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Evaluation of Fresh Groundwater Resources and Nitrogen Discharge to the Coastal Lagoon of an Atoll Island

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Abstract

Atoll island communities face socio-economic challenges together with environmental pressures imposed by climate change that are causing increased water quality degradation. Wastewater from domestic septic systems is released into the subsurface on Fongafale Islet, Tuvalu. It is unclear if this is contributing to high nitrogen (N) concentrations and eutrophication of the adjacent coastal lagoon. In this study, a variable-density groundwater flow and conservative contaminant transport model was developed in SEAWAT-2005 to simulate the salinity distribution and N transport through the Fongafale Islet aquifer. Model simulations evaluated the influence of tides, variable (interannual) recharge patterns, wastewater loading rates and wastewater source locations. The configuration of high and low permeability zones across the island strongly influenced the salinity distributions and N transport pathways. Tidal fluctuations were found to considerably increase freshwater-saltwater mixing and as a result reduce the zone of low salinity groundwater in the aquifer. In contrast, the impact of variable recharge was minor. The simulations indicated that maximum exit concentrations of N in the groundwater discharging to Fongafale Lagoon may be higher than concentrations previously measured in the lagoon, but subsurface travel times were found to be sufficiently long for N attenuation (denitrification) to be occurring in the subsurface. The findings contribute new conceptual understanding of processes influencing freshwater-saltwater dynamics and contaminant transport in an atoll island aquifer as needed to inform water quality and wastewater infrastructure programs.

Keywords

Atoll Island, Hydrogeology, Numerical Modeling, Nitrogen Transport, Fresh Groundwater Lens, Tides, Fresh Groundwater Resources, Submarine Groundwater Discharge

Summary for Lay Audience

Many developing small-island communities such as those of atoll islands face environmental and socio-economic challenges due to rapid urbanization and climate change. One consequence of these vulnerabilities is reduced availability and quality of groundwater, which is often an important source of fresh water in atoll islands. Another consequence is the degradation of the surrounding coral reef ecosystem, which is important resource supporting coastal fisheries critical to the livelihoods of atoll island residents. This degradation is a result of human waste and fertilizers entering water resources and causing high input of contaminants such as nutrients into groundwater and adjacent coastal waters. High nutrient levels, particularly nitrogen (N), leads to a process known as eutrophication that causes the excess growth of algae and loss of biodiversity in the marine environment. The sources of N, its groundwater pathways, and concentrations discharging to coastal water of atoll islands remains largely unstudied. Poorly functioning septic systems on the atoll island of Fongafale, Tuvalu have been releasing wastewater into the subsurface and it is unclear if this is contributing N to the lagoon.

A computer model was developed based on Fongafale Islet, Tuvalu to simulate the movement and interaction below the surface between saltwater, freshwater from rainfall infiltrating into the ground, and wastewater and associated N injected into the ground from septic systems. Impacts of tides, rainfall, amount of wastewater entering the ground and septic systems locations were evaluated. It was found that tides increase mixing of freshwater and saltwater preventing the formation of a low salinity zone in the shallow aquifer and lowering N concentration discharging to the lagoon. Septic systems located further from the lagoon also led to low N concentrations in the groundwater entering the lagoon and the longest subsurface travel times. The N exit concentrations found by the model represent an upper value as subsurface reactions were not considered in this study. The findings in this study provide new conceptual understanding of atoll island groundwater processes and contaminant transport that will contribute to future atoll island computer modelling, field studies, and planning and management strategies for wastewater infrastructure and pollution control.

Co-Authorship Statement

The candidate, James Gale, is responsible for the development of numerical model simulations, post-processing results and writing the drafts of all chapters of this thesis. Dr. Clare Robinson provided the initial motivation for this research, suggestions for model simulations, and scripts for post-processing model results. Revisions for improvement of the thesis were also provided by Dr. Clare Robinson and Dr. Sabina Rakhimbekova. Peter Sinclair from the Secretariat of the Pacific Community (SPC) also provided data and advice for the modelling simulations performed in this thesis.

