaluation of the effects of climate change on seawater intrusion in coastal aquifers: a case study from Richibucto, New Brunswick

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1. INTRODUCTION

Lower population densities along Canadian coastlines, compared to those of many other countries, means that the problem of seawater intrusion is seldom exacerbated to the extent commonly reported in the scientific literature. However, the history of local well fields such as the one in Richibucto, New Brunswick, demonstrate that this problem does exist in the Atlantic region. Municipal wells have been used to supply water for the town (population of approximately 1300) since 1991. During the past 10 years, some of these wells have experienced increases in chloride concentrations suggestive of seawater intrusion (e.g. Maritime Groundwater Inc. 2004; Stantec Limited 2009). The town currently supplies about 600 m^3 day^{-1} from three wells; one located in an unconfined sandstone aquifer that is about 20 m in thickness, and two located in a deeper (~ 40 m below sea level) confined or semi-confined sandstone unit.

The objective of this study was to assess the effects of climate change on the coastal aquiferutilized by the Town of Richibucto. A conceptual model was developed based on detailed topography and bathymetry, stratigraphic information from borehole and geophysical investigations, and hydrogeologic properties from previous investigations (e.g. Maritime Groundwater Inc. 2004; StantecLimited 2009). To simulate the possible future evolution of conditions in the aquifer, a three-dimensional variable-density groundwater flow modelcoupled with solute transport was developed. Climate change was considered by investigating the effect of changes in recharge and sea level boundary conditions on the groundwater salinity through the year 2100. Potential increases in the pumping rates from the municipal well field were also investigated.

Although there have been several recent studies dealing with the effect of climate change on coastal aquifers (e.g. Feseker et al. 2007; Carneiro et al. 2010; Oude Essink et al. 2010; Werner and Simmons 2009), this is one of the first case studies to be conducted in Atlantic Canada. The southern Gulf of St. Lawrence coastal area is generally characterized as being highly sensitive to sea level rise (Shaw et al. 1998), and the adjoining Maritime provinces have a large percentage of the population that is reliant on groundwater. Because many similar groundwater supplies exist in the Carboniferous sedimentary rocks along the shoreline of the Northumberland Strait in New Brunswick, Prince Edward Island, and Nova Scotia, it is expected that this study will provide useful insight for other locations in the region with similar hydrogeological conditions.

2. STUDY AREA DESCRIPTION

The conceptual hydrogeological model for the Richibucto area includes a number of stratigraphic units. The Saint Charles peat deposit located in the centre of the study area consists mainly of poorly humified (i.e. poorly decomposed) peat of Sphagnum genus, with well-humified peats making up less than 10 to 15% of the deposit (New Brunswick Department of Natural Resources 1993). The thickness of this surficial layer varies from 0 m to 7.6 m with an average of approximately 2 m. Underlying the peat deposit, and covering much of the rest of the region, is a layer of poorly sorted Wisconsinan aged sandy till with an average thickness of 3 m.

The sole source of groundwater in the study area is the Carboniferous-aged Richibucto Formation that underlies the surficial materials. This predominately grey and minor brownish red, trough cross-bedded medium- to coarse-grained sandstone includes discontinuous inter-stratified red and grey very fine-grained sandstone, siltstone and mudstone layers (St. Peter and Johnson 2009). The channel fill sandstones can be upwards of 20 m thick, and the overbank finer sandstone, siltstone and mudstone are typically only a few metres thick (van de Poll 1973). This model of aquifer architecture is supported by pumping tests that reveal evidence of discontinuous layers of lower permeability sediments (e.g. Rivard et al. 2008).

Recharge to the groundwater system is provided by rainfall and snowmelt infiltration. Jacobs (2011) modeled the recharge to the Richibucto aquifer using past climate data and the characteristics of the surficial sandy loam and peat materials with the United States Environmental Protection Agency code HELP (Schroeder et al. 1994).Based on climate parameters averaged from 1971 to 2000, values of recharge were predicted to be, on average, 322 mm $year^{-1}$ and 331 mm $year^{-1}$ through the peat and sandy loam materials, respectively (Jacobs 2011).

3. NUMERICAL MODEL

Although analytical solutions exist for approximating the location of the freshwater-seawater interface (FSI) (e.g.Bobba 1993; Mahesha and Nagaraja 1996; Werner and Simmons 2009), they are limited to conditions of homogeneous hydraulic conductivity, simple domain shapes and boundary conditions, and are often two-dimensional. Numerical models are better suited for predicting the salinity distribution in more complicated hydrogeological settings such as encountered in the current study. Although several numerical models are available to simulate density-dependent groundwater flow and solute transport, the SEAWAT code (Guo and Langevin 2002; Langevin et al. 2003; Langevin et al. 2008) was selected for this investigation because it is well established and has been frequently applied to coastal aquifers to predict the position of the FSI (e.g. Scheider and Kruse 2005; Lin et al. 2009; Chang et al. 2011; Masterson and Garabedian 2007; Praveena and Aris 2010; Rozell and Wong 2010; Webb and Howard 2010; Sanford and Pope 2010). SEAWAT iteratively couples a variable density form of the groundwater flow model MODFLOW (McDonald and Harbaugh 1988; Harbaugh et al. 2000) and the solute transport application MT3DMS (Zheng and Wang 1998).

3.1 Grid properties

Since it was assumed that local groundwater discharge occurs to the tidal rivers that effectively surround the study area the boundaries of the model were chosen to include the natural constant head and salinity conditions imposed by the Northumberland Strait, the Richibucto River and the St. Charles River (Figure 1). The model domain was 15000 m long and 9500 m wide. Horizontal cell dimensions varied from 300 m by 300 m on the outside edges of the model, to 100 m by 100 m near the well field. This discretization scheme provided the resolution needed for capturing major surface features and for simulation of the coastal freshwater-seawater transition zone, while still providing efficient simulation times. The model cells represented the hydrogeological units as equivalent porous media and, as such, the results should not be used to infer local conditions near individual pumping wells, which may be largely controlled by fractures in the sandstone bedrock.

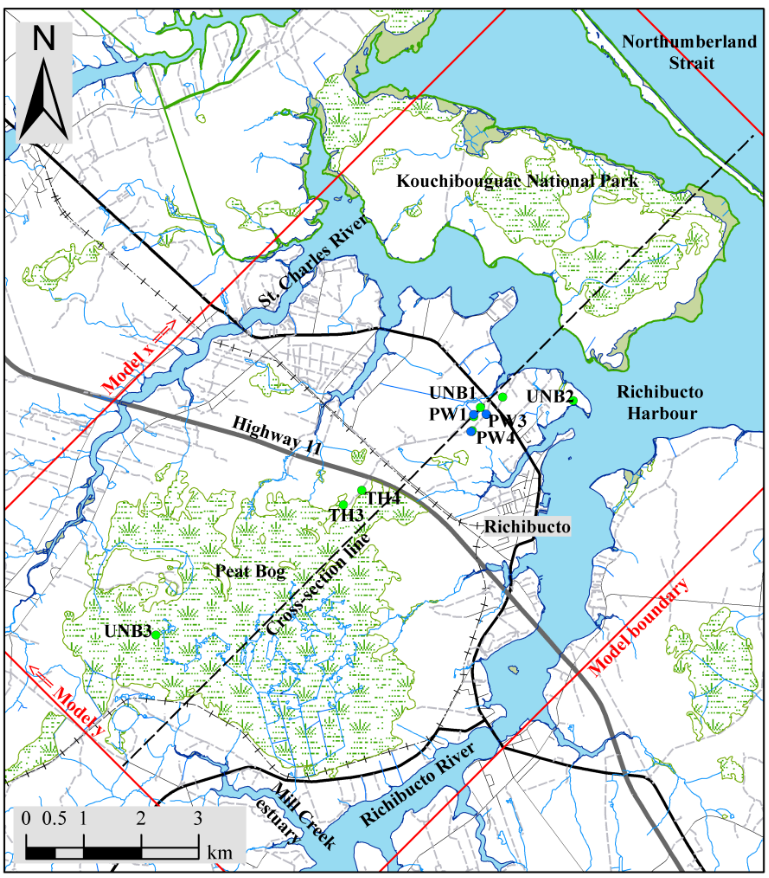

Figure 1. Regional map of the study area bounded by the tidal St. Charles and Richibucto Rivers, showing model boundaries, location of monitoring and pumping wells and location of cross-section (Figure 2).

The topography of the surface layer of the model was created from digital elevation data (Service New Brunswick 2010; Service New Brunswick 2001) and bathymetric data for the Richibucto harbour and the Northumberland Strait (Canadian Hydrographic Service 1988a, 1988b). Deformed grid layers were discretized initially to match local stratigraphy from records of the town well field (Water Management Services 1986; Maritime Groundwater Inc. 1991, 1992, 2004; Stantec Limited 2009) and 218 private water wells (Annie Daigle, New Brunswick Department of Environment, pers. comm.) and refined vertically to improve the resolution of the FSI The model domain extended to an elevation of approximately -125 m, which is thought to be the base of the Richibucto Formation (Ball et al. 1981).

The hydraulic conductivity and effective porosity of the model layers are summarized in Table 1. Where stratigraphic layers pinched out or were not present, the model layer was set at 1 m thick and assigned the properties of the layer above it. Since little is known about the hydraulic properties of the sandstone at elevations other than those encountered by the well field, the hydraulic conductivity was decreased with depth due to increasing lithification and decreasing fracture frequency (Rivard et al. 2008). Due to poor knowledge of the aquifer properties in areas away from the well field, the model parameters were propagated laterally through the entire model domain (i.e. aquifer properties are constant throughout the same model layer). Other model properties and SEAWAT solution parameters are listed in Table 2.

Table 1. Model hydrogeological properties.

MODFLOW layer #	Material and label	$K_x = K_y$ (m day^{-1})	K_z(m day^{-1})	n_e	Source
1	Peat	1	0.5	0.9	based on Mesri et al. 1997; Letts et al. 2000; Carrier et al. 2002; Radforth 1977
2	Surficial till	0.2	0.1	0.1	based on Rivard et al. 2008
3	Sandstone 1	20	5	0.2	
4	Shale (C1)	1×10^{-6}	1×10^{-7}	0.35	
5	Sandstone 2	15	3	0.2	
6	Shale (C2)	1×10^{-6}	1×10^{-7}	0.35	
7-9	Sandstone 3	10	1	0.15	Water Management Services 1986; Maritime Groundwater Inc. 1991, 1992, 2004; Stantec Limited 2009
10	Shale (C3)	1×10^{-6}	1×10^{-7}	0.35	
11-13	Sandstone 4	5	0.5	0.15	
14	Shale (C4)	1×10^{-6}	1×10^{-7}	0.35	
15-17	Sandstone 5	2	0.2	0.15	
18-21	Sandstone 6	1	0.1	0.1	

Table 2. Model transport and numerical parameters

Parameter Name (units)	Value
Longitudinal dispersivity (m)	1
Horizontal transverse dispersivity (m)	0.1
Vertical transverse dispersivity (m)	0.01
Molecular diffusion (m^2 day^{-1})	0
Specific Storage (m^{-1})	0.00016
Freshwater density (kg m^{-3})	1000
Seawater density (kg m^{-3})	1025
Seawater salinity (mg L^{-1})	35,000
Change in density over salinity (-)	0.7143
Flow Solution	Preconditioned Conjugate Gradient
Head change criterion (m)	1×10^{-5}
Transport Solution	
Advection	Explicit Total Variation Diminishing (TVD) solver
Dispersion and source/sink terms	Implicit Generalized Conjugate Gradient solver
Initial time step	Automatically calculated
Maximum time step	Automatically calculated based on stability criteria for TVD scheme

3.2 Boundary and initial conditions

A constant head boundary was applied to the southwestern, or inland, boundary of the study area (e.g. left side of Figure 2). It was assigned a value so that the piezometric surface of the boundary cells represented a subdued replica of the land surface at approximately 3 m depth. In an attempt to match measured groundwater chemistry throughout freshwater portions of the domain, constant total dissolved solids (TDS) concentrations specified at this boundary varied from 200 mg L^{-1} at the surface to 400 mg L^{-1} at the base of the domain. At this location, hydraulic head and salinity boundary conditions remained unchanged throughout the climate change sensitivity simulations. Freshwater was also supplied to the model from recharge (mentioned previously).

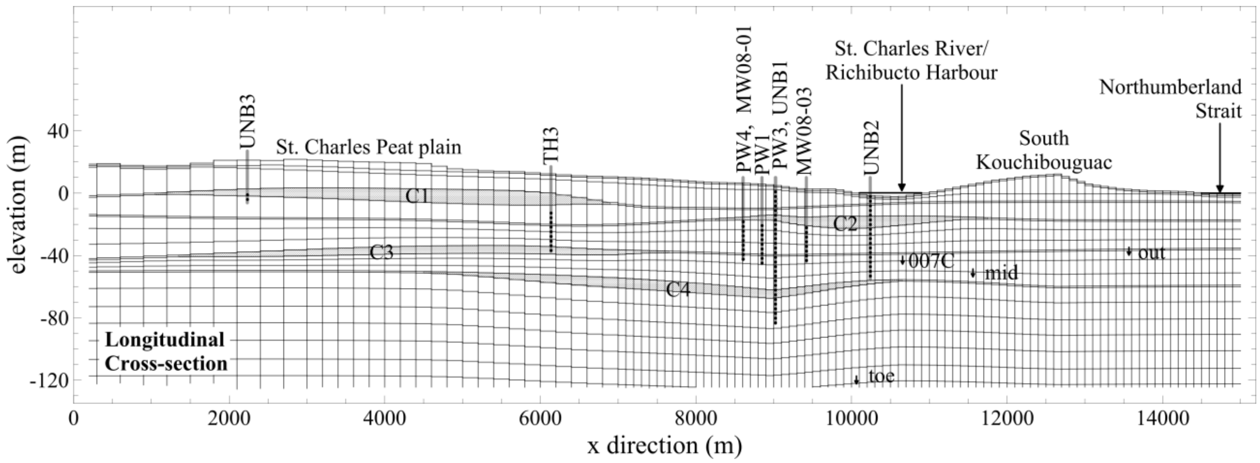

Figure 2. Cross-section of model domain showing grid discretization, shale layers (shaded grey and labelled C1-C4), well locations and surface waters (constant head above cells). Open interval of wells are shown by dotted lines. Vertical exaggeration is 20. Note that UNB2 appears to be in the harbour because of the projection onto the cross-section.

Constant head values of 0.22 m and 0.01 m were also applied to selected cells in layer 1 to represent rivers and the Northumberland Strait, respectively.. These elevations reflect the average tidal water levels (above mean sea level) as predicted by Fisheries and Oceans Canada (2010) for Richibucto Harbour and Richibucto Bar, respectively, and were confirmed by comparison with continuously monitored water levels in each river. The constant head boundaries in the rivers and harbour were also set as constant concentration boundaries. For example, salinity at the mouth of the Mill Creek estuary was set to 21 000 mg L^{-1} TDS, and this value was gradually increased to that of seawater (35 000 mg L^{-1}) offshore of Kouchibouguac National Park (Koutitonsky et al. 2004). The entire depth of the model in the eastern-most column (e.g. right side of Figure 2) was also set to a constant head and concentration equal to that of seawater. All constant concentration boundaries remained unchanged throughout the climate change sensitivity simulations.

The northwestern and southeastern sides of the model domain were assigned as zero flux boundaries. This was deemed appropriate since they are aligned roughly parallel to that of regional groundwater flow (e.g. Rivard et al. 2008), and their distance from the well field meant they should not have a large effect on this area.

Short term (i.e. daily) tidal water level fluctuations were not included in the simulations. Studies in which the influence of tidal fluctuations have been shown to be important are often performed at much finer temporal and spatial resolutions with numerical codes that can simulate unsaturated conditions (e.g. Robinson et al. 2006; Mulligan et al. 2011), or are concerned with tidal amplitudes larger than those observed at Richibucto (e.g. an amplitude of 6.4 m in Carey et al. 2009; Werner and Gallagher 2006). Because the tidal range of the Richibucto area is approximately 1.15 m, the exclusion of daily tidal water level fluctuations was not expected to have a significant influence on the long-term results that are of primary interest in this study.

3.3 Data limitations

The spatial distribution of hydraulic conductivity is one of the key controls on aquifer flow and the position of the FSI in a coastal aquifer (e.g. Ranjan et al. 2006; Ranjan et al. 2009; Narayan et al 2007; Rozell and Wong 2010; van der Kamp 1981). Although every attempt was made to accurately reproduce the location of low hydraulic conductivity layers (e.g. identified in borehole logs) in the numerical model, stratigraphic data was not available in large areas of the model domain including under the Richibucto harbour and Northumberland Strait, which are two of the potential sources of saline water. As a result, there is uncertainty in how far offshore low hydraulic conductivity layers extend. Because salt water intrusion (SWI) is especially sensitive to the sea-aquifer connections, usually associated with the presence of preferential flow paths (Carrera et al. 2010), the simulated location and shape of the FSI may be different if different shale unit extents had been used. For example, Green (2012) presents results for a simulation in which the C2 confining layer (Figure 2) did not extend as far under the harbour.

Although some spatially-limited groundwater salinity information was collected in the Richibucto area (Mott and Butler, this report), we were unable to confidently extrapolate field results to assign initial TDS concentrations to the entire domain, which is a requirement of the numerical simulations. The potential occurrence of remnant seawater, emplaced following the retreat of glaciers from this region approximately 13,000 years ago when relative sea level was significantly higher (Mott and Butler, this report; Webb 1982), has likewise not been considered in the current model for similar reasons. Instead, initial conditions were established by allowing the model to reach a quasi-steady state (or dynamic equilibrium) with prescribed present-day boundary conditions. This was achieved by running the model for very long periods of simulated time (e.g. Schneider and Kruse 2005; Bobba 2002; Carneiro et al. 2010; Comte and Banton 2006; Feseker et al. 2007; Kopsiaftis et al. 2009; Masterson and Garabedian 2007; Mulligan et al. 2011; Narayan et al. 2007; Park et al. in press; Watson et al. 2010; Yechieli et al. 2010).

The initial conditions, along with simplifications required by the conceptual and numerical model, imply that the results obtained are unlikely to accurately depict conditions at all locations in the aquifer. This uncertainty may in part contribute to the inability of the model to reproduce current groundwater TDS concentrations at several monitoring locations near the well field. The limitations imposed by sparse hydrogeological data, combined with the inherent uncertainties in predicting future climate conditions, mean that the simulation results are best suited for investigating the general response of the aquifer to changing boundary conditions, rather than predicting local hydraulic heads or TDS concentrations.

4. CLIMATE CHANGE SCENARIOS

4.1 Recharge, sea level rise, and pumping

Estimates of changes in precipitation and air temperature have been determined for the Richibucto area by the Environment and Sustainable Development Research Centre et al. (ESDRC 2010). Using data from the Canadian Climate Change Scenarios Network (www.cccsn.ca), ESDRC (2010) selected the results from 10 global climate models and two emission scenarios (A1B and A2) to calculate mean deviations for climate indices for the 2020’s (2011-2040), 2050’s (2041-2070), and the 2080’s (2071-2100). For example, a mean increase in annual air temperature of 3.7°C (standard deviation of 1.2°C) was determined for the 2071 to 2100 period. Conversely, ESDRC (2010) estimated minimal and fluctuating changes to seasonal precipitation, with mean ratios consistently between 0.9 and 1.1 (i.e. ratio of future seasonal precipitation to normal precipitation for the 1971 to 2000 period).

Jacobs (2011) used the predictions of ESDRC (2010) to estimate recharge through the peat and sandy loam surficial materials in the Richibucto area for the 2020’s, 2050’s, and the 2080’s. For the current study two scenarios were defined for recharge based on the results of Jacobs (2011): one reflecting the predicted changes developed by Jacobs (2011), and a more pessimistic one that doubled the percent change of those predictions relative to 1971 to 2000 conditions (Figure 3). It should be noted that the precipitation and temperature changes estimated by ESDRC (2010) result in an increasing recharge trend through the 2050’s; however, by 2100 there is a net decline in recharge for both scenarios relative to current conditions. For example, for the more pessimistic scenario the annual recharge rate for the peat at 2100 is 87 mm yr^{-1} less than estimated for 2011. All recharge was assigned a TDS concentration of 0 mg L^{-1}.

R.J. Daigle Enviro (2011) provided predictions of sea level rise for Richibucto based on global sea level rise trends and estimates of local vertical motion (crustal subsidence). The anticipated change in relative sea level for the Richibucto area is a rise of 0.93 m ± 0.38 m by the year 2100. As shown in Figure 4, one model scenario conformed to these predictions and one doubled these predictions for sea level in the Richibucto harbour and Northumberland Strait.

Changes in sea level were implemented consistently across all surface water boundaries. Since no information was available on how increases in sea level would be propagated inland into estuaries or harbours, it was assumed that an increase of 0.93 m by 2100 in the Northumberland Strait would result in an equivalent increase of 0.93 m in the tidal portions of the Richibucto and St. Charles Rivers. It should be noted that sea level rise was simulated by increasing the hydraulic head of spatially-constant surface water boundaries; the current study did not consider the effects of land inundation by seawater.

In the current study, the municipal wells were represented by point locations of specified flux from cells intersected by the open intervals shown in Figure 2. Future pumping scenarios were applied as shown in Figure 5. One model scenario assumed that the current pumping rates for the three municipal wells remained constant until 2100, while the other considered increased pumping based on the assumption that a portion of the neighbouring community would also be supplied from the Richibucto well field, in addition to a 0.5% per year increase in water consumption. Possible alternative well locations outside of the area of the current well field were not considered.

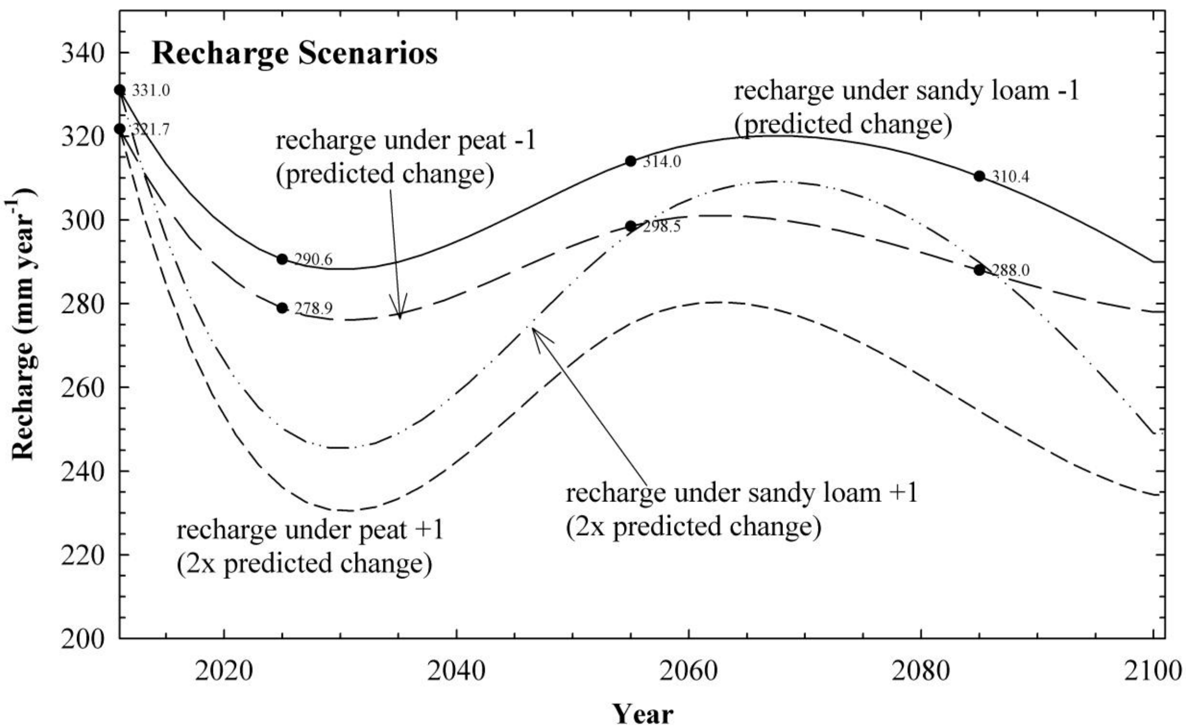

Figure 3. Groundwater recharge scenarios for different soil types. Values for current and predicted recharge presented by Jacobs (2011) are indicated by the data points at 2011, 2025, 2055 and 2085. A cubic spline function was used to interpolate. "+1" and "-1" denote high and low levels, respectively, for the recharge factor.

