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COASTAL NEW ENGLAND SEPTIC SYSTEM DRAINFIELDS: GROUNDWATER
TABLE AND GREENHOUSE GAS CYCLING DYNAMICS

BY

ALISSA HELEN COX

A DISSERTATION SUBMITTED IN PARTIAL FULFILLMENT OF THE
REQUIREMENTS FOR THE DEGREE OF
DOCTOR OF PHILOSOPHY

IN

BIOLOGICAL AND ENVIRONMENTAL SCIENCES

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DOCTOR OF PHILOSOPHY DISSERTATION

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2020

Abstract

Septic systems, or onsite wastewater treatment systems (OWTS), are a common means of treating wastewater across the world in areas where centralized wastewater treatment is not a feasible option. In the US, one in five households are served by septic systems, with a quarter to over half of New England households relying on this technology to treat wastewater. A well-functioning septic system protects public health by reducing pathogen loading to ground and surface waters, and attenuating nutrients and pollutants found in wastewater before it is recharged to local groundwater. However, not all wastewater constituents are removed during these processes, and thus OWTS can serve as a source of nutrients and other pollutants. Poorly designed systems whose components are situated without sufficient vertical separation distance to the groundwater table increase the risk of nutrient and pathogen pollution to local ground and surface water resources, with implications for both human and environmental health. Onsite wastewater treatment systems in general can also be sources of greenhouse gases, including carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O) — all by-products of microbial transformations of wastewater constituents.

I examined historic and current dynamics of coastal groundwater tables along the southern Rhode Island (RI) shore, in an effort to determine whether near-shore OWTS — which are ubiquitous in these coastal communities — are at risk for compromised separation distance. Inadequate separation distance below

OWTS components among the ~8,000 OWTS located within 1 km of coastal water bodies in this area puts these communities at risk for pollution of groundwater and sensitive coastal ecosystems, both of which are valuable resources to these communities. To understand historic groundwater table dynamics, I used depth to groundwater table data submitted to the RI Dept. of Environmental Management — the agency responsible for OWTS permitting — as part of permit applications for individual properties. Based on these data, groundwater tables along the southern Rhode Island coast have been rising at a rate of 14 mm/year since 1964. Areas where potable water is imported and recharged to local groundwater have higher rates of rising groundwater tables, up to 17 mm/year. Precipitation, human wastewater inputs, and sea level rise are the major factors elevating coastal groundwater tables, whereas evapotranspiration, discharge of groundwater to coastal water bodies, and drinking water extraction are the major components lowering groundwater tables. As residents in the coastal areas continue to increase groundwater recharge (e.g. via OWTS) while importing potable water (reducing local groundwater extraction), these observed rates of groundwater table rise are likely to continue or become exacerbated, especially in light of climate change, marked by more frequent and intense precipitation events, as well as sea level rise. As groundwater tables in southern RI's coastal communities rise, onsite wastewater treatment at the ~1,500 to 2,000 properties within 200 m of a tidal water body will become increasingly compromised, posing serious risks of nutrient and pathogen pollution to both ground and surface water resources.

I also assessed how present-day groundwater tables affect OWTS installed along the southern RI coast. Data from long-term shallow groundwater monitoring wells and ground-penetrating radar surveys of 10 drainfields in the southern RI coastal zone were used to determine whether septic system drainfields have adequate separation distance from the water table. My results indicate that only 20% of tested systems are not impaired by elevated groundwater tables, while 40% of systems experience compromised separation distance at least 50% of the time. Surprisingly, 30% of systems in this study do not meet separation distance requirements at any time of the year. Neither age of system nor a system's landscape position relative to a tidal water body were correlated with compromised separation distance. The compromised separation distances may be a result of inaccurate methods — specified by the regulations — to determine the height of the seasonal high water table. Improved methods of determining seasonal high water table elevations include (1) creating a network of long-term groundwater monitoring wells in the near-shore area, to which individual measurements during the wet season at different properties could be referenced; (2) using inexpensive indicator of reduction in soils (IRIS) tubes in addition to soil profile descriptions or individual well measurements at each site; and/or (3) increasing the required separation distance between OWTS components and the seasonal high water table to account for inaccuracies of the current approach, as well as anticipated long-term rises in groundwater table elevation.

The results of these two studies suggest that, as water inputs in southern RI continue to change in the coming decades, rising groundwater tables will reduce OWTS functionality by wide-spread reduction in separation distance, which appears to be occurring already at a fast rate. A network of long-term groundwater monitoring wells in the near-shore area across the southern RI coast would ensure that regulatory and municipal officials, as well as OWTS designers, have an accurate understanding of both long-term and short-term groundwater table dynamics in the region, and provide a reference to which individual measurements at private properties could be compared. A simpler, less data-driven solution would be to increase the required separation distance to the groundwater table, to account for the fact that current methods are inaccurate, and incorporate a buffer of sorts for the continued rise in groundwater tables along the coast. Failure to enact these proposed changes — or similar ones — in the methods of determining the groundwater table elevation at time of system design and installation may threaten coastal drinking water aquifers and coastal ecosystems with nutrient and pathogen pollution from poorly designed systems — even at the beginning of a system's lifespan.

Onsite wastewater treatment systems also represent a source of greenhouse gases as microorganisms transform wastewater constituents. To understand how microbial communities in the surface soils above shallow drainfields contribute to CH₄ and N₂O consumption, I measured greenhouse gas surface flux and below-ground concentrations and compared them to the microbial communities present using functional genes *pmoA* and *nosZ*. These

genes encode portions of particulate methane monooxygenase and nitrous oxide reductase, respectively, serving as a potential sink for the respective gases. I assessed the surface soils above three drainfields served by a single household: an experimental layered passive N-reducing drainfield, a control conventional drainfield, and a reserve drainfield not in use but otherwise identical to the control. Neither GHG flux, below-ground concentration or soil properties varied among drainfield types, nor did methane oxidizing and nitrous oxide reducing communities vary by drainfield type. I found differences in *pmoA* and *nosZ* communities based on depth from the soil surface, and differences in *nosZ* communities based on whether the sample came from the rhizosphere or surrounding bulk soil. Type I methanotrophs (Gammaproteobacteria) were more abundant in the upper and middle portions of the soil above the drainfield. In general, we found no relationship in community composition for either gene based on GHG flux, below-ground gas concentration, or soil properties (bulk density, organic matter, above-ground biomass). This is the first study to assess these communities in the surface soils above a full-scale experimental drainfield.

Overall, my research indicates that coastal communities relying on OWTS must find better ways of understanding short- and long-term groundwater table dynamics to ensure that onsite wastewater systems remain an effective part of the critical water infrastructure in sensitive coastal areas. A substantial proportion of systems in this community are already at risk of poor performance based on compromised separation distance, and the situation is likely to become worse unless stakeholders begin taking a more proactive approach to ensuring system

functionality over its lifespan. Designing and installing systems to be more resilient and robust in the face of climate change, while accounting for the accompanying changes in groundwater and precipitation dynamics is critical, as is developing innovative cost-effective solutions to improving wastewater treatment performance. The layered N-removing drainfield I studied is promising in that regard, though my research on microbially mediated GHG emissions highlights the need to consider OWTS' environmental impacts holistically. Ultimately, we must find affordable and effective wastewater treatment solutions that reduce anthropogenic pollution to both the hydrosphere and atmosphere, or our communities will no longer have access to drinking water resources, nor will its members be able to enjoy the ecosystems they currently rely on for economic, recreational and aesthetic purposes.

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appreciate your willingness to join me in the field and that you always made the time to help me gain a better understanding of all things wastewater-related – I would not be the “wastewater professional” I am today without you.

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I'd also like to thank my Laboratory of Soil Ecology & Microbiology family, especially Sara Wigginton and Bianca Ross. Sara, without your friendship, empathy and laughs throughout all the ups and downs of research and early motherhood, I would not have made it this far. Bianca, your bubblyness, compassion and humor were invaluable to keeping me going, and I hope to someday approach your mad professional people skills. I hold very dear the time the three of us spent together, cursing while coding, presenting all the wastewater things, enjoying various delicious foods, walking, occasional knitting, and the general camaraderie of being in CI 024 together. Thank you, also, to the fabulous undergrads who worked in the lab with us: Natasha, Jon, Alicia, Fidaa, Connor, Lincoln and Owen. You made sure we always had clean dishes available, helped generate a substantial portion of our data, and gave us the opportunity to learn from your questions and ways of doing things. Thank you for joining the lab and allowing me to mentor you through our research projects and for your cheerfulness, and adept assistance in both the lab and field.

Finally, thank you Matt for always being there for me, and thank you to my extended family for your support, encouragement and baby-entertainment. I also have a very special place in my heart for Allison Steele and the wonderful staff at the South County YMCA for the time they spent with Stella so I could get a little baby-free time during the week and actually get this dissertation written!

Dedication

This dissertation is dedicated to my daughter, Stella Cox, without whom my life would be boring. Stella, you have helped me discover and fill parts of my heart I never knew existed, and helped me learn to embrace the uncertainty and adventure every day brings. May your spirit of independence, curiosity and determination serve you well as you grow into yourself and find meaning and fulfillment in life. Stay true to yourself, remain brave and strong, learn from your challenges, surround yourself with people who nurture and support you and your dreams, and you will find joy in a life well-lived.

Ein Stern

*Ein Stern leuchtet auch für mich
ein Stern
der mir den Weg weist
ein Stern
der mir Hoffnung schenkt
ein Stern
ein Licht
inmitten der Dunkelheit
erinnert mich
ich bin nicht allein
Liebe begleitet mich
auf all meinen Wegen.
(Anke Maggauer-Kirsche)*

Preface

This dissertation has been structured to meet the University of Rhode Island's Graduate School guidelines for manuscript format. Five sections make up this dissertation, including an introduction, three manuscripts formatted to meet respective journal requirements, and a conclusions section. Manuscript 1, titled "Preliminary Evidence that Rising Groundwater Tables Threaten Coastal Septic Systems" was published in the *Journal of Sustainable Water in the Built Environment* in 2019, and is presented here as accepted for publication. Manuscript 2, "Temporal variability in the vertical separation distance of septic system drainfields along the southern Rhode Island coast", was published in the journal of *Water, Air & Soil Pollution* in 2020. The final manuscript (3) is titled "Structure of Greenhouse Gas-Consuming Microbial Communities in Surface Soils of a Nitrogen-Removing Experimental Drainfield", and is under review at the journal of *Science in the Total Environment*. All manuscripts are presented as submitted for publication. Additional information on authorship and publication details can be found on the first page of each manuscript.

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Introduction

Septic systems, also known as onsite wastewater treatment systems (OWTS), are a common method of wastewater treatment in the US and across the globe, especially in communities where centralized wastewater treatment is not an option as a result of siting constraints or long distances between individual sources of wastewater. Across the US, about one in five of households rely on septic systems, and in the northeastern US, septic systems are the only means of wastewater treatment for 25-55% of households (U.S. Environmental Protection Agency, 2002). Conventional septic systems begin with a septic tank, which receives all the wastewater generated in the building and provides primary treatment (separation of solids by gravity). The clarified wastewater flows by gravity to a distribution box which delivers the effluent evenly to the drainfield, where final treatment of wastewater occurs (Amador and Loomis, 2018). In the drainfield, physiochemical and biological processes in the soil attenuate pathogens and some nutrients as wastewater percolates through drainfield components and native soil before being recharged to groundwater (Amador and Loomis, 2018).

Many coastal communities, like those in southern RI, rely solely on septic systems for wastewater treatment. Poorly sited and hydraulically failing OWTS are significant sources of pollution along the eastern US coastline, from Florida (Lapointe et al., 2017) to North Carolina (Humphrey et al., 2010, 2011, 2013a), as well as to the Chesapeake Bay (Lusk et al., 2017) and Buzzard's Bay in Massachusetts (Williamson et al., 2017; Costa et al., 2018).

Adequate treatment of wastewater as it moves through the soil requires sufficient vertical separation distance, the distance between the bottom of the drainfield – the infiltrative surface – and the seasonal high groundwater table (RIDEM, 2018). Reductions in separation distance due to rising groundwater tables can significantly reduce a system's ability to adequately renovate wastewater (Manda et al., 2015; Cooper et al., 2016a; Humphrey et al., 2017). The coastal lagoons – or salt ponds – along the southern RI coastline are threatened by nutrient pollution (Postma et al., 1992; Slomp and Van Cappellen, 2004; Moran et al., 2014), likely from the widespread use of septic systems. This issue may be exacerbated as coastal groundwater tables rise as a result of climate change.

In coastal areas, the groundwater table – and thus the vertical separation distance – is influenced by tidal cycles, sea level rise, precipitation, evapotranspiration, storms, and groundwater extraction and recharge (Healy and Scanlon, 2010). Information about vertical separation distance temporal dynamics in coastal properties is important for stakeholders in these communities to plan more holistically for coastal resilience. Though OWTS are an important, permanent part of the water infrastructure in many coastal areas, both in the US and globally (Amador and Loomis 2018), OWTS are rarely explicitly addressed in coastal resilience planning. Understanding how groundwater table dynamics affect OWTS, and what the future impacts are likely to be, is critical to designing resilient systems and policies so that OWTS continue to serve crucial water treatment functions in nutrient and pathogen-sensitive areas.

To understand how groundwater table dynamics in southern RI affect separation distance in coastal septic systems, I took a two-pronged approach: (1) characterize historical groundwater table dynamics from the 1960s to 2018 using data on depth to groundwater table submitted to the Rhode Island Department of Environmental Management (RIDEM) as part of the septic system permitting process, and (2) assess current groundwater table dynamics in 10 drainfields across southern RI using groundwater table monitoring wells paired with ground-penetrating radar surveys to understand how variations in groundwater table depths over the course of several years affect separation distance in these drainfields.

The first approach is described in Manuscript 1 of this dissertation. Briefly, I used historic RIDEM water table data to quantify changes in depth to the groundwater table and in separation distance over the past half-century. I assessed the extent to which groundwater tables and separation distances changed over time by landscape position (relative to coastal water bodies) and community, and evaluated the relative contribution of factors known to drive changes in groundwater table. Because the northeastern United States is subject to higher than average rates of sea level rise (Horton et al. 2014; Sweet et al. 2017), we expected that sea level rise would be a major factor influencing coastal groundwater tables, similar to the dynamics in Cape Cod described by Walter et al. (2016). I hypothesized that data from properties bordering the ocean or salt ponds would demonstrate a clear sea level rise signature, whereas groundwater tables farther inshore (and thus more distant from tidally-influenced water bodies)

would be less affected by sea level rise. Because groundwater aquifers along the coast could be affected by additional factors, including changes in population dynamics along the coast, accompanying changes in groundwater extraction and recharge, as well as different precipitation patterns over time, I also explored the potential contributions of these factors to historic changes in groundwater tables.

The second approach — to determine present day separation distance dynamics in coastal drainfields — is described in Manuscript 2 of this dissertation. This study considered how shallow groundwater tables in coastal aquifers rise and fall in the short term, in response to normal tides and coastal events, such as storms, extreme tidal fluctuations and large precipitation events (Nielsen, 1999; Cartwright et al., 2004; Morgan and Stolt, 2006; Manda et al., 2015; Healy, 2017). These short-term increases in water table elevation result in the separation distance becoming compromised temporarily, and present a recurring threat as climate change-related storm and precipitation events become more frequent and extreme (Horton et al., 2014; Lin et al., 2016; Karegar et al., 2017; Marcos et al., 2019). To understand the extent to which individual coastal OWTS are impacted by current groundwater table dynamics, I determined the vertical separation distance in 10 systems along the southern RI coast from April 2017 to May 2019. I used long-term groundwater monitoring wells and ground-penetrating radar (GPR) surveys of the drainfield at each site to estimate the vertical separation distance. GPR relies on electromagnetic pulses at microwave frequency reflecting off soil particles and interfaces and objects, such as septic system components (Annan, 2009; Humphrey et al., 2013b). By analyzing wave reflection and

attenuation patterns, subsurface stratigraphic and water table information can be visualized in real time, without the need for invasive, labor-intensive methods like hand augering and excavation. I compared field-derived values for current separation distance in each system to separation distance and water table depths indicated on the OWTS permit design submitted to RIDEM to assess whether separation distance had changed in the intervening time period. Based on the findings presented in the first manuscript, I hypothesized that older systems would show greater reduction in separation distance than younger systems, since groundwater tables would have had longer to approach the elevation of the drainfield infiltrative surface, decreasing separation distance. I also hypothesized that the shallow nature of freshwater aquifers on the sandy barrier island systems would result in measurable, short-term increases in groundwater table elevation in response to coastal events (storms, extreme high tides, large precipitation events; Manda et al., 2015). These events and the associated rises in groundwater would therefore result in temporary compromising of separation distance, especially during the wet season (January to April; RIDEM, 2018), when the seasonal high water table is at its highest elevation for the year.

Nutrient and pathogen pollution from inadequate separation distance is not the only risk that septic systems pose to the environment. The microbial processes integral to the treatment of wastewater, in both the septic tank and the drainfield, produce greenhouse gases (Diaz-Valbuena et al., 2011; Truhlar et al., 2016; Fernández-Baca et al., 2019; Somlai et al., 2019), including carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O). Manuscript 3 describes efforts to

characterize microbial communities in surface soils known for consuming CH₄ and N₂O above three different types of drainfields: a control conventional Massachusetts Title V type drainfield, a reserve drainfield identical to the control but not in active use, and an experimental layered passive N-removing drainfield. The latter is a promising alternative to the proprietary advanced N-removing septic system technologies often required in coastal areas to mitigate nitrogen loading to these N-sensitive groundwater and ecosystems (Deschutes County, 2008; BCDHE, 2014; MDE, 2016; RIDEM, 2018). The experimental N-removing drainfield is a hybrid of a conventional drainfield and a N-removing bioreactor common in groundwater remediation and agricultural settings (Robertson et al., 2000; P. W. van Driel et al., 2006; Schipper et al., 2010; Addy et al., 2016). It is composed of a 45-cm layer of medium sand (identical to conventional Title V drainfields) located immediately below the wastewater distribution laterals, and is intended to serve as an aerobic zone promoting nitrification of the ammonia-rich influent wastewater, below which there is a 45-cm layer of sand mixed 1:1 (by volume) with sawdust, designed to promote saturated, anoxic conditions and provide a C source for heterotrophic denitrification. The saturated C-rich zone is likely to impact GHG produced in the system, by creating conditions favorable for methanogenesis and potentially incomplete nitrification giving rise to N₂O. To assess whether the system's ability to mitigate N pollution to groundwater might inadvertently give rise to atmospheric pollution by these potent greenhouse gases, I measured below-ground GHG concentration and surface GHG flux, and collected soil samples from surface soils above the drainfield to characterize the

CH₄ oxidizing and N₂O reducing microbial communities. Previous work in this experimental drainfield suggested that there are differences in below-ground GHG concentration, but not surface flux among drainfield types (Wigginton et al., 2020), which led me to hypothesize that there might be differences in these microbial communities. To explore these GHG consuming communities, I sequenced the extracted DNA from surface soil samples for functional genes encoding portions of particulate methane monooxygenase (*pmoA*) and nitrous oxide reductase (*nosZ*) genes.

I hypothesized that surface soil community structure would vary in response to differences in below-ground CH₄ and N₂O concentration and drive differences in surface GHG flux. Given that different types of aerobic methanotrophs favor different concentrations of methane and oxygen (Knief et al., 2003; Knief, 2015), we expected that the structure of the methanotrophic community would vary by drainfield type based on differences in belowground methane concentrations. We expected similar patterns for nitrous oxide reducing communities (denitrifiers), based on differences in below-ground N₂O concentrations. Additionally, we expected to find methanotroph communities to vary by depth in the surface soils of these drainfields, which likely experience a gradient of methane concentration, favoring high-affinity type I methanotrophs in the surface, and lower affinity type II methanotrophs deeper in the soil profile. Finally, we expected community structure to be different between rhizosphere and bulk soil in response to higher C availability and lower O₂ levels in the former, likely providing conditions favorable for heterotrophic denitrifiers.