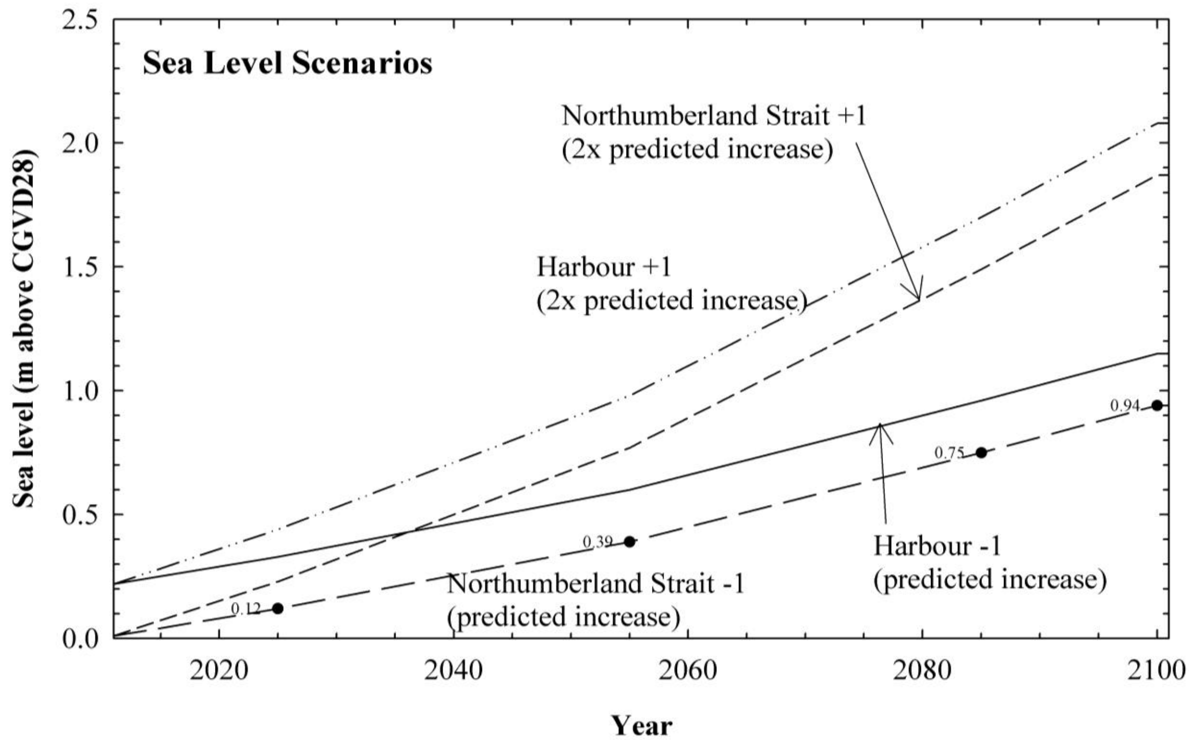

Figure 4. Sea level scenarios. Data points reflect increases reported by R.J. Daigle Enviro (2011) at times of 2025, 2055, 2085 and 2100 over current conditions. "+1" and "-1" denote high and low levels, respectively, for the sea level factor.

4.2 Simulation design and analysis

The relative importance of changing groundwater recharge, sea level and municipal well field pumping was assessed using a 2^3 factorial analysis (e.g. Levine et al. 2001; Green 2012). This analysis evaluates the results from a series of experiments (i.e. simulations) to determine the effect of increasing individual factors, or combinations of factors (interaction effects), from a "low" setting to a "high" setting. In this case, the factors considered in the numerical simulations were the transient boundary conditions of recharge, sea level, and pumping rates. The "low" setting was chosen to be the current conditions, while the "high" setting was the predicted future conditions (Table 3). Since two future scenarios for recharge and sea level were tested, this required two separate factorial analyses: Group I investigated the increase of factors from current to less pessimistic future conditions, and Group II investigated the increase of factors from current to more pessimistic future conditions (Table 3 and Table 4). The result, or response variable, used to assess the effects of the changing boundary conditions was the maximum increase in TDS at specific monitoring locations (i.e. "toe", "mid", "out", "007C" in Figure 2) during the simulation period of 2011 to 2100. Location "007C" was chosen to act as a sentinel location for seawater intrusion into the well field, while the "toe", "mid" and "out" locations were chosen to monitor the effects along the FSI. In all, two factorial analyses were performed for each location, for a total of eight analyses.

Table 3. Coding used for 2^3 factorial experimental design (see Table 4).

Factor	Group I Simulations		Group II Simulations	
	"Low" level (-1)	"High" level (+1)	"Low" level (-1)	"High" level (+1)
A: Recharge (Figure 3)	Current	Predicted change	Current	2x predicted change
B: Sea level (Figure 4)	Current	Predicted increase	Current	2x predicted increase
C: Pumping (Figure 5)	Current	Predicted increase	Current	Predicted increase

Table 4. 2^3 scenarios for studying the effects of recharge, sea level and pumping on TDS. "+1" denotes the high level of the factor, while "-1" denotes the low level. Note that Simulations 1 and 9, and 5 and 13 are identical.

Simulation Number		Factors and Levels		
Group I	Group II	A: Recharge	B: Sea level	C: Pumping
1	9	-1	-1	-1
2	10	+1	-1	-1
3	11	-1	+1	-1
4	12	+1	+1	-1
5	13	-1	-1	+1
6	14	+1	-1	+1
7	15	-1	+1	+1
8	16	+1	+1	+1

5. RESULTS AND DISCUSSION

5.1 Development of initial conditions for climate change simulations

As previously mentioned, initial conditions for the climate change simulations were developed using present-day boundary conditions, and running the model to quasi-steady state. This was accomplished by assigning initial conditions of freshwater in the entire model domain and, using present-day recharge rates and sea level, running a transient flow and transport simulation for 5000 years until a stable TDS distribution and mass were obtained. To obtain quasi-steady state conditions under the influence of pumping, an additional 120 year simulation including pumping from the municipal

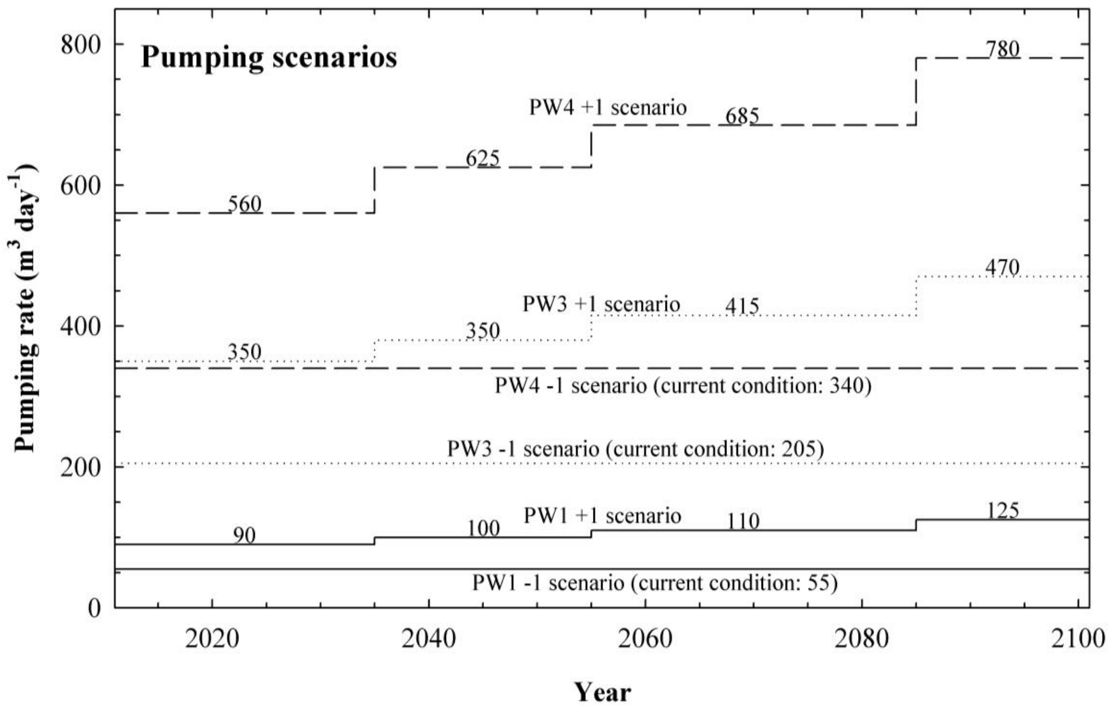

Figure 5. Pumping scenarios for municipal wells PW1, PW3 and PW4. In the scenario with increased pumping rates, the increases were applied stepwise so that rates during 2085 to 2100 were 2.3x the current values. “+1” and “-1” denote high and low levels, respectively, for the pumping factor.

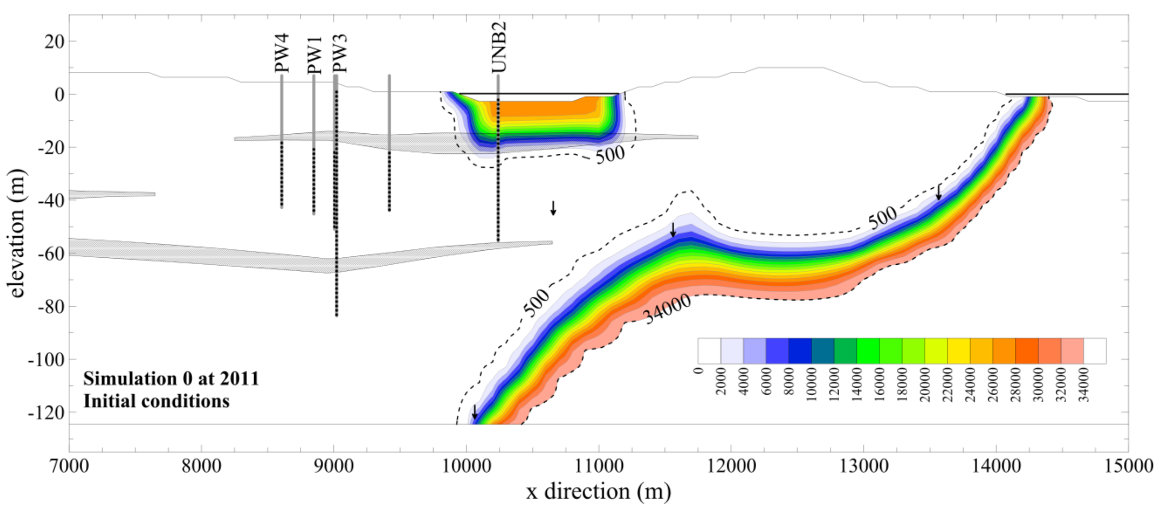

Figure 6. Initial TDS conditions for all simulations. Note that the results are for an enlarged portion of the model cross section shown in Figure 2. The source of the shallow zone of elevated TDS is the Richibucto harbour, while the deeper salt water wedge represents intrusion from the Northumberland Strait. Vertical exaggeration is 20.

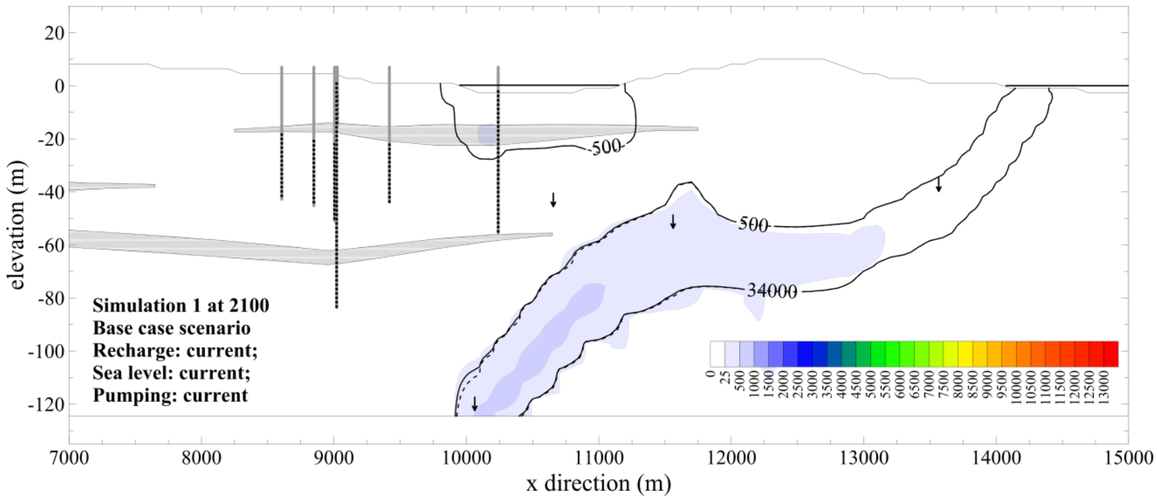

Figure 7. Base case scenario results for TDS. The colour scale shows the ΔTDS at the end of the simulation (2100) compared to the initial conditions (Figure 6). The dashed concentration contour lines represent the absolute TDS for the initial condition (Figure 6), while the solid concentration contour lines represent the absolute TDS at end of the simulation. Vertical exaggeration is 20.

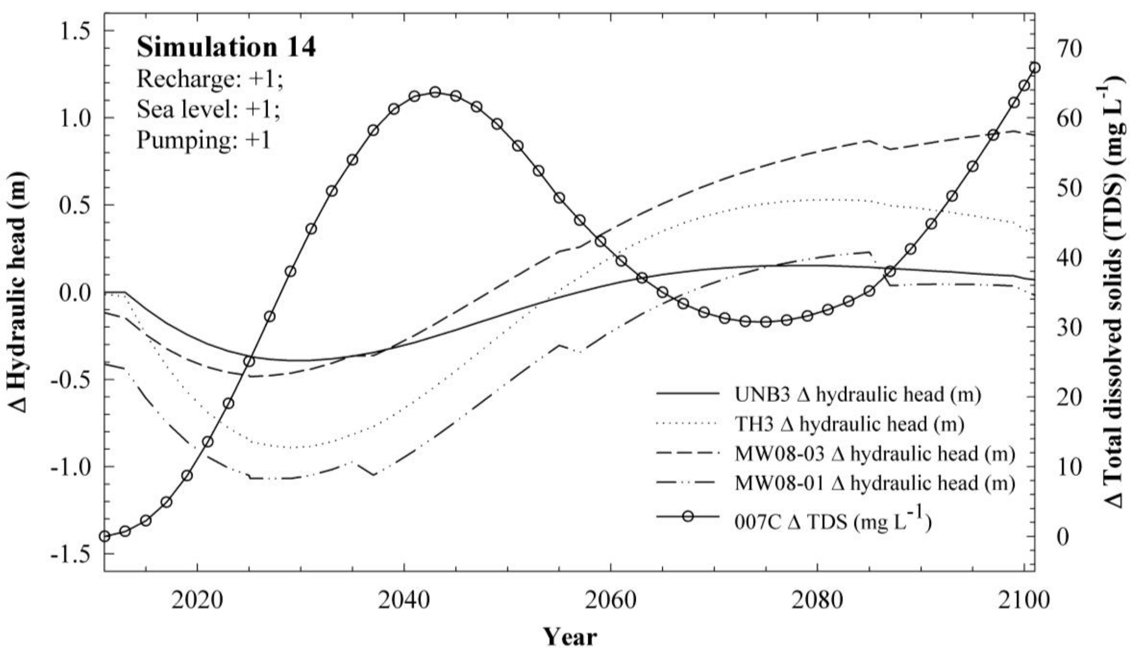

Figure 8. Example of the temporal variations of Δhydraulic head at selected locations and ΔTDS at 007C during Simulation 14 (Table 4). Note that this particular simulation had the combination of the highest variations in recharge, the highest sea level rise, and increased pumping rates.

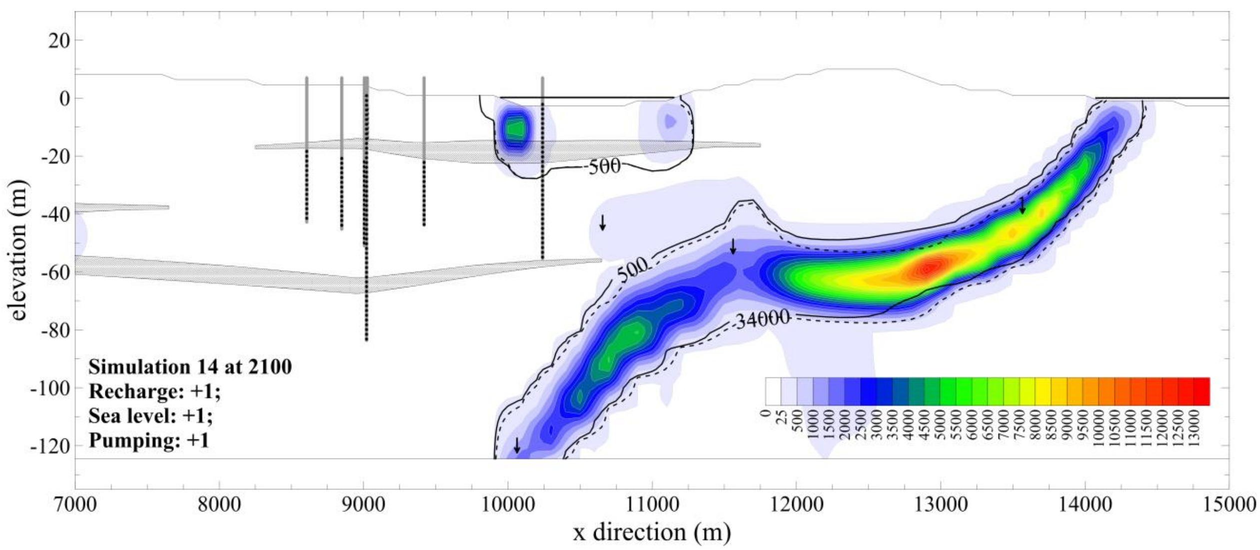

Figure 9. Example of the spatial changes in TDS for Simulation 14 (Table 4). The colour scale shows the ΔTDS at the end of the simulation (2100) compared to the initial conditions (Figure 6). The dashed concentration contours represent the absolute TDS for initial conditions (Figure 6), while the solid concentration contours represent the absolute TDS at the end of the simulation. Vertical exaggeration is 20.

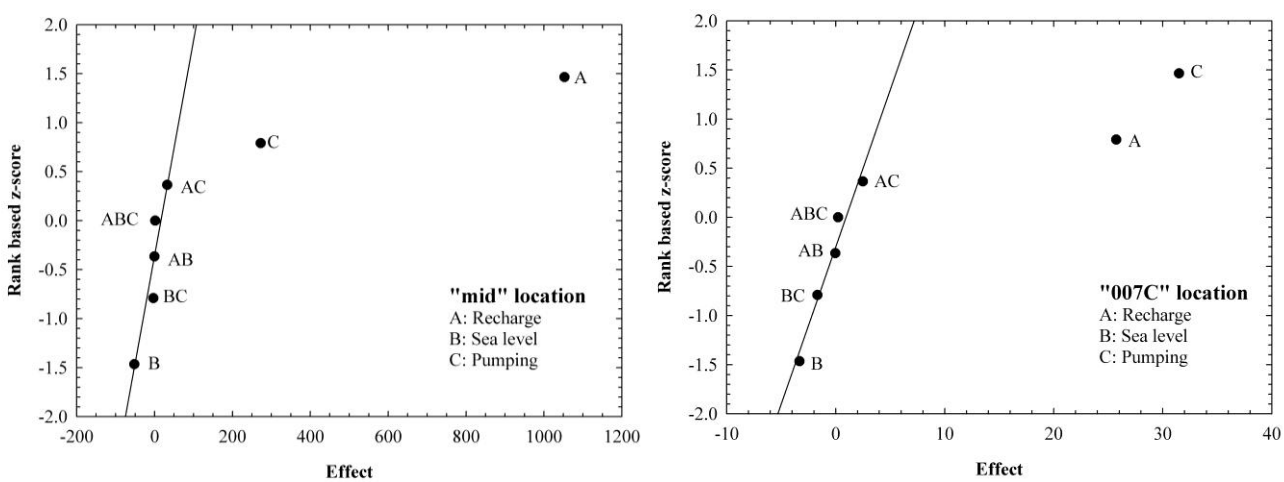

Figure 10. Normal probability plot of the effects of the factors on ΔTDS for location "mid" and "007C" (see Figure 2 for locations.

wells was then performed. A generally favourable comparison between simulated hydraulic heads and available field data was found (Green 2012). The spatial distribution of TDS, at the end of the 5120 year simulation, is shown in Figure 6. A base case simulation (Simulation 1) was then run to produce results for the period of 2011 to 2100 using the Figure 6 results as initial conditions and with the present-day boundary conditions unchanged. When the initial and final conditions for this simulation were compared, it was found that there were minor TDS changes occurring within the area of the FSI. To remove the influence of these changes from the simulations that included climate change scenarios, the results for Simulation 1 were subtracted from the results obtained for each of the Simulations 2 to 8, and 10 to 16. Thus, TDS changes (ΔTDS) were obtained relative to those assuming present-day boundary conditions during the period 2011 to 2100 (i.e. Simulation 1).

5.2 Climate change scenarios

Several key locations ("007C", "mid", "out" and "toe"; were chosen to illustrate the temporal variations in TDS through each simulation (e.g. Figure 7); spatial changes in TDS were also visualized along cross sections (e.g. Figure 8). It can be noted, by comparing Figures 7 and 8, that the temporal trends presented for the four monitoring locations do not necessarily represent the locations of maximum TDS change. For example, the maximum TDS change of approximately 13,000 mg L^{-1} for Simulation 16 occurs within the FSI at an elevation of about -60 m (Figure 8).

Table 5 presents the maximum ΔTDS for each simulation at the four monitoring locations. The results show that for the "007C" location the maximum changes in TDS were less than 100 mg L^{-1}, while changes as large as 5600 mg L^{-1} were obtained at the "out" location. As expected, the Group II simulations produced larger TDS changes because larger changes in sea level and groundwater recharge were imposed.

Table 5: Maximum change in TDS (mg L^{-1}) for the four selected monitoring locations through the 89 year simulation period. Note that Simulations 1 and 9, and 5 and 13 are identical and that the results are normalized to those for Simulation 1.

	Simulation	"007C"	"toe"	"out"	"mid"
Group I maximum ΔTDS results	1	0	0	0	0
	2	14	223	2031	846
	3	-3	1051	8	-48
	4	13	1307	2025	794
	5	27	-178	18	227
	6	49	48	2056	1093
	7	22	869	25	170
	8	44	1122	2052	1037
Group II maximum ΔTDS results	9	0	0	0	0
	10	38	463	5631	1858
	11	-6	2179	15	-97
	12	35	3116	5622	1766
	13	27	-178	18	227
	14	76	287	5664	2173
	15	18	1989	31	111
	16	67	2857	5654	2067

The results for the factorial analyses, including normalization of the results presented in Table 5 and probability plots of the effects, are presented by Green (2012). The factorial analyses indicated that all three of the factors investigated were significant to some extent, but their relative importance changed with location (i.e. "007C", "mid", "out" or "toe"). The effect of variations in groundwater recharge was greatest at shallow to intermediate depths (i.e. less than 60 m below sea level) in the aquifer, which are the depths typically targeted for groundwater extraction (Maritime Groundwater Inc. 1992; Stantec Limited 2009). Sea level rise appeared to be a significant factor only near the toe of the salt water wedge that is associated with intrusion from the Northumberland Strait.

The effect of the pumping rates was most significant relatively close (< ~ 2 km) to the well field (i.e. "007C", Figure 2). At this location the maximum increase in TDS was 76 mg L^{-1} for the parameter combination of Simulation 14 (Table 5). This represents a relatively small increase in TDS compared to the TDS range of 21,000 to 35,000 mg L^{-1} in the FSI; however, increases of this magnitude may be associated with increased production of brominated disinfection by-products in treated drinking water (e.g. Zhai et al. 2010; Sohn et al. 2006). No significant seawater intrusion into the well field was found for the current set of simulations.

The spatial variability concerning the relative importance of groundwater recharge, sea level rise, and pumping is likely related to the type of hydraulic control on the freshwater flow. Head controlled freshwater systems are dominated by specified head, or Dirichlet, boundaries (e.g. non-tidal surface water features). Flux controlled systems are dominated by specified flux, or Neuman, conditions (e.g. recharge). Numerical models of confined and unconfined aquifers dominated by flux controlled freshwater boundaries have been found to be relatively insensitive to the effects of sea level rise when compared to those where freshwater flow is controlled by specified head boundaries (Werner and Simmons 2009; Chang et al. 2011).

Because the tidal rivers in the current numerical simulations are connected to the Northumberland Strait, their elevations were assumed to increase at the same rate as the sea level rise predictions. This resulted in similar increases to freshwater heads in the interior of the model domain, indicating that the model tends to be flux controlled. This may explain the relative insensitivity of the TDS results to sea level rise. For the majority of the modeled aquifer thickness, it appears that sea level rise by itself is not causing a decrease in the freshwater flow. This factor only becomes significant at a depth where the influence of recharge flux diminishes and constant head boundaries (simulating regional inflow of freshwater on one side and sea level on the other side) become more important.

A number of recent studies have revealed that freshwater boundary conditions can control how an aquifer will respond to climate change, and that sea level rise may have a relatively insignificant effect on the position of the FSI. For example, Loáiciga et al. (2012) used a numerical model, in which the freshwater boundaries were controlled by constant flux conditions, to model seawater intrusion near Monterey, California. They found that groundwater extraction was the predominate cause of seawater intrusion in their study area, and that sea level rise had little effect. Although Loáiciga et al. (2012) did not test the effect of declining recharge, their results support the current finding that pumping may be more significant to SWI than sea level rise. Similarly, Carneiro et al. (2010) investigated the effects of decreasing recharge and rising sea level on a surficial aquifer in Morocco. They simulated three future climate and sea level rise scenarios and found that the reduction in freshwater flow volume responsible for the increased salinity was attributed mainly to a combination of declining recharge and declining contributions from an adjacent aquifer, rather than to the predicted increase in sea level.

6. CONCLUSIONS

A conceptual and numerical model of variable density groundwater flow coupled with solute transport was developed using available hydrogeological data for Richibucto, NB. Although there are limitations with the model, notably a lack of hydrogeological properties and groundwater TDS data in regions outside the municipal well field, it is considered a suitable tool for assessing the general response of the sandstone aquifers to climate change and sea level rise.

We considered two scenarios each for future changes in recharge and sea level and one scenario for pumping for the period from 2011 to 2100. Scenarios of groundwater recharge were based on predictions made by Jacobs (2011); one scenario had a net decline in groundwater recharge of 40 mm year^{-1}by 2100, while the other had a net decline of approximately 85 mm year^{-1} by 2100.One sea level scenario considered a net rise of 0.93 m, while the other considered a net rise of 1.86 m during the simulation period. Pumping rates from the municipal well field increased by a factor of 2.3for the future scenario. Various combinations of these scenarios were simulated in two groups and the maximum change in total dissolved solids at selected locations within the model during the 2011-2100 simulation period was used as the model response.

The results indicate that the relative importance of the three factors (i.e. groundwater recharge, sea level rise and pumping) changes depending on the specific location considered. The effect of generally declining recharge is the most significant at shallow to intermediate depths (i.e. less than 60 m below sea level) along the transition zone, while the effect of increasing pumping rates was most important for a location relatively close to the well field and at a depth similar to the depth of groundwater extraction. The effect of sea level rise was found to be significant only at the much deeper inland toe of the transition zone. Not surprisingly, the more pessimistic changes to boundary conditions resulted in larger effects on the seawater distribution, but the relative significance of the factors was similar to that obtained for the less pessimistic scenarios. The variability of the most significant factors with position can at least in part be explained by the spatial variability of the controls on freshwater flow. For example, the importance of recharge diminishes with depth, while the overall hydraulic gradient imposed by the boundary conditions becomes more significant (i.e. increasing the significance sea level rise).

Although sea level rise is currently receiving significant attention in developed coastal areas because of the consequences for infrastructure and coastal erosion, this investigation suggests it has the least significant effect (of the three factors considered) on future seawater intrusion in shallow to intermediate aquifers similar to those of the Richibucto region. This finding may be encouraging to coastal water resource planners. Indeed, local and regional districts will not be able to directly control changes in climate and sea level. However, because groundwater recharge is a process that is influenced by land use, and pumping rates and locations can be managed, steps to control these two factors may prove to be beneficial in protecting coastal fresh groundwater supplies.

Prince Edward Island Case Studies

Prince Edward Island is unique among the four Atlantic Provinces, and indeed the country, by virtue of its total dependence on GW as a source of potable water. This combined with its small landmass and resulting proximity of important GW resources to the coast makes the issue of SWI a key area of concern to water managers. Salt-water intrusion has been studied by various investigators (Carr, 1968, van der Kamp, 1981, Jacques Whitford and Associates Ltd, 1990b), however the extent and severity around the province is highly variable.

While SWI has only been an issue for one municipal water supply system in the Province (City of Summerside) it has frequently been an issue for smaller residential or cottage developments, generally immediately adjacent to the shore. Somewhat ironically, some of the best information on the occurrence of SWI stems from efforts to explore for and exploit salt-water wells for use by the aquaculture sector (see for example study by Jacques Whitford Environment, 1990b). This underlines the fact that the occurrence and severity of SWI in the Province is highly variable, in spite of relatively uniform geological and climatic environments. Wells for aquaculture or fish processing purposes have been constructed that supply GW with reported salinities of up to 25 parts per thousand or more (Jacques Whitford and Associates Ltd, 1990), while other wells constructed at the shoreline have been known to produce fresh water.

The principal study area in Prince Edward Island under the current program is in the Summerside area, located on a narrow isthmus between the Northumberland Strait and the Gulf of St. Lawrence, and where GW supplies the drinking water and industrial/commercial needs of the Province's second largest city. Salt-water intrusion has, in the past, affected older municipal wells near the harbour.

The case study presented for this area, *Simulating Saltwater Intrusion in a Changing Climate, Summerside, PEI, d*ocuments the results of a program of test drilling, geochemical assessment and numerical modelling to assess the potential impact of CC with respect to sea level rise and changing recharge conditions. A brief evaluation of potential CC impacts is also described for the First Nations community of Lennox Island. Lennox Island is located just off the north coast of Prince Edward Island, and has recently upgraded its water supply infrastructure. With the communities small land area and relatively low elevation above sea level, changes to sea level and implications for water supply are of key interest to the community.

Case Study:
Simulating Saltwater Intrusion in a Changing Climate, Summerside, PEI

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1. BACKGROUND INFORMATION AND SITE SELECTION

Over the years, a growing number of geological and hydrogeological studies have investigated the Prince Edward Island's (PEI) coastal aquifer and its interaction with the sea. The geology of the province is described as an alternating succession of red beds ranging in age from Upper Pennsylvanian to lower-middle Permian (van de Poll, 1981). Surficial deposits for the Summerside area are described as a sand and clay phased ground moraine with an average thickness of less than 6 m. Off-shore deposits are of similar composition, derived from local tills and bedrock (Bartlett, 1977).

Hydrological analyses suggest the aquifer behaves in an unconfined, semi-confined manner (Carr, 1971; Tremblay et al., 1973). Predominant groundwater flow is via fractures; however, the aquifer does exhibit moderately high inter-granular porosity in the order of 17 to 23% (Delcom Consultants, 1994). Pump tests of municipal wells located throughout the province produced an average hydraulic conductivity of 1.9×10^{-5} m/s (Carr, 1969); however, values have ranged from 10^{-3} to 7×10^{-7} m/s across the island (Parsons, 1972; Carr, 1971; Lapcevic and Novakowski, 1988; Francis, 1989; Jacques, Whitford Consulting Engineers and Scientists, 1990; Paradis et al. 2006).

A study by Tremblay et al. (1973) investigated the influence of saltwater intrusion (SWI) into the coastal aquifer surrounding the city of Summerside in the late 1960s. By sampling spring-water from a number of small springs and streams in the Summerside area and by assuming that the chemical composition of the samples are influenced predominantly by groundwater base-flow, it was possible to determine a rough approximation of the spatial distribution of SWI in the area (Figure 1). At the time of sampling, spring-water from the most western and eastern streams appeared uninfluenced by saltwater contamination as suggested by the low chloride concentrations. Assuming these values represented the natural chloride content of base-flow for the area it is evident that the streams within or close to the city limits were elevated in chloride. This would suggest the presence of saltwater contamination and SWI of the shallow aquifer which was the conclusion reached by Tremblay et al., (1973). However, without further chemical analysis it is not possible to definitively reach this conclusion.

Over the past 50 years, a number of pumping wells within the city limits have either penetrated the saltwater-fresh water interface or have begun pumping brackish to saline water, ultimately leading to the abandonment of

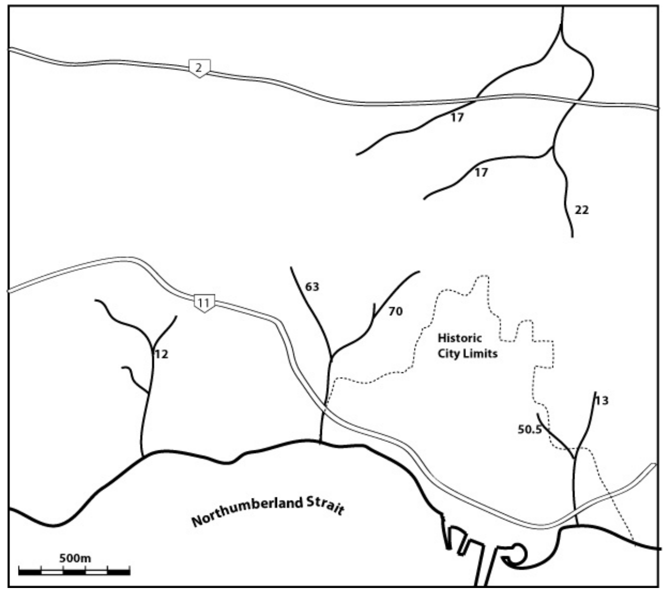


Figure 1. Location and chloride concentration (ppm) of springs and streams in the Summerside area (Derived from Tremblay et al., 1973).

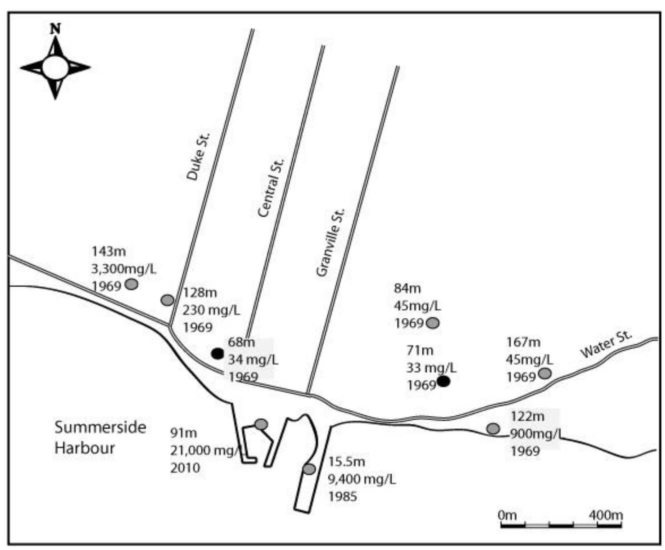
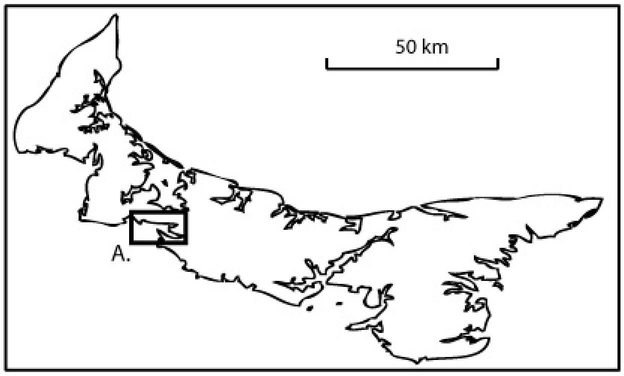


Figure 2. Approximate location of a number of pumping and observation wells within the city limits which have historically produced saline water. The solid black marks represent two industrial wells monitored during pumping by Tremblay et al. (1973). The approximate depth, chloride concentration and year of sampling is given for each well.



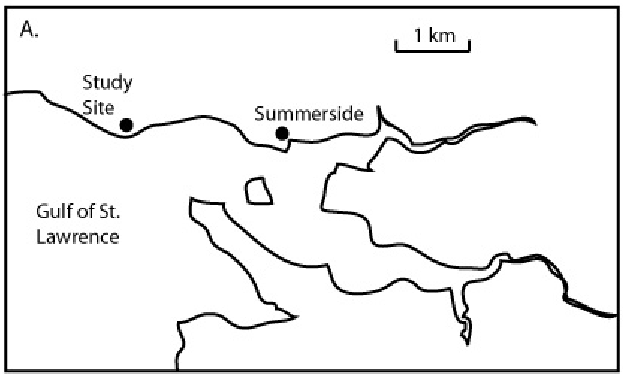


Figure 3. Map of Prince Edward Island. Highlighted box (A) indicates the position of the study site west of the City of Summerside.

two city wells. In addition to these contaminated wells, a number of saltwater wells along the coast have purposely been developed for industrial needs. The approximate location and depth of these wells along with the chloride content and year of testing has been provided in Figure 2. Chloride concentrations for the two wells identified by solid black markers (Figure 2) were monitored over 8 hours of pumping (Tremblay et al., 1973). During this extraction period the chloride concentration in the two wells fluctuated greatly, ending with the chloride concentration nearly tripling in the western well and almost doubling in the eastern well.

In addition, saltwater encroachment and up-coning has been identified and mapped within the city limits of Summerside, presumably due to high pumping rates (Tremblay et al., 1973). Saltwater intrusion in the province has also been documented in areas of minimal pumping due to tidal fluctuations, changing recharge rates and low-lying estuaries penetrating far inland. Saltwater intrusion has been discovered at Eliot River (Carr, 1969) and York Point (van der Kamp, 1981) despite low groundwater withdrawals and anthropogenic influence. A study of the Rustico Harbour identified the position of the saltwater-fresh water interface and found that the specific location was highly variable within even short distances (e.g. at one location, saline water could not be found at shallow depth, while only 400 metres away the interface was located 200 metres inland) (Rivard, et al. 2008).

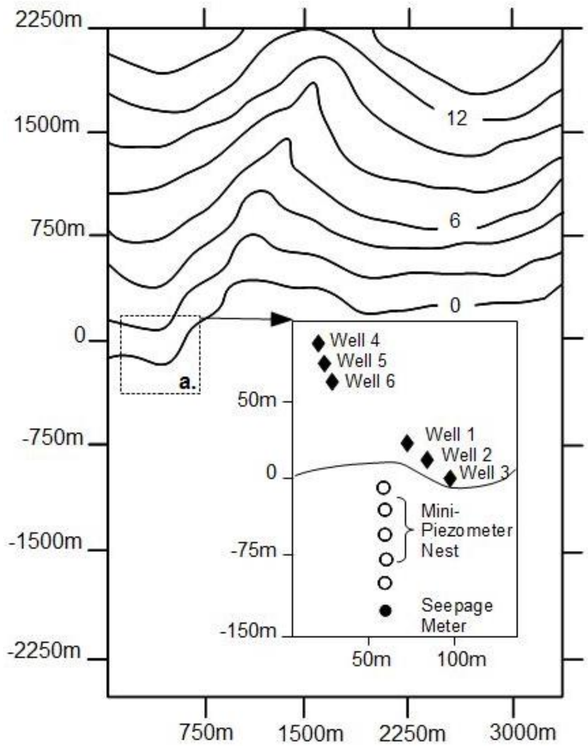
Studies have also been conducted for the province to investigate salt water wells as a commercial fishery resource (Jacques Whitford & Associates Limited, 1990a) and to evaluate the potential impact of sea level rise on such wells (Jacques Whitford & Associates Limited, 1990b). Currently, salt water wells are used commercially at both Souris and Victoria-by-the-Sea.

The chosen study site consists of a small acreage of coastal land encompassing the majority of a small watershed discharging into the Gulf of St. Lawrence at the mouth of the Summerside estuary (Figure 3). The watershed is approximately 4.9 km^2 and characterized as a gently sloping terrain reaching a maximum elevation of approximately 40 m in the northeast. The small brook comprising the watershed is an inconspicuous unnamed first order stream originating from a small marsh in the upper reaches of the watershed. The location of the site was chosen to best represent natural conditions as little anthropogenic influences exist, yet remains near historical occurrences of SWI and numerous hydrogeological analyses.

2. SITE CHARACTERIZATION

A total of six monitoring wells were installed in close proximity to the coast at varying depths (Table 1) to create a rough landward transect (Figure 4). Further inland, another two observation wells (62 m) previously drilled for the Dynasty Spa Resort continue this transect from the coast. Electrical resistivity profiles were conducted for two of the wells (wells 1 & 4) and to further characterize the geologic formation, a down-hole camera was lowered into these wells. By combining geophysical, video and field observations an accurate depiction of the lithology of the study area was developed which compared well with previous geological descriptions of the surrounding area (Tremblay et al., 1973; van de Poll, 1981).

Water samples were collected from each of the 6 coastal wells and analyzed for major ions, $\delta^{18}O$ and $\delta^{2}H$ stable isotopes. Clear trends in the major ion chemistry of the water samples depict a drastic rise in the chloride content and a reduction of calcium with depth and proximity to the coast (Figure 5). The saltwater interface was located at approximately 70 m below surface through electrical conductivity measurements and stable ion ratios (Table 2). A linear mixing relationship between seawater and freshwater was found with depth in the coastal wells by comparison of $\delta^{18}O$ with Cl, and deviations from the local meteoric water line (Leybourne, et al. 2006) in deep coastal wells further support this mixing relationship.



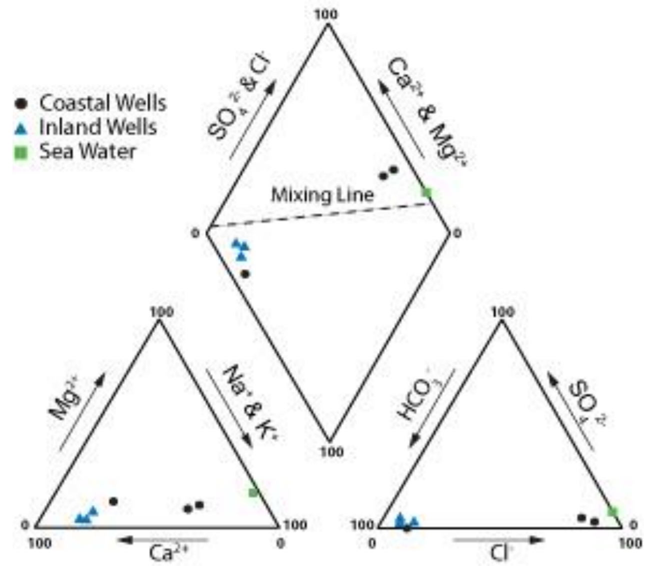


Figure 4. Model domain of the small watershed including 2m topography and a) study site including location of coastal and terrestrial wells, potentio-manometer sampling transect and seepage meter location.

Figure 5. Piper diagram depicting water chemistry of samples extracted from the observation wells and sea water. Including theoretical saltwater-freshwater mixing-line (Appelo and Postma, 2005).

Table 1. Total depth below ground level and screened intervals for the six observation wells.

Well I.D	Well Depth	Screened Interval (m)
Well #1	82.3 m	79.3-82.3
Well #2	41.8 m	38.8-41.8
Well #3	24.4 m	No Screen
Well #4	83.8 m	80.3-83.8
Well #5	26.2 m	23.2-26.2
Well #6	18.3 m	No Screen

Table 2. Ionic ratios for water samples taken from coastal observations wells on site.

Ion Ratios	Well 3	Well 2	Well 1	Seawater
Cl/Br	NA	288	293	297
Na/Cl	0.84	0.44	0.40	<0.86
Ca/ (HCO_3+SO_4)	0.23	0.98	1.04	>1.00

Hydraulic head measurements were recorded over a period of 12 months using an LTC Levelogger programmed with a 15 minute sampling interval. By averaging head throughout the duration of this study period it was determined that the predominant direction of groundwater flow was towards the coast with a hydraulic gradient of 0.002. Vertical hydraulic gradients suggest a recharging environment in both coastal and inland wells with the exception of well 5, due to the presence of a clay-stone confining layer and horizontally occurring fractures associated with bedding planes. Fractures were also identified and recorded at various depths in wells 1 and 4 though the use of the down-hole camera log.

Hydraulic head measurements varied both spatially and temporally throughout the sampling period, exhibiting various degrees of seasonal variations and strong tidal oscillations. However, the occurrence of this tidal pumping had little influence over the conductivity of the wells, with the exception of the deep coastal well (well 1).

To further characterize the hydrogeology of the study site, a number of investigations were conducted to classify the inter-tidal zone. A modified Lee type seepage meter (Lee, 1977) was constructed and installed 150 m off-shore (Figure 4) to directly measure seepage flux across the sediments for the study site. After a sampling period of 12 hours, 515 mL of water discharged into the collection bag. The diameter of the seepage meter is 87 cm and therefore, the seepage rate of the sediments 150 m offshore is 0.014 m/day. The intertidal zone located between the seepage meter and the coast was sampled using a potentio-manometer to determine the vertical hydraulic gradient 60 cm below the sediment. The results of this analysis are given in table 3. By determining the hydraulic gradient at the location of the seepage meter, the hydraulic conductivity of the sediment may be calculated using Darcy's law (1.67×10^{-7}). If it is assumed that this hydraulic conductivity measurement is representative of the off-shore sediments of the study site, the seepage rate may be calculated from each measurement of vertical gradient (Table 3).

Table 3. Offshore hydraulic gradient measurements and seepage rates calculated using Darcy's law.

Distance Offshore (m)	Hydraulic Gradient (*dh/dl*)	Seepage Rate (m/day)
30	0.22	0.013
50	0.06	0.037
75	1.33	0.082
115	0.07	0.004
150	0.12	0.007

3. NUMERICAL MODELING

A three-dimensional variable-density numerical model was constructed to simulate flow and transport in the coastal aquifer. The finite difference model was built using the USGS code SEAWAT (Langevin and Guo, 2006). SEAWAT is a coupled version of MODFLOW (Harbaugh et al., 2000) and MT3DMS (Zheng and Wang, 1999; Zheng, 2006) designed to simulate three-dimensional, variable-density saturated ground-water flow and transport (Langevin, et al., 2007) and has previously been applied to study variable-density flow problems such as salt water intrusion (Bakker 2003; Shoemaker, 2004) and submarine groundwater discharge into the ocean (Langevin 2003; Mao et al. 2006; Robinson et al. 2006).

3.1 Boundary Conditions and Model Parameters

The model domain consists of 29 rows and 16 columns, encompassing the entirety of the small watershed and extending 2.5 km into the Gulf of St. Lawrence. The x-z co-ordinate origin was located at mean shoreline at an elevation of 0 m relative to the model datum. The model domain extended 2.3 km landward and 2.5 km seaward from the shoreline at mean sea level. The aquifer depth was assumed to be 200 m, representing a shallow coastal groundwater system. Ten vertical layers were used to describe the density variations with depth. Grid dimensionality was uniform throughout the model, in exception to along the seashore where the grid was refined for more accurate measurements. The uppermost layer of grid cells was assigned elevations by importing 2 m GIS topographical data. Only steady state boundary conditions were applied to the model, fluctuations in sea level and water table due to changing tides and seasonal oscillations in recharge were neglected. A recharge boundary of 400 mm/yr was uniformly specified on all terrestrial cells forming the upper boundary. All upper-most active offshore cells were designated as constant head boundaries (0 m) along with a constant concentration boundary of 35,000 mg/L and a corresponding fluid density of 1025 kg/m^3. Initial conditions for salinity distribution for the simulated aquifer was obtained using a transient simulation from specified initial conditions and letting the model run until a stable position of the saltwater interface develops. The model was then calibrated using geochemical and head measurements from eight observation wells located throughout the watershed.

3.2 Model Parameters and Calibration

The uppermost portion of the island's red bed formation forms an unconfined/semi-confined aquifer which extends across most of the province. Numerous field investigations have been conducted to characterize the aquifer (Carr, 1969; van der Kamp, 1981; Francis, 1989) and hydraulic properties determined from these studies have been incorporated into model development to define the limits for parameter estimation during calibration. The model was calibrated against chloride concentrations and average hydraulic head values for the observation wells on site. A trial and error method was performed and recharge and hydraulic parameters were tuned to improve the match between observed and simulated values.

The simulated aquifer was assumed homogeneous with horizontal and vertical hydraulic conductivities of $4x10^{-5}$ and $4x10^{-6}$ m/s, respectively. Longitudinal dispersivity was designated as αL = 10 m and transverse dispersivity as αT = 0.05 m. The specific yield of the simulated aquifer was set to 0.1 with a specific storage value of $1x10^{-06}$ 1/L and an effective porosity of 0.17.

4. SIMULATIONS

After calibration, the model was run under transient conditions until head and concentration values stabilized. The simulated groundwater flow from the active model cells into the constant-head, constant-concentration cells is assumed to represent SGD into the Northumberland Strait. This steady-state simulation was used as the baseline simulation, representing current conditions, in which all other simulations were compared.