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Manuscript 1 :
Preliminary Evidence that Rising Groundwater Tables Threaten
Coastal Septic Systems

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Abstract

Many communities along the southern Rhode Island coast rely on onsite wastewater treatment systems (OWTS), known as septic systems, to treat and disperse wastewater. System design requires sufficient vertical separation distance between the bottom of the drainfield and the seasonal high groundwater table to ensure an adequate volume of unsaturated soil to treat wastewater before it reaches the groundwater. Based on depth to groundwater table data submitted to the state regulatory agency as part of OWTS permit applications, groundwater tables along the southern Rhode Island coast have been rising at a rate of 14 mm/year since 1964. Communities where potable water is imported have greater rates of rising groundwater tables, up to 17 mm/year. From a mass balance perspective, precipitation, human wastewater inputs, and sea level rise represent the major factors elevating coastal groundwater tables, whereas evapotranspiration, discharge of groundwater to coastal waterbodies, and drinking water extraction are the major components lowering groundwater tables. As water inputs continue to change in the coming decades, rising groundwater tables may reduce OWTS functionality, threatening coastal drinking water aquifers and coastal ecosystems with nutrient and pathogen pollution.

Keywords

wastewater; onsite wastewater treatment; septic system; separation distance; coastal resilience; coastal groundwater; rising groundwater table; coastal communities; coastal resilience planning; historic dataset.

Introduction

Poorly sited and hydraulically failing onsite wastewater treatment systems (OWTS), i.e., septic systems, are significant sources of pollution along the eastern US coastline, from Florida (Lapointe et al. 2017) to North Carolina (Humphrey et al. 2010, 2013; Humphrey and O'Driscoll 2011) to Chesapeake Bay (Lusk et al. 2017) to Buzzard's Bay in Massachusetts (Costa et al. 2018; Williamson et al. 2017). A conventional OWTS consists of a septic tank, which receives all the wastewater generated in a building and provides primary treatment, and a soil treatment area or drainfield, where the wastewater is dispersed to soil in trenches or bed systems for final treatment. This design is typical of the vast majority of OWTS across the United States. Homes along the southern Rhode Island coast, which consists of barrier beach complexes derived from glacial landforms (Soil Survey Staff 2016), primarily rely on conventional systems to treat residential wastewater.

Adequate treatment of wastewater as it percolates through the soil before reaching the groundwater table is predicated on adequate vertical separation distance, defined as the distance between the bottom of the drainfield—the infiltrative surface—and the seasonal high groundwater table (RIDEM 2018c). Reductions in separation distance due to rising groundwater tables can significantly reduce OWTS' ability to adequately renovate wastewater (Cooper et al. 2016; Humphrey et al. 2017; Manda et al. 2015). Coastal lagoons—or salt ponds—in southern Rhode Island are threatened by nutrient pollution (Moran et al. 2014; Postma et al. 1992; Slomp and Van Cappellen 2004). It is likely that

OWTS are contributing to this problem in this area, and this issue may be exacerbated as coastal groundwater tables rise.

In coastal areas, the groundwater table—and thus the vertical separation distance—is influenced by tidal cycles, sea level rise, precipitation, evapotranspiration, storms, and groundwater extraction and recharge (Healy 2017). Additionally, the landscape position of the measurement (e.g., the backslope of the barrier beach dune versus the low-lying area bordering a salt pond) is likely to influence the depth to groundwater table measurement. This study evaluated the impact of these factors on southern Rhode Island's coastal groundwater tables and the vertical separation distance in OWTS drainfields. To identify changes in groundwater tables along the coast, historical seasonal high groundwater table information available through the Rhode Island Department of Environmental Management (RIDEM) was assessed. This data set allowed quantification of changes in depth to the groundwater table and in separation distance over the past half-century. The data were used to assess the extent to which groundwater tables and separation distances have changed over time by landscape position (relative to coastal water bodies) and community, and the relative contribution of factors known to drive changes in groundwater table were explored. Because the northeastern United States is subject to higher than average rates of sea level rise (Horton et al. 2014; Sweet et al. 2017), it was hypothesized that sea level rise would be a major factor influencing coastal groundwater tables, similar to the dynamics in Cape Cod described by Walter et al. (2016). The extent to which groundwater tables might be affected by sea level

rise was expected to depend on landscape position; the authors hypothesized that measurements made on properties bordering the ocean or salt ponds would demonstrate a clear sea level rise signature, whereas groundwater tables farther inshore (and thus more distant from tidally influenced waterbodies) were likely to be less affected by sea level rise. However, groundwater aquifers along the coast could be affected by additional factors, including changes in population dynamics along the coast, accompanying changes in groundwater extraction and recharge, as well as different precipitation patterns over time. The contributions of these factors to historic changes in groundwater tables, in addition to sea level rise, also were investigated.

Methods

Hydrogeology of the Southern Rhode Island Coast

The hydrogeology of the southern Rhode Island coast was characterized and modeled by Masterson et al. (2007) (Figure 1.1(a)). The region is characterized by glacial deposits of varying thickness (3–30 m) stratified over granitic bedrock, with postglacial tidal delta sands and fine-grained sediments from coastal lagoons making up the uppermost layers directly along the coast (Masterson et al. 2007).

The northernmost boundary of the south coastal basin is made up of the Charlestown Moraine. The basin includes eight major coastal lagoons—salt ponds—that receive surface runoff and groundwater from individual subwatersheds (Figure 1.1 (a)). These salt ponds are separated from the Atlantic Ocean by barrier beach complexes. The basin also contains a long-term USGS groundwater monitoring well (Site No. 412214071394001, RI-CHW 18), in which

distance from the ground surface to the groundwater table is recorded monthly (Fig. S1). The south coastal basin aquifer system is bounded vertically by crystalline bedrock, and laterally by surface water bodies: rivers to the north and west, and Block Island Sound (Atlantic Ocean) to the south. The salt ponds receive freshwater from both surface water and groundwater inputs, and ponds containing breachways (artificial channels that connect the ponds to the ocean) also exchange water with Block Island Sound during tidal cycles. All along the southern Rhode Island shoreline, groundwater from this coastal aquifer system also discharges to Block Island Sound, although the specific mechanics of this are poorly understood (Masterson et al. 2007).

Figure 1.1. (a) Map of the watershed of the South Coastal Basin across southern Rhode Island, including subwatershed boundaries and the location of the long-term USGS groundwater monitoring well in Ninigret Park (USGS 412214071394001 RI-CHW 18; triangle on map) (adapted from Masterson et al. 2007); and (b) map of the southern portion of Rhode Island showing only parcels in the towns of Westerly, Charlestown, and South Kingstown located more than 61 m (200 ft) from a sewer line (orange lines) and with built structures (as identified by the E 911 site layer on RIGIS). These parcels with built structures are assumed to also rely on OWTS for wastewater treatment. Parcels in purple are within 1 km of the coastline.

Site Selection

To quantify changes in the depth to groundwater tables along the southern Rhode Island coast, the authors collected seasonal high groundwater table information submitted to RIDEM as part of the OWTS permitting process (RIDEM 2018a, b). To simplify the analysis of groundwater table dynamics, the authors focused on data from properties located on sandy soils (including Hooksan, Succotash, Fortress, and Beaches soil series, based on the SoilWeb online soil survey (Soil Survey Staff 2018)). These sandy soils are categorized as moderately well to excessively drained and are unlikely to contain confined aquifers or exhibit complex groundwater dynamics. Homes located on glacially derived till landforms or other headlands were excluded from this analysis. All the homes in this study are located within 200 m of the southern Rhode Island coastline—including tidally flushed salt ponds along the shore. This places all properties within RIDEM-designated OWTS critical resource areas, which are regions within coastal watersheds deemed to be particularly sensitive to nitrogen and pathogen pollution from OWTS, and have special regulatory protections and requirements in place (RIDEM 2015). This analysis of near-shore groundwater tables encompassed an area of 23.6 km² that included 5,027 properties (Figure 1.1(b)).

Groundwater Table Depth Data

Using the georeferenced tax assessor's maps from the towns of Westerly, Charlestown, and South Kingstown, Rhode Island (referred to collectively as southern RI), homes within 200 m of a coastal feature (Atlantic Ocean or salt pond) and located on sandy soils were identified. The owners of record for each parcel were then cross-referenced with RIDEM's online OWTS permit database (RIDEM 2018a, b). For each parcel where an owner of record matched a submitted OWTS permit, the values for depth to groundwater table measurements reported to RIDEM were extracted. The depth to groundwater table measurements were made from the ground surface during the wet season (January–April), when seasonal high groundwater tables generally are highest (RIDEM 2018c). The data set included repeat measurements made over the course of multiple years at 52 of the 385 properties in this study.

Roughly 60% of OWTS permit applications submitted prior to 1990 had groundwater table readings that were verified and/or witnessed by a RIDEM staff member. The method of measuring depth to seasonal high groundwater tables approved by RIDEM changed between 1964 and the present. Before 2002, the depth to the seasonal high groundwater table was measured between January and April (the wet season) using a temporary 10-cm-diameter (4-in.) slotted/perforated groundwater test well, generally less than 3 m (10 ft) deep (RIDEM 1978). In 2002, regulations changed to allow the use of soil evaluations to estimate seasonal high groundwater tables, based on redoximorphic characteristics visible in the soil (RIDEM 2002). According to the regulations, this

method is considered by RIDEM to be comparable to test-well groundwater table depth measurements. All OWTS regulations since the 1980s require a RIDEM staff person to verify soil evaluations and depth to groundwater table measurements. Examination of permit applications submitted after 1990 found that ~56% of the measurements of depth to groundwater table were verified either by test-well measurements or by soil descriptions verified by RIDEM staff. In contrast, groundwater table measurements associated with permit applications submitted for alterations or repairs to an existing (failed or failing) system were rarely verified, because the regulations at the time did not make this an explicit requirement of approval (RIDEM 2018c). Overall, the verification data suggest that the groundwater table data used in this analysis represent the groundwater table dynamics observed at the time of the permit application.

Geospatial Analysis and OWTS Distribution

ArcGIS (ESRI v 10.5) was used to analyze the geospatial distribution of property parcels, OWTS, depth to groundwater table, as well as the location of sewer areas in this study. Any parcel located within 61 m (200 ft) of a sewer line was not considered in this study (Fig. S2), since regulations require homes within 61 m (200 ft) of sewer lines to connect to public sewers whenever possible (RIDEM 2018a). Only parcels known to contain a structure – based on E-911 Site information (RIGIS 2017) – were considered in this analysis (Figure 1.1(b)), as vacant parcels are unlikely to have OWTS in place.

Population and housing data were extracted from the 1990, 2000 and 2010 U.S. Census Data (RIGIS 1991, 2007, 2012) and analyzed with ArcGIS. To

calculate population and number of houses along the coast, the reported number from the census data was divided by the area of the census block. The resulting ratio was multiplied by the region of the census block that intersects with a 1-km buffer from all coastal features along the southern RI coast. Although this assumes that the population and housing units are evenly distributed over a large rural census block, these area-corrected calculations appear to be reasonable, given that comparison of the GIS-generated population estimates were very close to historic town-wide data from the Rhode Island Division of Statewide Planning (RIDSP 2010).

The number of non-sewered parcels for the towns of South Kingstown, Charlestown and Westerly are shown in Table S1 and Table S2. Of the 21,822 total parcels across all three towns, over half are located within 1 km of the coast, spanning an area of 61.5 km². Along the coast, 8,804 of those parcels currently have built structures on them, which the authors inferred to mean that these parcels most likely also have OWTS (RIGIS 2017; Figure 1.1(b)).

Calculations concerning groundwater imports (per capita and per OWTS) were based on converting volumes of wastewater entering the drainfield (Amador et al. 2018) into linear (vertical) length equivalents, by dividing the total volume of wastewater generated by the parcels in question by the area occupied by all the parcels, as calculated in ArcMap. Linear annual groundwater import and numbers (Table S7) therefore represent mean per-parcel rates of groundwater additions (recharge) or subtractions (extraction).

Results & Discussion

Water tables along the southern RI Coast between 1964 and 2017

Based on the historic depth to seasonal high groundwater table measurements reported to RIDEM in OWTS permits between 1964 and 2017, groundwater tables in southern Rhode Island appear to be rising over time. A linear regression of the entire data set ($n = 427$), which included some properties with multiple groundwater table measurements made in different years ($n = 52$ of 385), indicated that the groundwater table along the southern Rhode Island coast has been rising at a rate of 14 mm/year (Figure 1.2; regression significance $p < 0.001$) since the 1960s. From 1964 to 2017, this amounted to a net increase of almost 69 cm in the average groundwater table height along the coast. When the depth to the groundwater table measurements are grouped by decade, Kruskal–Wallis rank sums and pairwise Wilcoxon rank sum tests indicated that groundwater tables in the 1970 and 1980s were significantly lower ($p < 0.001$) than groundwater tables in the 1990, 2000, or 2010s (Figure 1.3). This analysis supports the overall trend described by the regression of the entire data set (Figure 1.2), indicating that groundwater tables are rising over time. The rate of change in groundwater table levels reported in this study is less than that in a USGS groundwater well (CT-NHN 235) monitored in New Haven, Connecticut located approximately 550 m from the bay (Bjerklie et al. 2012): from 1940 to 1975, a rise of ~ 1.2 m (4 ft), a rate of change of approximately 34 mm/year, was observed in this well (Bjerklie et al. 2012). Another long-term USGS groundwater monitoring well (G-852) ~ 3 km from the coast in Miami-Dade County, Florida

indicates groundwater tables rising at a rate of 2.8 mm/year since 1974 (Sukop et al. 2018), which is lower than the rate observed in southern Rhode Island. On the other hand, in New Hanover County, North Carolina, groundwater levels between 1964 and 2012 lowered by as much as 3 m (10 ft) in response to pumping stresses, which is equivalent to rates of groundwater tables lowering by as much as 60 mm/year (McSwain et al. 2014). The 14-mm/year rate of groundwater rise reported in this study falls well between previously reported rates of groundwater table change in coastal communities in the eastern United States.

Figure 1.2. Depth to groundwater measurements submitted to RIDEM as a part of the OWTS permitting process along the southern Rhode Island coast between 1964 and 2017 ($n = 427$). On the y-axis, 0 represents the ground surface. The red line indicates the minimum depth to

water table in critical resource areas, as per RIDEM regulations, which changed in 2008. The blue line represents the linear regression (shaded grey region indicates the 95% confidence interval).

Figure 1.3. Groundwater table depths grouped by decade (based on data from Figure 1.2). Boxes with the same letter are not significantly different from one another, based on Kruskal-Wallis rank sum and pairwise Wilcoxon rank sum tests.

Current OWTS regulations include provisions establishing both minimum vertical separation distance between the infiltrative surface and the groundwater table, as well as minimum “depth to groundwater table”, as measured from the ground surface (RIDEM 2018a). The first regulations to specify either of these distances were introduced in 1968, by the RI Department of Health (RIDOH). Since then, the vertical separation distance from the soil treatment area

infiltrative surface to the groundwater table must be at least 91 cm (3 ft; RIDOH 1968). These regulations were in effect in 1978, when RIDEM became the regulating body for OWTS (RIDEM 1978), and still apply today in many areas (RIDEM 2018a). In Critical Resource Areas, which includes the salt pond watersheds (RIDEM 2015), the minimum depth to the groundwater table from the soil surface must be 152 cm (5 ft; red line in Figure 1.2), while minimum vertical separation distance in the area must be 122 cm (4 ft; RIDEM, 1978, 1996, 2002, 2018).

Since it is difficult to accurately determine the vertical separation distance from some of the plans in the RIDEM database, an assessment was made of compliance with depth-to-groundwater table regulations based on reported data. Nearly 50% (n=213) of reported measurements are noticeably shallower than the post-1974 minimum depth to groundwater table regulations in Critical Resource Areas (152 cm (5 ft); Figure 1.2), as required by RIDOH (RIDOH 1974), and later by RIDEM. Beginning in 1974, regulations allowed for exemptions to the 152 cm minimum groundwater table requirement. For example, parcels with seasonal high groundwater tables between 60 and 91 cm (2 – 3 ft) could be developed if the drainfield was oversized by twofold (RIDOH 1974). Therefore, it is possible that some of the shallow groundwater table measurements represent applications for variances, or applications for OWTS permits that may have been denied by RIDEM after submission. Beginning in 1996, advanced treatment technologies with better nutrient and/or pathogen reduction capabilities than the standard

OWTS design may have been installed at sites with shallow groundwater tables, potentially lowering the risk of groundwater contamination (RIDEM 1996).

The correlation between time and depth to the groundwater table, although statistically significant, was not particularly strong (Figure 1.2), suggesting that factors other than time contribute to the variation in this parameter. The area included in the analysis included three different towns with differing demographics and temporal and spatial water use patterns, as well as a variety of landscape positions, all of which can contribute to variations in depth to the groundwater table. The role of these factors was therefore examined next.

Water Table Trends by Landscape Position

As mentioned previously, the southern RI coast is characterized by barrier beach complexes, where portions of the coastal headland are interrupted by salt ponds (Figure 1.1, Figure 1.4 (f)). These salt ponds are separated from Block Island Sound by spits of land running roughly parallel to the mainland, called barrier beaches. Barrier beaches often contain natural and/or man-made channels – or breachways (Figure 1.4 (f)) – that allow these ponds to flush (albeit poorly) during tidal cycles with near-shore ocean waters.

Figure 1.4. (a-e) Depth-to-water table measurements by landscape position; and (f) examples of landscape position. The blue lines represent linear regressions, and the shaded grey areas indicated the 95% confidence intervals of the regression model. Only the Barrier Beach and Salt Pond landscape positions had significant rises in groundwater tables; the other landscape positions had no significant change in groundwater tables over time.

When examining groundwater table depths at a more refined spatial scale, at which location is taken into account, as opposed to the analysis of the entire data set for a much larger area that does not differentiate proximity to surface water, different trends emerge. Groundwater table depths grouped by landscape position indicate that groundwater tables in the back barrier and along the salt pond are closest to the ground surface, whereas groundwater tables for homes located on the barrier beach facing the ocean have slightly greater depth to groundwater from the soil surface (Figure 1.4). This was expected, since homes

located on the barrier beach are often on the backslope of the dune, and thus have a higher elevation relative to sea level, than the homes closer to the salt pond, where the elevation difference between the ground and the salt pond water surface is rarely greater than 1.5 to 2 m (5 to 6.5 ft). When examining groundwater table depth trends over time by landscape position, parcels located on the barrier beach and along the salt pond experienced the steepest rise in groundwater, with values of 22 mm/year ($p < 0.001$) and 27 mm/year ($p < 0.01$), respectively (Figure 1.4). These findings partially support the initial hypothesis that sea level rise is likely to be impacting coastal groundwater tables, as properties along the barrier beach and salt pond are in close proximity to the ocean. The lack of significant change in the “near salt pond” and “near beach” landscape position also supports the hypothesis that parcels farther from the tidally influenced water bodies would have groundwater table dynamics that are less affected by less sea level rise, serving as a control. This is further supported by data from a long-term USGS groundwater monitoring well ~1 km north of Ninigret pond (RI-CHW 18; Figure 1.1), which does not indicate that there has been a significant change in groundwater table heights in that region over the same time period (Fig. S1). Together, these support the hypothesis that groundwater farther from the coast is less likely to be affected by sea level rise. Nevertheless, these rates of groundwater change far exceed the rate of sea level rise, and suggest that sea level rise is not sufficient to account for all of the observed changes in groundwater tables. Therefore, other factors affecting

groundwater table dynamics may be driving these changes and warrant consideration.

Water Table Trends by Town

Towns along the southern RI coast have different home occupancy and tourism patterns, as well as differing sources of potable water (imported vs. municipal vs. private well), leading to an examination of differences among historic groundwater table trends by town. When the data are analyzed by town – disregarding landscape position – Charlestown and Westerly show statistically significant, shallower depths to water table over time, at rates of change of 17 and 12 mm per year, respectively, while South Kingstown shows no significant rate of change in depth to groundwater table over time (Figure 1.5). Because these towns are all located on the same coastline, and therefore subject to the same sea level rise, these data suggest that other factors are driving changes in groundwater table.

Figure 1.5. Depth-to-water table measurements submitted to RIDEM between 1964 and 2017 separated by town. Blue lines represent linear regressions, and the shaded gray region

indicates the 95% confidence interval of the regression model. Only Charlestown and Westerly had significant rises in groundwater tables over time.

Drivers of Changes in Groundwater Tables

Water tables have likely been changing along the southern RI coast over the past decades for a variety of reasons, including: (i) sea level rise, (ii) development and population growth, (iii) changes in precipitation and evapotranspiration, (iv) increased importation of potable water, and (v) changes in export of water to coastal water bodies.

Sea level rise. The Newport, RI tidal station – the closest station to the study area where sea level rise is recorded – is about 20 km from the easternmost portion of South Kingstown coast, and roughly 50 km from the westernmost portion of the Westerly coast (Figure 1.1(b)). There has been a documented 3.18 mm/year rise in sea level at this station (NOAA / National Ocean Service 2017) since 1964 (Figure 1.6(a)). According to Karegar et al. (2016), the southern RI region shows little subsidence due to groundwater extraction. In addition, isostatic rebound in this area results in a change of ~ -1 mm/year of the ground surface (Karegar et al. 2016). Since these signatures are captured by relative sea level rise observed at the Newport, RI tidal station, these drivers are not considered separately. Given the 3.18 mm/year rate of sea level rise (Figure 1.6(a).), coastal groundwater tables should have risen by roughly 1 mm/year, assuming one unit of sea level rise corresponds to a 35% increase in the same units for groundwater table height, as observed for Cape Cod, MA by Walter et al. (2016). Modeling of groundwater aquifers in New Hampshire indicates that near-

shore groundwater tables increase 54 to 88% for every unit of sea level rise (Knott et al. 2018). However, Walter et al. (2016) also found that groundwater tables directly along the coast can rise as much as sea level. Similarly, modeling of the New Haven, CT area shows that near-coastal groundwater tables are expected to rise at the same rate as sea level, as long as recharge to the groundwater does not increase (Bjerklie et al. 2012). Therefore, groundwater tables could have risen as much as 3.2 mm per year along the southern RI coast. Based on the hypothesis that sea level rise is contributing to rising coastal groundwater tables, all parcels directly adjacent to salt ponds and the ocean should respond to sea level rise similarly, as the groundwater aquifers in these areas are all subject to the same oceanic and tidal influences. However, the rates of change in groundwater tables in Figure 1.4 are several times to an order of magnitude greater than would be expected based solely on the groundwater-sea level interactions described by Walter et al. (2016), Knott et al. (2018), or Bjerklie et al. (2012). As noted previously, while the groundwater tables at the salt pond and barrier beach landscape positions show a comparable increase in groundwater tables, the groundwater tables along the back barrier do not behave in the same manner (Figure 1.4). Sea level rise might account for ~3 mm of the annual rise in groundwater tables along the southern Rhode Island coast, leaving ~11 mm of the observed changes unaccounted for. Thus factors in addition to sea level rise must affect coastal groundwater tables.

Figure 1.6. (a) Monthly mean sea level recorded at the Newport, Rhode Island tidal station from 1964 to 2017; dots represent monthly means, the blue line represents the linear regression for the data, and the grey shaded region is the 95% confidence interval for the

regression (data from NOAA/NOS 2017); and (b) mean annual precipitation for Rhode Island from 1964 to 2017; the red horizontal line represents mean annual precipitation (1,147 mm), the blue line represents the linear regression for the data (rate of change = 2.7 mm/year; $p = 0.057$, $R^2 = 0.047$), and the grey shaded region is the 95% confidence interval for the regression; solid triangles indicate years that were wetter than average (more than 1,147 mm of precipitation), and solid circles correspond to drier-than-average years.

Development and population growth. The population along the southern RI coast has more than doubled in the past half century (Table S3). Over this time period, the number of homes built in the region has also increased, though many of these residences are used seasonally (Table S4, Table S5). Analysis of these data did not find strong correlations between development indices (e.g., US Census data) and changes in coastal groundwater tables (Table S6). Regressions did not account for more than fractions of millimeters of groundwater change, representing a very small fraction of the overall change in groundwater table depths shown in Figure 1.3. As such, neither development nor population growth was considered to be an important factor contributing to changes in groundwater tables.

Water imports and recharge via OWTS. Another potential driver of groundwater table changes along the coast are water inputs from wastewater percolating through the soil, especially when coupled with import of potable water from outside the watershed. For example, modeling of groundwater dynamics in New Haven, CT shows that a 12% increase in recharge to coastal surficial aquifers, when coupled with 0.9 m (3 ft) of sea level rise, can raise groundwater tables by an additional 30 cm (1 ft) (Bjerklie et al. 2012). In areas where homeowners rely on private drinking water wells to supply their potable

water needs, and OWTS to treat the wastewater they generate, a balance of groundwater extraction and recharge is expected. This scenario describes the majority of the town of Charlestown, where there are no public sewers or municipal water systems. However, in homes located directly on the barrier beach it is quite commonplace to have potable water cisterns (>4000 L capacity) on the property, which are filled regularly by tanker trucks with potable water sourced from areas outside the watershed, rather than relying on private wells to supply a home's water needs. Therefore, properties in the town of Charlestown directly along the southern coast are importing water, which is then recharged to local groundwater aquifers by OWTS dispersal. Since a good number of the groundwater table measurements came from these regions of Charlestown, it is possible that this groundwater import is reflected in the rate of groundwater table rise for all of the Charlestown sites (Figure 1.5), even though homes located on and near the salt pond may rely on individual private wells for their water needs.

In contrast, the entire town of Westerly is served by a public water supply, so that all the water entering a coastal home results in a net import of groundwater to the region as the wastewater treated by the home's OWTS percolates into the groundwater. Therefore, Westerly groundwater aquifers receive constant inputs – even though there is no drawdown of groundwater by individual wells – leading to elevated groundwater tables (Figure 1.5). Additionally, the quality of the coastal groundwater may not be of immediate concern to homeowners or municipal officials, as it does not present a health risk

were it to be contaminated by pathogens or nutrients from poorly functioning OWTS.

In South Kingstown, coastal homes are also served by municipal water, although unlike Westerly, this water district is smaller and localized to the general coastal area. On the surface, this would suggest that South Kingstown is also importing groundwater, though this is not reflected in the groundwater table rise trends as it is in Westerly (Figure 1.5). This may be due to the coastal morphology of South Kingstown, where the aquifers serving the local coastal water district are both being drawn down by regional municipal wells and recharged by the OWTS in the region (Figure 1.1(a)). This could explain why South Kingstown's coastal groundwater tables do not appear to be rising at the same rate as those in Westerly or Charlestown (Figure 1.5).

Estimates of the amount of water being imported per capita that is recharged to local groundwater aquifers via OWTS yields some surprising insights. Assuming a mean daily flow for systems in this area of about 500 L of wastewater per day for a conventional OWTS, corresponding to roughly 200 L per capita wastewater generation (Amador et al. 2018), one can estimate both the contributions from all OWTS in the coastal RI area, as well as the per capita contributions to groundwater, using U.S. Census data from 2010 (RIGIS 2012). If one considers all of the OWTS in the region (which includes both seasonally and year-round occupied homes), roughly 6 to 56 mm (depending on year, data source, town, and area of interest; Table S7) of OWTS-treated wastewater are percolating into the groundwater on an annual basis. If one focuses on the area

within 200 m of a coastal waterbody (representing the only the region covered by the dataset in this analysis), and account for the 40% pore space in the region's soils, the input to groundwater per OWTS ranges from 50 to 125 mm per year (Table 1.1). Because this includes a large number of seasonal systems (between 26 and 49% of homes, depending on the town; Table S5), these values are a clear overestimate, since seasonally-occupied homes would not be recharging groundwater during the months of September to May. However, this does represents a peak flow scenario, where summer homes are occupied by large numbers of seasonal residents and their visitors. When considering groundwater inputs per capita, a more conservative value of 4 to 22 mm (again depending on year, data source, town, and area of interest; Table S7) of water being recharged to groundwater emerges. Constraining this estimate to the area within 200 m of a coastal water body, the annual below-ground inputs to the groundwater table range from 15 to 27.5 mm (Table 1.1). However, the U.S. Census population data that this calculation is based on presumably only includes permanent residents, so it likely represents normal, non-peak flow background recharge rates.

Unfortunately, it is difficult, if not impossible to adequately quantify summer home occupancy in the southern RI region – and associated groundwater inputs from OWTS – because no entity tracks seasonal occupancy or the number visitors to the region. Nevertheless, seasonal and tourist activity in the area is likely to substantially impact the water inputs along the coast, so it is likely that the true value of annual inputs to coastal RI groundwater aquifers lies somewhere

between the calculated values of total OWTS (peak-flow) and total population (non-peak flow) groundwater inputs.

Table 1.1. Summary of factors contributing to rising groundwater tables along southern Rhode Island coast.

	Factor	Low Estimate	High Estimate
		Annual Groundwater Rise (mm / yr)*	Annual Groundwater Rise (mm / yr)*
Imports	Recharge from Wastewater [Per Capita] {Per OWTS}	[15] {50}	[27.5] {125}
	Sea level rise ¹	1.0	3.2
	Precipitation Increase (1960 – 2017) ²	6.8	6.8
	Mean Precipitation (RI) ²	2868	2868
Total Imports		[2891] – [2905] {2926} – {3003}	
Exports	Evapotranspiration ³	1147	1405
	Groundwater Discharge to Salt Ponds ^{4*}	1250	1500
Total Exports		2397 – 2905	

Note: For per capita and per OWTS groundwater imports, volumes were converted to length measurements by dividing the total volume for all parcels within 200 m of a coastal water body by the area occupied by all the parcels with OWTS in the near-shore area, as calculated in ArcMap (Table S7). Values for sea level rise, precipitation, and evapotranspiration are based on calculations described in text. Vertical lengths of groundwater additions or subtractions (imports, precipitation, evapotranspiration, and export to coastal waters) were converted to represent the belowground vertical distance by which each factor would raise or lower the groundwater table, considering an average soil porosity of 40% for the region's sandy soils (Weil and Brady 2016).

¹(NOAA / National Ocean Service 2017; Walter et al. 2016) ²(NOAA National Centers for

Environmental Information 2018) ³(Sanford and Selnick 2013) ⁴Ninigret & Greenhill Pond regions (Scott and Moran 2001) *Does not account for discharge directly to Atlantic Ocean

Precipitation and evapotranspiration. Long-term changes in precipitation patterns along the coast could also contribute to the overall rise in groundwater tables. Mean annual precipitation for Rhode Island is 1147 mm or 45.17 inches per year, averaged over the 20th century (NOAA National Centers for Environmental Information 2018). However, a linear regression of precipitation data from 1964 to 2017 indicates that the average annual precipitation has increased by 2.7 millimeters per year (Figure 1.6(b)).

Differences in annual precipitation patterns may have large effects on wet season groundwater tables. When this dataset is analyzed by focusing on properties (n=52 out of 385) where multiple permit applications were submitted to RIDEM in different years (Figure 1.7), groundwater table behavior shows a great deal of interannual variability. A linear regression of this dataset indicates that groundwater tables are rising at rate of 21 mm/year ($p < 0.00$; $R^2 = 0.11$), which is faster than the overall historic dataset indicates (Figure 1.2). Of these 52 sites, 8 sites (15%) showed an absolute increase in depth to the groundwater table ("lowering"; range 5cm to 129.5 cm), 10 sites (19%) exhibited no change in groundwater table depth over time, and 32 sites (65%) showed a decrease in depth to the groundwater table ("rising"; range -198 cm to -5 cm; Figure 1.7). For groundwater tables that rose, the rate of change was from -76.2 to -0.6 cm/year; for groundwater tables that lowered, the rate of change ranged from 0.3 to 55.9 cm/year (Figure 1.7). The timing of these measurements – e.g., whether repeat

measurements were made during exceptionally dry or wet months of the wet season (January – April) – may drive the sometimes extreme rates of change (e.g., letter 'c', Figure 1.7). When site-specific changes in groundwater table depth are compared with precipitation data, it is clear that “wet” (above average annual precipitation) or “dry” (below average precipitation; Figure 1.6(b) and Figure 1.7) years often appear to influence the groundwater table measurements. Comparing these annual precipitation trends with site-specific repeat groundwater table measurements shows that in some cases, “wet years” result in shallower groundwater table measurements than measurements made during dry years (Figure 1.7). This suggests that the differences in depth to the groundwater table indicated by repeat groundwater table data reflect, in part, the natural variation of groundwater tables in response to precipitation. The data in this analysis indicate that in some cases, differences in annual precipitation can account for the change in the groundwater table (letters 'a' and 'b', Figure 1.7) but not in others (letters 'c' and 'd', Figure 1.7).

Figure 1.7. Repeat measurements of depth-to-water table over time at individual sites ($n = 52$ of 385) along the southern Rhode Island coast, organized by overall trend (lowering groundwater tables, no change

in groundwater tables, and rising groundwater tables). Each line represents a single site with depth to groundwater table measurements reported to RIDEM in different years. Solid circles indicate measurements taken during dry years, and solid triangles indicate measurements taken in wet years (Figure 1.6(b)). Letters a and b indicate sites where the difference in annual precipitation might account for the changes in groundwater table depths from year to year, whereas letters c and d denote sites where differences in annual precipitation cannot account for the changes in groundwater table depths.

From a groundwater recharge perspective, a fraction of the precipitation is lost to the atmosphere by evapotranspiration, which in Rhode Island is between 40 and 49% of annual precipitation (based on data for 1971-2000; Sanford and Selnick 2013). Recharge rates for the glacial materials in the southern RI region are reported to be 686 mm or 27 inches per year, which represent 60% of the annual precipitation (Masterson et al. 2007). In general, precipitation falling upon the region's sandy soils infiltrates quickly, resulting in elevated groundwater tables within hours of a precipitation event, which recede to pre-precipitation event levels within a few days (Morgan and Stolt 2004). One can therefore assume that between 40 and 50% of the mean annual precipitation (1147 mm; Table 1.1) is being returned to the atmosphere via evapotranspiration. This calculation suggests that 458 to 562 mm of the average annual precipitation are therefore exported from the southern RI groundwater aquifers. Since evapotranspiration includes surface as well as below-ground water losses to the atmosphere, this length of water being lost via evapotranspiration can be converted to a below-ground in-soil vertical distance: considering that sandy soils are comprised of ~40% pore space (Weil and Brady 2016), 458 mm of water being lost by evapotranspiration would take up 1147 mm of vertical distance in soil, since only 40% of the soil consists of pores that can store water. The same

logic can be applied to convert the mean annual precipitation of 1147 mm, as well as the 2.7 mm of extra precipitation the region has been receiving every year to account for 2868 and 6.8 mm of additional below-ground groundwater table height, respectively. Therefore, precipitation could account for around 2873 mm of groundwater table height additions over the course of the year, of which 1147 to 1405 mm are lost by evapotranspiration (Table 1.1). Net precipitation alone could therefore account for as much as 1726 mm of the groundwater table height over the course of a year. However, these additions to groundwater table are also likely to be balanced by groundwater flow exports to coastal water bodies, in addition to human draw-down of aquifers to supply potable water needs, which are discussed next.

Groundwater flow and exports to coastal waterbodies. Coastal aquifers consist of complicated combinations of confined and unconfined aquifers, and submarine groundwater discharge is ubiquitous in coastal zones. Rates of groundwater discharge can vary significantly, and be affected by seasons, coastal sediments, tidal flow influence and other oceanic factors (Burnett et al. 2006). Based on radionuclide tracer analysis of groundwater flux, the influx of groundwater to the salt ponds in southern RI ranges from 10 to 30 mm per day (equivalent to a total of $2-5 \times 10^7$ L per day of water entering the ponds from the entire watershed; Scott and Moran, 2001). This volume is equivalent to a linear value of 365 to 1095 mm per year of groundwater discharging into the ponds and/or Atlantic Ocean from coastal groundwater aquifers. By focusing on the two ponds that are in close proximity to many of the properties in this study (Ninigret

and Green Hill Ponds; Figure 1.1), the annual groundwater discharge values can be further constrained to roughly 500 to 600 mm per year (Scott and Moran 2001). When pore space is taken into account, these values for groundwater export convert to a below-ground annual drop in groundwater tables of 1250 to 1500 mm. It is reasonable to expect these values to vary annually with precipitation patterns, and that these exports exist in a state of dynamic equilibrium with some of the groundwater imports discussed above. Furthermore, to the authors' knowledge there are no estimates of how much groundwater is discharged directly to the Atlantic Ocean in this region, in addition to the discharge to the salt ponds – therefore, it is likely that some of the groundwater exports remain unaccounted for in this analysis.

Summary of factors contributing to rising groundwater tables

The preliminary analysis of the data presented here indicates that the rise in coastal groundwater tables is influenced by several different import factors of different magnitudes: potable water imports, recharge from OWTS, increases in precipitation, and sea level rise. Two of these – precipitation and sea level rise – are out of human control; the only factors that can be changed are the out-of-basin water imports and recharge to local aquifers via OWTS drainfield infiltration. Although the magnitude of the contribution of precipitation and SLR to rising groundwater tables is small, it is continuous and unlikely to abate in the near future. In fact, recent studies indicate that sea level rise is likely to accelerate (Nerem et al. 2018) and that the magnitude of SLR along the Atlantic coast of the US may be 3 to 4 times higher than the global average (Sallenger et al. 2012).

Thus, changing the water infrastructure along the southern RI coast may be the only means of managing human-derived rises in groundwater tables.

Implications

Using a single measurement from a test pipe to verify groundwater tables, as per the RIDEM regulations (RIDEM 2002), may not be the most accurate method of determining the depth to the seasonally-high groundwater table. For example, Morgan and Stolt (2004) found that weekly monitoring of groundwater table wells may underestimate the actual highest level a third of the time, by as much as 55 cm. The alternative – relying on the position of redoximorphic features – may significantly underestimate the actual highest seasonal groundwater table elevation, especially in coarse soils or soils with high sand proportions (Morgan and Stolt 2006). In these types of soils (which are very similar to the soils in this research area) the soil below the drainfield of an OWTS designed based on a separation distance to redoximorphic features may be saturated up to 21% of the time (Morgan and Stolt 2006). Therefore, both types of depth-to-groundwater table measurements reported to RIDEM are likely underestimate how shallow the groundwater tables actually are in coastal areas with sandy soils, such that coastal OWTS may be under greater threat than represented by these data.

Additionally, whether measurements were made during “wet” or “dry” years (Figure 1.7(b)) can also have serious implications for OWTS performance: if actual groundwater tables are higher than assumed based on measurements in dry years, there will be a reduced separation distance between the drainfield base

and the groundwater table. This could lead to substantially reduced nutrient and pathogen attenuation before the wastewater enters groundwater aquifers. This is compounded by the rise in groundwater tables, since many systems designed and installed in drier years in the past several decades may now have inadequate separation distance and be contributing significantly to ground and coastal water pollution in southern RI.

Over the next century, both sea level rise and the frequency and magnitude of intense precipitation events are expected to increase (IPCC 2013), exacerbating the current trend of rising groundwater tables along the coast. Population growth in southern RI is also likely to increase in the near future, resulting in more OWTS being installed in near shore areas. Returning to the use of individual groundwater wells might help reduce the impact of the water import problem. However, poor groundwater quality from nutrient pollution and saltwater intrusion initially led to widespread cistern use in the region, and will likely make a return to groundwater wells untenable, and cistern use will likely increase. In addition, current housing density, which may increase with further development pressure, may make it impossible to site groundwater wells destined for potable water use with adequate setback distances to septic systems.

If the communities along the coast continue to develop their municipal water distribution systems – in response to poor groundwater quality and/or salt water intrusion – the effects of water imports into the coastal zone will be exacerbated. These concerns have been raised by Bjerklie et al. (2012) in New Haven, CT: increasing recharge via improved stormwater management aimed at

increasing infiltration is expected to result in higher groundwater tables. In addition to resulting in an increasingly shorter vertical separation distance, another consequence of changing the potable water source may be that, because potable water is imported, poorly functioning OWTS could be considered exclusively as an environmental threat, rather than both a human health and environmental problem. The resulting lack of urgency in addressing OWTS failure from shrinking separation distance could result in resistance among coastal property owners to investing in more resilient and efficient wastewater treatment options, and eventually cause serious water quality problems in coastal ecosystems. However, once coastal waters become degraded, the desirability of this region as a residential and tourist destination will diminish, potentially resulting in sinking property values. The proactive management of this issue is critical to maintaining the public and environmental health, and scenic quality of the region, and calls for OWTS designs and installations in at-risk areas along the coast that are more resilient to rising groundwater tables.