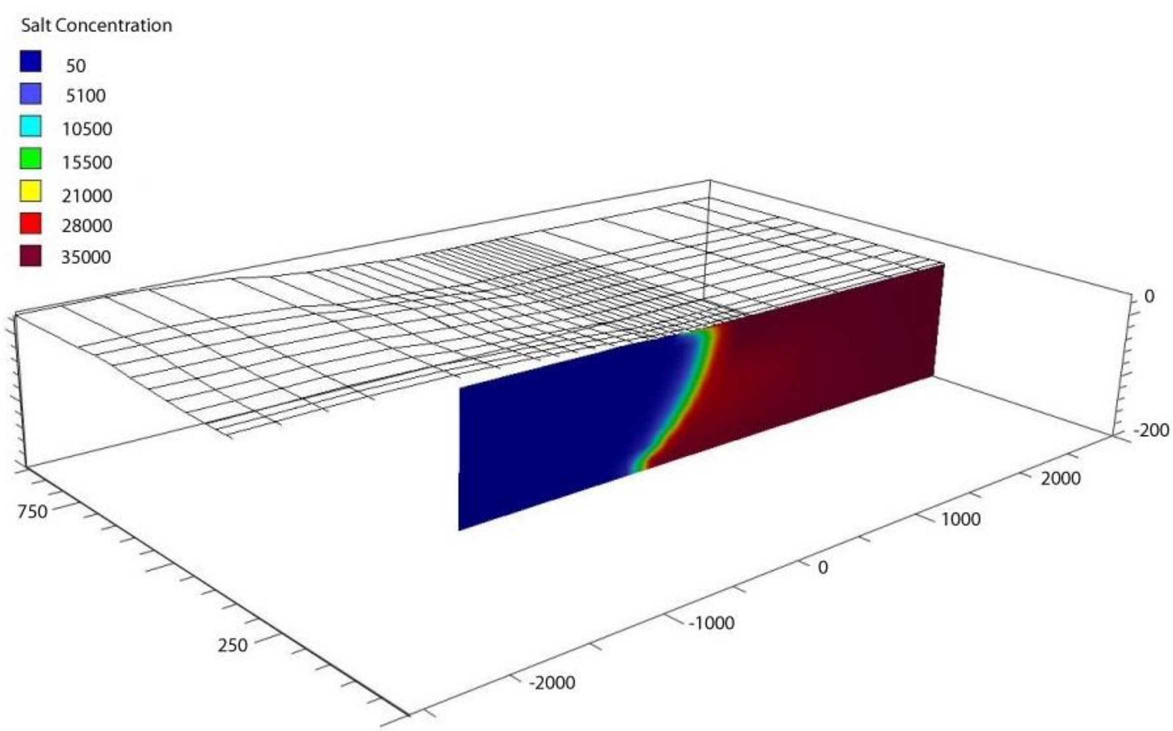


Figure 6. Model domain of the small watershed including 2m topography and a) study site including location of coastal and terrestrial wells, potentio-manometer sampling transect and seepage meter location.

Input parameters for climate change simulations were taken from the HadCM3 coupled oceanic-atmospheric climate model (Gordon et al, 2000). Four emission scenarios were chosen to give a representation of the predicted variability of current simulations (Table 4). From the A1 family, the A1F1 variant was chosen to describe the upper limit or "worst case scenario" for global emissions. This scenario describes a fossil intensive, consumer based world with rapid economic growth and globalization rates. On the opposite side of the spectrum is the B1 family, chosen to describe the "best case scenario". This scenario describes a global population working towards economic, social and environmental

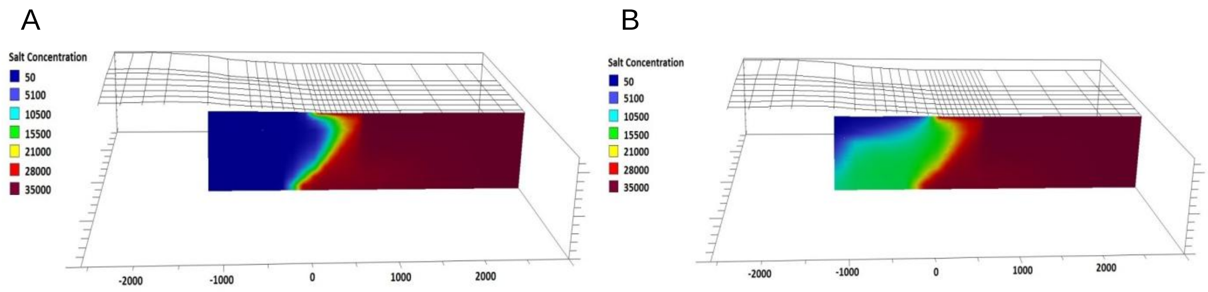


Figure 7. Comparison of baseline simulation (A) with pumping simulation (B) after 50 years of continuous pumping at 70 L/s (Concentration units are in mg/L and distance in m).

sustainability, relying on clean and efficient technologies. The middle ground was represented by the A1B emission scenario. This scenario describes a consumer based world similar to A1F1; however, with the population relying on a combination of renewable and fossil based energy sources. Finally, the A2 scenario describes a segmented world of divided, self-reliant nations with an increasing global population and a reduced rate of technical innovation. This scenario falls short of the A1F1 scenario but still represents the upper limits of potential global emissions. For each of the four emission scenarios the percent change in recharge was calculated by determining the percent change in precipitation from current conditions and assuming a similar percent change in recharge from current conditions.

For comparative purposes another climate change simulation was conducted using input data obtained through an ensemble approach for predicting future climate variables corrected for Summerside, PEI (Table 4). Data was obtained through a multi-model average of 20 climate models using the A1B and A2 emission scenarios.

Pumping simulations were conducted by importing well field data from the currently abandoned Linkletter well field, located in the North-West section of the watershed (Delcom Consultants, 1994). A pumping well was initiated at a depth of 62 m and pumped at a continuous rate of 70 L/s for a period of 50 years. Throughout the simulations, the position of the saltwater-freshwater interface was observed and recorded, along with rate of discharge to the constant head-constant concentration boundary cells.

Table 4. Summary of simulations conducted to determine the effects of climate change and pumping on SWI and SGD for the study site.

Simulation	Description	Result
Base-line simulation	Calibrated to represent present conditions	Position of the interface: 0 Total fresh SGD: 0.18 m day^{-1}
HadCM3 A1B1	-139 mm increase in annual recharge -0.59 m linear rise in sea-level	Position of the interface: +20 m Total fresh SGD: 0.24 m day^{-1}
HadCM3 A1B	-33 mm increase in annual recharge -0.48 m linear rise in sea-level	Position of the interface: 0 Total fresh SGD: 0.19 m day^{-1}
HadCM3 A2	-29 mm increase in annual recharge -0.38 m linear rise in sea-level	Position of the interface: 0 Total fresh SGD: 0.19 m day^{-1}
HadCM3 B1	-31 mm increase in annual recharge -0.48 m linear rise in sea-level	Position of the interface: 0 Total fresh SGD: 0.19 m day^{-1}
Pumping	- Pumping rate of 70 L/s for 50 years	Position of the interface:-1000 m Total fresh SGD: 0
Multi-ensemble approach	-33 mm increase in annual recharge -1 m linear rise in sea-level	Position of the interface: 0 Total fresh SGD: 0.19 m day^{-1}

5. RESULTS

Geological investigations of the study site show that the geological succession is similar to that of previous investigations in the region. Important geological features identified which pertain to saltwater intrusion are the presence of horizontal fractures and fractured bedding plains associated with clay-stone layers. These conduits can provide direct pathways for the lateral encroachment of saltwater, and therefore, increase the susceptibility of the aquifer to saltwater contamination. The influence of these fractures on hydrogeological processes is exhibited in the strong tidal signal observed in well 5. If pumping was to commence in this well or any similar well, the presence of these fractures would rapidly lead to saltwater contamination.

Results from the geochemical analysis indicate that the saltwater-fresh water transition-zone begins somewhere between 20 and 40 m below the surface and extends past 80 m in depth. The position of this interface varies slightly due to tidal influence on a daily basis and seasonally due to seasonal changes in recharge and the position of the water-table. Due to the relatively low anthropogenic influences and pumping stresses within the aquifer it is assumed that these conditions represent natural conditions for this coastal aquifer. Although it has been suggested that geological conditions are similar enough that hydrogeological interpretations for one area of the island may be applicable to other locations (Carr, 1969), the spatial variability and dynamic nature of the saltwater-fresh water interface act to discourage this practise for anything more than rough estimates for other locations in the region. However, it may be assumed that under similar conditions the shape and position of the interface will be similar for other locations within the aquifer.

Simulation results suggest that groundwater discharges into the Strait at a rate of approximately 1.2 x 10^{-6} m^3/s along the coast of the study site. Upon comparison with measured surface run-off rates for the watershed, simulated SGD constitutes approximately 13% of the total freshwater discharge to the sea. Results from this simulation indicate that the magnitude of SGD decreased exponentially with distance from the coast, with approximately 50% of total discharge occurring within the first 75 m. The position of the simulated saltwater-fresh water interface begins at approximately 30 m below surface at the shoreline, with a zone of diffusion spanning almost 150 m at the bottom of the aquifer (200 m below surface). These findings compare well to on-site measurements with SGD estimates varying within an order of magnitude between seepage meter analyses and modelling results at 150 m offshore and an R^2 value of > 0.90 between simulated and measured chloride concentrations in observation wells.

Analysis of climate change simulations indicate that alterations in sea-level and recharge rates lead to slight changes in head and the position of the saltwater- freshwater interface. The A1F1 emission scenario leads to the greatest overall change, resulting in a shift of the saltwater interface by approximately 20 m. In terms of SGD rates, varying degrees of SGD occurred under the different emission scenario simulations (Table 4). The A1F1 scenario exhibited the greatest overall change in rate of discharge (approximately 30% increase), followed by the A2, A1B and B1 emission scenarios (< 1% change). With the exception of the A1F1 scenario, the climate change simulations for 2080-2100 did not lead to a significant change in the position of the interface from the steady-state baseline simulation (Table 4).

Pumping simulations produced the greatest overall change in SGD rates for the study site (Figure 7). At a pumping rate of 70 L/s brackish water was being pumped into the well after 25 years and after 50 years the well was pumping water with salt concentrations in excess of 15,000 mg/L (Figure 7).

6. CONCLUSIONS AND DISCUSSION

The seepage rate of 0.18 m day^{-1} for the study site lies just above the reported global average of 0.1 m day^{-1} (Taniguchi et al., 2002); however, this rate may be far above the true global average as there exists a strong bias in sampling areas in which SGD is expected to be high. Regardless, 13% of total freshwater discharge is a significant contribution of the local water budget and it may be assumed that this source of water constitutes an important pathway for nutrients and potential contaminants into the local coastal aquatic environment.

In terms of the influence of climate change on saltwater intrusion and submarine groundwater discharge for the study site, the findings of this study suggest that the hydrological conditions put forth from the HadCM3 simulations of three of the four IPCC emission scenarios investigated had no significant effect. However, there was a significant change in the position of the saltwater interface for the A1F1 emission scenario, which supports the evidence that climate change has the potential to exacerbate (Sherif and Singh, 1999) or possibly even negate saltwater intrusion in certain locations. It appears however, that if the HadCM3 simulations of climate change prove accurate, climate change poses little threat in terms of the relative security of this coastal aquifer to SWI. It must be noted however that these simulations do not account for the depletion of shoreline due to coastal erosion resulting from sea-level rise, which could result in SWI and which is not investigated in this study. It must also be noted that even small alterations in the total volume of SGD may have significant effects on the total nutrient budget due to the magnitude of TDS in SGD (Krest, et al., 2000; Testa et al., 2002).

Upon comparison of climate change simulations with pumping simulations, it becomes evident that the effects of climate change on SGD and SWI are greatly shadowed by the effects of even moderate pumping conditions in coastal aquifers. However, in arid climates, the effects of climate change may be much more pronounced and in combination with the effects of heavy pumping, the outcome could be much more drastic (Sherif and Singh, 1999). Furthermore, as global population levels and densities are expected to increase in the future, it appears intuitive that coastal water resource management look not just towards the effects of climate change but rather the coupled effects of climate change and increased pumping rates.

This study gives specific information about the nature of groundwater flow into the Northumberland Strait and provides valuable insight towards the potential influence of climate change on SGD and the relative safety of the island's water resources. The results of this study indicate that if simulations prove accurate, the effects of climate change on the study site appear to have little consequences and that the effects of heavy pumping rates are of a greater concern. However, questions remain pertaining to the coastal groundwater flow pattern of the site and future work is needed to draw definitive conclusions. In the event that the Linkletter well field was put into operation, the presence of the existing observation wells would be ideal in tracking any potential changes in the position of the transition zone and SWI. Similarly, the present groundwater model used to simulate climate change and pumping simulations could easily be manipulated to determine optimal pumping rates which minimize SWI and any effects on SGD.

Case Study:
Investigation of the Risk of Salt Water Intrusion to Water Supply Infrastructure, Lennox Island, PEI

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Similar to the rest of Prince Edward Island, fresh groundwater is plentiful on Lennox Island. It is underlain by a fractured sandstone aquifer with relatively high transmissivities and numerous productive wells have been completed to depths between 8 and 30 m. The average thickness of the glacial materials is approximately 6 m. There have been few, if any, serious saltwater intrusion problems reported on the island, despite a relatively low water table. The static water level was found to be about 1 metre above mean sea level in the new well field area (Jacques Whitford Ltd., 2005).

References have been found for 17 existing wells on Lennox Island and historically another 24 wells prior to 1953. No information was obtained to indicate any wells with significant salt water intrusion. One well in the wharf area did show chloride and conductivity levels which would suggest some influence from sea water in the aquifer.

A new water supply system was recommended in the March 2007 report with wells located north of Sweetgrass Trail as shown in Figure 1. The ground elevation in this area is approximately 6.5 metres above sea level. Four production wells and 4 observation wells were drilled in 2007 and the production wells were put into service on 2008. The water system has an estimated capacity of 50,000 USGPD and the pumps are on a rotating schedule to limit drawdown times for individual wells. (Travis Dyment, personal communication, July 14, 2010).

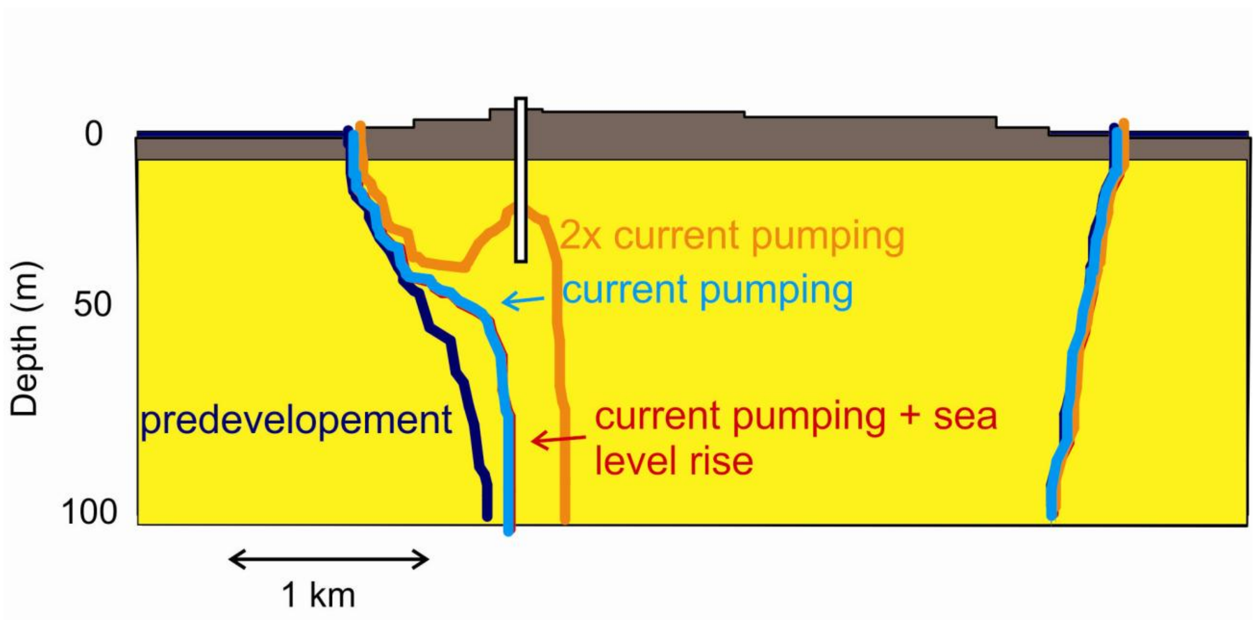

Figure 1. East-west cross-section passing through the existing Lennox Island well field. Glacial till is shown in grey and sandstone in yellow.

Models were constructed using SEAWAT, a module of MODFLOW that allows for coupling of solute transport and groundwater flow equations to simulate variable density flow. These models investigated the effects of both groundwater extraction and sea level rise on Lennox Island. There has not been extensive hydraulic testing on Lennox Island but the limited information available indicates that both the glacial till and sandstone are similar to units found throughout Prince Edward Island. In the simulations produced here, the sandstone was assigned a hydraulic conductivity of 2×10^{-5} m/s and the till was assigned a hydraulic conductivity of 10^{-6} m/s. Both of these values are within the ranges of studies examined by Hansen (2012).

The surveying associated with this data did not allow for rigorous calibration of this model but was able to produce a reasonable approximation of the position of the water table on Lennox Island and produced a saltwater-freshwater interface beneath wells known to produce freshwater under current conditions. Additional models were produced to explore the possible effects of pumping twice as much water from the aquifer and those of expected sea level rise over the next century. Increasing sea level by 1 m would have a minimal effect on the position of the freshwater-saltwater interface beyond the impact of pumping at the current rate. Doubling the amount of water produced resulted in saltwater intrusion to the point where the Lennox Island's water supply wells are likely to see an increase in salinity. Saltwater would encroach laterally and some upconing would also occur.

The findings here are quite similar to those of Hansen (2012), where it was found that groundwater use is a more important consideration in the management of saltwater intrusion issues. It should be noted that neither that study nor the current study fully investigated the impact of inundation of coastal erosion. These processes should cause a migration of saltwater inland by approximately the same amount as the movement of the coastline.

Nova Scotia Case Studies

Approximately 50% of Nova Scotians rely on GW for their water supply, with over half of the population residing within 20 km of the coastline. Salt-water intrusion into coastal aquifers driven by rising sea levels and changes in GW recharge resulting from climate change is therefore a key issue for water resource management in the province. An improved understanding of the effects of CC on water resources was identified as a strategic action under the province's recently released Water Resources Management Strategy (2010). The current distribution of salinity in Nova Scotia's coastal aquifers due to SWI, however, is not very well understood.

Although SWI problems are known to exist in Nova Scotia (e.g. Cross, 1980), few of these cases have been documented in scientific literature. Several early regional GW assessment reports attributed elevated salinity in coastal aquifers to SWI (e.g. Trescott, 1968 and Hennigar, 1964), However these interpretations were based only on the proximity of the aquifers to the coast. There are no known large scale regional SWI issues in Nova Scotia, although elevated chloride in some communities serviced by individual wells (e.g. Village of Pugwash, Porter, 1980), and in some pumping wells of municipal wellfields (e.g. Town of Pictou, Kennedy, personal communication, 2011) have been attributed to this phenomenon.

In the current study, a case study is presented that documents the results of field programs involving test drilling and hydrogeological, geophysical and geochemical techniques aimed at assessing sources of saline GW in the communities of Pugwash in Cumberland County and Wolfville in Kings County. In addition an approach to a more regional assessment of the risk of SWI of coastal aquifers in the province is presented.

Case Study:
Atlantic Climate Adaptations Solutions Field Report: Pugwash and Wolfville, Nova Scotia

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1. INTRODUCTION

Coastal groundwater supplies are particularly important in Nova Scotia. Roughly 50% of Nova Scotia's population is reliant on groundwater (Kennedy 2008), 40% are reliant on private wells, and 70% live within 20 km of the coast (Government of Nova Scotia 2005). Many small coastal communities are in rural areas that are not serviced by municipal water supplies. In these locations simple, cost effective tools are favored for sustainable management of resources, which is crucial to ensuring their availability. Population in urban areas of Nova Scotia is increasing and leading to new development, which is causing concerns over water resources in these areas as well. The population of Halifax rose 4.6% from 1996 to 2001 (Vasseur 2007). Increasing urban population means increasing density of withdrawal from groundwater, and increases the risk of saltwater intrusion (SWI) in urban areas, which could have severe ramifications.

Stress on fresh groundwater supplies and risk of SWI will be compounded in coastal regions over the next century due to sea-level rise (Melloul and Collin 2006). The IPCC (2007) estimates that global eustatic sea-level rise by A.D. 2100 will be between 18 cm to 59 cm depending on the rate at which global warming progresses. Some more recent estimates have been as high as 120 cm (Rahmsdorf 2007). Peltier (2004) estimates that present day regional crustal subsidence due to postglacial isostacy in Nova Scotia varies from south to north across the province, and is between 8 and 20 cm per century. Evidence for local subsidence in Nova Scotia is supported by Sella et al. (2007) using GPS observations, and by observations from the tide gauge at Halifax Harbour. The combined effect of eustatic and isostatic sea-level rise will be an average increase of approximately 70 cm across Nova Scotia by A. D. 2100 (Forbes et al. 2009; CBCL 2009).

Climate change may have additional effects on coastal groundwater resources. Vasseur (2007) reports that increasing intensity of storms, such as hurricanes and winter cyclones, will increase storm surges, flooding coastal areas with seawater. Coastal flooding can speed up the salinization of coastal aquifers. Vasseur (2007) also reports that, although precipitation is expected to remain the same or increase over the same period of time, more precipitation falling in briefer, more intense storms, more precipitation falling as rain during the winter months while the ground is frozen and running off, and higher temperatures year round are likely to all contribute to a decrease in total recharge in Nova Scotia. This would result in a decrease in the availability of coastal groundwater and, along with increased demand especially

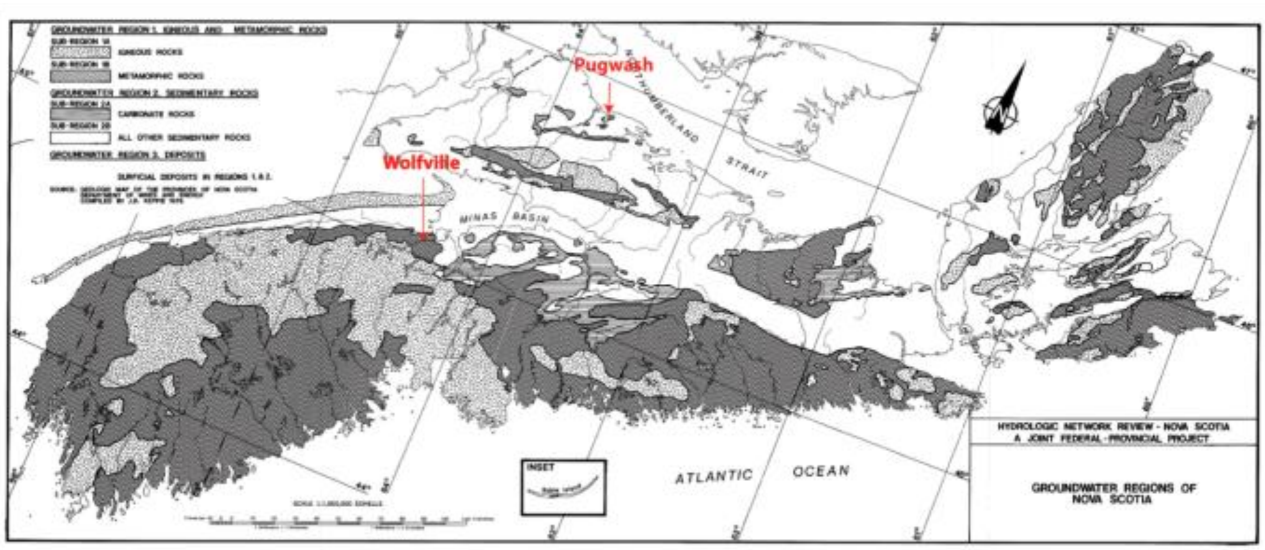

Figure 1. A map of the Province of Nova Scotia showing various hydrostratigraphic regions, and the locations of the two field sites chosen for detailed investigation for this project (modified from: NS Environment, 2009).

during the summer months when aquifers are at their lowest, further increase the risk of SWI in coastal aquifers.

Few investigations of saltwater intrusion in Atlantic Canada occur in literature. Investigation of saltwater intrusion in the Canadian Maritimes dates back to Carr's (1968) study of layered saltwater intrusion, which at the time was due primarily to large withdrawals from canning and food processing plants in New Brunswick and Prince Edward Island. Later Carr (1969), Tremblay et al. (1973), and van der Kamp (1981) found evidence of saltwater intrusion at York Point and Summerside PEI. Documentation of SWI in Nova Scotia is limited to a handful of reports by the government and by private consultants (H.J. Porter & Associates 1979; H. J Porter & Associates 1980; H. J Porter & Associates 1982).

This study attempts to gain information on the occurrence and nature of SWI in Nova Scotia in the interest of developing tools for managing the risk of SWI, and creating awareness to the problem. A focused hydrologic investigation at two sites in Nova Scotia was conducted in an effort help to broadly define factors that may be associated with increased risk of SWI throughout the province (Figure 1). Data from these investigations will be subjected to further analysis and comparison to other sites outside the province to develop tools to assist in identification of "high vulnerability" sites. These tools will be used to direct further investigation and to inform policy for management of coastal groundwater resources. Site selection was based on preexisting concern of the threat of saltwater intrusion. Two field sites, the Village of Pugwash and the Town of Wolfville, Nova Scotia were selected for a focused investigation of seawater intrusion. A combination of methods was used for this study including: evaluation of pre-existing data, installation and assessment of monitoring wells, and geochemical analysis.

2. VILLAGE OF PUGWASH

2.1 Site Description

The rural Village of Pugwash (population ~ 800) is part of the Municipality of Cumberland County on Nova Scotia's Northumberland Shore. The property of the Pugwash Thinkers' Lodge was chosen for installation of new monitoring wells to investigate the occurrence of SWI in the village. This site, which is approximately 10,000 m^2 in area, is located on a small peninsula jutting westward into Pugwash Harbour on the northwestern edge of the village near the intersection of King Street and Water Street. Ground elevation at the site ranges from sea level to approximately 7 m above sea level along a ~150 m eastward transect. There are three buildings on the site: the Thinkers' Lodge, the adjacent caretakers house, and a closed lobster cannery.