Extrapolating groundwater table rise rates (Table S7) over the next century suggests that a system meeting the 152 cm (5 ft) minimum depth to groundwater measurement in 2018 might, depending on assumptions, have the groundwater table at the ground surface by 2030 (extreme 118 mm/year rise scenario) or by 2050 (moderate 46 mm/year rise scenario; Figure 1.8). In both these cases, the entire OWTS will have been inundated and cease to function. If the current rate of 14 mm per year rise continues, that same 152 cm groundwater table might be as shallow as 135 cm in 2030, 109 cm in 2050 and 44 cm in 2100 (Figure 1.8).

Figure 1.8. Anticipated changes in groundwater table depths, based on historic groundwater table behavior and different estimated values for rates of groundwater inputs based on factors described in Table S7. The grey line represents no change in groundwater table height (152 cm).

The current proposed solutions to the coastal hazards threatening properties along the shore are completely inadequate in dealing with OWTS function and contribution to coastal nutrient and pathogen pollution. Raising buildings to a certain height above a forecasted flood level might protect homes from flood events; however, these plans do not include measures to prevent inundation or damage to critical below-ground OWTS infrastructure which may be seriously damaged or destroyed by these same flood events (Allen et al. 2018;

Fisher et al. 2016). There appear to be two ways forward to prevent negative impacts to coastal surface and groundwater resources from rising groundwater tables: (1) Design OWTS with drainfields that account for rising groundwater tables over several decades, and can withstand and function during and after large storm impacts along the coast, and/or (2) Change the way wastewater is managed in coastal communities, which would require significant investment in new infrastructure. Research from the Virginia Coastal Policy Center has highlighted how coastal communities in Virginia are currently grappling with coastal hazards today (Moore and Acker 2018). Decisions on whether to implement sometimes costly resilience strategies, including “hard” and “soft” infrastructure, are fraught with complications, as are decisions in communities regarding relocation farther inland (Moore and Acker 2018). The Isle de Jean Charles in Louisiana is another example of a coastal community facing tough choices in light of increasing threats posed by coastal hazards (Usborne 2018). However, neither of these locations adequately addressed how to make OWTS infrastructure more sustainable and resilient. If coastal communities across the U.S. do not begin to seriously consider OWTS as part of their coastal resilience plans, property values in these regions will likely diminish as pollution reduces the attractiveness and recreational potential of these coastal areas.

Conclusions

The results of this preliminary analysis indicate that groundwater tables in coastal aquifers are rising in southern RI. Two of the factors that appear to contribute to rising groundwater tables – sea level rise and increases in annual

precipitation – are essentially outside of human control, and will continue to shorten the vertical separation distance between the drainfield infiltrative surface and the groundwater table for the foreseeable future. In contrast, imports of out-of-basin potable water that is recharged to coastal aquifers via OWTS along the coast represent a factor that is squarely under human control. As such, changing potable water delivery and treatment infrastructure may provide opportunities to mitigate rising groundwater tables.

At-risk coastal regions face continuing threats from growing populations, contaminants entering the groundwater from OWTS, and climate-related coastal changes and hazards (e.g. sea level rise, storms, increased intensity of precipitation). Coastal community policy and decision makers need to develop new ways to ensure that current and future OWTS are climate-change resilient in order to protect these areas. To achieve this, a better understanding of the behavior of coastal groundwater tables is critical. A network of long-term groundwater monitoring wells would yield valuable information in this regard and could lead to more accurate and detailed modelling and understanding of groundwater table dynamics along the coast. The preliminary results presented here are a starting point to help improve the understanding of coastal groundwater table behavior along the southern RI barrier beach complex. This will benefit local and regional communities, as well as 10% of the world's coastlines that are also classified as barrier beach complexes (Stutz and Pilkey 2011), especially if they rely on OWTS to treat domestic wastewater.

Data Availability Statement

Some or all data, models, or code generated or used during the study are available from the corresponding author by request:

- Water table depth data by parcel
- R code for statistical analyses and figure generation

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Supplemental Data

Data from the long-term USGS groundwater monitoring well (Site no. 412214071394001 RI-CHW 18) indicate no statistically significant change in groundwater levels from 1960 to present (Figure S1).

Figure S1. Monthly data from long-term USGS groundwater monitoring well (Site no. 412214071394001 RI-CHW 18) in Ninigret Park, Charlestown, RI (location shown in Fig. 1.1). The blue line is the linear regression, surrounded by the 95% confidence interval (grey).

Figure S2. Map of the southern portion of Rhode Island showing all the parcels (from West to East) in Westerly, Charlestown and South Kingstown located more than 61 m (200 ft) from a sewer line. Orange lines indicate location and extent of sewer areas, primarily in Westerly, South Kingstown and neighboring Narragansett (the eastern most portion of the map). Parcels in pink are within 1km of the coastline (which includes salt ponds permanently connected to the Atlantic Ocean via breachways or man-made channels, as well as tidally-influenced rivers (e.g. Narrow River in the northeastern extent of the map)).

Table S1. Number of parcels in southern Rhode Island (here referring to the towns of South Kingstown, Charlestown and Westerly). Built structures refer to buildings identified by the E 911 site layer on RIGIS.

Town	All southern RI Parcels >61 m from sewer		All southern RI Parcels >61m from sewer with built structures	
	No. Parcels	Total area (km ²)	No. Parcels	Total area (km ²)
South Kingstown	8,345	117.7	6,852	80.6
Charlestown	6,040	90.7	4,830	52.8
Westerly	7,437	60.9	6,108	33.2
TOTAL	21,822	269.4	17,790	166.6

Table S2. Parcels in southern RI within a 1-km radius of the coastline. Salt ponds permanently connected to the Atlantic Ocean via breachways or man-made channels are counted as part of the coastline. Built structures refer to buildings identified by the E 911

Town	Coastal RI parcels >61 m from sewer		Coastal RI parcels >61 m from sewer with built structures	
	No. Parcels	Total area (km ²)	No. Parcels	Total area (km ²)
South Kingstown	3,873	23.5	2,913	14.9
Charlestown	3,239	17.4	2,333	10.4
Westerly	4,374	20.6	3,558	11.7
TOTAL	11,486	61.5	8,804	37.0

Development and Population Growth

The population along the southern RI coast has more than doubled in the past half century (Table S3), undoubtedly because of the region's popularity as a picturesque summer vacation destination. The nearby university (the University of Rhode Island) also makes it lucrative for property owners rent out homes previously occupied only during the summer months to college students during the academic year. The town of Charlestown shows the greatest population increase over this period of time (more than tripling in 50 years), followed by South Kingstown (nearly tripling) and Westerly (not quite doubling). Today, South Kingstown is the most populous town, followed by Westerly and Charlestown.

Table S3. Historic town-wide population in southern RI from 1960 to 2010. Source: US Census Bureau, as compiled by the RI Division of Statewide Planning.

	1960	1970	1980	1990	2000	2010
Charlestown	1,966	2,863	4,800	6,478	7,859	7,827
South Kingstown	11,942	16,913	20,414	24,631	27,921	30,639
Westerly	14,267	17,248	18,580	21,605	22,966	22,787
Total Population	28,175	37,024	43,794	52,714	58,746	61,253

Using more detailed datasets from the 1990, 2000 and 2010 digitized geo-referenced U.S. Census reports (RIGIS 1991, 2007, 2012), the authors were able to both confirm these town-wide estimates of population, and calculate population for those parcels within 1 km of the coast (Figure S2; Figure S3). Because Charlestown is more sparsely populated than Westerly, and some of the census blocks might overlap other less populous neighboring towns, it is possible that this area-based approach to calculating population led to the slight underestimate of the town-wide Charlestown data. However, the authors believe that this estimate is therefore conservative, and the good agreement with calculated and reported town-wide population numbers for South Kingstown and Westerly make them confident that these calculations for the populations within 1 km of the coast are reasonable (Figure S3).

Figure S3. Population estimates for southern RI, by town. Points connected by lines represent data for town-wide population estimates from the RI Division of Statewide planning. Lines spanning from 1990 to 2010 without points represent data calculated using geo-referenced US Census reports from 1990, 2000 and 2010 (RIGIS 1991, 2007, 2012). Note that the calculated town-wide population estimates for South Kingstown and Westerly agree well with the other dataset, while the calculated values for the town-wide Charlestown data appear to be a slight underestimate. Also depicted are the population estimates for persons living within 1 km of the coast; these estimates seem to be proportional overall population estimates for the entire town.

Using the geo-referenced U.S. Census data, the authors were able to ascertain the number of total houses, occupied houses and the total population in both southern RI and within 1 km of the coast (Table S4). Again, even along the coast, there is clear evidence of growing population and development over the past 30 years.

Table S4. Calculated number of houses (total and occupied) and total population in southern RI (town-wide and within 1 km of the coast), based on geo-referenced US Census data from 1990, 2000 and 2010. Note that the calculated town-wide population numbers agree well with the data from Table S3.

Town	1990			2000			2010		
	Total Houses	Total Occupied Houses	Total Population	Total Houses	Total Occupied Houses	Total Population	Total Houses	Total Occupied Houses	Total Population
Westerly	10,304	8,347	21,312	11,231	9,367	22,889	12,288	9,654	22,756
South Kingstown	9,552	7,377	24,498	11,288	9,263	27,907	13,205	10,308	30,619
Charlestown	3,273	2,069	5,506	4,460	2,995	7,433	4,627	2,971	71,87
Westerly - Coast	2,514	1,210	2,896	2,619	1,349	3,062	2,887	1,353	3,041
South Kingstown - Coast	2,758	1,289	3,215	3,143	1,810	4,341	3,444	1,592	3,728
Charlestown - Coast	1,231	471	1,054	1,880	792	1,693	2,034	759	1,552

When comparing development along the coast (represented by the number of houses built within 1km of the coast) to the historic water table data, it is clear that in Westerly and Charlestown, the number of built houses corresponds to a slight shallowing of groundwater tables (decrease in depth-to-groundwater table), whereas in South Kingstown, there is a very slight deepening of groundwater tables in response to the number of houses built in the area. The same trends are evident, when water table depth is plotted against the number of occupied homes along the southern RI coast, based on US Census data, though the rates of change are twice those of the built houses (Table S6). Presumably the occupied houses represent homes that are occupied year-round, and do not include homes that are only used seasonally or as second homes. Seasonal homes might contribute

significantly to groundwater dynamics in the summer months, though this effect may not be visible from January to April, when wet-season depth-to-groundwater table measurements are made.

Seasonal home occupation also varies rather drastically by Town (Table S5), ranging from 56 to 84% of houses being occupied year-round. Westerly appears to have the largest proportion of seasonally occupied homes, while permanent South Kingstown residents appear to vastly outnumber the seasonal homes. When occupancy rates are considered for individual homes, Charlestown seems to average about two persons per occupied house, while Westerly occupancy rates suggest that nearly each occupied home hosts a single person.

Table S5. Comparing the number of occupied houses by town. Values were calculated from 2010 U.S. Census Data and analyzed spatially in ArcGIS (RIGIS 2012). Values in bold are statistically significant to $p < 0.001$, underlined vales are statistically significant to $p < 0.01$

	Total vs Occupied Houses		Occupied Houses vs Total Population	
	<i>Ratio</i>	<i>R²</i>	<i>Ratio</i>	<i>R²</i>
Charlestown	0.61	0.91	<u>0.51</u>	0.99
South Kingstown	0.84	0.89	0.84	0.89
Westerly	0.56	0.95	0.91	0.98

Table S6. Effect of different US Census development indices of the coastal portion (<1 km from the coastline) of southern RI on depth-to-water table. Values in bold are statistically significant: '***' = $p < 0.001$; '**' = $p < 0.01$; '*' = $p < 0.05$; '.' = $p < 0.1$.

Town	Slope: WT change per Built House (cm / house)	R ² & Signific- ance	Slope: WT change per Occupied House (cm / house)	R ² & Signific- ance	Slope: WT change per Person (cm / person)	R ² & Signif ic- ance
Westerly	-0.032	0.03*	-0.068	0.05**	-0.064	0.00
South Kingstown	0.007	0.00	0.012	0.00	0.038	0.05 .
Charlestown	-0.030	0.02	-0.067	0.04*	0.034	0.01
Overall	-0.007	0.00	-0.012	0.00	0.010	0.01 *

Population along the southern RI coast seems to have a different relationship to the depth to groundwater table: Only westerly's increasing population appears to be causing groundwater tables to become slightly shallower, though this effect is not statistically significant (Table S6).

However, the rates of change for all of these indices of development (water table depth versus houses, occupied houses and population) are on the order of fractions of a millimeter, therefore it is unclear how strong an effect development along the coast really has on groundwater table dynamics.

Table S7. Millimeters of water entering groundwater aquifers from OWTS in southern RI each year, considering all the OWTS in the region and the population of the area (based on RIGIS E911 Site data, as well as US Census Data; RIGIS 1991, 2007, 2012, 2017). The SRI region was analyzed over different spatial scales, including town-wide, coastal (within 1 km of coastal water bodies) and near-shore (within 200 m of a coastal water body). Calculations were made assuming a per-system daily flow rate of 500L and per-capita daily flow rate of 200L (Amador, et al., 2018). Volumes were converted to length measurements by dividing the total volume for all parcels by the area occupied by all the parcels in question, as calculated in ArcMap

Year (Data Source)	2017 (E911 sites)	1990 (US Census)	2000 (US Census)	2010 (US Census)			
Source of groundwater additions	All OWTS in area (mm / yr)	All OWTS in area (mm / yr)	All populat ion in area (mm / yr)	All OWTS in area (mm / yr)	All populat ion in area (mm / yr)	All OWTS in area (mm / yr)	All populati on in area (mm / yr)
Charlestown	16	6	4	9	6	9	6
Charlestown - Coast	39	12	4	19	7	20	7
Charlestown - Shore	55	15	5	24	8	21	5
South Kingstown	15	10	11	12	13	14	14
S. Kingstown - Coast	34	17	8	21	12	21	10
S. Kingstown - Shore	43	20	8	31	15	30	10
Westerly	32	22	20	25	22	26	21
Westerly - Coast	53	22	11	26	13	24	11
Westerly - Shore	56	23	9	28	11	18	5
Total Towns	18	12	11	14	13	16	13
Total (1km) Coast	41	17	8	22	11	22	9
Total (200m) Shore	51	21	8	26	11	20	6

Manuscript 2 :
**Temporal variability in the vertical separation distance of septic
system drainfields along the southern Rhode Island coast**

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Abstract

Many coastal communities rely on individual onsite wastewater treatment (i.e. septic) systems to treat and disperse wastewater. Proper wastewater treatment in these systems depends on sufficient volume of unsaturated soil below the drainfield's infiltrative surface. This is governed by the vertical separation distance – the distance between the groundwater table and the drainfield infiltrative surface – which is specified in (regulatory jurisdictions' onsite wastewater system) regulations. Groundwater tables along the southern New England coast are rising due to sea-level rise, as well as changes in precipitation and water use patterns, which may compromise the functioning of existing septic systems. We used long-term shallow groundwater monitoring wells and ground-penetrating radar surveys of 10 drainfields in the southern Rhode Island coastal zone to determine whether septic system drainfields have adequate separation distance from the water table. Our results indicate that only 20% of tested systems are not impaired by elevated groundwater tables, while 40% of systems experience compromised separation distance at least 50% of the time. Surprisingly, 30% of systems in this study do not meet separation distance requirements at any time of the year. Neither age of system nor a system's geographical relationship to a tidal water body were correlated with compromised separation distance. The observed compromised separation distances may be a result of inaccurate methods, specified by the regulations, to determine the height of the seasonal high water table. Our preliminary results suggest that enacting

changes in the regulatory permitting process for coastal zone systems may help protect coastal drinking and surface water resources.

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Introduction

Coastal onsite wastewater treatment systems (OWTS; i.e. septic systems) are threatened by climate change. Increased frequency and intensity of coastal storms, coupled with rising groundwater tables, are already taking their toll on coastal OWTS' ability to renovate wastewater, a trend that will become exacerbated in the coming decades (Cox et al. 2019, 2020). Extensive portions of coastal communities in southern Rhode Island (RI, USA) rely on individual OWTS to treat domestic wastewater. A previous analysis of data from the RI Dept. of Environmental Management (RIDEM) indicates that near-shore groundwater

tables have been rising since 1964 at an average rate of 1.4 cm per year (Cox et al. 2019). However, the rate of groundwater rise depends on both a parcel's landscape position relative to tidal water bodies and its longitudinal position along the coast. Analysis of multi-year water table data showed that a majority of properties (65 %) exhibit rising groundwater tables, while some properties had little change in water tables (19%), and others showed deepening groundwater tables (15%; Cox et al. 2019). As groundwater tables rise, the vertical separation distance between the bottom of the drainfield's infiltrative surface and the seasonal high water table shrinks, seriously compromising the system's ability to attenuate nutrients and pathogens from the percolating wastewater before it intercepts the groundwater (Cooper et al. 2016; Humphrey et al. 2011, 2017). In addition to long-term changes, shallow groundwater tables in coastal aquifers also rise and fall in the short term, in response to normal tides and coastal events (e.g. storms, extreme tidal fluctuations and large precipitation events; Cartwright et al. 2004; Healy 2017; Manda et al. 2015; Morgan and Stolt 2006; Nielsen 1999). These short-term increases in water table elevation result in separation distance becoming compromised temporarily, and present a recurring threat as climate change-related storm and precipitation events become more frequent and extreme (Horton et al. 2014; Karegar et al. 2017; Lin et al. 2016; Marcos et al. 2019).

Information about vertical separation distance temporal dynamics in coastal lots is important for stakeholders to plan more holistically for coastal resilience, since OWTS are an important and likely long-term or permanent part

of the water infrastructure in many coastal areas, both in the US and globally (Amador and Loomis 2018). Understanding how groundwater table dynamics affect OWTS, and what the future impacts are likely to be, is critical to designing resilient systems and policies so that OWTS continue to serve crucial water treatment functions in nutrient and pathogen sensitive areas.

To understand the extent to which individual coastal OWTS are being impacted by current groundwater table dynamics, we determined the vertical separation distance in 10 systems along the southern RI coast from April 2017 to May 2019. The systems varied in landscape position (geographic relationship to tidal water bodies) and longitudinal position along the coast, as well as in system design and age (Table 2.1). We used long-term groundwater monitoring wells and ground-penetrating radar (GPR) surveys of the drainfield at each site and to estimate separation distance. GPR works by transmitting electromagnetic pulses at microwave frequency from an antenna into the ground, while a receiver records wave reflections off the sediments or soil and objects, such as septic system components, as a function of two-way wave travel time (Annan 2009; Humphrey et al. 2013). By analyzing wave reflection and attenuation patterns, subsurface stratigraphic and water table information can be visualized in real time, without the need for invasive, labor-intensive methods like hand-augering and excavation. In other studies GPR has been used effectively in sandy soils, like those characteristic of the southern RI coastal region (Masterson et al. 2007), to visualize the sub-surface conditions in real time (Doolittle et al. 2006; Manda et al.

2015; Walter et al. 2016), and to visualize structures of OWTS and water tables below drainfields in North Carolina (Humphrey et al., 2013). However, previous research indicates that the precision of GPR-based predicted water table elevations has an error margin of ~6 to 60 cm (Bentley and Trenholm 2002; Doolittle et al. 2006), which may be problematic for sites where the separation distance is less than 1.5 m. Field-derived values for today's separation distance in each system were compared to separation distance and water table depths indicated on the OWTS permit design submitted to RIDEM to assess whether separation distance has changed in the intervening time period. We hypothesized that older systems would show greater losses in separation distance than younger systems, based on the findings of Cox et al. (2019), as groundwater tables would have had longer to approach the elevation of the drainfield infiltrative surface, decreasing separation distance and the drainfields' ability to renovate wastewater. We also hypothesized that the shallow nature of freshwater aquifers on the sandy Rhode Island barrier island systems would result in measurable, short-term increases in groundwater table elevation in response to coastal events (storms, extreme high tides, large precipitation events; Manda et al. 2015). These events and the associated rises in groundwater would therefore result in temporary compromising of separation distance, especially during the wet season (January to April; RIDEM 2018a), when the seasonal high water table is at its highest elevation for the year.

Table 2.1. Summary of site characteristics for OWTS used in this study. See Figure 2.1 for the location of each well. Soil series for each parcel are based on online soil maps (Soil Survey Staff 2018), and verified during well installation. ^aRIDEM issues a “Certificate of Conformation” once the system has been installed and inspected by RIDEM to ascertain that the system conforms to the submitted plan’s design, and enables owners to begin using the system. ^bSHWT stands for “seasonal high water table”. ^cCurrent OWTS regulations require a separation distance of 122 cm from the infiltrative surface to the seasonal high water table (RIDEM 2018a)

Site ID	Town	Landscape location	Soil series	Year OWTS conformed ^a by RIDEM	Year of SHWT ^b measurement
CH01	Charlestown	Barrier Beach	Hooksan	1977	1976
CH02	Charlestown	Back Barrier	Fortress	1988	1982
CH03	Charlestown	Back Barrier	Hooksan	1977	1975
CH04	Charlestown	Salt Pond	Merrimac	1983	1982
CH05	Charlestown	Near Pond	Merrimac	1969 / 1994	1969
CH06	Charlestown	Salt Pond	Merrimac	1982	1981
CH07	Charlestown	Barrier Beach	Bridgehampton	1994	1991
CH08	Charlestown	Back Barrier	Fortress	2012	1993
CH09	Charlestown	Barrier Beach	Hooksan	1988	1982
WE01	Westerly	Back Barrier	Succotash	1984	1982
WE02	Westerly	Back Barrier	Succotash	2012	2011

Site ID	Plan ^c vertical separation distance (m)	Date groundwater monitoring well installed	Distance from well to center of drainfield (m)	Depth of well below ground surface (m)	Date GPR Survey
CH01	1.52	4/23/2017	8.12	2.97	4/11/2017
CH02	1.22	6/30/2017	8.44	2.77	1/3/2018
CH03	1.22	9/30/2017	4.48	2.51	1/3/2018
CH04	0.91	9/30/2017	18.34	1.76	1/3/2018
CH05	0.61	9/30/2017	27.00	2.37	1/29/2018
CH06	1.07	9/30/2017	14.00	1.85	1/29/2018
CH07	0.91	10/30/2017	16.40	1.89	1/3/2018
CH08	0.91	10/30/2017	43.90	2.44	12/6/2016
CH09	1.22	10/30/2017	30.98	2.78	1/3/2018
WE01	0.91	4/13/2018	22.64	2.08	4/9/2019
WE02	1.22	10/3/2018	35.34	1.80	4/9/2019

Methods

Site Selection

The southern Rhode Island coast is characterized by granitic bedrock overlain with stratified glacial deposits with thicknesses ranging from 3 to 20m (Masterson et al. 2007). Large portions of the coast are comprised of outwash material from post-glacial tidal deltas and coastal lagoons (i.e. salt ponds), interspersed with dense till-dominated headlands. The southern RI coastal basin is broken up into many smaller watersheds ultimately discharging into the salt ponds and Atlantic Ocean (Masterson et al. 2007).

Study sites were selected based upon (1) proximity to coast (within 100m of a tidal water body; corresponding to an estimated 1,500 to 2,000 systems (Cox et al. 2019)), and (2) soil type: sandy soils including Hooksan, Succotash, Fortress and Beaches soil series were preferred for suitability of GPR investigation (Humphrey et al. 2013; Soil Survey Staff 2018), though some sites (CH04, CH06, CH07) mapped as such were discovered to be finer in nature (e.g. Merrimac and Bridgehampton soil series) during well installation (Table 2.1). Property ownership history was cross-referenced with OWTS permit applications submitted to RIDEM. We solicited voluntary participation in our study from homeowners whose permit application plans were sufficiently detailed to allow us to assess depth-to-water table and separation distance at the time of OWTS installation. Properties located on dense till or with extremely rocky soils were excluded from this analysis, as they precluded well installation.

The plans for each system, redacted to maintain participants' anonymity, are available by request from the authors. The oldest system in this study was installed in 1969. It had a 3rd trench added in 1994 without other modifications to the existing system (CH05). The most recently installed system (WE02) was an advanced-treatment nitrogen removal system installed in 2012. Drainfield design varied among systems, but included conventional pipe-on-stone trench design (sites CH02, CH04, WE01), bed-type drainfield design (sites CH01, CH03, CH05), concrete galleys (sites CH06, CH07, CH08), and site WE02 had a bottomless sandfilter design. Sites CH01, CH02, CH04, CH07, CH08, WE01 and WE02 had designs where the surface of the drainfield was elevated relative to the original contours to create sufficient vertical separation distance between the infiltrative surface and seasonal high water table as required by the regulations at the time.

Groundwater Monitoring Wells

Groundwater monitoring wells consisted of 5 cm (2 in) inner diameter schedule 40 PVC pipe, which was slotted on the lower 1.2 m (0.010 in slot width; Atlantic Screen & Manufacturing, Inc. Milton, DE). Wells were installed to a maximum depth of ~3 m using a power soil probe (5 HP mechanical earth drill with carbide tip; Little Beaver, Livingston, TX) as close to the drainfield as possible without interfering with homeowner use of the property (Figure 2.1). A 5-10 cm collar of bentonite clay was poured around the top of the well to prevent preferential flow from entering the well hole during precipitation events. A 2-m capacitance water logger (Odyssey Dataflow Systems Ltd, Christchurch, New Zealand) was suspended in each well, set to collect water table height every 10

minutes. Loggers were calibrated using the bucket method, following the manufacturer's instructions. Wells were in place for a minimum of one wet season (January – April, according to RIDEM permitting procedures; RIDEM 2018a). The oldest well was in place for 25 months (CH01), while the most recent well was in place ~7 months (WE02; Table 2.1). One site had two wells on the same property: well CH02 was located in the back barrier position approximately 8.4 m from the center of the drainfield, while well CH09 was placed in the barrier beach dune approximately 31.0 m southwest of the same drainfield. All other sites were instrumented with only one groundwater monitoring well (Table 2.1).

Data from the loggers were corrected using the calibration curve described above. Every two to three months, the water table depth in the well was manually verified using a water level meter (Model 102M; Solinst Canada Ltd., Georgetown, Ontario). Water tables measured in this fashion were 5-10 cm lower than the level indicated by the logger (data not shown). The difference in measurements can be attributed to the water table falling after the brass cylinder (approx. volume of 17.9 cm³) at the end of the logger was removed from the well during the verification measurement, corresponding in a ~9cm drop in water table height in the well. Therefore, the discrepancy between the actual water table in the well measured manually and the value recorded by the data logger is ~5cm.

Figure 2.1. Diagram of a typical site layout: Plan view of site on left, cross section of the drainfield and well on right. Monitoring wells with groundwater monitoring loggers were installed in proximity to the drainfield, often at the edge of the property to minimize disturbance to homeowners and protect the well.

At each site, the relative elevations of the well, drainfield surface and location of the temporary well or soil pit from OWTS permit application were surveyed using an automatic self-leveling rotary laser level (Lasermark LMH series and LD-400 detector; CST/Berger (Robert Bosch Tool Corporation, Mount Prospect, IL)) and stadia rod. When possible, these elevations were tied into the benchmark indicated on the plans for the OWTS (Table 2.1). Relative elevation data, coupled with the plan elevation specifications, enabled us to calculate the well's vertical relationship to the infiltrative surface of the drainfield. In most cases, the plan's benchmark (an object with a known or assumed elevation that all plan elevation are relative to) was still accessible, which allowed us to accurately compare the groundwater monitoring well's exact elevation to the drainfield

components. For site WE01 the benchmark had been removed, however we were able to use the nearby benchmark for WE02 and apply a correction of 0.096 m for WE02's North American Vertical Datum of 1988 (NAVD88) elevation to correspond to 1984's mean sea level (MSL) elevation for nearby Watch Hill Point (NOAA et al. 2008). The plan for CH05 did not specify a benchmark, so elevation measurements were made relative to the center of the drainfield's ground surface elevation. The benchmark for CH08 was destroyed when the building was renovated, but the southwest corner of the distribution box was exposed to grade, and so elevations were tied into this component's calculated elevation from the plan. We verified these assumed elevations by examining the GPR transects to the plans (see below), and determined that the plan's relative elevations appeared to correspond with the components in the ground.

Ground-Penetrating Radar

We performed GPR surveys of the drainfields by laying out a rectangle on the ground surface calculated to enclose the drainfield, based on the OWTS permit plan information. We employed an SIR (Subsurface Interface Radar) 4000 unit, a ground-penetrating radar data acquisition system manufactured by Geophysical Survey Systems, Inc. (GSSI; Nashua, NH). We used a 400 MHz antenna with a calibrated survey wheel to collect the radar data in distance mode. GPR settings included a scan rate of 50 scans/m, a sampling rate of 512 samples/scan and a sampling window of 50 ns.

Drainfield surveys were run in 0.5 or 1-m parallel transects perpendicular to the direction of the drainfield trenches or components that extended at least 1

m to outside the drainfield edge, enabling us to characterize the sub-surface stratigraphy in 3-D. At each site, we used the GPR antenna to track the water table from the well (where we measured the depth to the water table manually at the time of the GPR survey) to the drainfield, to ensure the manually specified dielectric constant in the SIR unit produced a radar profile with accurate below-ground distances for each site (Doolittle et al. 2006; Manda et al. 2015).

Drainfield GPR surveys were performed in the winter or early spring to coincide with elevated seasonal high water tables and dormant backyard vegetation, and to minimize inconvenience to homeowners for seasonally-used vacation properties.

RADAN for Windows (version 7; GSSI, Nashua, NH) was used to process the GPR transect records. Processing included setting the initial pulse to time zero, color table and transformation selection. Additionally, drainfield components (laterals, infiltrative surface) and the water table were annotated using RADAN's target pick toolkit, based on refraction patterns in the GPR cross sections and comparison to each site's OWTS plan specifications.

Precipitation and Storm Events

Precipitation is measured daily in Kingston, RI (station KGNR1; Iowa Environmental Mesonet 2019a) and Block Island, RI (station BIDR1; Iowa Environmental Mesonet 2019b), which bracket the southern RI coast from North to South (Figure 2.2). The average annual precipitation recorded at the Kingston station from 2000 to 2018 amounts to 1373 mm (range: 1101 in 2016 to 1666 mm in 2018; (NOAA 2019)). From 2000 to 2018, the average precipitation during

the wet season (Jan – April) amounts to 472 mm (range: 346-622 mm), representing about 34% (range: 22-37%) of the annual precipitation (NOAA 2019). Our study occurred between January 2017 and May 2019, which appears to have been slightly wetter than usual, with annual precipitation amounts of 1391 and 1666 mm for 2017 and 2018, respectively, and wet season precipitation accumulations of 544, 622 and 584 mm for 2017, 2018, and 2019, respectively.

Figure 2.2. (A) Regional context of this study. The state of Rhode Island is colored dark green in all panels. Blue box in A shows extent of panel B. (B) Locations of the weather stations for precipitation data, as well as the location of the offshore wave buoy. Black box in B indicates the extent of C. (C) the geographic extents of the towns of Westerly (blue) and Charlestown (light green), as well as the locations of the study site (consisting of a groundwater monitoring

well adjacent to a drainfield). Study site CH02 is not labeled, but is close to study site CH09. Table 1 lists the details for each site.

Because coastal groundwater tables in sandy soils respond quickly to precipitation events (Morgan and Stolt 2006), we focused on daily precipitation in this study. From January 2017 to May 2019, daily precipitation ranged from 0 to 87.4 mm in Kingston RI, with a mean of 4.1 mm, while daily precipitation on Block Island ranged from 0 to 63.2 mm, with a mean of 3.7 mm. We focused on daily precipitation total accumulations greater than 24.5 mm (1 in; representing the top 4.5% of observations ($n = 79$; Online Resource 1), since these are likely to be reflected rapidly (within hours to less than a day) in the groundwater table in sandy coastal soils (Manda et al. 2015; Morgan and Stolt 2006), overshadowing peaks resulting from background “noise” of tidal influence.

We also considered the effects of storm events in the region. Since there is no single repository of information on small coastal storms affecting the southern RI coast, we used a variety of sources to compile this information. The National Oceanic and Atmospheric Administration (NOAA) compiles a storm events database, where extreme weather events of “sufficient intensity to cause loss of life, injuries, significant property damage, and/or disruption to commerce” and rare, unusual or significant meteorological events can be searched by county (NOAA National Centers for Environmental Information 2019). However, events of lesser intensity may also cause temporarily elevated groundwater tables, and thus this dataset may fail to describe other events throughout the year. To augment our data, we used the RI MyCoast app and website, which solicits

photos and documentation from the community following large coastal storm events, as well as during “King Tide” events – the highest seasonal tides that occur each year when a spring tide coincides with perigee (RI MyCoast et al. 2019). Events of this magnitude should affect coastal groundwater tables, but the data are limited to information uploaded by volunteers, and thus may not capture all such events. In addition, we used offshore wave buoy data (Scripps buoy No. 44097 southeast of Block Island; NOAA National Data Buoy Center 2019) to identify deep-ocean weather conditions that might cause larger than usual wave action along the coast, resulting in elevated coastal groundwater tables. We selected dates that had waves greater than 3.1 m ($n = 113$, Online Resource 1), corresponding to the top 5% of the overall observations for the time period, to filter out small events that may not have had noticeable near-shore effects.

Statistics & figure creation

We used R version 3.6.1 (R Core Team 2019) to perform linear regressions and descriptive statistics on water table, site, precipitation and storm data. To create figures, we used ArcMap v. 10.5 (Esri, www.esri.com), Radan 7 (GSSI), and the ggplot2 (version 3.2.1) and ggpubr (version 0.2.2) packages in R.

Results & Discussion

GPR investigations of drainfields

In all cases, the GPR survey confirmed that drainfields were installed according to the permit application plans. Figure 2.3 shows an example of how the GPR survey corresponds to the plan’s design specifications for site CH02. Panel C shows a GPR transect of the drainfield, whose approximate location is

indicated by the grey line (marked with T and T') in the plan in panel A. The transect shows the 10-cm (4-in) diameter PVC pipe (green circle) and the approximate location of the infiltrative surface (white bar). The water table is marked in blue. At each site, drainfield components and the water table can be identified, especially when comparing transects to the OWTS plan from the permit application. GPR has also been successfully used by others to identify the components of drainfields, including tanks, distribution boxes and trenches (Humphrey et al. 2013).

Figure 2.3. An example (site CH02) of drainfield plans in (A) plan view, and (B) cross-section, as well as a cross section of the drainfield made by the GPR unit (C; approximate location of transect noted by grey line and "T-T'" in A), and an annotated 3-D view of the integrated GPR transects (D). The green circle in C represents the 10 cm (4 in) diameter PVC pipe, the white line below the green circle represents the infiltrative surface and the blue line represents the

water table, corresponding to the same color lines and zigzags in D. The grey rectangle shows the approximate location of the transect in C.

However, due to the nature of the GPR-generated electromagnetic waves' distortion as they travel through different layers with varying dielectric constants (Doolittle et al. 2006), it is not possible to use GPR transects to definitively calculate the exact separation distance below the infiltrative surface: As the waves travel through the gravel trench, or concrete drainfield structures, both of which contain large air-filled spaces, the propagation time changes because of the slightly different dielectric properties (permittivity and conductivity) of the material, and thus the travel time appears to become shortened, since the Radan GPR processing software assumes the entire profile has the same dielectric constant. These phenomena cause a distortion in the apparent "distance" (a calculation based on the travel time and dielectric properties) directly below the trench. This is clearly visible in Figure 2.3C, where the water table appears mounded directly below the infiltrative surface. This apparent mounding is an artifact, as there were no wastewater inputs to the system at the time of the GPR survey in January of 2018 (Table 2.1), because this is a seasonally-occupied home and used only from April-October. Additionally, salts from the long-term application of wastewater to the soil below the trench can cause signal attenuation, occasionally making it impossible to identify the groundwater table directly below the drainfield structure, a phenomenon also described by Humphrey et al. (2013). Previous work in coarse-textured soils indicates that the discrepancy between the actual water table and the apparent water table based

on GPR transects can range up to 69 cm, though average difference is usually 16 to 20 cm (Bentley and Trenholm 2002; Doolittle et al. 2006). Though these values are useful in field investigations of water table depths, the resolution of GPR transects is not fine enough to make definitive assessments of whether a system's separation distance is adequate or not.

However, GPR surveys are still a valuable tool in drainfields, especially when plans are very old or are missing information: In our study, the GPR surveys were particularly useful in sites where the plan's original benchmark was no longer available, indicating that this may be a valuable non-invasive tool for researchers, regulators or municipalities to locate systems and assess below-ground drainfield conditions in real time (Humphrey et al. 2013). Employing a tile probe to validate drainfield component elevations may also be helpful at the time of the survey, though this was not possible in most cases in our study, because the ground was frozen at the time of the survey and/or had large rock fragments in the overburden material (Table 2.1).

Groundwater table logger data and separation distance

Figure 2.4 shows boxplots of the distribution of water table measurements based on logger data, as well as the mean water table for each site. The range of groundwater table depths observed at each site varies considerably, with CH01, CH03 and CH09 displaying the greatest range in depths (162.8, 177.5, and 183.3cm, respectively), and WE01, WE02, and CH08 having the narrowest range (54.3, 52.7 and 55.6cm, respectively). Sites CH01, CH03 and CH09 are located on Hooksan soils, which likely respond to changes in groundwater tables quickly.

Sites WE01, WE02 and CH08 represent the smallest datasets (Table 2.2).

Additionally, sites WE01 and WE02 are located on the southern shore of a smaller waterbody – with a different tidal range – than the majority of the remaining sites farther to the east (Figure 2.2), which may explain the differences in observed water table ranges. Site CH08 is located in the back barrier area in close proximity to a small brackish pond without a consistent connection to the ocean, which likely affects the local hydrologic dynamics.

Figure 2.4. Boxplots depicting the range of depths to the groundwater table for each study site. Boxes represent the interquartile range, with the included horizontal lines representing the median value. Whiskers extend 1.5 times the interquartile range beyond the upper or lower quartile. Colors of boxes correspond to the landscape position of the site. Black “x” represents the mean value for the water table depth. Note: the well at CH07 was installed during a coastal storm – when the water table was unusually high, and thus remained dry for parts of the year that did not experience elevated groundwater levels – explaining the apparently skewed distribution of the water table depths.

Table 2.2. Site-specific logger-based water table measurements and their relationship to the drainfield's infiltrative surface. The difference in the number of days the water table was logged and the number of days the well was in place is a result of logger malfunctions or low-battery events at sites CH01, CH03, CH08 and WE01. Plan-specified separation distances and dates of well installation can be found in Table 1. Since 1992, RIDEM regulations in these near-shore areas require a 122 cm (4 ft) separation distance for new systems being installed (RIDEM 1996). *One-year subset describes logger data in wells from May 10, 2018 to May 9, 2019.

Site ID	Entire dataset				One-year subset*	
	No. days well in place	No. days water table logged	Days with impaired plan-specified separation distance (%)	Days with impaired separation distance (current regulation; %)	Days with impaired plan-specified separation distance (%)	Days with impaired separation distance (current regulation; %)
CH01	673	538	32.6	3.0	14.8	0.3
CH02	622	622	1.0	1.0	0.0	0.0
CH03	558	474	29.8	29.8	28.1	28.1
CH04	587	587	14.7	44.4	13.5	43.5
CH05	586	586	2.7	55.8	1.6	54.5
CH06	586	586	98.6	100	100.0	100.0
CH07	500	500	9.1	28.1	7.7	25.5
CH08	384	212	100	100	100.0	100.0
CH09	500	500	2.1	2.1	0.0	0.0
WE01	316	201	100	100	100.0	100.0
WE02	77	77	24.2	24.2	24.7	24.7

Figure 2.5 shows the data from the groundwater table logger referenced to critical elevations of each site's drainfield. Overall, there is considerable variability in the extent of compromised separation distance below the drainfield (Figure 2.5). In 20% of drainfields (CH05 and CH02/CH09), the plan-specified separation distance is only briefly compromised during storm events during the course of the year (Table 2.2), and thus likely pose a low risk of contaminating groundwater and nearby coastal water resources. Site CH01 is unusual, in that the 1977 plan specified a 152 cm (5 ft) vertical separation distance, greater than the current

required separation distance. However, today the vertical separation distance based on the plan is compromised over 32.6% of the time. At sites CH03, CH04, CH07 and WE02 the plan-specified separation distance is compromised from ~9 to 30% of the time (Table 2.2). When assessed based on current regulations (required 122 cm (4 ft)) vertical separation distance is compromised several times over the course of the year (24-44% of time) at some sites (CH04, CH07, WE02), and for sites CH03, CH05, CH06, CH08 and WE01, the separation distance is compromised more than 50% of the time (Figure 2.5, Table 2.2). The water table at site CH04 reaches the ground surface regularly (Figure 2.5), however, the well is located in a depression on the site, and is ~1.5 m lower than the elevation of the drainfield surface. Alarming, water table elevation for sites CH06, CH08 and WE01 is high enough that vertical separation distance is compromised almost constantly throughout the year, based both on the plan-specified separation distance and current regulations. Even more concerning is the fact that the water table reaches the infiltrative surface 1.4 % of the time at site CH08. This means that during storms or extreme precipitation events, the groundwater near this system is mixing with untreated septic tank effluent, which could result in considerable nutrient and pathogen pollution of the groundwater and nearby coastal water bodies.