2.2 Site History

There is a history of water quality issues in Pugwash, mainly related to high salinity in private wells. This salinity has been attributed to a combination of road salting, SWI, and high-salinity formation waters associated with evaporite deposits, which are being actively mined just south of the village (Porter and Associated 1980). H.J. Porter & Associates' (1979) report on the Village of Pugwash to the Municipality of Cumberland County noted that total dissolved solids (TDS) in Pugwash from Windsor Group formation brines might cause significant contamination to cause groundwater to fail drinking water standards. They also noted that wells, which once pumped fresh water, have had to be abandoned due to saltwater intrusion.

All of the houses in Pugwash are dependent on private wells. H.J. Porter & Associates' (1980) stated that approximately one third of all private wells in Pugwash were contaminated with salt to the extent that salinity could be tasted (~250 mg/L Cl), and that high levels of salt from the Windsor Group are prevalent in the southern portion of the village (Figure 2). They also concluded that likely sources for this contamination included seawater intrusion, Windsor Group deposits, and road salting. H.J. Porter & Associates (1979; 1980) reported SWI along Water Street and, CBCL Ltd. (2005) reported that 10%-12% of wells tested in Pugwash exceed Canadian Drinking Water Standards for Na and Cl and that only 65% of residents used their wells for drinking water.

Figure 2. A aerial photo of the Rural Village of Pugwash showing the field site and the newly installed monitoring wells. The white line at the bottom right shows the mapped contact of the Cumberland and Windsor Groups (G.W. Kennedy).

2.3 Geology

The surficial geology in Pugwash is sandy glacial till, which is approximately 5 to 10 m deep (H.J. Porter & Associates 1982). There are two primary bedrock units of interest in Pugwash; the Windsor Group and the Malagash Formation. The Carboniferous Windsor Group limestone, gypsum, salt, anhydrite, shale, and sandstone, underlies the southern part of the village. It provides the ore for the Pugwash Salt Mine, which is located in the southern end of the village (H.J. Porter & Associates 1979). Formation brines from the Windsor Group are saline, with TDS above recommended limits for some solutes, and a cause for significant water quality issues in the Village of Pugwash (H.J. Porter & Associates 1979).

The primary aquifer in Pugwash is the Carboniferous Malagash Formation of the Pictou Group, which underlies the northern part of the village (H.J. Porter & Associates 1979). This sedimentary aquifer is composed mainly of alternating braided stream, arkosic sandstone, and red mudstone with rare coal and lacustrine limestone (Ryan et al. 1990b). Water quality in the Malagash Formation is better than in the Windsor Group, but high TDS also occurs as a result of groundwater flow from the Windsor Group deposits (H. J. Porter and Associates 1979). Two previous studies that have attempted to determine the hydrologic properties of the Malagash Formation in Pugwash and found the transmissivity to be between 6 and 150 m^2/day, and that the storativity was roughly 10^{-4}; indicating semi-confined or leaky aquifer conditions (CBCL 2005) (H.J. Porter & Associates 1982).

2.4 Investigation

Three new monitoring wells were installed along an east-west transect in late September 2010. Well #1 (NS-2010-TH4), Well #2 (NS-2010-TH5), and Well #3 (NS-2010-TH3) were drilled to depths of 55.7 m, 69.1 m, and 56.4 m respectively. Well #1 is furthest from the coast at approximately 100 m, Well #2 is closest to the coast at a distance of approximately 10 m, and Well #3 is approximately 40 m from the coast. Wells are located in the Malagash Formation approximately 300 m north of the contact with the Windsor Group.

2.5 Hydrogeology

Stratigraphy encountered during drilling was roughly continuous through all three wells, and consisted of upper and lower silt shale layers, with a coarser middle sandstone aquifer. Correlation of stratigraphy from the three new boreholes showed that all units dip steeply to the north/northwest. This is consistent with previous interpretations of the bedrock geology in this location, which suggests that bedding steepens progressively upwards toward the fault with the Windsor Group (Ryan et al. 1990b). Pump testing of (NS-2010-TH5) using the Theis method (Theis 1935) showed a transmissivity of ~ 30 m^2/day. A recovery test showed the hydraulic conductivity was approximately $3x10^{-5}$ m^2/s. It also showed that the effective thickness of the aquifer was approximately 12 m, which is consistent with observations made during drilling. Storativity is approximately 10^{-4} indicating semi-confined conditions. This is verified by previous studies (H.J. Porter & Associates 1982), and explains the high hydraulic head in each of the wells, which was found to be in excess of 4 m above sea level. High potential near the coast indicates that the aquifer is under confined or semi-confined conditions, and likely discharges some distance offshore (Kooi and Groen 2001). Monitoring of water level, temperature, and resistivity with data loggers is ongoing, and is being performed in collaboration between Saint Francis Xavier University, the Nova Scotia Department of Natural Resources and Nova Scotia Environment.

2.6 Analytical Solution

Kooi and Groen (2001) developed an analytical solution that very nicely describes confined flow discharging a distance offshore, and uses onshore conditions to calculate the distance offshore that the freshwater tongue reaches (Figure 3). Here it is used to characterize flow in the Malagash Formation. The following input parameters were calculated for Pugwash based on the interpretation of well logs and the results of pump testing: hydraulic gradient is approximately 0.02, the thickness of the confined aquifer is roughly 20 m, the thickness of the overlying confining layer is roughly 10 m, the horizontal hydraulic conductivity of the confined aquifer is 3×10^{-5} m/s, and the vertical hydraulic conductivity is estimated to be approximately one third of the vertical conductivity or 1×10^{-5} m/s.

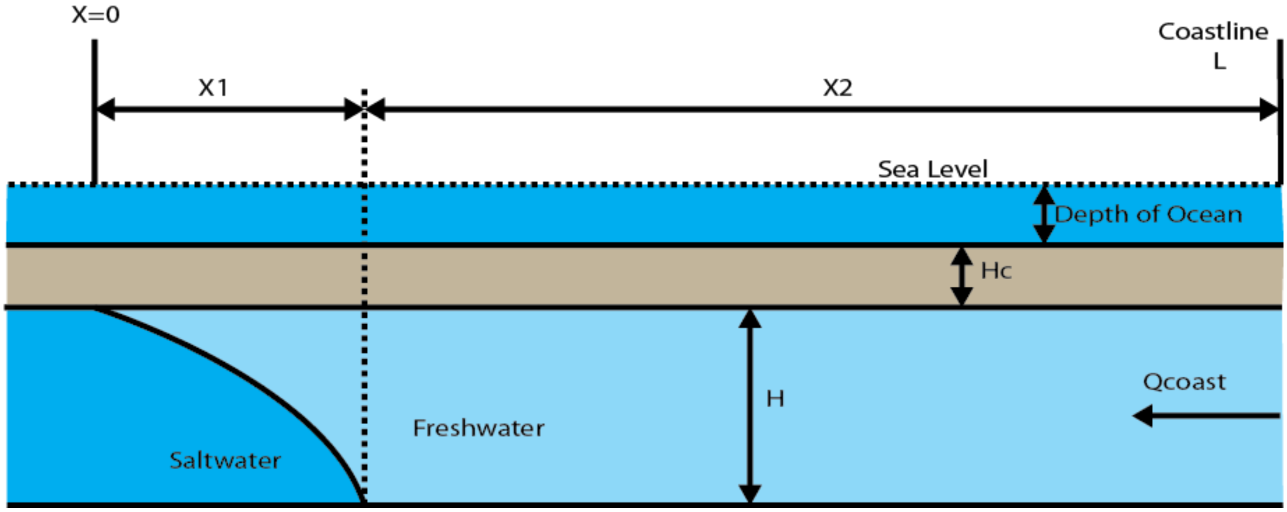

Figure 3. A conceptual diagram of the analytical solution from Kooi and Groen (2001).

Kooi and Groen's (2001) solution calculates the distance the freshwater tongue extends beyond the coastline. In the case of the Thinkers' Lodge site in Pugwash the solution shows the freshwater tongue extends approximately 200 m beyond the coast. Figure 3 shows that the seaward extent of discharge is roughly in the middle of Pugwash Harbour. With high hydraulic head at the coast, and submarine groundwater discharge (SGD) discharging under confined conditions roughly 200 m offshore there is little likelihood of SWI into the Cumberland Group aquifer under present day conditions.

2.6 Electrical conductivity

A conductivity profile was taken from all three newly installed wells in December 2010. Sharp stratification within the well at various depths can be seen. Stratification within the well should not be considered analogous to content of the formation at various depths due to the effects of mixing and density stratification of fluids in the well (Robins 1989). Values of electrical conductivity for most fresh water ranges from 0 µS/cm to 2000 µS/cm, brackish water ranges from 2000 µS/cm to 20000 µS/cm, and seawater is ~40,000 µS/cm. Readings from Pugwash ranged from approximately 500 µS/cm at the top of each well to as high as 2250 µS/cm near the bottom. This is indicative of brackish water, however geochemical analysis was necessary to determine the source of this salinity.

2.7 Geochemistry

A map of the Village of Pugwash from the H.J. Porter & Associates (1980) report shows marine salt-water intrusion affecting the Thinkers' Lodge site. This was based on Ca/Na ratios, which were then spatially approximated. Results of chemical analysis done for this study show high Ca (120 mg/L) and SO_4 (270 mg/L) not Cl (26 mg/L) or Sodium Na (40 mg/L). This along with the absence of Br, which is a conservative ion and used as a indicator for SWI (Whittemore 1988), suggests that salinity at the Thinkers' Lodge site is the result of dissolution of minerals in the formation by water rock interaction, principally gypsum and anhydrite associated with saline fluids from the Windsor Group. The moderate topographic gradient and high head at the coast also indicate that SWI should not be a concern at this location, however groundwater in the southern part of Pugwash, associated with the Windsor Group, have Cl contents as high as 1,600 mg/L with lower SO_4 and Ca, making them geochemically more similar to intruded seawater. Although this suggests that seawater may be an issue for portions of Pugwash, it is inconclusive due to lack of Br or isotope analysis. Furthermore, reanalysis of chemistry data from CBCL's (2005) Groundwater Quality Assessment shows that the highest salinity groundwater appears to be limited to locations within the Windsor Group or along the contact between the Cumberland and Windsor Groups (Figure 4). This suggests changes in major ion chemistry may be due to local variation from gypsum and anhydrite to Halite in the Windsor Group, especially considering the fact that halite is mined nearby.

2.8 Summary of investigation in Pugwash, NS

The result of investigation in the rural Village of Pugwash showed the existence of significant water quality issues, primarily due to high solute content. Although previous investigations pointed to SWI as one of the causes of this high solute content, the results of this investigation suggest that most of this issue may be attributed to formation fluids from the Windsor Group. Chemical analysis of groundwater supports this hypothesis, as does high hydraulic head and semi-confined conditions found in the Cumberland Group at the study site.

Figure 4. Chloride concentrations from a domestic water quality survey performed in Pugwash (CBCL, 2005) show high chloride associated with the mapped location of the Windsor Group evaporate deposit in the southern part of Pugwash.

3. TOWN OF WOLFVILLE

3.1 Site History

The Town of Wolfville is located at the northern end of the Annapolis Valley along the Cornwallis River, and adjacent to the Bay of Fundy. The town sits on the side of a 100 m high bedrock ridge that slopes downward towards the Cornwallis River at a grade of approximately 4 %. Wolfville has two municipal wells that supply water for its roughly 3,700 residents (Hennigar and CBCL 2006; Statistics Canada). An additional 3,300 students occupy Wolfville between September and May for a total year-round weighted population of approximately 6,100 persons (NSDNR Unpublished Data). Two municipal wells, the Cherry Lane Well and the Wickwire Street Well, supply approximately 2700 m^3/day of water to meet the town's needs. A geothermal doublet at Acadia University pumps between 400 m^3/day and 2000 m^3/day, but is a looped system and returns all but a small portion of the groundwater removed. The difference is estimated to be approximately 80 m^3/day (Hennigar and CBCL 2006).

Wolfville was selected for detailed investigation due to concern over increasing salinity in the town's two municipal supply wells, and increasingly low water level in the Nova Scotia Department of Environment observation well located 750 m from the town's municipal two supply wells. Water level in the Nova Scotia Department of Environment's observation well has been decreasing since monitoring began in 1969 at a rate of ~2.5 cm/yr. In that time the level has dropped from 1.1 m to an average of 0.4 m above sea level in 2008. Annual fluctuation of approximately 1 m means that the head is occasionally below zero during the early summer (NS Environment 2009). This indicates potential for risk of SWI and salinization of the aquifer. Modeling of the entire groundwater flow system in Wolfville conducted by Hennigar and CBCL (2006) suggested that SWI might be occurring to some extent during the summer months but, for the area surrounding the municipal water supply, this has yet to be confirmed. The goal of this site investigation was to gain more information about the likelihood of SWI affecting the town's supply wells.

3.2 Geology

Glacial deposits ranging from 5 - 50 m in thickness make up the majority of the surficial geology in Wolfville (Trescott 1968). The principal aquifer in Wolfville is a semi-confined surficial sand and gravel aquifer known as the Cornwallis aquifer (Trescott 1968), which was identified by Hennigar and CBCL (2006) as a glaciofuluvial deposit. This unit trends from Kentville to Wolfville along a paleo bedrock channel. The Cornwallis aquifer is overlain in many places by a semi-confining layer of glacial till. Transmissivity of this aquifer in the vicinity of the town's two supply wells is roughly 1,489 m^2/day. Hydraulic conductivity is estimated to be roughly 4 x 10^{-4} m/s in this surficial aquifer (Hennigar and CBCL 2006).

Bedrock in the Wolfville area is made of up three units. The Wolfville Formation of the Newark Supergroup or Fundy Group makes up the valley floor in Wolfville. It is composed of triassic fluvial sandstone, and conglomerate, aeolian sandstone, minor deltaic-lacustrine deposits (Crosby 1962). This formation is the primary aquifer for much of the Annapolis Valley. It underlies the Cornwallis aquifer, and the two aquifers are in hydraulic connection (Jacques Whitford 1991). Values of hydraulic conductivity in the Wolfville Formation range from 10^{-8} to 10^{-2} (Rivard et al. 2007). The Wolfville Formation dips 5 to 10 degrees to the northwest and overlies the Halifax Formation, which makes up Wolfville Ridge and is part of the Meguma group, composed of Ordovician slope-outer shelf shale, siltstone, minor sandstone, and Fe-Mn nodules, in places metamorphosed to schist (Crosby 1962). This unit has a transmissivity of ~4.5 m^2/day (Trescott 1968; Hennigar and CBCL 2006). The Horton group underlies the eastern end of the Town of Wolfville, and is comprised of shale, siltstone, conglomerate, breccia, and minor dolostone-limestone (Crosby 1962). The Horton group is a secondary aquifer (JWEL 1991), but is not an important groundwater source in Wolfville or located at the study site.

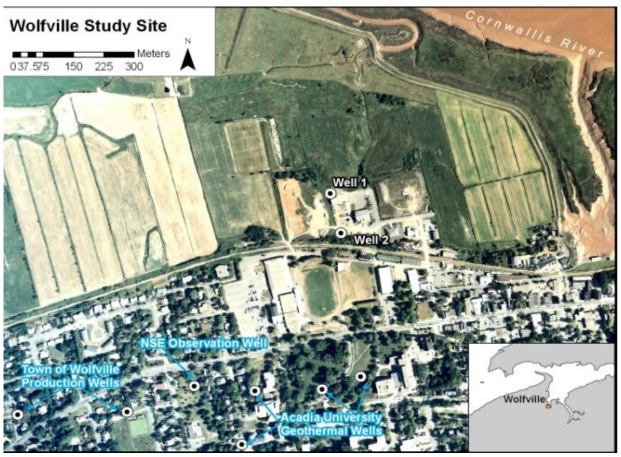

Figure 5. A aerial photograph of the Town of Wolfville, NS, showing the position of the Town's two production wells, the geothermal doublet at Acadia University, and the two wells installed for this project (Well 1 and Well 2).

3.3 Investigation

Two new monitoring wells, NS-2010-TH6 and NS-2010-TH7, were installed in the Wolfville Public Works Yard in November of 2010 (Figure 5). The wells were installed in close proximity to where the town stores its road salt for the winter, and disposes of excess snow from the streets. Both wells were drilled to a depth of 92 m. Stratigraphy encountered during drilling NS-2010-TH6 was roughly in accordance with previous investigations, and found undifferentiated till on top of sandy till overlying red sandstone conglomerate bedrock. Bedrock was identified as being part of the Wolfville Formation. NS-2010-TH7 encountered similar sandstone, but notably harder fine-grained, grey bedrock between 80 m and 90 m interpreted as Halifax Formation slate. Both wells were cased with 3" PVC. NS-2010-TH6 was screened from 79.2 m to 82.3 m and NS-2010-TH7 was screened from 76.2 m to 79.2 m. These wells along with the geothermal wells for Acadia University, and a Nova Scotia Environment monitoring well help form a transect between the Cornwallis River and the Town of Wolfville's two municipal supply wells. A recovery test, used to determine the hydraulic conductivity of both wells, indicated a hydraulic conductivity for of about 5 x 10^{-5} m/s (Theis 1935).

3.4 Geochemistry

The Nova Scotia Department of Environment (2009) reported that salinity in the Nova Scotia Department of Environment Observation well in the Cornwallis aquifer had risen from 78 mg/L in 2004 to 87 mg/L in 2008. This is well below the 250 mg/L aesthetic objective (AO) set by Nova Scotia Environment. Further they reported that the bromide concentration was 0.06 mg/L. A Br/Cl mass ratio of 0.00077 from this study suggests road salt rather than seawater was the leading cause of contamination. Typical Br/Cl ratios for seawater are 0.0035 (Morris and Reily 1966). The presence of Bromide does leave open the question of whether or not some fraction of the salinity is related to SWI. Tritium (^{3}H) levels of 4.7 T.U., from the same study, suggested that the source of contamination is either modern or a mixture of modern (post 1952) and old (pre 1952) (NS Environment 2009).
Analysis for major chemical constitutes from NS-2010-TH6 and NS-2010-TH7 showed ion ratios similar to seawater, but enriched in Ca consistent with intruding seawater. This in addition to depleted ^{3}H indicates that this higher salinity water is much older than water in the Corwallis aquifer, predating 1952, which would be consistent with intruded seawater. NS-2010-TH6 showed significant chloride 480 mg/L and trace bromide 0.5 ± 0.5 mg/L; a ratio of Br/Cl of 0.001. Chloride in NS-2010-TH7 was only slightly higher at 490 mg/L, and bromide was 0.8 ± 0.5 mg/L; a Br/Cl ratio of 0.0016. This is significantly lower than seawater, indicating possible Cl contamination from other sources. Partial depletion in Na (Na/Cl 0.4) and Mg (Mg/Cl 0.04), total depletion of SO_4, and enrichment of Ca (Ca/Cl 0.2) found are consistent with intruding seawater. Figure 6 shows a Piper plot of groundwater chemistry from Wolfville. The diamond shows a trend similar to what one would expect if the early stages of SWI were occurring.

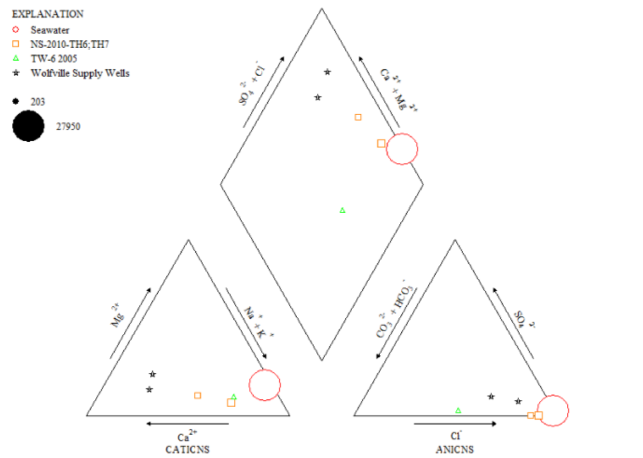

Figure 6. A piper plot showing the distribution of composition and salinity from several wells in and around the Wolfville study site. The ion ratios in the two wells installed for this project NS-2010-TH6 and NS-2010-TH7 are consistent with those found in intruding seawater.

Based on the concentration of Cl in these samples seawater may contribute roughly 2.8% of the bulk composition in the aquifer. However, stable $\delta^{18}O$ and $\delta^{2}H$ isotopes plot above the meteoric water line, and are generally consistent with values from other wells in the Annapolis Valley (Figure 7). This suggests that this salinity is not due to seawater intrusion. However, the lack of notable difference from other fresh groundwater samples may be due to the low concentration of seawater found in the new wells.

3.5 Summary of Investigation in Wolfville, NS

The result of this investigation shows that the Town of Wolfville's municipal supply wells may be at risk for SWI. Bulk ion composition in the Wolfville aquifer is consistent with that of an intruding seawater front. Chemical analysis from several previous studies shows that increasing salinity in the Cornwallis aquifer is probably due to road salting, and may have a formation component. Based on the findings in this study it appears possible that the Cornwallis aquifer may also be vulnerable to upconing of slightly more saline water from the underlying Wolfville aquifer. The wells installed for this project will help provide a monitoring tool to help detect a saltwater wedge moving inland if aquifer conditions allow this to occur. Continuous monitoring, and additional sampling of the Wolfville formation is recommended to ascertain the source salinity, extent of contamination, and to determine the risk it may pose to the Cornwallis aquifer.

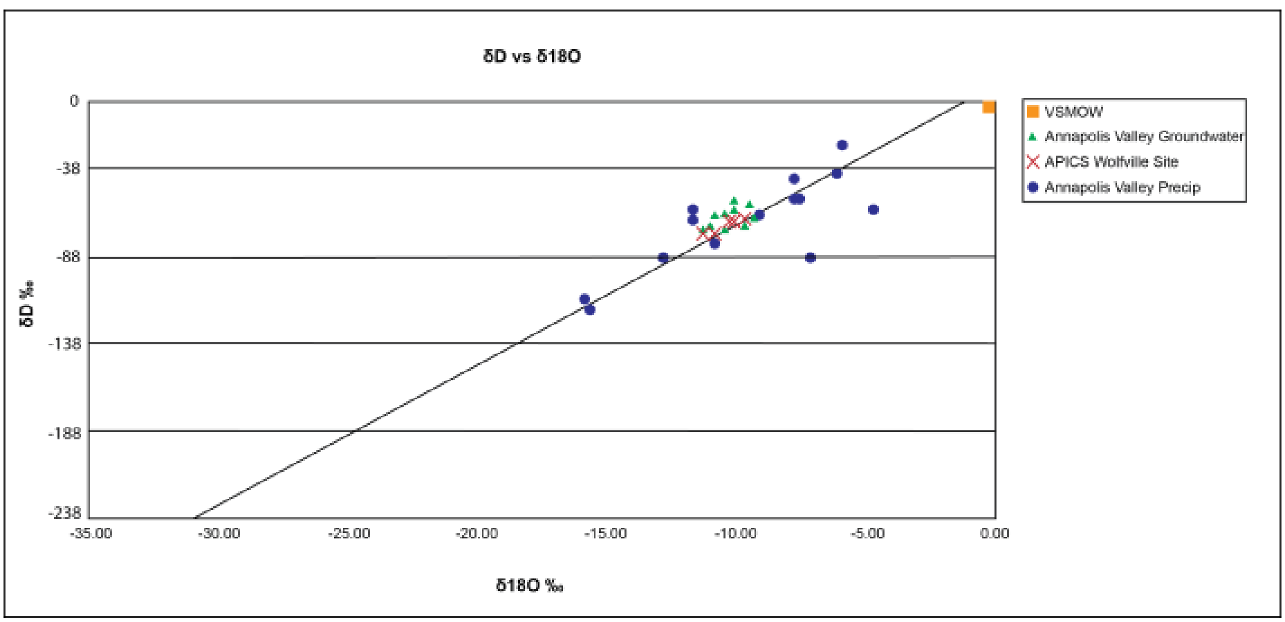

Figure 7. Graph of stable $\delta^{18}O$ and $\delta^{2}H$ isotopes from samples taken in the Town of Wolfville and other sites in the Annapolis Valley superimposed over the 'meteoric water line'.

4. OVERVIEW OF SWI IN NOVA SCOTIA

There are few well-documented cases of SWI in Nova Scotia. This study showed that evaporite deposits are the primary cause of concern over water quality in the Village of Pugwash, a site that was thought to be impacted by SWI. It was also shown through hydrogeological investigation and analytical modelling that there is low risk of SWI in most of Pugwash due to the hydraulic conditions of the aquifer.

In the Town of Wolfville previous studies showed no evidence of SWI, and previous modelling only hinted that it may be an issue during the summer months. The results of this study suggested that the extent of SWI in the lower Wolfville aquifer might be significant. Further monitoring of the chemistry throughout the Cornwallis and Wolfville aquifers is necessary to determine whether or not the presence of this saltwater wedge poses a threat to potable water supply in Wolfville.