Figure 2.5. Groundwater table dynamics relative to the infiltrative surface and required separation distance. For each panel, the thick blue line represents the elevation at which the plan-specified separation distance (Table 2.1) should be maintained – if the water table, as measured by the logger, crosses above this line, separation distance is compromised. The green

line, if present, indicates the height of the water table required to meet the 122 cm (4 ft) vertical separation distance currently required for new systems in this area. The dashed grey line indicates the elevation of the infiltrative surface; for plots lacking this line, the line's elevation was out of range for the axis, but can be approximated by subtracting the separation distance specified in Table 1 from the location of the thick blue line. Black circles above peaks correspond to "King Tide" events, and black triangles correspond to coastal storms (RI MyCoast et al. 2019). Colored dots correspond to the magnitude of precipitation for events exceeding 24.5mm (1 in) Kingston or Block Island, RI (Iowa Environmental Mesonet, 2019a; b; Fig. 1, Online Resource 1). The well at CH07 was installed during a coastal storm – when the water table was unusually high, and thus remained dry for parts of the year that did not experience elevated groundwater levels – explaining the "bottoming out" of the line in the summer months of 2018.

Examination of the water table data from May 10, 2018 to May 9, 2019 – the period for which the majority of the wells have full complements of data (Figure 2.5) – shows that the trends for this period are consistent with those of the entire dataset. With the exception of one site, the percentage of days with compromised separation distance in the final year of the study is within a few percent of that calculated from the entire dataset (Table 2.2). At site CH01, the compromised separation distance based on plan specifications fell from 32.6 to 14.8 % of days in the one-year subset (Table 2.2), likely because this period had fewer winter and late-spring storms than previous years (Online Resource 1). No system experienced groundwater tables intersecting the infiltrative surface during this period either, likely for the same reason. Overall, our findings are in line with research conducted at a much larger scale in the eastern region of Bogue Bank, NC, which indicated that seasonal high water tables reach the infiltrative surface of drainfield trenches in 13-35% of the modeled area, and another 5-7% of the land area has less than 0.3 m of separation distance based on GPR investigations and geo-spatial modeling (Manda et al. 2015).

Figure 2.6 shows the response of the water tables in different landscape positions to various coastal events, including storms and/or large precipitation events during the 2018 wet season. Precipitation amounts of at least 24 mm result in a clear, short-term spike in the groundwater table, as do king tides and storm events (Figure 2.6). Groundwater table behavior analysis and modeling in Miami-Dade County (FL) has shown similar trends for near-shore water tables (Sukop et al. 2018). In this study, large precipitation events correspond to peaks in groundwater table heights in all instances, while large waves and other events do not always correspond to obvious changes in water tables (Figure 2.6). As climate change continues to affect the region, another important consideration is sea level rise and increasingly intense and frequent storm events, which will continue to affect coastal water tables (Sukop et al. 2018), likely leading to more frequent and possibly serious compromises in vertical separation in coastal drainfields (Mihaly 2017).

Our findings likely underestimate the extent of compromised separation distance because the assumption that the plan-specified elevation of various components is accurate is conservative: The components could not have been installed at higher elevations, because this would have resulted in hydraulic failure in these passive, gravity-fed systems (except site WE02, which is pressurized). Therefore, systems still operational today must have been installed with components at least at the elevation specified by the plan, but could have been installed deeper in the soil (lower elevation), resulting in less separation distance than specified by the plan. Our assessment of separation distance at each site

therefore represents the maximum possible distance, based on the plan's information and current groundwater dynamics measured at each site.

Figure 2.6. Groundwater table dynamics during the 2018 wet season, broken down by landscape location. Each panel shows the water tables for each site in a given landscape position, with different sized dots above the peaks corresponding to the amount of precipitation in 24 h, and the symbols corresponding to the type of coastal event.

Time since installation does not appear to be a determining factor in predicting which systems experience compromises in separation distance, based on linear regressions of the change in separation distance as well as the change in water table elevations over time (Online Resource 2), although in all cases our data indicate that the water table rises to compromise the plan's separation distance multiple times in a year (Figure 2.5). These findings suggest that the

rising groundwater regional trends described in Cox et al. 2019 are not necessarily representative of dynamics at the individual parcel scale. The findings in the study reported here indicate that water tables in half the sites have risen over time (range: 23.2 to 53.6 cm rise since time of measurement submitted to RIDEM), while the other sites experienced apparent drops in groundwater tables over time (range: 4.6 to 61.5 cm drop), based on the difference between the mean water table elevation at each site (averaged over entire period recorded by the logger) and the water table elevation submitted to RIDEM as part of the permitting process. These findings are in line with those of the Cox et al. (2019) study, who observed that 65% of parcels with repeated water table measurements showed rising groundwater tables, with the rest remaining the same or falling over time. When the changes in water table height in this study are corrected for time, rates of water table rise range from 0.6 to 3.8 cm/yr, and rates of falling water tables range from 0.1 to 1.7 cm/yr, which fall within reported ranges of coastal groundwater table changes along the southern RI shore (Cox et al. 2019), in New Haven, CT (Bjerklie et al. 2012), Miami-Dade County, FL (Sukop et al. 2018) and are less drastic than the lowering water tables reported by (McSwain et al. 2014) in New Hanover County, NC. As discussed in Cox et al. (2019), changes in groundwater tables are likely a result of changes in groundwater use and recharge patterns. For example, some areas of the coast (e.g. Westerly and the barrier beach complex in Charlestown) import drinking water, resulting in elevated groundwater tables as this water is recharged to local aquifers via drainfields.

Neither a system's landscape position nor designer of record – the design professional who submitted the plans to RIDEM – seemed to predict the rate or extent of compromised separation distance (Online Resource 2). The depth of the groundwater table relative to the ground surface (as reported to RIDEM or as measured in this study) is also not a predictor of compromised separation distance (Online Resource 2). It may be that the method of establishing the elevation of the seasonal high water table required by the regulations is inaccurate, leading to chronic underestimates of the elevation of the seasonal high water table (Cox et al. 2019; RIDEM 2002, 2018a). Before 2002, the seasonal high water table was measured using a temporary, 10-cm (4 in) diameter test well installed for a minimum of 48 h during the wet season (January to April; RIDEM 1996). After 2002, RIDEM also permitted the use of a soil evaluation, where the presence of soil redoximorphic characteristics was used to estimate the elevation of the seasonal high water table (RIDEM 2002). Both of these approaches are likely to underestimate the actual elevation of the seasonal high water table, as discussed in Cox et al. (2019). Individual well observations in the wet season are likely to underestimate the water table up to a third of the time, and/or by as much as half a meter (Morgan and Stolt 2004), and redoximorphic features are a poor indicator of the seasonal high water table in coarse or sandy soils (Humphrey and O'Driscoll 2011), typical in RI's near-shore regions, since soil above the redoximorphic features could be saturated 21% of the time (Morgan and Stolt 2006). Thus, current regulations governing seasonal high water table determination may be partially responsible for the observed patterns of compromised separation

distances, which are particularly worrisome in coastal areas with shallow groundwater tables and fragile coastal ecosystems.

Implications

The patterns of compromised separation distances in our study have serious implications for human health (Amador and Loomis 2018; Cooper et al. 2015, 2016), and likely contribute to the impaired water quality of many of the salt ponds (i.e. coastal lagoons) in the region. An estimated >1,500 septic systems are located within 100 m of a tidal water body along the southern RI coast (Cox et al. 2019), and thus our results suggest that a large number of systems in the region may be experiencing compromised separation distance. This may explain why nutrient loading has been an issue for many of the poorly flushed coastal salt ponds in the region (Nixon and Buckley 2007). Two of the coastal water bodies along the southern RI shore – Ninigret and Green Hill Ponds – have total maximum daily load (TMDL) management plans for bacteria that are monitored by the RIDEM, in compliance with the federal Clean Water Act (RIDEM 2018b). Septic systems in the area are major contributors to these pollution issues (Amador et al. 2018; Lapointe et al. 2017; Masterson et al. 2007). To add to the concern, every septic system on the barrier beach complex adds to the nutrient and pathogen loading of these coastal ponds, because wave action and other dynamics cause groundwater to flow from the beach face toward the salt pond (Nielsen 1999; Urish and Ozbilgin 1989). Compromised separation distance means that nutrients and pathogens cannot be retained or processed as

efficiently in the unsaturated zone of the soil (Amador et al. 2018; Amador and Loomis 2018; Cooper et al. 2016), resulting in increased loading to the ponds.

The southern RI coast is densely populated and there are no large-scale plans to expand the existing municipal sewer networks. Thus, onsite wastewater treatment will remain a critical component of the water treatment infrastructure in this region for the foreseeable future, as is the case in many other coastal communities in the US and abroad. These communities should therefore consider how best to improve individual systems' ability to maximize wastewater treatment to maintain groundwater and environmental quality. Our findings suggest that compromised separation distance is a real concern in many of the sites we investigated, raising questions about how widespread this issue is in the region and/or in similar communities worldwide. To our knowledge, there is no entity tracking separation distance in systems that are in place with conformed installations, and so there is currently no data available to evaluate this outside this study.

It is likely that climate change will exacerbate the observed compromises in separation distance, as more frequent and intense storm events affect coastal groundwater tables in acute episodes, and sea-level rise raises coastal groundwater tables over the coming decades (Emanuel 2017; Kopp et al. 2014; Lin et al. 2016). The region's seasonal occupancy patterns may partially protect the region's ground and coastal water bodies from some of the nutrient and pathogen loading associated with elevated groundwater tables (Cooper et al. 2016), as storm events resulting in short-term elevated groundwater tables during

the wet season coincide with the portion of the year when most seasonal residents are not occupying their properties. However, year-round residents in these communities will still be subject to these episodic events, and as groundwater tables rise in the coming decades, these instances of compromised separation distance could become more frequent or chronic, even outside the wet season.

To protect the health of both groundwater resources and the nearby coastal ecosystems, it might be necessary to update the current regulations for septic systems installed in the region to either (1) change the method of determining the seasonal high water table, which currently relies on redoximorphic features observed during a soil evaluation, or individual depth-to-water table measurements during the wet season; or (2) increase the required separation distance from the infiltrative surface to the groundwater table in coastal properties, to account for the lack of accuracy of current methods of determining the seasonal high water table, and that coastal groundwater tables are projected to rise with sea level in the coming decades (Cox et al. 2019; Mihaly 2017; Walter et al. 2016).

Currently, the only long-term groundwater table monitoring data publicly available in the region is from a USGS groundwater monitoring well (Site No. 412214071394001, RI-CHW 18) over 600 m from the nearest northern margin of Ninigret Pond in Charlestown, RI, one of the largest salt ponds along the southern RI shore. Though there have been efforts to quantify groundwater inputs into the salt ponds from the surrounding headlands (Moran et al. 2014;

Scott and Moran 2001), to the authors' knowledge, little research is available describing the region's current freshwater aquifer surface elevation dynamics along the northern shores within 100 m of the tidally-influenced salt ponds in the southern RI region. Furthermore, although there are models (Nielsen 1999) and data (Urish and Ozbilgin 1989) describing the shape and elevation of the freshwater aquifer's surface and lower boundaries in a barrier beach complex, there is little current data describing long-term fluctuations of the fresh groundwater table surface over the course of a year or its response to coastal storm events in the southern RI region.

To increase our understanding of coastal groundwater dynamics, and to improve the accuracy of seasonal high water table measurements, the development of a long-term groundwater table monitoring well network along the coast could prove useful, as these data could be used to track seasonal and annual variations, and document the effects of extreme events on the region's groundwater. These data would be useful in evaluating whether a seasonal high water table measurement at an individual property along the coast corresponds to a representative water table elevation, or if it was taken during an unusual water table phenomenon. Data from these wells might also prove useful in assessing the water table's response to precipitation events at particular sites. Tying each monitoring well to soil profile descriptions and the elevation of redoximorphic features could also help cross-validate these methods for use in the field.

Periodic re-assessment of an in-place system's vertical separation distance, if it is located within a certain distance of the coast could also be helpful.

Currently, RIDEM considers a system to have failed if a system shows evidence of not being able to accept effluent, or if it has demonstrably contaminated a nearby well or water course, triggering a requirement for system replacement to meet current regulations (RIDEM 2018a). Adding a provision of separation distance being less than a certain value qualifying a system as having failed might help trigger the replacement of poorly sited systems that are putting groundwater and coastal ecosystem health at risk.

Failure to address compromised separation distance could lead to considerable negative consequences for human and environmental health in the region, contribute to further undermining of coastal property values (McAlpine and Porter 2019) and possibly affect the region's tourism industry. As is often the case, preventing environmental contamination will likely be much more cost-effective than trying to remediate it once it has become an extensive problem.

Conclusions

We investigated the separation distance under drainfields in 10 waterfront sites in coastal Rhode Island to assess whether changes in groundwater tables have compromised the available volume of unsaturated soil beneath drainfields required to adequately treat wastewater. The vertical separation distance from the infiltrative surface of the drainfield to the seasonal high water table has been required to be at least 122 cm (4 ft) since 1992 in coastal areas (RIDEM 1996, 2018a). Our data indicate that only 20% of evaluated systems have adequate separation distance, except during infrequent extreme events that temporarily elevate coastal groundwater tables. In contrast, half the sites in this study had

compromised separation distance at least 50% of the time, and 30% of sites in this study did not meet separation distance requirements specified by the system's plan at any point in the year. At one site, the groundwater table reached the infiltrative surface at the drainfield 4/281 days. Neither age of system, landscape position nor system designer were correlated with rate or extent of compromised separation distance.

Our data suggest that the current method of assessing the seasonal high water table in coastal sites is inaccurate, and may result in compromised wastewater treatment performance regardless of how faithfully installation follows the plans. We recommend that RIDEM reexamine how the seasonal high water table is established in near-shore sites, as single observations of test wells during the wet season and reliance on redoximorphic characteristics in sandy soils are likely to underestimate the position of the seasonal high water table and therefore compromise a system's vertical separation distance. Furthermore, more research is needed to understand how generalizable the trends described in this study are to the New England coastal region, as well as to other coastal communities across the globe. Our approach to assessing the relationship between groundwater table elevations and critical septic system infrastructure can be easily adapted to meet the needs of communities in a wide range of geographic regions, and thus may be useful to inform policies across the US and abroad. Failure to address the issue of inadequate separation distance in near-shore and inland areas could lead to and/or exacerbate nutrient and pathogen loading problems for drinking water and surface water resources.

Conflict of Interest

The authors declare that they have no conflict of interest.

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Online Resources

Online Resource 1 Large precipitation and storm events along the southern RI coast from January 2017 to May 2019. Top: Rain events exceeding 24.5 mm (1 in) in a 24-h period, as measured by the Block Island (purple dots) and Kingston (green dots) weather stations (Iowa Environmental Mesonet 2019a, 2019b). Bottom: Extreme events in Washington County, RI reported by NOAA (NOAA National Centers for Environmental Information 2019), wave heights greater than 3.1 m as measured by the Scripps Buoy southeast of Block Island (station No. 44097) by the National Buoy Data Center (NOAA National Data Buoy Center 2019)

and MyCoast app user reports from Westerly, Charlestown and South Kingstown, RI (RI MyCoast et al. 2019). Figure 1 displays the location of each station

Online Resource 2 *Comparison of water table elevation change trends with time, mean water table depth, landscape location and system designer. Chi-squared, df and p values depicted in panels E and F are results from Kruskal-Wallis testing.*

Note: Redacted system design plans and 3-D renditions of the ground-penetrating radar transects of each drainfield are available from the authors by request.

Manuscript 3 :
**Structure of greenhouse gas-consuming microbial communities in
surface soils of a nitrogen-removing experimental drainfield**

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Highlights

- Greenhouse gas concentrations and flux not linked to CH₄ or N₂O consuming communities
- Methane oxidizing communities (*pmoA* gene) varied with depth in soil above drainfields
- N₂O reducing communities (*nosZ* gene) varied with depth and soil type above drainfield

Graphical Abstract

[Drainfield Cross-section]

Abstract

Septic systems represent a source of greenhouse gases generated by microbial processes as wastewater constituents are degraded. Both aerobic and anaerobic wastewater transformation processes can generate nitrous oxide and methane, both of which are potent greenhouse gases. To understand how

microbial communities in the surface soils above shallow drainfields contribute to methane and nitrous oxide consumption, we measured greenhouse gas surface flux and below-ground concentrations and compared them to the microbial communities present using functional genes *pmoA* and *nosZ*. These genes encode portions of particulate methane monooxygenase and nitrous oxide reductase, respectively, indicating microbes that may serve as a potential sink for the respective greenhouse gases. We assessed the surface soils above three drainfields served by a single household: an experimental layered passive N-reducing drainfield, a control conventional drainfield, and a reserve drainfield not in use but otherwise identical to the control. We found that neither GHG flux, below-ground concentration or soil properties varied among drainfield types, nor did methane oxidizing and nitrous oxide reducing communities vary by drainfield type. We found differences in *pmoA* and *nosZ*-containing communities based on depth from the soil surface, and differences in *nosZ* communities based on whether the sample came from the rhizosphere or surrounding bulk soils. Type I methanotrophs (Gammaproteobacteria) were more abundant in the upper and middle portions of the soil above the drainfield. In general, we found no relationship in community composition for either gene based on GHG flux or below-ground concentration or soil properties (bulk density, organic matter, above-ground biomass). This is the first study to assess these communities in the surface soils above an experimental working drainfield, and more research is needed to understand the dynamics of greenhouse gas production and consumption in these systems.

Introduction

Onsite wastewater treatment systems (OWTS; also known as septic systems) are a common means of treating household wastewater in the United States and around the world. In the northeastern U.S. at least 25% of homes rely on OWTS (U.S. Environmental Protection Agency, 2002). Greenhouse gas (GHG) emissions are a byproduct of microbial transformations of wastewater constituents in wastewater (Truhlar et al., 2016; Brannon et al., 2017; Fernández-Baca et al., 2018, 2019). Production and consumption of GHG is driven by microbial processes that occur in septic tanks and drainfields (Truhlar et al., 2016; Amador and Loomis, 2018; Fernández-Baca et al., 2019; Somlai et al., 2019). In addition to carbon dioxide (CO₂), OWTS emit both methane (CH₄) and nitrous oxide (N₂O), potent GHGs with ~25 and ~300 times, respectively, the global warming potential of CO₂ (IPCC et al., 2014). Two particular processes are involved in the consumption - and thus possible mitigation - of GHGs:

1. Methane oxidation to CO₂ by methylotrophs and methanotrophs, which can be aerobic or anaerobic (Wang et al., 2017). Most aerobic methanotrophs rely on particulate methane monooxygenase (partially encoded by the gene *pmoA*) to perform the first step in methane oxidation, and are subdivided into type I and type II, which tend to have low and high affinities for methane, respectively (Chistoserdova et al., 2009; Knief, 2015).

2. Nitrous oxide reduction, which converts N_2O to harmless nitrogen (N) – carried out by the enzyme nitrous oxide reductase (partially encoded by *nosZ*).

Microbial consumption of GHGs is thought to happen in the soil above conventional drainfield infiltrative surfaces, and both *nosZ* and *pmoA* are ubiquitous in drainfield surface soils (Fernández-Baca et al., 2018).

Grasses growing in lawns above shallow drainfields take up water and nutrients from wastewater dosed to the drainfield, resulting in lush green grass above laterals in stark contrast to grassy areas not located above a drainfield. The rhizosphere of these grasses has a different microbial community structure than the surrounding bulk soil, driven partly by the hypoxic regions surrounding roots, as the roots consume oxygen, and by C-rich root exudates (Philippot et al., 2013). Hypoxia, coupled with additional C sources, favors heterotrophic denitrification, so that the rhizosphere has a measurable impact on denitrifier community structure (Henry et al., 2008; Hamonts et al., 2013; Philippot et al., 2013) and thus may affect the flux of N_2O .

In addition to GHGs, OWTS can contribute substantial quantities of N to the environment, posing threats to groundwater and surface water quality (Hoghooghi et al., 2016). Conventional OWTS are not designed to remove N from wastewater before it reaches groundwater (Cooper et al., 2015, 2016b; Lancellotti et al., 2017), which results in high N loads delivered to coastal ecosystems (Amador et al., 2018). To address this problem, engineered advanced

N-removal technologies are commonly required in coastal areas (Deschutes County, 2008; BCDHE, 2014; MDE, 2016; RIDEM, 2018). These proprietary technologies can cost homeowners more than twice as much as a conventional system, which presents a barrier to widespread adoption. A less costly, non-proprietary layered drainfield designed to remove N from septic tank effluent has been tested in Canada, Brazil and the US (Robertson et al., 2000; NY State Center for Clean Water Technology, 2016; Suhogusoff et al., 2019), and is currently undergoing field testing in Barnstable County, MA (NY State Center for Clean Water Technology, 2016). In contrast to conventional gravity-fed drainfields used in Massachusetts – which consist of sand media overlying native soil – the layered drainfield is pressure-dosed with septic tank effluent, and consists of layered media designed to facilitate passive N-removal. Nitrification takes place in an upper sand layer, and denitrification takes place in a lower layer consisting of a 1:1 mix (by volume) of sand and sawdust, providing a C source that is typically lacking in the deeper portions of conventional drainfields. In similar systems comprised of woodchips rather than sawdust, about half of the reactivity of woodchips is lost during the first year as soluble C compounds elute out, but stable rates of N-removal persist for many years once established (Robertson, 2010). This layered design, which borrows design principles from C-amended bioreactors commonly used in agriculture and groundwater remediation (P. W. van Driel et al., 2006; Schipper et al., 2010; Addy et al., 2016), may change GHG flux patterns when compared to conventional systems, as the saturated C source

could support methanogenesis, and could act as a source of N_2O from incomplete denitrification.

We examined the relationship among above- and below-ground GHG emissions and CH_4 - and N_2O -consuming microbial communities as determined by *pmoA* and *nosZ*, respectively, in the surface soil above three types of drainfields serving the same household. One drainfield treats pressure-dosed septic tank effluent via a 90-cm sand layer (conventional system), and the other uses the layered experimental N-removing media described above. A third drainfield is identical to the conventional system, but is not in use. We hypothesized that surface soil community structure would vary in response to differences in below-ground CH_4 and N_2O concentration and differ with differences in surface GHG flux. Given that different types of aerobic methanotrophs favor different concentrations of methane and oxygen (Knief et al., 2003; Knief, 2015), we expected that methanotroph community structure would vary by drainfield type based on differences in methane concentrations. Additionally, we expected to find methanotroph communities to vary by depth in the surface soils of these drainfields, which likely experience a gradient of methane concentration, favoring high-affinity type I methanotrophs in the surface, and lower affinity type II methanotrophs deeper in the soil profile. Finally, we expected community structure to be different between rhizosphere and bulk soil in response to higher C availability and lower O_2 levels in the rhizosphere, likely providing conditions favorable for heterotrophic denitrifiers.

Methods

System description

The OWTS in our study serves a single family (two adults, one young child) in Acushnet, MA, and was installed in April 2018. The system (Figure 3.1) consists of a two-chamber, 5,680-L septic tank, a 3,785-L pump chamber with a mechanical pump that delivers a consistent dose of effluent (187.3L per dose; ~870 L per day) to a GeoMat™ 1200 Leaching System (Geomatrix Systems, LLC, Old Saybrook, CT), a shallow pressurized network of PVC pipes above a 30-cm wide bed of plastic filaments, surrounded by geotextile fabric (Wigginton et al., 2020). The drainfield was divided into three sections (Figure 3.1): (1) a reserve area, consisting of 3 GeoMat™ laterals above 90cm of medium sand that is not dosed with wastewater; (2) a control area, consisting of 3 GeoMat™ laterals above 90cm of sand dosed an average of 4.7 times a day; and (3) an experimental area, consisting of 3 GeoMat™ laterals on a 45-cm layer of sand underlain by a 45-cm layer of a 1:1 (by volume) mix of sand and sawdust (mainly *Quercus spp.*), dosed at the same time and with the same volume as the control area. An impermeable vertical plastic liner separates the lowest 45 cm of the control and experimental areas (Figure 3.1). The entire drainfield is underlain by a 5-cm layer of peastone, designed to retain water above this interface and increase saturation and anoxic conditions of the lower layer of the drainfield.

Mean values for septic tank effluent constituents measured from system startup to the time of sampling fall within established values for residential wastewater (Loomis and Kalen, 2014; Amador and Loomis, 2018): 16.5 °C, 191.6

mg/L 5-day biochemical oxygen demand (BOD₅), 80.4 mg/L total suspended solids (TSS), 1.5 mg/L dissolved oxygen (DO), 55.1 PPM total nitrogen (TN), 228.8 mg alkalinity, and pH of 7.6.

Figure 3.1. Diagram of septic system layout for a single family dwelling (not to scale). Left: An aerial or plan view shows how wastewater moves from the dwelling to the septic tank, then to a pump chamber and is dosed to the drainfield. The drainfield is subdivided into three areas: Experimental (layered), Control (conventional) and Reserve (identical to conventional but not dosed with wastewater). Right: A cross section of a single lateral in the control and experimental layered drainfields. Both drainfields are dosed with wastewater via the GeoMat™, which is located approximately 25 cm below the soil surface. The control drainfield media consists of a 90-cm layer of sand, while the experimental side consists of a 45-cm sand

layer above a 45-cm layer containing a mix of sand and sawdust. Both sides are underlain by a 5-cm later of peastone, which sits above undisturbed native soil.

Below-ground GHG measurements

The control and experimental areas were instrumented with below-ground gas sampling chambers consisting of 17.1-cm-long and 2.5-cm-i.d. capped PVC well screen (Atlantic-Screen Inc., Milton, DE) that was connected to the surface via nylon tubing (Wigginton et al., 2020; Figure 3.2). Chambers were installed approximately 15cm below the GeoMat™ laterals, corresponding to a depth of ~38 cm from the ground surface. A three-way stopcock valve at the end of each tube, closed to the atmosphere when not being sampled, was used to collect gas samples (Wigginton et al., 2020).

Figure 3.2. Sampling scheme for greenhouse gas measurements, soil properties and microbial community analysis. Diagram is not to scale.

Before we sampled below-ground gases, we purged the gas tubing and chambers. A 300-mL volume of gases was extracted with an air-tight 60-mL syringe and expelled to the atmosphere. Next we mixed pore space gases five times by drawing up a 60-mL gas sample and expelling the gases back into the gas chamber. A sample of the mixed gases was collected and stored in evacuated 20-mL glass vials with rubber septa that were submerged in water (Wigginton et al., 2020). Gas samples were analyzed on a Shimadzu Gas Chromatograph-2014 Greenhouse Gas Analyzer fitted with a methanizer (Shimadzu Scientific Instruments, Columbia MD), using either a flame ionization detector (250°C, CO₂ and CH₄) or an electron capture detector (325°C, N₂O). The column was kept at

80°C and the injection port operated at 100°C. The carrier gas was ultrapure N₂, flowing at a rate of 25 mL/min (Wigginton et al., 2020).

Surface GHG flux measurements

We measured greenhouse gas fluxes from the soil surface in three locations per drainfield type (Figure 3.2) using a dark, closed PVC chamber (26.5 cm i.d., 13 cm height; Wigginton et al., 2020) connected to a model G2508 real-time cavity ring down spectroscopy analyzer (Picarro, Santa Clara, CA) via nylon tubing. The chamber, outfitted with a rubber gasket, was placed on a beveled ring made of 11.5-cm tall, 24.5-cm i.d. section of PVC pipe, which was pounded into the ground to a depth of ~10 cm. The chamber was left in place for 8 to 10 minutes at each sampling location (Brannon et al., 2016). Air temperature inside the chamber was measured continuously using a Hobo™ data logger (Onset, Bourne, MA).

Gas flux data were processed as described in Brannon et al., (2017, 2016) and Martin and Moseman-Valtierra (2015). Non-significant slopes ($p < 0.05$) were reported as no flux.

Surface soil properties

To characterize surface soil properties, we collected three soil cores in each drainfield type using aluminum cores (4.55-cm i.d., 15.1-cm tall). Cores were taken within 1 hour after GHG flux measurements in the same area where these measurements were made (Figure 3.2), transported in sealed bags on ice, and stored at 4°C until analysis. For each core, we determined soil bulk density by dividing oven-dried (105°C) core weight by the core volume (267.5 cm³). We used

the loss-on-ignition method (550°C for 5 hours) to determine organic matter content. To characterize the above-ground plant biomass, we cut vegetation at the top of each core at the soil surface and dried the vegetation at 60°C for 2 weeks.

Soil samples for microbial analysis

Soil samples for microbial analyses were collected with a flame-sterilized coring device (2-cm i.d., 23-cm length) within 1.5 hours of making greenhouse gas flux measurements in the same area where flux chambers were placed (Figure 3.2). Cores were inserted into the ground vertically until resistance indicated we had reached the top of the geotextile fabric of the GeoMat™. Three cores were collected per treatment area (one core corresponding to each flux measurement location, see Figure 3.2).

We used flame-sterilized tools to subdivide each core into upper (0-5 cm), mid (5-10 cm) and lower (10-15+ cm) sections. From each section we collected soil samples from the center of the core (to avoid cross-contamination) and placed them into sterile Whirlpack bags. We also separated visible roots and soil particles clinging to roots from “bulk” surrounding soils for each depth. Soil samples were transported on ice to the lab and stored at -80°C until analysis.

Microbial Community Analysis

Community DNA was extracted from soil samples using the DNeasy PowerSoil Kit (Qiagen, Hilden, Germany) following the manufacturer’s instructions. 50 of the 54 possible microbial core subsamples collected in the field were included in our analysis, because four of the rhizosphere samples did not

contain enough soil to extract at least 5 ng/ μ L DNA as determined using a NanoDrop™ spectrophotometer (ThermoFisher Scientific, Waltham, MA). The DNA extracted from these 50 samples was used to amplify target gene sections.

We amplified *nosZ* and *pmoA* gene fragments in separate, single 50- μ L reactions using primers *nosZ*1F and *nosZ*1662R (Throbäck et al., 2004; Wigginton et al., 2018), and *pmoA*A189cgF and *pmoA*Amb661R (Costello and Lidstrom, 1999; Fernández-Baca et al., 2019), respectively. Primer pairs included a sequencing overhang (forward primer: TCGTCGGCAGCGTCAGATGTGTATAAGAGACAG, reverse primer: GTCTCGTGGGCTCGGAGATGTGTATAAGAGACAG) at the 5' end. Supplemental Material 1 contains primer sequences and PCR conditions for amplifying gene fragments. We examined PCR products on a 1% (w/v) ethidium bromide-stained agarose gel.

Amplicons were purified (Agencourt AMPure XP, Beckman Coulter, Indianapolis, IN), quantified (Invitrogen Qubit™ Flex Fluorometer, Thermo Fisher Scientific, Waltham, MA), indexed, and pooled equimolarly. Sequencing was performed on paired-end reads of 300 bp on an Illumina Miseq Next Generation Sequencer at the University of Rhode Island's Genetic Sequencing Center.

Sequencing data analysis & statistics

We used QIIME 2 version 2019.10 (Bolyen et al., 2019) to process sequence reads and perform bioinformatic analyses. We demultiplexed and quality filtered raw reads with the q2-demux plugin, and then separated the pooled gene reads using the q2-cutadapt plugin. We used PCR primer sequences (Supplemental Material 1) and their reverse complements for each gene to find

and separate matching reads and trim off the primers. After this step, each gene amplicon was analyzed separately. For each gene amplicon, trimmed demultiplexed reads were joined, filtered and denoised using the q2-dada2 plugin (Callahan et al., 2016) and chimeras were removed, resulting in unique amplicon sequence variants (ASVs). We used the q2-alignment plugin to align ASVs with MAFFT (Katoh et al., 2002) to produce rooted and unrooted phylogenetic trees with FastTree 2 (Price et al., 2010).

To assign taxonomy with the q2-feature-classifier plugin (Bokulich et al., 2018), we created custom *nosZ* and *pmoA* databases. Briefly, we downloaded all FASTA sequences from the NCBI Nucleotide database matching the gene query in question ("*nosZ*" or "*pmoA*"), then converted the files to 'DNAFASTAFormat' to import into QIIME 2. We also downloaded a list of all accession numbers matching the same gene query from the NCBI Nucleotide database, and used Entrez Direct (E-utilities on the UNIX Command Line; Kans, 2013) to fetch taxonomy strings for each accession number. Taxonomy strings were formatted for import into QIIME 2. Using the imported sequences from NCBI for each gene, we extracted reference reads (via q2-feature-classifier) matching our PCR primer sequences (Supplemental Material 1), and then trained the feature classifier with the appropriate taxonomy using a naïve Bayes approach. To assign taxonomy to our sequenced reads, we used the classify-sklearn command from the q2-feature-classifier plugin using our custom classifier for each gene.

We used the phyloseq package (v 1.30) in R (v 3.6.2; R Core Team, 2019) to explore alpha diversity and create taxonomic barplots and heatmaps in ggplot2

(v 3.2.1) from the QIIME2 outputs. We used the vegan package (v 2.5-6) to calculate Bray-Curtis dissimilarity matrices, and perform permutational multivariate analysis of variance (PERMANOVAS) using these distance matrices (with 999 permutations) to assess beta diversity via the Adonis function. We created dendrograms based on hierarchical clustering (hclust) with “average” distances, and visualized these with the gg dendro (v 0.1-20) package. The complete code for this analysis, including parameters used in each step, can be found at <https://github.com/alissacox/GHG-cycling-genes>.

We used R (v 3.6.2) to perform statistical analyses of soil properties, greenhouse gas fluxes and concentrations, and their relationships, which were plotted using the ggplot2 and ggpubr (v 0.2.4) packages. P values less than 0.05 were used as a cutoff to determine statistical significance.

Results & Discussion

Surface soil properties

There were no significant differences among drainfield types for bulk density, organic matter content or above-ground biomass (Table 3.1), although visual inspection of each drainfield area showed much more lush grass above the control and experimental. Bulk density and above-ground biomass showed a statistically significant inverse relationship.

Table 3.1. Mean and standard deviation (n=3) of soil property values for the upper 15 cm of soil above the drainfield collected at each gas sampling point. There were no statistical differences among drainfield types based on Kruskal-Wallis testing.

Drainfield type	Bulk density (g/cm ³)		Organic matter (%)		Above-ground dry biomass (g/cm ²)	
	Mean	SD	Mean	SD	Mean	SD
Control	1.22	0.06	3.81	0.52	0.07	0.02
Experimental	1.27	0.05	4.15	1.41	0.08	0.01
Reserve	1.37	0.02	3.49	0.12	0.01	0.02

Greenhouse gases

Figure 3.3. LEFT: Greenhouse gas flux measurements at the ground surface. The dashed line in the middle plot indicates 0 flux. RIGHT: Greenhouse gas concentrations in the control and experimental drainfields collected ~38cm below the ground surface. The reserve drainfield lacked gas sampling chambers.

All three drainfield types were net emitters of CO₂ and N₂O and net sinks for CH₄ (Figure 3.3). Similar results have been reported for drainfields from conventional OWTS (Truhlar et al., 2016; Fernández-Baca et al., 2018, 2019; Somlai et al., 2019). The reserve area had the lowest CO₂ and N₂O emissions, and consumed methane at about the same rate as the control and experimental areas. Although the control and experimental drainfields had elevated N₂O and CH₄

concentrations relative to atmospheric levels 37 cm below the ground surface (Figure 3.3), N₂O flux ranged from 0.01 to 0.02 $\mu\text{mol}/\text{m}^2/\text{h}$ and CH₄ flux ranged from -1.73 to -0.53 $\mu\text{mol}/\text{m}^2/\text{h}$, suggesting consumption of these gases in surface soil. Previous studies of GHG fluxes from drainfield soils above conventional drainfields report ranges of -12 to 99 $\mu\text{mol}/\text{m}^2/\text{h}$ for methane, and 0.14 to 14.11 $\mu\text{mol}/\text{m}^2/\text{h}$ for nitrous oxide (Fernández-Baca et al., 2018, 2019).

Microbial community richness, diversity and composition

Microbes with methane oxidizing potential as detected by pmoA

Our samples contained 161 unique *pmoA* sequences (or sequence variants). The 161 *pmoA* sequences were found a total of 3,136 times across 36 samples (samples with 0 sequences were dropped from analysis). Particulate methane monooxygenase amplicons in our dataset ranged from 265 to 440 base pairs (bps), with a mean length of 317.5 bps and a median length of 297 bps. The mean frequency per sample was 64 (median = 25), with maximum frequency per sample of 550. Five of the 36 samples had fewer than 10 sequences, but were retained in the analysis. Rarefaction curves (data not shown) indicate that we had sufficient sampling depth, despite the low numbers of unique sequence variants. Fernández-Baca et al. (2018) reported similar numbers of *pmoA* sequences for communities in drainfield soils.

Analysis of alpha diversity shows that the number of observed sequence variants ranges from 1 to 27 across all samples (Supplemental Material 2). The Shannon index, a measure of richness and diversity that accounts for both abundance and evenness of the taxa present (Shannon and Weaver, 1949), ranges

from 0 to 2.8, and Simpson index values, which measure the sample richness by assessing the relative abundance of individual sequences (Simpson, 1949), range from 0 to 0.9. There were no significant differences in alpha diversity measures for drainfield type, depth, or soil type (bulk vs. rhizosphere), nor was alpha diversity significantly correlated with any soil properties or GHG flux or concentration values. In an analysis of upland methanotroph communities using the same *pmoA* primer pair in this study, only pH was found to have a significant influence on community distribution (Knief et al., 2003). Methane and oxygen concentrations, nutrient availability, temperature and soil moisture have also been described as drivers of methanotroph community structure (Knief, 2015). In contrast, Fernández-Baca et al. (2018) found no relationship between methanotroph communities and soil properties including volumetric water content, temperature or conductivity in a study of methane oxidizing communities in soil above conventional drainfields.

We also examined differences in the beta diversity of our samples using a Bray-Curtis distance matrix (Sørensen, 1948). Examination of distances using permutational multivariate of analysis of variances (Adonis) based on 999 permutations indicates that there were no significant differences in beta diversity among drainfield types. There were, however, significant differences by depth, exemplified by tendency of samples to cluster together based on depth in the dendrogram (Figure 3.4 left) No other variable (soil properties, below-ground GHG concentration or above-ground GHG flux) showed a significant relationship with beta diversity patterns for *pmoA*.

pmoA

nosZ

Depth a Lower a Middle a Upper Soil Type a Bulk Soil a Rhizosphere

Figure 3.4. Beta diversity dendrograms based on “average” method of hierarchical clustering using Bray-Curtis distance matrices for *pmoA* (left) and *nosZ* (right). The length of the branches

indicates distance between samples. *pmoA* samples are colored by depth and *nosZ* samples are colored by soil type. Sample codes indicate drainfield type (Control, Experimental or Reserve), replicate number, sampling depth (Upper, Middle or Lower) and soil type (Bulk or Rhizosphere).