In general the risk of SWI in Nova Scotia is not severe. Factors such as moderate hydraulic gradient, high annual recharge rates, and low coastal population densities mitigate the risk of SWI in the province. Nevertheless, investments in research and monitoring of groundwater supplies can help identify and track potential sources of contamination especially as sea-level rise increases stress on coastal aquifers.

Case Study:
Overview of seawater intrusion in Nova Scotia – Regional vulnerability assessment

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1 INTRODUCTION

Seawater intrusion (SWI) into coastal aquifers as a consequence of overpumping, coupled with the effects of climate change and sea-level rise, is a key issue for water resource management in the Province of Nova Scotia. Nova Scotia has over 10,000 km of coastline and approximately 70% of Nova Scotians reside within 20 km of the coast (Government of Nova Scotia 2005). It is estimated that over 50% of Nova Scotians living in coastal areas depend on groundwater as their primary source of potable water, and many of the province's coastal regions are experiencing residential growth, especially in suburban areas of Halifax (Figure 1) which experienced 7.5% population growth from 2006-2011 (Statistics Canada 2012).

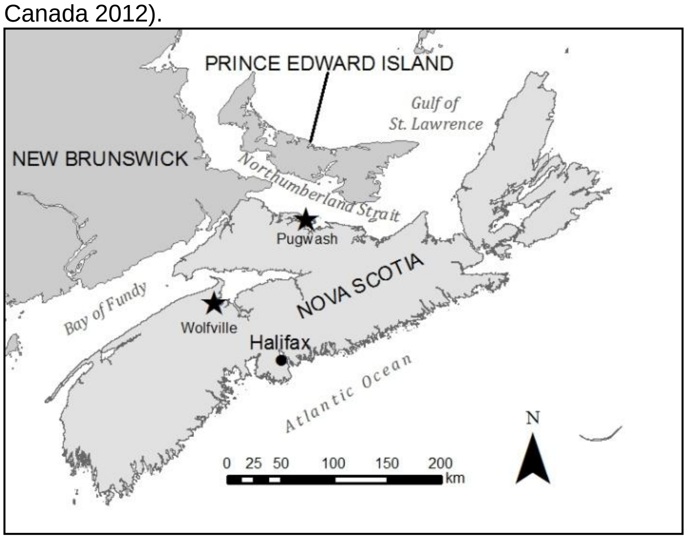

Figure 1. Location map showing Pugwash and Wolfville Atlantic RAC project sites.

The combined effect of eustatic and isostatic sea-level rise will produce an estimated 70 cm increase in relative sea level across Nova Scotia by 2100 (Forbes et al. 2009). Vasseur and Catto (2007) report that although precipitation is expected to remain the same or increase over the same period of time, the timing of precipitation events may contribute to an overall decrease in groundwater recharge in the province. This would result in a decrease in the availability of coastal groundwater and, along with projected increases in demand, especially during the summer months when groundwater levels are at their lowest, exacerbate the risk of SWI.

A study was recently completed under the Atlantic Regional Adaptation Collaborative (RAC) to provide an initial assessment of the potential impact of climate change on coastal aquifers in Atlantic Canada through a series of case studies consisting of site-specific field investigations coupled with groundwater modeling under various climate change scenarios. The location of the Nova Scotia case studies is shown in Figure 1. Historical documentation of SWI in Nova Scotia is limited to a handful of reports by the government and private consultants (e.g. H.J. Porter & Associates 1979; H. J Porter & Associates 1980, Cross 1980, Briggins and Cross 1995, CBCL Limited 2005). Based on the Atlantic RAC case study findings, Ferguson and Beebe (in prep.) reported that the province's aquifers are not very susceptible to SWI. Nova Scotia receives high rates of precipitation, and projected changes in recharge will have a minor impact on coastal aquifers over the range of hydraulic gradients observed in the province (Ferguson and Gleeson 2012). In addition, the low permeability of glacial till material overlying much of the province was interpreted to result in a high water-table slope, forcing the freshwater-saltwater interface off the coast (Ferguson and Beebe in prep.). Key findings of the case study work under the Atlantic RAC also suggest that coastal aquifers are most sensitive to changes in groundwater withdrawals, highlighting the importance of managing water demand in coastal areas (Beebe 2011, Ferguson and Beebe in prep.).

1.1 SWI Vulnerability Characterization

Mapping the relative vulnerability of seawater intrusion throughout the province was proposed as a useful tool for managing SWI risk and prioritizing groundwater management activities. Quantitative methods of vulnerability characterization (e.g. Werner et al. 2012) cannot be meaningfully applied at the provincial scale due to the lack of detailed hydrogeological information available at this scale. The relative vulnerability of bedrock aquifers can be qualitatively characterized based on the hydrogeologic characteristics of the province's five major bedrock groundwater regions (Table 1). The province's bedrock groundwater regions are shown in Figure 2 and a detailed description of their hydrogeologic characteristics can be found in Kennedy and Drage (2009).

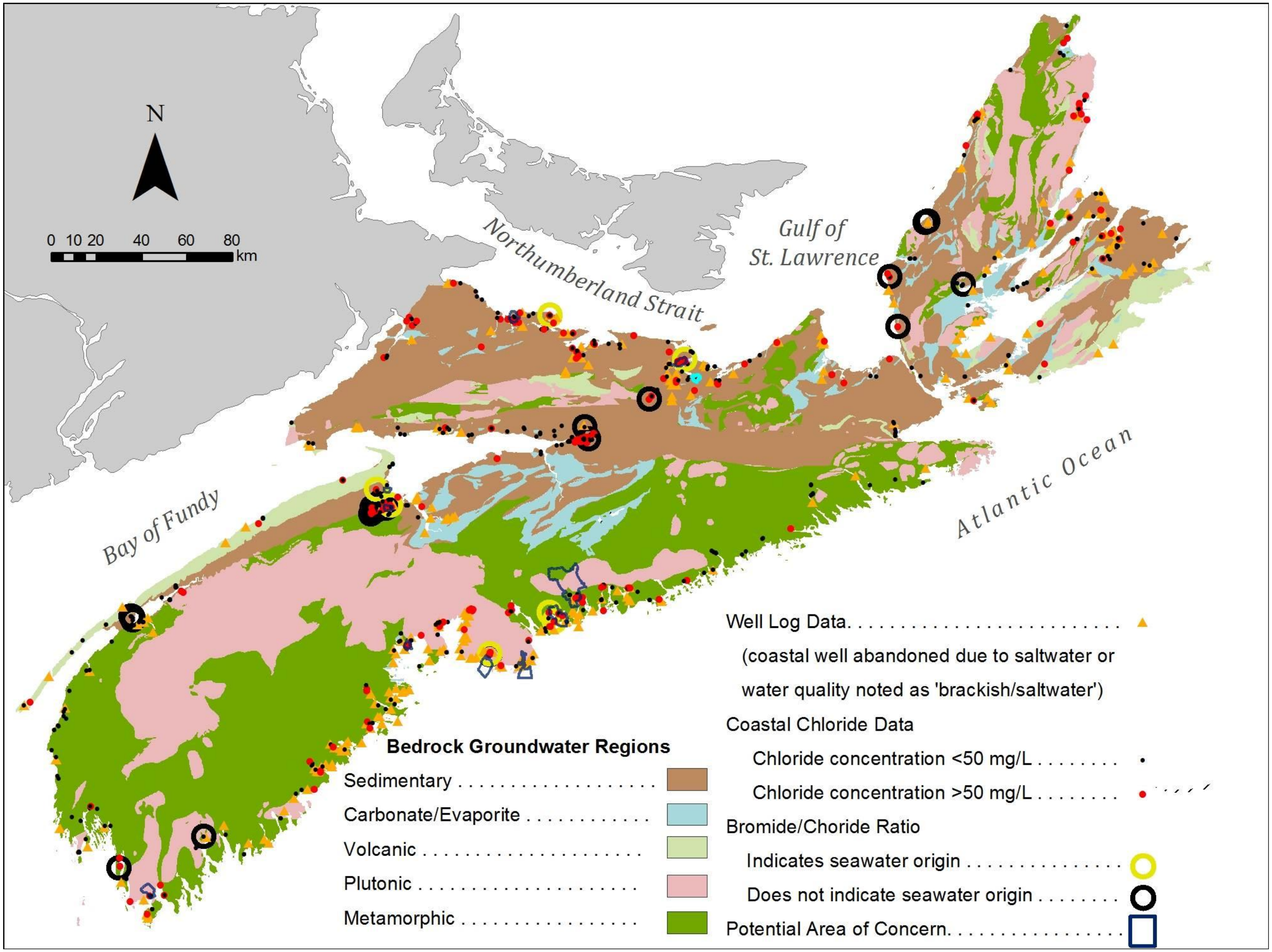

Figure 2. Map showing various indicators of potential saltwater intrusion.

This approach, however, does not account for water use patterns and other site-specific information, and does not provide vulnerability characterization at the scale needed for land-use planning decisions. Qualitative vulnerability assessment techniques developed by others were reviewed (e.g. GALDIT, Chachadi and Lobo-Ferreira 2005), however, a customized tool tailored to the available spatial datasets in Nova Scotia was considered the most advantageous approach.

The adaptive capacity of the water user is an important consideration in SWI vulnerability characterization. Private well users have limited adaptive capacity because they have few low cost remedial options available to them if their water well becomes impacted by SWI. Remedial options might include the installation of a dug well or cistern system, modifications to the existing well or pumping configuration, water deliveries, reverse osmosis treatment, or water servicing. It is estimated that over 30% of Nova Scotians in coastal areas are supplied by private wells, and based on provincial trends over 90% of these wells are drilled wells intercepting fractured bedrock aquifers (Kennedy and Drage 2009). Reliance on private well water supplies in coastal areas is increasing in some parts of Nova Scotia. For example, 6 of the 10 subdivision applications received by the Halifax Regional Municipality over the past several years for new subdivisions serviced by private wells have been located within 1500 m of the coastline.

Table 1. Qualitative relative vulnerability assessment of major bedrock groundwater regions in Nova Scotia.

<table>
<tr><th>Groundwater Region</th><th>Bedrock Unit</th><th>Risk Characterization</th></tr>
<tr><td>Sedimentary (mostly siliciclastic units)</td><td>Cumberland, Fundy, Horton, Mabou, Morien, and Pictou groups</td><td>Extent of SWI can be more widespread due to higher hydraulic conductivity of sedimentary rock aquifers</td></tr>
<tr><td>Carbonate/ Evaporite</td><td>Windsor Group</td><td>Groundwater already associated with high TDS and often not suitable for consumption</td></tr>
<tr><td>Volcanic</td><td>Stirling and Fourchu groups, and North Mountain Formation of Fundy Group</td><td>Predominantly vertical fracturing may limit extent of SWI</td></tr>
<tr><td>Plutonic</td><td>All granitic bedrock types</td><td rowspan="2">Extent of SWI dependent on connectivity of water bearing fractures with seawater, lower hydraulic conductivity may result in more localized effects, although greater drawdowns in these rock types may result in increased upconing of seawater</td></tr>
<tr><td>Metamorphic</td><td>Meguma Supergroup, Georgeville Group</td></tr>
</table>

There are few instruments available to water managers to evaluate the sustainability of groundwater withdrawals and manage SWI risk in these unserviced areas of the province. SWI vulnerability is also more difficult to characterize in these areas due to a lack of available well water chemistry and water level monitoring data relative to the extent of unserviced coastline. Nova Scotia currently has a total of 17 observation wells within 1500 m of the coastline and does not routinely collect private well chemistry data.

Although municipal groundwater supplies can be more susceptible to SWI than private well water supplies due to concentrated pumping of larger water volumes, areas serviced with municipal groundwater can be considered to have lower inherent risk because the resource is actively being managed by water utilities and subject to regulatory oversight, including monitoring protocols and routine chemistry analyses. Water utilities also have greater capacity to characterize risk and implement adaptive management (e.g. drill new wells farther inland) relative to private well users. Quantitative, physically based SWI assessment is appropriate for these systems (e.g. chemical fingerprinting, analytical/numerical modeling).

For these reasons the relative SWI vulnerability characterization work here focused on coastal bedrock aquifers located in unserviced areas of the province.

1.2 Study Objectives

The objectives of the study were to develop a simple GIS-based tool for broadly evaluating the relative vulnerability of bedrock coastal aquifers to SWI in unserviced areas of the province.

2 METHODS

2.1 SWI Indicator Map

Various indices of seawater intrusion were compiled and mapped within 1500 m of the coastline to help assess the potential extent of SWI in Nova Scotia. These indicators include groundwater chemistry data (such as chloride and bromide concentrations and ratios), water well drill locations where well drillers have reported encountering seawater, and areas where available reporting or anecdotal information have identified potential SWI.

For the purposes of the indicator map, chloride concentrations greater than 50 mg/L were considered to represent elevated levels above background. Kennedy and Finlayson-Bourque (2011) reported a median chloride concentration of 24 mg/L in the province's bedrock aquifers, with approximately 70% of the levels falling below 50 mg/L. In addition to SWI impacts, elevated chloride levels in groundwater in Nova Scotia have been attributed to sources such as road salt, on-site wastewater discharges and bedrock formation salt (e.g. halite bedrock units within the carbonate/evaporate groundwater region, Figure 2). A limited number of groundwater samples collected throughout the province also include measurements of bromide. The Br/Cl ratio has been used previously in Nova Scotia (e.g. Briggins and Cross 1995) and other jurisdictions (e.g. Snow et al. 1990) to effectively differentiate marine sources of chloride from road salt (e.g. halite). Br/Cl ratios of approximately 3.4×10^{-3} indicate chloride of seawater origin, whereas bromide tends to be present at lower concentrations in road salt, resulting in lower Br/Cl ratios.

2.2 SWI Vulnerability Map

2.2.1 GIS Approach

The assessment area was defined as a 250 m x 250 m grid within 1500 m of the 1:10 000 provincial coastline layer. The few documented cases of SWI in Nova Scotia are within 500 m of the coast, and therefore 1500 m inland from the coastline was considered to be a conservative distance for the relative vulnerability assessment. Key provincial spatial data layers were clipped to the grid and five variables considered to influence SWI vulnerability were derived from these layers (see Section 2.2.2 to 2.2.6).

The relationship between these variables and SWI vulnerability is conceptually illustrated in Figure 3 according to the Ghyben-Herzberg approximation, which relates the position of the saltwater-freshwater interface to the densities of freshwater and saltwater and the distribution of hydraulic head (Ghyben 1888; Herzberg 1901). This relationship predicts that the depth of the interface below sea level is equal to approximately 40 times the elevation of the water table above sea level.

Variables were classified and rankings were assigned to each grid cell. The rankings were summed to produce an overall SWI relative vulnerability map, which was then compared to available chemistry data to evaluate the reliability of the vulnerability assessment. Grid cells were compared based on a ranking relative to each other, and not upon a numerical indicator used to indicate a physical threshold. The ranking system for each of the five input variables is qualitative and based on a five-category defined-interval classification. The overall vulnerability ranking uses a three-category defined interval classification intended to generally classify SWI vulnerability into high, medium and low categories.

The GIS approach developed here is intended to be used as an indexing tool that can be readily re-applied to expanded or refined input spatial datasets as they become available. SWI vulnerability refers to the relative likelihood of an existing private well in a given area to be impacted by SWI, based on a simple treatment of factors known to influence SWI.

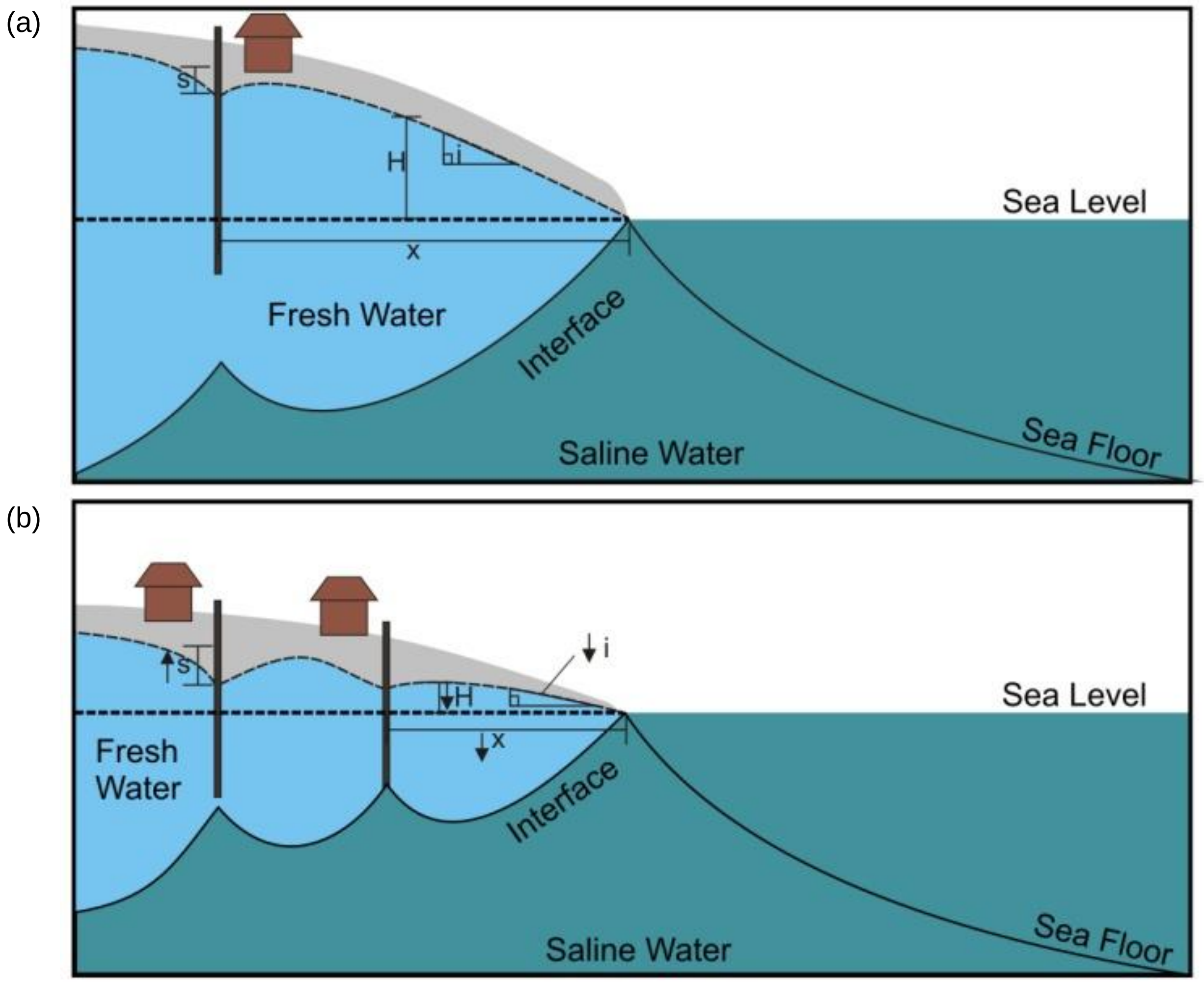

Figure 3. Conceptual illustration showing how vulnerability to SWI impacts can increase according to the Ghyben-Herzberg relationship. In (b), the water supply wells are at a greater risk due to lower freshwater head above sea level (H) as a consequence of their closer position to the coast, a lower hydraulic gradient (i) and increased drawdown (s).

2.2.2 Distance to the Coastline

SWI risk is greatest near the coastline and decreases with distance inland (x, Figure 3). The distance from each grid centroid to the 1:10 000 coastline layer of the province was calculated, and the grid cell was assigned a ranking according to the criteria shown in Figure 4a.

2.2.3 Topographic Slope

Coastal aquifers with low hydraulic gradients are associated with increased SWI risk (i, Figure 3). Topographic slope was selected as a surrogate variable for hydraulic gradient since the bedrock water table in Nova Scotia tends to follow a subdued pattern of topography due to high precipitation rates and the presence of low-permeability surficial cover (Gleeson et al. 2011). The slope was calculated from the 20 m digital elevation model (DEM) of the province and grid cells were assigned a ranking according to the criteria shown in Figure 4b.

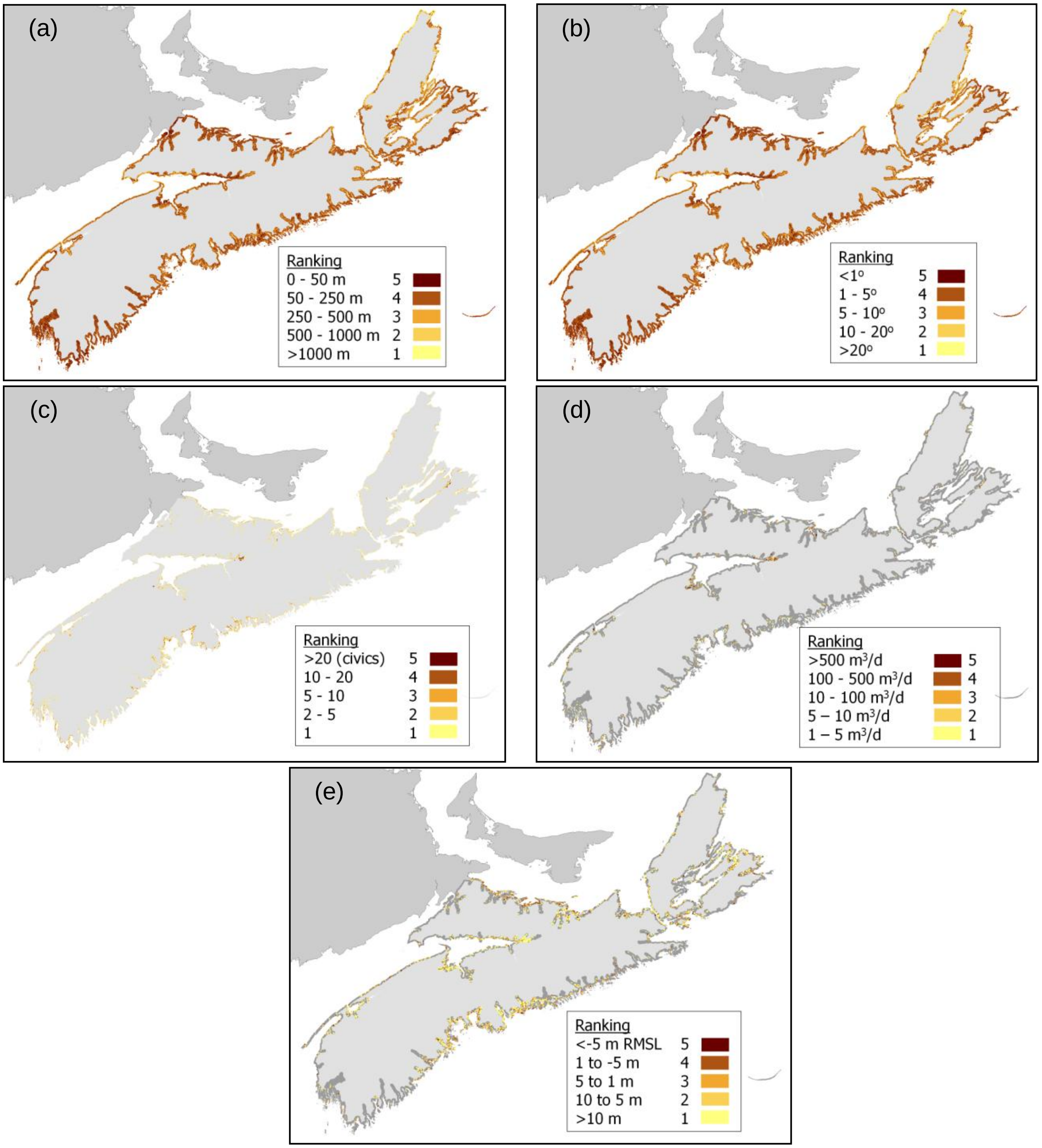

Figure 4. Input layers used to estimate overall relative SWI vulnerability, including (a) distance to the coastline, (b) topographic slope, (c) civic point (residential) density, (d) large groundwater users and (e) water level elevation relative to mean sea level (RMSL).

2.2.4 Civic Point (Residential) Density

Areas of intensive groundwater withdrawals have greater risk of SWI because increasing drawdown (s, Figure 3) will cause increased upconing of saline water according to the Ghyben-Herzberg relationship. Nova Scotia maintains a spatial layer of civic points throughout the province (Service Nova Scotia 2012). This layer was clipped to exclude civic points located within municipal water distribution zones, and the resulting civic point layer was used to indicate domestic groundwater use in areas not serviced with municipal water. A count was performed of civic points within each grid cell and rankings were assigned according to the criteria shown in Figure 4c.

2.2.5 Large Groundwater Users

In addition to concentrated domestic groundwater use, large non-domestic groundater users in unserviced areas can increase drawdown (s, Figure 3) and SWI risk. Nova Scotia maintains a spatial layer of large groundwater users and estimated daily withdrawal rates throughout the province (Kennedy et al. 2010). Daily non-domestic groundwater withdrawal rates were summed for each grid cell and classified according to the criteria shown in Figure 4d.

2.2.6 Water Level Elevation

An aquifer's susceptibility to seawater intrusion is influenced by pressure in the freshwater zone relative to sea level, since adequate freshwater pressure must be maintained to prevent SWI (H in Figure 3). Where the bedrock static water level is at or below sea level, SWI risk is greater.