The majority of the *pmoA* sequences (31.6%) are unidentified Bacteria, Methylococcaceae (27.4%), and unidentified Proteobacteria (17.7%; Figure 3.5), in agreement with previously published *pmoA* sequence analyses (Sengupta and Dick, 2017). Unidentified Gammaproteobacteria (10.3%) and unidentified Bacteria from environmental samples (8.6%) are the next most abundant. Another 3.0% are unidentified members of the Methylococcales order, and 1.2% are in the Methylocystis family. *Methylobacter* sequences represent five of the unique *pmoA* sequence variants (0.2%) from our samples, and are only found in sample R3MB (Figure 3.5). These findings are in line with those of (Fernández-Baca et al., 2018, 2019), who found similar taxonomic groups for methane oxidation in surface soils of conventional drainfields and drainfield mesocosms. These phylogenetic groups have also been reported for soils above capped landfills (Rai et al., 2019). Type I methanotrophs (gammaproteobacterial; Hanson and Hanson, 1996) in this study appear to be more prevalent and abundant than type II methanotrophs, as described by Rai et al. (2019) and Sengupta and Dick (2017), including those in a conventional drainfield soil (Fernández-Baca et al., 2018). Our results differ from those for a conventional drainfield soil, where the proportion of type II methanotrophs was greater than the proportion of type I methanotrophs (Fernández-Baca et al., 2018). However, our drainfield surface soils are quite

shallow (< 25cm), and thus may have different gas diffusion dynamics than the soils above the deeper conventional systems (likely 90-120 cm) described by Fernández-Baca et al. (2018). This may explain why we had more type I methanotrophs (gammaproteobacterial). Type I methanotrophs are adapted to grow at low concentrations of methane, while type II (Alphaproteobacteria) are adapted to oxidize higher levels of methane and are more tolerant to low oxygen levels (Nazaries et al., 2013). *Methylocystis* spp., the only type II methanotroph present in our study, are oligotrophic (Knief and Dunfield, 2005), and were only present sporadically in our samples (Figure 3.5).

While there were no differences in taxonomic composition among drainfield types, per sample taxonomic composition does appear to vary somewhat by depth: with the exception of sample R3LB (the bulk soil from the lower portion of the 3rd core taken in the reserve drainfield), Gammaproteobacteria (Type I *pmoA*; Hanson and Hanson, 1996) appear to be restricted to middle and upper portions of the soil cores, mirroring patterns described in rice paddy soils (Lee et al., 2015) and in surface soils above drainfield mesocosms (Fernández-Baca et al., 2019). Fernández-Baca et al. (2019) hypothesized that decreasing oxygen levels below 8 cm depth may be responsible for the decline in aerobic methanotroph communities, which may explain what is occurring in this study. Anaerobic methane oxidizers who couple methane oxidation to nitrite reduction were not examined in this study, as the *pmoA* primers used do not amplify *pmoA* sequences from these anaerobes (Luesken et al., 2011). Our study also did not follow anaerobic methane-oxidizing archaea

(ANMW-2), which are thought to use a form of reverse methanogenesis (Wang et al., 2017) and don't have *pmoA*. The soil immediately above the GeoMat™ distribution system is likely to fluctuate between oxic and hypoxic/anoxic states during every dose of septic tank effluent, as the nutrients and carbon are degraded by aerobic organisms at the infiltrative surface, depleting oxygen temporarily. This cycling of oxygen concentration may make it difficult for type I methanotrophs to thrive in the deeper portions of the surface soil above the drainfield. This is in contrast to the oligotrophic *Methylocystis* spp. (Knief and Dunfield, 2005), whose distribution appears to be less constrained by depth, especially in the reserve portion of the drainfield (Figure 3.5).

Figure 3.5. Relative taxonomic abundance of *pmoA*-containing methane oxidizing bacteria. Sample codes indicate drainfield type (Control, Experimental or Reserve), replicate number, sampling depth (Upper, Middle or Lower) and soil type (Bulk or Rhizosphere). One sequence variant in sample C1LB assigned to “*Streptomyces*” with 77% confidence was reclassified manually as “*Bacteria*”.

The composition of the methane oxidizing community does not vary among drainfield types; only sample depth is correlated with community composition of the methane oxidizers in drainfield surface soil. At installation, the surface soil microbial community across all three drainfield areas was similar, since the same soil was used to grade and landscape the surface. Although we

measured elevated GHG concentrations – relative to background atmospheric levels – below the drainfield's PVC distribution network in the experimental and control drainfields, these differences do not appear to drive the surface methane oxidizer communities to diverge from the reserve portion of the drainfield.

Although there was not much difference in methane flux in the three drainfield types (Figure 3.3), it is unclear whether this is a function of the composition of the methane oxidation community or of other factors. The fact that the community is stratified by depth indicates that enough time has elapsed between system installation and our sampling event to establish stable, divergent communities. Monitoring of bacterial community dynamics in soil under controlled laboratory conditions indicates that soil microbial communities can become relatively stable in as few as 3-5 weeks, and maintain this stable composition for months thereafter (Zegeye et al., 2019). Therefore it is likely that the communities we sampled in this study represent communities that have achieved relative stability in the ~14 months between installation and sampling.

There are several possible explanations for why the methane oxidizing community appears to vary with sampling depth, but not among drainfield types. The *pmoA* community stratified to reflect patterns typically found in drainfield soils, with more methanotrophs present at shallow depth and fewer occurring deeper in the profile (Fernández-Baca et al., 2018, 2019), but flux values may not vary sufficiently among drainfield types to result in differences in community structure among drainfields. It may be that the community we characterized via amplification of *pmoA* is not shaped or driven by methane levels in the sub-

surface, a finding previously described by Fernández-Baca et al. (2018). Methane levels may not shape the communities in this study because the organisms containing *pmoA* are not relying on this gene for major metabolic functions. Our findings are in contrast to a study examining expression of *pmoA* in drainfield mesocosms, which did find communities to be shaped by methane levels (Fernández-Baca et al., 2019), though conditions in our study were likely more variable than in a mesocosm experiment using artificial wastewater. It is possible that our amplification and analysis methods (primer choice, analysis pipeline) inadvertently did not capture major players in this community. Our primers are specific to aerobic methanotrophs, and anaerobic methane oxidizers may be important members of methane oxidizing functional community in these drainfields. As mentioned earlier, *pmoA* in anaerobic methane oxidizers using DAMO process are not amplified with the primer set used in this study (Luesken et al., 2011), nor can our methods identify anaerobic methane-oxidizing archaea (ANMW-2) relying on reverse methanogenesis. Finally, it is also possible that the ammonia volatilization following dosing of septic tank effluent may inhibit methane oxidizers directly above the infiltrative surface. This would only occur in the control and experimental drainfields that receive regular doses of wastewater, and may explain why more samples in the “lower” portion of our microbial cores in the control and experimental drainfields did not contain *pmoA* (Figure 3.5).

Bacteria with denitrification capacity as detected by nosZ

Our *nosZ* amplicons contained 5,739 unique sequences, appearing a total of 673,960 times across all 50 samples. Sequence length ranged from 250 to 422 bps, with mean and median lengths of 408 and 414 bps, respectively. The per sample mean frequency was ~13,500, with a median of ~13,900 and a minimum and maximum frequency of about 3,000 and ~31,000, respectively. All samples were retained in the subsequent analyses.

Alpha diversity measures indicate a range of observed sequence variants from 67 to 380 per sample. Shannon's index ranges from 2.13 to 5.26, and Simpson's index ranges from 0.78 to 0.99 (Supplemental Material 2), indicating greater diversity and richness of *nosZ* genes than for *pmoA*. This is to be expected, as Bacteria with *nosZ* are far more common, spanning a much wider number of phylogenetic groups than *pmoA*. There were no statistically significant differences in alpha diversity among drainfield type, or depth, nor was there a statistically significant relationship with GHG flux or below ground concentration of these gases. Though values for both alpha diversity indices are somewhat lower in the rhizosphere samples than in the bulk samples, these differences were not significant.

Based on PERMANOVA analysis with 999 permutations, there were no differences in beta diversity among drainfield types. There were, however, significant differences in beta diversity between bulk soil and rhizosphere *nosZ* communities, as well as among communities at different sampling depths. Previous research in conventional drainfield soils found greater *nosZ* abundance

and variants in surface soils (Fernández-Baca et al., 2018). The dendrogram in Figure 3.4 (right) shows clear clustering patterns based on soil type for *nosZ*. There were no other statistically significant relationships between *nosZ* beta diversity and any other variables.

In contrast to the *pmoA* taxonomic assignments, the vast majority of the sequence variants for our *nosZ* amplicons are classified as unknown Bacteria (46.9%) or unclassified Bacteria identified in environmental samples (40.6%; Figure 3.6). The remaining sequences are all in the phylum Proteobacteria. The bulk of these sequences are in the Alphaproteobacterial order, with many identified to genus: *Mesorhizobium* (5.3%), *Bradyrhizobium* (4.2%) and *Afipia* (1.9%). *Thauera* spp. (Zoogloeaceae) make up another 0.4% of the overall sequences, while *Azospirillum* spp. from environmental samples make up another 0.5%. Other taxa each account for less than 0.05% of the remaining sequences, and include the genera *Aeromonas* (the only Gammaproteobacterium in our samples), *Oligotropha*, *Zoogloea*, *Achromobacter*, *Pleomorphomonas*, *Hydrogenophaga*, *Sinorhizobium/Ensifer*, *Acidovorax*, *Agrobacterium*, and *Azospirillum*. Both genera in the Zoogloeaceae are common to wastewater treatment plants (Wang et al., 2012), and are able to utilize nitrate, nitrite and nitrous oxide as terminal electron acceptors, in addition to oxygen (Boden et al., 2017). The other Betaproteobacterial genera are also members of denitrifying families, and have been found in soil and/or onsite wastewater treatment systems (Orellana et al., 2014; Ji et al., 2015; Wigginton et al., 2018; Qin et al., 2020). In terms of Alphaproteobacteria, both the Bradyrhizobiales and Rhodospirillales orders

contain genera occurring in soils and/or onsite wastewater treatment systems, and are known to complete at least some portion of the denitrification pathway (Palmer and Horn, 2012; Bai et al., 2012; Orellana et al., 2014; Wigginton et al., 2018).

Examining the taxonomic composition of the low abundance taxa shows that the reserve drainfield contains more samples with more low-abundance taxa than either of the other drainfield types (Figure 3.6). Enrichment experiments in soil have shown that the original microbial community is reduced by growth on chitin-enriched soils (Zegeye et al., 2019), which may mirror the soils above the control and experimental drainfields which are dosed with N- and C-rich wastewater, and may have enriched for community members able to tolerate and thrive in these conditions in these areas. The lack of enrichment for particular members in the reserve portion may better reflect the original community composition of the topsoil used to for system installation. Interestingly, *Zoogloea* spp., common organisms in wastewater, are found in all drainfield types, even the reserve drainfield which was not dosed with wastewater.

Figure 3.6. Relative abundance of all *nosZ* sequences and their taxonomic assignments (top) and those representing <10% of the overall abundance by taxon (bottom). Sample codes

indicate drainfield type (Control, Experimental or Reserve), replicate number, sampling depth (Upper, Middle or Lower) and soil type (Bulk or Rhizosphere).

nosZ is commonly found and expressed, even in organisms with incomplete or partial denitrification pathways (Philippot et al., 2011; Lycus et al., 2018; Albright et al., 2019). Thus many microbes are capable of performing the final step of denitrification, and potentially serve as a sink for N₂O (Conthe et al., 2019). The primers used in this study amplify a wider range of cultured and environmental *nosZ* organisms than other primers (Throbäck et al., 2004). The large number of unique sequence variants assigned to a variety of taxa support this. Our results indicate that the differences in this study between below-ground N₂O concentration and surface flux (Figure 3.3) are likely driven by active nitrous oxide reducers in surface soils, serving as an N₂O sink.

The ubiquity of *nosZ* among Bacteria is surprising, considering this conversion is not an energy-yielding process and there are many other electron-generating processes likely to be active in these soils. However, *nosZ* has been hypothesized to be able to effectively compete with other denitrifying reductases for electrons and thus be highly active and efficient, suggesting that organisms able to reduce N₂O may have a competitive advantage in environments with frequently fluctuating availability of electron donors and acceptors (Conthe et al., 2019). As described earlier, the soils directly above the infiltrative surface are likely to experience such cycling following every dosing event, as fresh anoxic wastewater makes contact with oxygen and the microbial communities in the

immediate vicinity of the GeoMat™ distribution network. Given that *nosZ* may provide a competitive edge in this variable environment, the lack of relationship observed in this study between community composition and either N₂O flux or below-ground concentration is surprising, but has been reported in conventional drainfield soil communities (Fernández-Baca et al., 2018, 2019). It is possible, however, that the regular doses of wastewater create a gradient of available electron acceptors and donors for microbes in the soils above the drainfield, and may be what drives differences in beta diversity by depth. Long-term application of sewage sludge shapes *nosZ* communities to be different than soils amended with non-organic fertilizers, which is hypothesized to be a function of C contained in the sludge (Enwall et al., 2005). Furthermore, qualitative observations made during sampling indicate that our soil cores generally had fewer root hairs in the lower portions of the soil cores, possibly reducing some of the optimal habitat for these heterotrophic denitrifiers, and contributing to the observed differences by depth. However, the observed differences in *nosZ* communities based on soil type did support our hypothesis, indicating that the root exudates in the rhizosphere, a hypoxic area where oxygen is consumed rapidly by roots (Philippot et al., 2013), likely provide important carbon compounds to heterotrophic nitrous oxide reducers in the drainfield surface soils in this study. Similar findings have been described in agricultural systems (Henry et al., 2008; Hamonts et al., 2013).

Contrary to our expectations, we did not observe differences in methane oxidizing or nitrous oxide reducing communities among drainfield types, but found that beta diversity varied significantly with depth (*pmoA* and *nosZ*) and with

soil type (and *nosZ* only). Given the complexity of microbial community dynamics across very fine scales in soils and soil aggregates (Wilpiseski et al., 2019), it is possible that syntrophy among various community members plays an important role in these drainfield soils, and that GHG consumption is a function of more than just methane monooxygenase or nitrous oxide reductase. It is also possible that the microbial community growing in the GeoMat™ and/or at the infiltrative surface also consumes GHGs generated in the drainfield below, affecting the concentration and flux of GHG in the soil above. Analyses of these communities and their functions could possibly help explain the observed difference between below-ground concentrations and net surface flux in this study.

Conclusions

We found the drainfield soils to be net emitters of CO₂ and N₂O, and net consumers of CH₄. Fluxes of each GHG were significantly correlated with soil bulk density and organic matter, but did not vary by drainfield type. When we examined methanotroph and nitrous oxide reducing communities in surface soils above drainfields, we found that methanotroph community composition varied with depth, with type I methanotrophs (Gammaproteobacteria) more abundant in the upper and middle portions of the soil samples. Neither drainfield type nor soil type (bulk vs. rhizosphere) were associated with significant differences in methanotroph communities, nor did GHG flux from the surface or GHG concentration below the infiltrative surface appear to have measurable effects on community composition. In contrast, nitrous oxide reducer community

composition varied with soil type (bulk vs rhizosphere soil), but was not associated with drainfield type or any of the other soil properties or GHG surface flux or below-ground concentrations measured in this study.

To our knowledge, this is the first study to explore GHG consuming communities above an experimental passive N-removing drainfield. Onsite wastewater treatment in coastal regions is becoming more and more focused on N removal, and a better understanding of N-removing technologies' impacts on GHG emissions and the mechanisms involved is critical to understanding the true environmental impact and life cycle costs of these critical components of our wastewater infrastructure.

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Author Contributions

A.H. Cox: Conceptualization; Data curation; Formal analysis; Investigation; Methodology; Project administration; Software; Validation; Visualization; Roles/Writing - original draft; Writing - review & editing.

S.K. Wigginton: Conceptualization; Data curation; Investigation; Methodology; Resources; Software; Validation; Writing - review & editing.

J.A. Amador: Conceptualization; Funding acquisition; Investigation; Methodology; Project administration; Resources; Supervision; Validation; Writing - review & editing.

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Supplemental Materials

Supplemental Material 1. Polymerase Chain Reaction (PCR) conditions for amplifying *nosZ* and *pmoA* gene fragments from soil-extracted DNA.

	<i>nosZ</i>	<i>pmoA</i>
Forward Primer (5'-3')	<i>nosZ</i> 1F (CGYTGTTCMTCGACAGCCAG)	<i>pmoA</i> A189cgF (GGNGACTGGGACTTCTGG)
Reverse Primer (5'-3')	<i>nosZ</i> 1662R (CGSACCTTSTTGCCSTYGCG)	<i>pmoA</i> Amb661R (CCGGMGCAACGTCYTTACC)
Fragment length (bp)	417	508
PCR mixture (50 µL reaction)	25 µL MyFi 21.75 µL Water 1 µL Forward Primer (10 µM) 1 µL Reverse Primer (10 µM) 1.25 µL extracted DNA template	20 µL MyFi 20 µL Water 2.5 µL Forward Primer (10 µM) 2.5 µL Reverse Primer (10 µM) 5 µL extracted DNA template
Thermocycler settings	94°C for 4 mins 35 cycles of: 94°C for 1 min 62°C for 1 min 72°C for 1 min 72°C for 10 mins	94°C for 4 mins 35 cycles of: 94°C for 1 min 54°C for 1 min 72°C for 1 min 72°C for 10 mins

pmoA

nosZ

Supplemental Material 2. Alpha diversity measures per sample for pmoA (top) and nosZ (bottom). Sample codes indicate drainfield type (Control, Experimental or Reserve), replicate number, sampling depth (Upper, Middle or Lower) and soil type (Bulk or Rhizosphere).

Conclusions

The research findings presented in the first two manuscripts indicate that nearshore coastal groundwater tables are rising more rapidly than can be accounted for by sea level rise alone. Averaged across the southern RI shore, groundwater tables have been rising at a rate of 14 mm/year since the 1960s, with rates ranging up to 17 mm/year in communities that are importing groundwater. These rates affect an estimated 1,500 to 2,000 southern RI OWTS located within 200 m of a coastal waterbody. Groundwater imports, especially in communities which are served by public water supply or individual cisterns storing out-of-basin water, likely contribute to the observed rates of rising groundwater tables along the southern RI coast. In addition, data from 10 drainfields in the same area indicates that 70% experience compromised separation distance at some point in the year, meaning only 30% of systems are not at risk of inadequately treating wastewater before it recharges to the shallow groundwater below. To our surprise, the time since system installation did not predict compromised separation distance; rather, it appears the methods used to determine the original seasonal high groundwater table used to design the system are not sufficiently accurate to ensure proper separation distance.

These findings have important implications. First, it is critical that we re-think how the wastewater community determines the seasonal high water table upon which OWTS designs are based, since current methods seem to be inadequate and fail to account for the observed long-term changes in

groundwater table elevations in the region. Furthermore, it is unlikely that this phenomenon is limited to the southern RI shore, but our studies seem to be the first of their kind to assess separation distance in coastal areas. Another area of concern is the absence of OWTS regulations addressing system resilience in terms of climate-change derived coastal changes, including sea level rise and extreme precipitation and flooding events. To maintain the integrity of OWTS as a critical piece of the water infrastructure in coastal communities, we must begin to be more proactive in the design and installation of systems, to ensure their resilience and robustness to withstand the inevitable challenges rising groundwater tables and climate change will cause in these regions.

Pollution of ground and surface water resources is not the only threat OWTS pose to the environment. My assessment of the greenhouse gas cycling dynamics in surface soils above different drainfield types (Manuscript 3) showed that the surface soil may reduce emissions of N_2O and act as a net sink for CH_4 , as indicated by above-ground flux values that were lower than expected based on the below-ground concentration of these gases. However, the microbial communities responsible for methane and nitrous oxide consumption in these surface soils did not vary among drainfield type. We found that the structure of the methanotrophic community varied with depth, with type I methanotrophs more prevalent in the upper and middle portions of the surface soils. The structure of nitrous oxide reducing communities also varied with depth, as well as between the rhizosphere and surrounding bulk soil.

These preliminary results suggest that the layered passive N-removing drainfield contributes to atmospheric pollution at comparable rates to a conventional drainfield, and more research is required to determine whether this drainfield design represents a promising option at significant cost savings relative to proprietary engineered N-removal systems required in coastal areas.

In general, we need to be more purposeful in considering how OWTS are viewed by the public, municipal, state and government officials, and professionals in the wastewater industry. Septic systems are a critical component of the water treatment infrastructure, protecting both public and environmental health, and will continue to play a major role in the coming decades. Poorly functioning systems (e.g. those with compromised separation distance) contribute to pathogen and nutrient pollution in groundwater and both fresh and coastal surface waters, lowering nearby property values as these resources degrade. Neglecting to consider GHG emissions in analysis of true “cost” or impact these systems have on the environment could inadvertently misrepresent OWTS’ contribution to anthropogenic atmospheric pollution. It is imperative that all stakeholders begin to think ahead and find innovative solutions to handle perturbations and changes resulting from climate change, including changing temperature and precipitation patterns in addition to coastal-related hazards. Part of these solutions must include robust and effective but also affordable innovations that maximize contaminant removal in onsite wastewater treatment.