Nova Scotia maintains a database of water well data, including static water level estimated by drillers during well construction (Nova Scotia Environment 2012a). Water wells located within the study area grid were filtered to exclude data with missing water level information. Moreover, the location accuracy of wells in the database is variable, and only well locations accurate to the property level were retained. The resulting data layer contained 8526 records, representing 15.5% of the total records located in the study area.

A ground surface elevation was assigned to each of these wells using the best available digital elevation model (DEM). Higher resolution LiDAR-derived DEMs (compared to the provincial DEM) were available in eight sub-areas of the province. The water level elevation was calculated as the depth to water level reported by the well driller on the log subtracted from the DEM estimated surface elevation. Where multiple wells were located within a grid cell, the minimum water level measurement was used for the vulnerability assessment. The more accurate water level elevation reported in provincial observation wells (Nova Scotia Environment 2012b) and pumping test reports superseded well log data where these data were available. Where water level data were sparse, the water level estimate predicted the water level in grid cells up to 500 m from the point measurement. Each grid cell was classified with respect to water level according to the criteria shown in Figure 4e.

2.2.7 Relative SWI Vulnerability

Relative SWI vulnerability was calculated as the sum of all five criteria to produce an overall score according to the criteria shown in Figure 5. Vulnerability was not calculated for cells with null water level measurements (i.e. no water level elevation estimate available).

2.2.8 Validation

A rigorous statistical validation of the relative SWI vulnerability mapping was not possible due to the limited availability of groundwater geochemistry data and confirmed cases of SWI in unserviced areas of the province. The results of the overall vulnerability mapping were compared to the SWI indicator mapping, however, to determine if there was general agreement between high vulnerability and existing indicators of SWI.

3 RESULTS

The SWI indicator map is shown in Figure 2. The map shows that many of the elevated chloride levels near the coast are located along the Atlantic coast in the south-central region of the province, and along the Northumberland Strait, where significant coastal populations exist and more groundwater information is available. The origin of the elevated chloride levels shown in Figure 2 is not known, although it should be noted that the median concentration of chloride in coastal regions (<1500 m from the coastline) is 43 mg/L compared to 15 mg/L for samples collected in the rest of the province. Areas where the bromide/chloride ratio is indicative of seawater intrusion to bedrock aquifers were identified in Prospect, Lawrencetown, Cow Bay, Wolfville, Canning, Pictou and Fox Harbour (Figure 2). Based on the data presented in the compilation map, suspected occurrences of SWI tend to be of local extent.

Figure 4 shows the results of the ranking of SWI vulnerability for the variables described in Sections 2.2.2 to 2.2.6. Figure 5 shows the relative SWI vulnerability map for the province. Since relative SWI vulnerability was calculated only

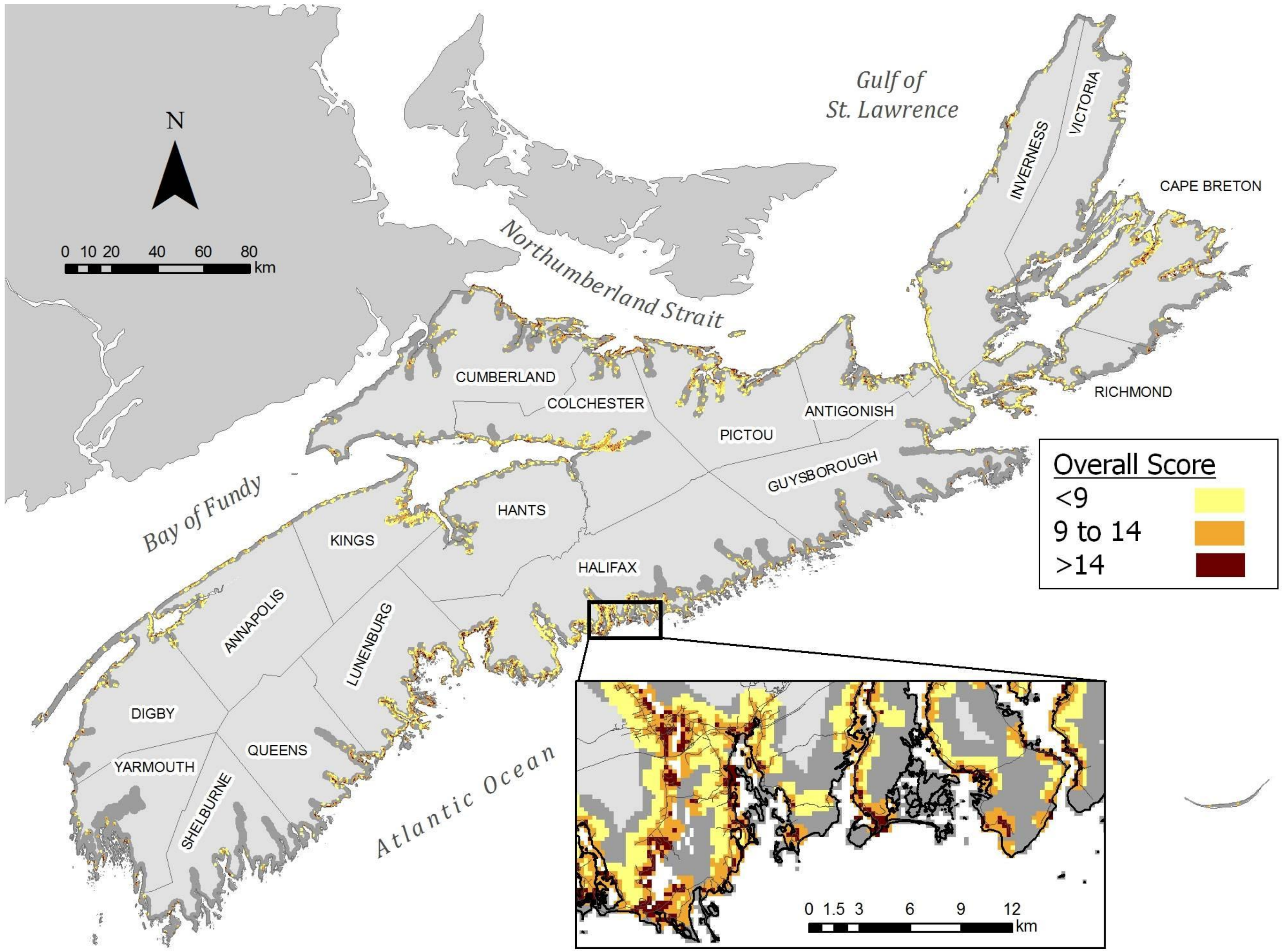

Figure 5. Overall relative SWI vulnerability map.

for cells with water level information (e.g. calculated only for cells with water level information (e.g. Section 2.2.6), an overall vulnerability score was calculated for only approximately 26% (49,564 cells) of the total study area. Table 2 provides a count of the number of cells in each of the high, medium, and low vulnerability categories for the province and for each county. The south-central (Halifax and Lunenburg counties) and Northumberland Strait (Cumberland and Pictou counties) areas of the province had the greatest relative vulnerability based on a count of cells in the high vulnerability category. Table 2 also shows that 86% of the locations with Br/Cl ratios indicating SWI, and 55% of the locations where drillers have reported encountering seawater, were located in grid cells with a high relative vulnerability score.

4 DISCUSSION

Although recent work (Ferguson and Beebe in prep.) indicates that Nova Scotia's coastal aquifers are not especially vulnerable to SWI compared to many other coastal aquifer systems around the world, a large segment of the population in Nova Scotia relies on private wells intercepting coastal bedrock aquifers for potable water, and residential development (and groundwater use) in some coastal areas, especially in suburban Halifax, is growing (Statistics Canada 2012).

Prevention of SWI is critical because the adaptive capacity of private well users is limited. Since SWI is sensitive to groundwater withdrawals, careful water management is needed to prevent SWI, but water managers have limited instruments available for managing water demand and assessing SWI risk. Climate change effects, such as eustatic sea-level rise and changes to groundwater flow dynamics, can increase the severity of this risk.

The GIS-based relative SWI vulnerability indexing tool identified unserviced areas that may already be experiencing SWI or are at greatest risk to sea-level rise and additional withdrawals by new developments using private wells. The results of the assessment may allow land-use planners to target land-use controls and more detailed groundwater study requirements in areas of high residential growth and SWI vulnerability. Groundwater managers may use the results of the analysis to identify suitable coastal aquifer monitoring well locations and to help prioritize areas for more detailed risk assessment using quantitative physically based methods, such as well surveys, analytical and numerical modeling, and chemical fingerprinting. Effective utilization of the relative vulnerability mapping would require routine update and improvement of input data sets (e.g. higher resolution DEMs, new well logs, improved georeferencing of logs), and the development of suitable mechanisms for the regular transfer of the map information to users such as municipal planners and water managers.

It is difficult to evaluate the reliability of this approach as a relative vulnerability indexing tool. There was reasonably good spatial agreement between relative vulnerability and geochemical indicators of SWI and salinity problems reported by well drillers (Table 2). The collection of additional groundwater chemistry data (especially Br) and comparison of other geochemical indicators of SWI to the mapping would provide opportunities for a more robust evaluation of the approach presented here.

The relative SWI vulnerability assessment tool has a number of fundamental limitations and should only be used for broad relative risk characterization as a first pass analysis. The subjective indexing approach used in generating the relative vulnerability map lacks the theoretical translation of hydrogeological characteristics into SWI vulnerability. The approach could be refined by improved consideration of the local hydrogeologic setting (e.g. confined vs. unconfined aquifer types, hydraulic conductivity and bedrock fracture patterns). These types of refinements, however, are likely predicated on a more detailed understanding of the coastal zone hydrogeology of the province and how it relates to SWI vulnerability. If this knowledge is generated, however, it may be feasible to apply analytical SWI vulnerability indicators (e.g. Werner et al. 2012) at the provincial scale.

5 CONCLUSION

A GIS-based approach was developed to help broadly characterize SWI relative vulnerability in unserviced areas of Nova Scotia. The tool provides a preliminary assessment of relative SWI vulnerability in Nova Scotia based on factors known to influence the position of the seawater – freshwater interface. The spatial analysis can be readily repeated as new data become available. There was reasonably good agreement between indicators of SWI and areas identified as having medium to high vulnerability. The south-central and Northumberland Strait areas of the province appear to have the greatest relative SWI vulnerability based on the results of the mapping. Improvements to the quality and resolution of input data layers and refinement of the indexing approach are recommended to improve the reliability of the relative SWI vulnerability map.

Table 2. Count of high, medium and low vulnerability scores broken down by jurisdiction (province and county) and compared to observed geochemical and well log data.

	Total Cells	Number of cells	Percent of total cells	Number of cells	Percent of total cells	Number of cells	Percent of total cells
		High		Medium		Low	
Nova Scotia	49564	4497	9%	20019	40%	25048	51%
Halifax	8926	1039	12%	3848	43%	4039	45%
Lunenburg	4266	592	14%	1869	44%	1805	42%
Cumberland	3403	421	12%	1595	47%	1387	41%
Pictou	3908	415	11%	1555	40%	1938	50%
Cape Breton	5111	408	8%	1973	39%	2730	53%
Colchester	3588	256	7%	1422	40%	1910	53%
Richmond	3220	244	8%	1442	45%	1534	48%
Queens	1236	175	14%	557	45%	504	41%
Guysborough	1683	161	10%	814	48%	708	42%
Digby	1649	142	9%	615	37%	892	54%
Inverness	2570	126	5%	791	31%	1653	64%
Victoria	1897	99	5%	585	31%	1213	64%
Antigonish	1684	98	6%	612	36%	974	58%
Yarmouth	625	89	14%	322	52%	214	34%
Kings	2290	75	3%	723	32%	1492	65%
Shelburne	510	74	15%	235	46%	201	39%
Annapolis	1475	43	3%	547	37%	885	60%
Hants	1523	40	3%	514	34%	969	64%
Drillers encountering saltwater[1]	96	53	55%	37	39%	6	6%
Br/Cl Indicating SWI	7	6	86%	1	14%	0	0%
Cl > 50 mg/L[1]	224	91	41%	97	43%	36	16%

1. Data intersecting the carbonate/evaporite groundwater region were excluded from the counts due to naturally occurring salinity

Acknowledgments

The writer would like to acknowledge Allie DeCoste, Jeff MacKinnon, Heather Cross and John Drage for their contributions to the research, and the Atlantic Climate Adaptation Solutions Association, the Halifax Regional Municipality, Nova Scotia Environment, the Applied Geomatics Research Group, and the Centre of Geographic Sciences for data support in terms of the provision of LiDAR-derived DEM data.

Newfoundland and Labrador Case Studies

As sea level continues to rise, SWI is a growing concern for coastal communities of Newfoundland and Labrador that rely on GW for drinking water supplies. Although there is a realization of potential SWI, the risk and rate of possible development is hard to quantify. At present, there has been no consistent record keeping of well abandonment in the province. In addition, samples for water chemistry of private wells are not typically collected during well construction and therefore cannot be used for investigative guidance and to date, there has been no known study that specifically addresses the presence or progression of SWI into private or public wells in the province.

Although no official studies or records have been found on SWI affecting wells, there are some adopted practices amongst drillers and contractors for communities in the study area known to contain a salt source into GW. With progression of sea level rise, in part due to CC, the reconnaissance study presented here is a first attempt to establish baseline data in this regard. The study focuses on selected areas identified by industry as at risk for encountering salt water at depth. The study evaluates specific wells through a combination of geophysical and geochemical techniques, using existing accessible wells in locations believed to be representative of those at risk to SWI. An in-depth paper with study particulars is available upon request.

Case Study:
Reconnaissance of Southwest Newfoundland: Examining Potential Sea-Water Intrusion in Past and Current Public Water Supply Wells, Southwest Newfoundland-NL

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1 PURPOSE / LOCATION

The small study completed by the Department of Environment and Conservation. Newfoundland and Labrador was intended to provide baseline data in regards to salt water intrusion along the southwest portion of the island and to also provide reconnaissance on government owned abandoned wells. Salt water intrusion is not known to have been studied in specific regard to climate change, and with the persistence of sea level rise, the need for one was apparent. Groundwater data contained within the provincial Drilled Well Database is inconsistent and does not provide an adequate means of assessing meaningful baseflow parameters. This study provided the opportunity to collect baseline data and supply a foundation to make community decisions on groundwater and engage further study and data collection.

Based on the International Panel for Climate Change predictions, Batterson and Liverman (2010) predicted local sea level rise for four zones in Newfoundland and Labrador. The southwest portion of the island falls within zone 2 where sea water for the region is expected to rise less than 2 mm per year. This equates to a projected sea level rise of 40 cm by the year 2049 and greater than 100 cm by the year 2099.

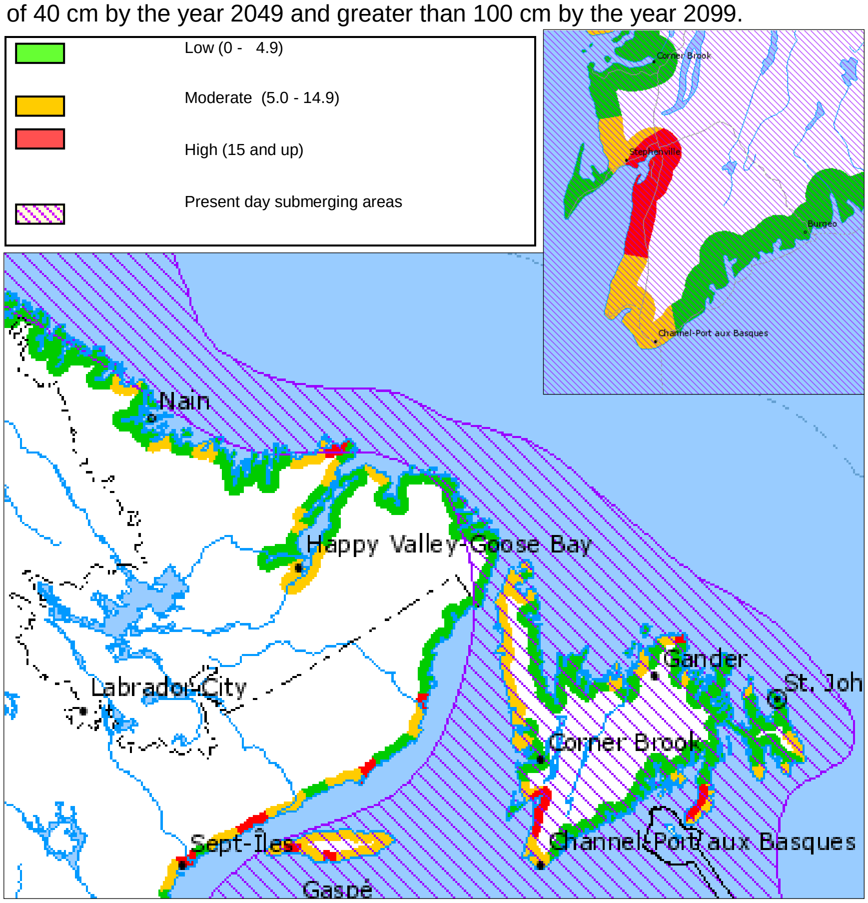

Figure 1. Coastal sensitivity to sea level rise.

Furthermore, a sensitivity index assembled by NRCan to provide a national atlas illustrating coastal sensitivities to sea level rise in Canada indicates a large region in the Bay of St. George area with high coastal vulnerability to sea level rise (Figure 1). The index was based on scores from 1 to 5 and relies on seven variables: relief, geology, coastal landform, sea-level tendency, shoreline displacement, tidal range and tidal height. The index showed that the majority of Newfoundland and Labrador is of moderate to low sensitivity to sea level change. The exception however, is the northwest coast of the Burin Peninsula, and the projects study area of St. George's Bay (Batterson and Liverman, 2010). To provide an overall reconnaissance of the southwest portion of the Island, the Port aux Port Peninsula and the Codroy Valley area were also included in the study. Although the index does not rate potential for sea water intrusion directly, it does relate to the variables used in the index.

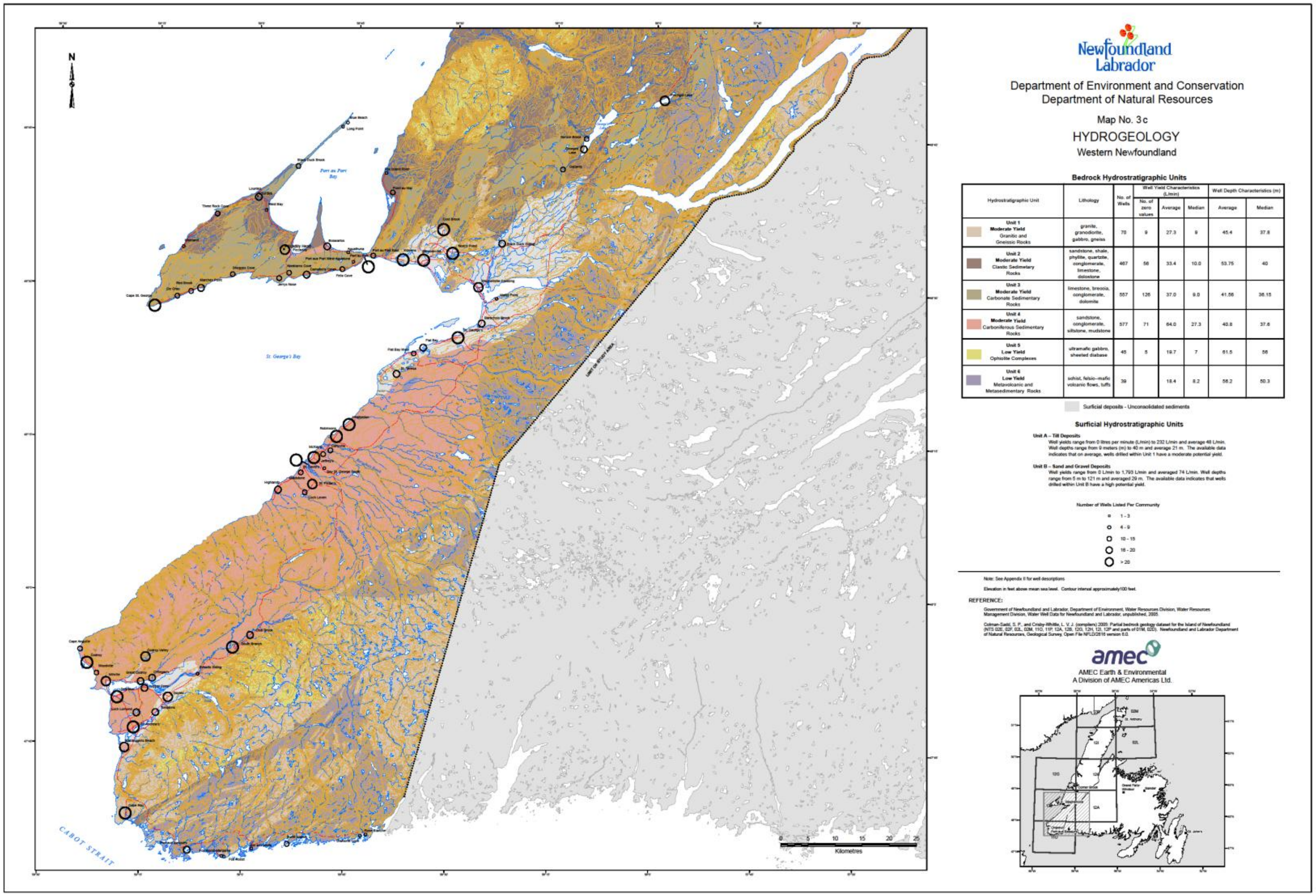

Figure 2. Hydrogeology of southwest Newfoundland.
(Source: http://atlas.nrcan.gc.ca/auth/english/maps/climatechange/potentialimpacts/coastalsensitivitysealevelrise

The southwest portion of the island is largely underlain by different types of carboniferous bedrock, and provides an average yield of 64 Lpm (Amec Environmental, 2008) (Figure 2). This is a relatively high yield in comparison with other areas of the province. The region has been heavily inundated by past glacial events and glaciofluvial surficial deposits of up to 50 metres in some areas. Coupled with elevations as low as 1 metre above sea level in some areas, bedrock is often found at some depth below sea level. The southwest portion of the island also has many communities that rely on groundwater as a public water source. Depending on usage, the fresh water- sea water interface could potentially be drawn in to current supplying aquifers. The seven variables listed previously may also accommodate storm surges which would also put groundwater at risk of sea water intrusion. One or a combination of these factors could play a part in sea water intrusion.

2 METHODOLOGIES / CHALLENGES

For the purpose of reconnaissance in the study area, the locations of abandoned government water wells were investigated and located, often with the assistance of communities during fieldwork. After locating each well, an assessment was made in a case by case scenario in regards to appropriateness of the wells. When deemed adequate as a study well, an initial well log was completed that included length of casing above ground, GPS coordinates, static water level, temperature profile, conductivity profile, and well depth. Static water level, well depth, temperature and conductivity were gathered using the Solinst Temperature, Level, and Conductivity Meter (TLC). The TLC meter was calibrated each morning in accordance with manufacturer's specification and periodically throughout each day. Spikes in conductivity were used as a basis for grab sample location, which were taken at various depths in the well using the Solinst Discrete Interval Samplier. For some wells assessed as having vulnerability of salt water intrusion, a borehole video was completed using the GeoVision Dual-Scan Micro Borehole Video System. This allowed an evaluation of the integrity of well construction and overall well condition. Using conductivity spikes and borehole video, the AquaVision Colloidial Borescope was used in select wells to evaluate aquifer velocity and direction. In wells that contained conductivity of roughly 1000 µS or greater, or contained appropriate bromide to chloride ratios from earlier water chemistry results, samples were collected and sent to the University of Illinois for enriched tritium analysis. The Grundfos Redi Flo-2 pump was used to obtain the 500 ml quantity needed for the analysis. Due to various reasons such as well accessibility, weather conditions, time constraints, permission of private property access, and relevance to the study, work completed on each well varied.

There were some challenges encountered in regard to the methodology. Identifying discontinued government drilled wells that were located in areas susceptible to salt water intrusion was very challenging. When wells were located in promising sites, either well depth did not allow opportunity to puncture the interface at depth or the well was obstructed by rocks, or a well plug. Considering these wells were originally drilled to avoid interaction with the interface, this scenario was anticipated.

Lack of prior knowledge regarding which communities had abandoned government water wells was a reoccurring issue. Much of the time spent in the study area was used traveling between communities and locating town officials or residents that could give insight on the topic. Much time was also spent in locating wells that were identified. Information regarding basic well construction details and pumping yield were not available for most of the study wells. The abandoned wells were largely unrecorded in the Drilled Well Database, which made community visits a necessary component of the reconnaissance. Permission of access was also an issue as the wells were often located on personal property.

Challenges regarding equipment were also experienced. The groundwater pump purchased for the study contained a defect that prevented proper function. The pump was sent back to the manufacturer for repair and could not be used during the first field trip. The pump was repaired prior to the second field excursion.

3 MAIN FINDINGS

Upon completion of the study, many things were revealed not only about the prospect of salt water intrusion for the study area, but also about fundamental data gaps in the Department of Environments Drilled Well Database.

There were a total of eleven wells sampled throughout the study area and eleven abandoned government drilled wells that were not sampled due to well obstruction or unsuitability for the study. This number is considered to underestimate the number of abandoned, government owned water wells in the study area. It was found that basic knowledge of the wells such as drilling year, depth of well, and problems encountered were in most cases not known by Town officials or residents. Neither are they recorded in the Department of Environments Drilled Well Database.

Table 5. Chloride and bromide source water results from selected study wells

Community	Well ID	Sample depth (m)	Cl- (ppm)	Br- (ppm)
Flat Bay West	Abd 15	16	66.8	0.188
Port aux Port East	Abd 16	37	23	0.221
Port aux Port East	Abd 16	39.5	26.2	0.132
Port aux Port East	Abd 16	49.5	26.9	0.14
Highlands	Abd 22	Whole	54.8	0.616
Mainland	Abd 25	8.6	171	1.42

Although the question of recent sea water intrusion/presence lingers for some of the study wells, the presence of bromide in wells suggest that at one time, sea water may have entered into the aquifer. Wells that contained various concentrations of bromide were the Flat Bay West (Abd 15), the two Port aux Port East wells (Abd 16 & 19), the Highland well (Abd 22) and Mainland well (Abd 25). Source water from the remaining wells that did not yield any detectable bromide are not considered to be influenced by sea water intrusion and are not included in these findings. Chloride was found in all sampled wells with concentrations varying from 10.7 ppm to 171 ppm, averaging at 42 ppm. These chloride levels do not support any strong influence of sea water intrusion, and may be reflective of formational or anthropogenic sources.

The highest concentration of chloride was found in the Mainland well, which also showed the highest level of bromide with a bromide to chloride ratio of 0.0083. Since the wells were not pumped prior to sampling; prolonged ambient conditions of the wells may have left chemical constituents higher in concentration compared to true aquifer chemistry. The presence of bromide most likely indicates the presence of seawater, either as present or relic sea water intrusion. At current ambient conditions and shallow depth, it is unlikely that this well is under considerable influence of current sea water intrusion. Although a local resident of Mainland recalls the well never being activated due to salt in the well, this has not been confirmed by the study. While ions are high in relevance to samples from other wells, concentrations still fall within drinking water parameters set forth by Health Canada. Taking all complexities such as site selection and water well chemistry into account, the Mainland well is at some risk of developing or currently experiencing some form of sea water intrusion. This does not indicate that the mainland well aquifer contains the highest risk of sea water intrusion, but rather the well is coincidently sited with the highest risk.

Source water samples were collected for tritium analysis in wells that are probable as having or developing some degree of current sea water intrusion. These samples were collected from the Mainland well, Tompkins well, Flat Bay East well and the Highlands well. Results indicated that the Tompkins, Flat Bay East, and Mainland well contained between 4.15 to 6.01 tritium units (TU). The Highland well however contained only 0.85 TU. Lack of correlating bromide content for the Tompkins and Flat Bay East well suggest that tritium is most likely a result of road salt contamination. The Mainland well was the only well to present both bromide and tritium levels to suggest recent sea water occurrence.

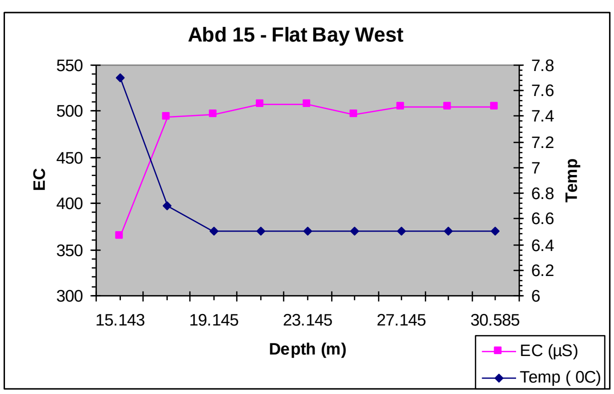

Figure 3. Conductivity profile for Flat Bay West (Abd 15).

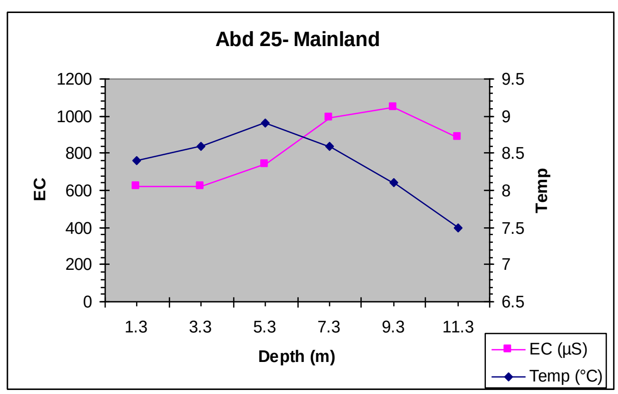

Figure 4. Conductivity profile for Mainland (Abd 25).

Conductivity profiles for each well containing bromide and that are within 500 meters from the coast are shown in Figures 3 and 4. Although the Highlands well met these parameters, a profile could not be completed for the well. Each profile reveals useful information about the well and aquifer that were unknown prior to the study.

1. As shown in the Figure 3, conductivity for the Mainland well peaked at roughly 1050 µS at 9.3 metres depth, while temperature increased incrementally after 5.3 meters depth. The Mainland well is constructed within clastic sedimentary bedrock and extends to a depth of 12 meters below surface. The well is roughly 68 metres from the coast with a 7 metre elevation above sea level. Due to time constraints and private land issues, equipment requiring a generator, such as the colloidial borescope, could not be used.

2. The Flat Bay West well showed moderate conductivity measured at a high of 507 µS at 21 metres depth where it remained fairly consistent throughout the entire depth of the well. Temperature also remained consistent after a depth of 17 metres where it remained at 6.5oC to the base of the well (31 metres). Total well depth was measured at 31 metres. The well is constructed in an area where surficial deposits have been measured up to 50 meters in depth (AMEC Environmental, 2008). Borehole video showed that well casing is extended to the bottom of the well with what appears to be strategically placed joints in the casing. This may or may not be an attempt at a well screen for a surficial well.

   The colloidial borescope was used at this well as a tool to assess aquifer flow direction and velocity. As shown in Figure 5, colloids suspended in the well water form a loose, but apparent pattern. The use of jointed casing to the entire depth of the well may be a cause of scatter amongst the data, and could prohibit accurate measurement. From the graph it is shown that the colloids are moving in and easterly direction (90.78 azimuths), with an average velocity of 99.84 um/sec. From these results it would suggest that the aquifer is moving inland as opposed to towards the shore.

3. The Port aux Port well is located within 500 meters from the shore but is not suspected to contain any recent sea water intrusion activity. Although the well extends roughly 25 metres below sea level, given it's elevation of 75 metres above sea level, and its 420 metre distance from the coast, the well is thought to have minimal risk of imminent sea water intrusion.

   The temperature, conductivity profile of the well indicates the well is inverted, with highest conductivity and lowest temperatures in the shallower portion of the well (Figure 4). This well in located at the base of a high grade, and is known to flow during high precipitation events. The inverted nature of the well may be explained by artesian properties. Pressures may cause heavier ions to accumulate at the top of the well as opposed to settling at the bottom.

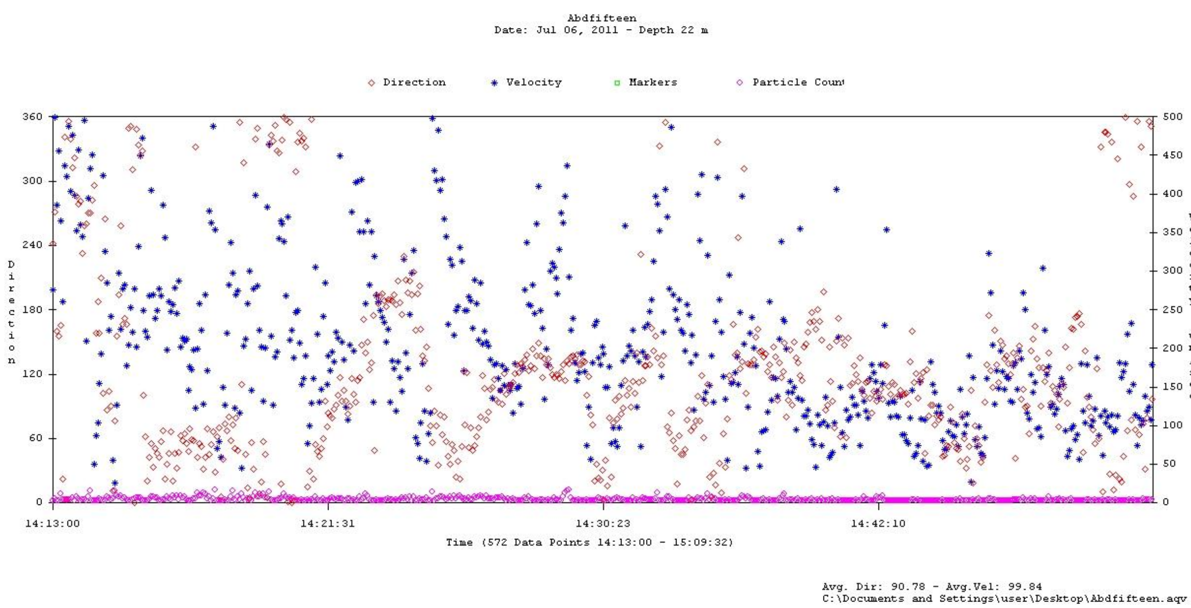

Figure 5. The colloidial borescope was used at Flat Bay West as a tool to assess aquifer flow direction and velocity. Colloids suspended in the well water form a loose, but apparent pattern.

The well water chemistry obtained from the study revealed some expected and unexpected results in regard to aquifer chemistry. Some wells located adjacent to the coastline with a relatively high conductivity in relevance to other wells in the study area, did not have any measurable amount of bromide (a signature element of sea water). Aquifers that have experienced sea water intrusion either at present or as relic would expect to yield this common signature ion. The Flat Bay East well (Abd 21), Tompkins (Abd 20) and St. Georges wells did not have any measurable bromide, in spite of favorable conditions and past sea level rise. Abandoned well 16 in Port aux Port East, did contain measurable bromide. The well however is over 400 metres from the coast and is located at the base of a high grade, 75 metres above sea level.

4 CONCLUSIONS

The study was a ground up attempt to shed light on the effects of climate change and sea level rise on coastal aquifers. Although chemical analysis revealed only one well (Mainland) contained tritium, bromide and chloride which may suggest recent sea water presence, the salt water-fresh water interface is not thought to have been fully intersected. The presence of bromide in six (Table 1) of the wells sampled, however, suggest sea water may have entered the aquifer during a past heightened sea level event(s). Bromide presence may therefore be useful as indicator for aquifers at risk of reoccurring sea water intrusion with ongoing sea level rise and isostatic submergence.

Only through studies and monitoring can proper adaptation take place and contribute to the long term preservation of coastal groundwater. Baseline studies such as these are crucial to not only recognize sea water intrusion but also to supply a foundation on how to protect coastal aquifers while still maintaining an accustomed standard of living. Long term monitoring of aquifer chemistry and conductivity would be most beneficial to evaluate the progression of sea water intrusion over time. This may be a possible avenue for future study.

Acknowledgments

Communities of St. Georges, Flat Bay Eats, Flat Bay West, Tompkins, Mainland, Cape St. George, Stephenville, Port aux Port East, Bay St. George South. A special thank to Eric Watton and Kim Bitterman for technical and field support. Department of Natural Resources Chemistry Lab. Dorothea Hanchar for supervision and guidance. The Department of Environment Water Resources Management Division and Groundwater Section for in kind contribution.

IV. Discussion / Summary

The studies presented here, conducted in each of the four Atlantic Provinces, assess the potential implications of CC with respect to SWI. Key factors in these evaluations include the current susceptibility to SWI, the potential impact of rising sea-level, altered recharge conditions, and the influence of GW extraction on the position of the SFI. It is important to note that none of the work presented here examines the effect of coastal erosion on SWI, and in areas where significant recession of coastlines is anticipated, the SFI can be expected to move inland in tandem with changes to the coastal shoreline.

The susceptibility to SWI varies considerably on a regional basis and on a local scale as a result of differing hydrogeological conditions. Furthermore, the degree of investigation of SWI differs from region to region, largely in response to the type of local water supply sources and infrastructure and historical problems with SWI. The majority of well documented cases of SWI examined in this study are located in areas underlain by sedimentary strata of the Maritimes Basin. As this group of geological formations is widespread within the study region, this correlation may be entirely coincidental. However, the hydrogeological characteristics of these areas in general are favourable for the development of GW supplies, thus they may be preferentially represented in coastal areas of the region.

Regional evaluations, such as those conducted in Nova Scotia and Newfoundland and Labrador, do suggest the potential for SWI in other geological / hydrogeological environments within Atlantic Canada. Further work is merited in these regions, in order to characterize in more detail the vulnerability to SWI.

Furthermore, the site specific studies conducted in Nova Scotia (Pugwash and Wolfville areas) highlight the importance of fully evaluating sources of saline GW using multiple approaches, and demonstrate that other sources of saline GW, such as those related to geological features (e.g. salt deposits in Pugwash), may also be important considerations in the development or maintenance of GW supplies, albeit via different strategies. In the Pugwash case for example, if an (incorrect) attribution of GW salinity to SWI was made for northern parts of the community, it could potentially lead to efforts to seek alternate sources of groundwater further to the south, where formation brines from the Windsor group pose a much greater threat to GW quality.

For the studies conducted in New Brunswick and Prince Edward Island, in documented areas of SWI, several conclusions can be drawn. First, sea level rise alone is unlikely to result in significant threats to coastal GW supplies. Numerical simulations for the coastal aquifer in the Richibucto area demonstrate that sea level rise of up to 1.86 m (twice the predicted increase) was the least significant factor in simulated increases in GW salinity in shallow to intermediate depth aquifers, and becoming significant only at much greater depths. Similar simulations for the Summerside region of PEI suggest only slight changes in the position of the SFI as a result of the combined effects of sea level rise and altered recharge rates, with a maximum displacement of the SFI of only 20 m under the most extreme CC scenario (A1F1) evaluated.

Secondly, changes to GW recharge rates could be expected to alter the position of the SFI as a result of changes to fresh GW fluxes toward coastal discharge points. Here the degree of uncertainty regarding future recharge rates is greater as a result of the complex array of factors involved. Predictions from the Richibucto simulations suggest that in the absence of significant GW demand (i.e. pumping activity), declining recharge is likely to be the most significant factor affecting the position of the SFI in shallow to intermediate portions of the aquifer except near the well field. While this suggests that for municipal water suppliers, management of water demand may be the key factor in maintaining water supplies, in other regions the impact of changes to GW recharge rates on the position of the SFI may be of greatest significance to coastal rural development relying on private, on-site water wells. Here, better awareness of the potential occurrence of SWI under existing conditions may be the primary consideration, given the greater uncertainty in estimating changes to recharge. However, it would suggest that planners and developers should err on the side of caution with potential changes in the position of the SFI being one of the considerations in evaluating the long term sustainability of coastal GW supplies. Furthermore it highlights the need for more investigation of CC impacts on GW recharge, in part in relation to SWI, but also from a broader GW management perspective.

Simulations conducted for the Summerside area of PEI employed a somewhat simpler approach to estimating future recharge conditions, assuming changes in recharge to be roughly proportional to projected changes in precipitation. While not assessed independently, as noted above, the combined factors of rising sea level and changed recharge conditions did not result in significant changes to the position of the SFI.

Finally, but most importantly, studies in both New Brunswick and PEI suggest that the greatest sensitivity is related to the rate of GW withdrawal from coastal aquifers. While it is noted that water demand and associated GW withdrawal rates are not necessarily directly related to CC, the investigations documented here clearly show that management of GW extraction is the most important factor in preserving the quality of GW in coastal aquifers. In the Richibucto case, simulations incorporating a yearly increase in demand of 0.5% (increase by a factor of 2.3 by the year 2100) suggest pumping activity to have the greatest impact on GW salinity. Significant effects are predicted primarily in the vicinity of the well field, although maximum simulated increases in total dissolved solids are 72 mg/L, a modest increase relative to concentrations expected at the SFI.

In the Summerside simulations, a hypothetical pumping scenario at a rate of 70 L/s from an inactive municipal well field in Linkletter is examined. The well field was partially developed (exploration work only) in anticipation of future demand by the City of Summerside. The simulations conducted as part of this work predict that under these pumping conditions, brackish water could reach the well field within 25 years, and salt concentrations in excess of 15,000 mg/L could occur after 50 years.

From each study it is clear that management of GW extraction rates, whether in response to increased demand as a result of CC, or simply as a matter of growth and development, appears to be the most important factor in preserving the integrity of coastal GW resources.

Furthermore, work in each study area highlights the importance of site specific factors affecting the vulnerability to SWI.

While the current work can provide regional generalizations on the potential impacts of CC on SWI, the effective management of SWI requires detailed, site specific information. Not only can the extent of SWI vary significantly from location to location, but as illustrated by the work in the communities of Pugwash and Wolfville in Nova Scotia, confirmation of the source of saline GW is a fundamental requirement in the development of potential mitigative strategies to occurrences of saline GW. Nevertheless, work conducted on a regional scale in Nova Scotia and Newfoundland and Labrador demonstrate valuable approaches to assessing the relative risk of SWI, using relatively standard geochemical approaches.

V. Conclusions / Recommendations

The case studies presented in this report utilize a variety of approaches to investigate the threat of SWI in the Atlantic region, and thus provide some guidance in relation to the impact of CC change on coastal aquifers. In those regions where the dynamics of SWI have been studied in some degree of detail, the effects of CC, by way of sea level rise, do not appear to be a significant concern, and generally management of water demand would appear to be the most critical factor under existing or future climatic conditions.

Firm conclusions on the effect of CC on future GW recharge rates are more difficult to formulate, however the work presented here does suggest some sensitivity of the SFI to changes to overall hydrogeological regimes. While these effects appear to be less critical than water demand considerations, they may be significant in areas characterized by low net water demand, but a reliance on marginal, individual on-site water supplies in the immediate proximity of the coast.

While these conclusions are drawn on the basis of site-specific case studies, the study sites are believed to be representative of hydrogeological conditions found throughout the Atlantic region, and it is likely that these general conclusions can be extended to the region as a whole. Notwithstanding the conclusions drawn above, regarding the effects of CC, SWI under current conditions presents a potentially serious threat to coastal GW supplies in some areas of the Atlantic Region, however the full extent of this vulnerability is not thoroughly documented. To a significant extent this may reflect the current geographic distribution of coastal communities depending on GW resources, and if there should be a trend toward greater utilization of GW in such settings, it will be important to consider the potential risks of SWI in planning for future water supply developments. Furthermore, simulations conducted for hypothetical water withdrawal scenarios from a well field in PEI demonstrate that prolonged periods of GW extraction could result in the gradual increase in salinity over a period of decades, even in cases where current water quality is acceptable.

Based on the findings presented here, it is recommended that a first priority for water managers, provincial regulatory authorities and planners is consideration of water demand management programs. While this is particularly relevant to the coastal settings relying on GW discussed here, reduction in overall water demand makes sense in any environment for a variety of reasons, and the strategies for water conservation are not unique to any specific source (GW or surface water) or location (coastal or inland).

Secondly, it is apparent that our knowledge of the occurrence of SWI in the Atlantic region is incomplete, and that the regional reconnaissance work noted in this report, assessing the potential for the occurrence of SWI in coastal areas, should be continued in order to assist planners and water managers in the development and management of water supplies, particularly where it heavy demand on coastal aquifers can be anticipated. Such investigations should be conducted using a variety of techniques, as is demonstrated in the cases of the Wolfville and Pugwash case studies, as multiple sources of saline GW are possible, and simplistic assumptions attributing saline GW to SWI simply on the basis of proximity to the coast, run the risk of mis-directing remedial efforts.

In cases of existing or anticipated high water demand, it would be prudent to establish long term monitoring strategies to detect the potential influence of SWI. Ideally these activities could be augmented by simulations to predict the long term impact of GW extraction on the SFI, and to assess the sustainability of current pumping rates, or the capability of supporting increased capacity in the future. Generally, provincial agencies, such as environment or natural resource departments are likely to be best positioned to conduct this type of investigation, where possible in collaboration with local water utilities and specialists within federal departments or academia.

In addition, as the state of the science evolves, further work on potential changes in GW recharge conditions is merited, based not only on possible changes in the position of the SFI, but in the broader context of GW resource management in general.

The effects of coastal erosion on SWI have not been examined in this report, however in areas with high erosion rates, inland movement of the SFI can be expected to mimic shoreline recession, representing additional factors that planners and water managers should take into account in developing and maintaining sustainable GW supplies in the immediate area of the coast.

In summary, the direct effects of CC on SWI as a result of sea-level rise do not appear to be cause for concern. Rather, overall management of water supplies depending on coastal aquifers needs to be conducted with caution, with particular consideration to GW extraction rates. Toward this end, water supply development should be conducted with due consideration to effects of GW withdrawals from coastal aquifers to ensure extraction rates are sustainable in the short and long term.

Publications and Proceedings

Much of the work completed over the course of this project has been or will be shared with the scientific community via publication and/or conference proceeding.

New Brunswick

- A final report summarizing the project work completed near Richibucto, NB, forms Appendix 1 at the end of this task report.
- Green, N., 2012, A case study of the effects of climate change on seawater intrusion in coastal aquifers in New Brunswick, Canada. M.Sc.E. thesis, Department of Civil Engineering, University of New Brunswick, Fredericton, NB, 165 p.
- LeBlanc, N., K. MacQuarrie, K. Butler, E. Mott, N. Green, and D. Connor, 2012, Hydrogeological observations from boreholes installed to investigate seawater intrusion near Richibucto, New Brunswick. Unpublished report prepared for the Town of Richibucto, 8 p. plus appendices.

- Green, N.R., E.B. Mott, K.T.B. MacQuarrie, and K.E. Butler, 2011, Preliminary hydrogeological data and numerical modeling for a seawater intrusion study at Richibucto, New Brunswick. Presentation at the 37th Atlantic Geoscience Society Colloquium, Fredericton, NB, February 2011.
- Green, N., and K. MacQuarrie, 2012, An evaluation of the importance of factors influencing seawater intrusion in coastal aquifers subject to climate change: A case study from Atlantic Canada. Poster presented at the 39th International Association of Hydrogeologists Congress, Niagara Falls, ON, September 2012.
- Mott, E.B., N. Green, K.E. Butler, and K.T.B. MacQuarrie, 2011a, Preliminary interpretation of electrical resistivity tomography (ERT) surveys investigating seawater intrusion at Richibucto, eastern New Brunswick. Presentation at the 37th Atlantic Geoscience Society Colloquium, Fredericton, NB, February 2011.
- Mott, E., K. Butler, N. Green, and K. MacQuarrie, 2011b, Application of ERT to the investigation of seawater intrusion at Richibucto, New Brunswick. Proc. of the Joint Mtg. of the Canadian Quaternary Association and the Canadian Chapter of the International Association of Hydrogeologists, Quebec City, August 2011, 6 p.
- Mott, E.B., and K.E. Butler, 2012, Assessing saltwater intrusion using electrical resistivity tomography and borehole geophysics in a coastal sandstone aquifer, New Brunswick, Canada. Presentation at the 39th International Association of Hydrogeologists Congress, Niagara Falls, ON, September 2012.

Nova Scotia

- Beebe, C., 2011. Investigation of Occurrence and Assessment of Risk of Saltwater Intrusion in Nova Scotia, Canada. M.Sc. thesis, Department of Earth Science, Saint Francis Xavier University, Antigonish, NS, 83 p.

- Beebe, C., Ferguson, G., and G. Kennedy, 2011. Analytical modeling of saltwater intrusion: tests from Nova Scotia and the eastern United States. Proc. of the Joint Mtg.

of the Canadian Quaternary Association and the Canadian Chapter of the International Association of Hydrogeologists, Quebec City, August 2011.

- Kennedy, G.W., 2012, Development of a GIS-based approach for the assessment of relative seawater intrusion vulnerability in Nova Scotia, Canada. Presentation at the 39th International Association of Hydrogeologists Congress, Niagara Falls, ON, September 2012.

Prince Edward Island

- Hansen, B., 2012. Simulating the Effects of Climate Change on a Coastal Aquifer, Summerside, Prince Edward Island. M.Sc. thesis, Department of Earth Science, Saint Francis Xavier University, Antigonish, NS, 95 p.

- Hansen, B., and G. Ferguson, 2011. Simulating submarine groundwater discharge in a changing climate, Summerside, PEI. Proc. of the Joint Mtg. of the Canadian Quaternary Association and the Canadian Chapter of the International Association of Hydrogeologists, Quebec City, August 2011.

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Appendix 1

A case study of coastal aquifers near Richibucto, New Brunswick: Saline groundwater occurrence and potential impacts of climate change on seawater intrusion

Final Project Report Prepared for the Atlantic Climate Adaptation Solutions Association (ACASA) and the Town of Richibucto

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