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Chapter 1

1 Introduction

1.1 Background

Atoll islands are small, low-relief islands that are often in remote settings and highly vulnerable to climate events including droughts and coastal flooding (Bailey et al., 2010; Werner et al., 2017). The impact of climate events are exacerbated in atoll island settings by high population densities that can reach up to 12,000 people/km^2 (White & Falkland, 2010). Further, fresh groundwater reserves on many atoll islands are being threatened by salinization due to rising sea levels and by over-abstraction of fresh groundwater to meet the demands of increasing populations (Robins, 2013). Increasing anthropogenic development under these environmental pressures is causing high inputs of contaminants such as nutrients into groundwater and the marine environment (Knee et al., 2016; Slomp & Van Cappellen, 2004). Pollution of groundwater on atoll islands is a major issue as groundwater is often an important potable freshwater source (Bailey et al., 2010; Robins, 2013; van der Velde et al., 2007). Pollution of the coastal lagoon and ocean adjacent to atoll islands is also a major issue as degradation of the marine ecosystem including fish habitats and coral reefs impacts the communities that reside on these islands as they are often heavily reliant on coastal fisheries for their livelihoods (Gillett, 2016; Klassen & Allen, 2017b; McMahon & Santos, 2017; Osawa et al., 2010). It has become well documented that submarine groundwater discharge can be important for fluxes of freshwater and nutrients from terrestrial sources such as sewage and fertilizers to coastal and oceanic island waters (Beusen et al., 2013; Knee et al., 2016; Moosdorf et al., 2015). Improved understanding of nutrient sources and delivery pathways to coastal waters are needed by doing continuous, long-term monitoring together with predictive modeling (Borja et al, 2010). These can help to develop effective management programs and policies to protect coastal and atoll waters.

On atoll islands, a fresh groundwater lens forms when there is adequate volume of fresh recharge that is able to “float” above the more dense saline groundwater (Ketabchi et al., 2014; Underwood et al., 1992). Understanding the dynamics of the freshwater lens in atoll islands is important for predicting the availability of fresh groundwater resources. The size, shape and hydrogeology of an atoll island together with the impacts of transient effects such as interannual recharge patterns, oceanic forcing (tides, storm surges), and sea level rise can all impact the thickness of the fresh groundwater lens (Falkland et al., 1991; Nakada et al., 2012; Underwood et al., 1992; Werner et al., 2017). Oceanic forcing including tides are known to considerably influence the water table elevations, groundwater flow patterns and salinity distribution in continental coastal aquifers, but their impact on atoll island groundwater systems are not well understood despite the high connectivity between the ocean and groundwater system in atoll island settings (Werner et al., 2013). Numerical models developed to predict the fresh groundwater lens on atoll islands typically account for tides by using high dispersivity values that can reproduce the wide freshwater-saltwater mixing zones (Werner et al., 2017). The few studies that have explicitly simulated the effect of tides on the fresh groundwater lens dynamics have only considered generic atoll island settings and have not explored interacting factors (Bailey et al., 2009; Bailey & Jenson, 2014).

In addition to water insecurity, eutrophication of coastal lagoons adjacent to atoll islands is a major issue for atoll island communities (Graves et al., 2021; Osawa et al., 2010; van der Velde et al., 2007). Eutrophication is triggered by high inputs of nutrients, in particular N, into the lagoon. It is well established that submarine groundwater discharge can be important for delivering N from land-based sources (e.g., from wastewater and fertilizer) to coastal waters in continental (mainland) coastal environments (Beusen et al., 2013; Knee et al., 2016; Moosdorf et al., 2015). However, there is limited understanding of the importance of groundwater in delivering nutrients to adjacent coastal waters in atoll island settings including the subsurface delivery pathways. The flux of N from groundwater to the coastal waters is governed by the land-based N contamination source, subsurface transport pathway, travel (residence) time, and geochemical processes along the discharge pathway (Anwar, Robinson, & Barry, 2014; Slomp & Van Cappellen, 2004). While numerous studies have examined N groundwater contamination in continental coastal aquifer systems and factors influencing N delivery to coastal waters via groundwater discharge (eg. Valiente et al., 2018; Wang et al., 2017; Xue et al., 2009), little is known in atoll island settings (eg. Graves et al., 2021; Haßler et al., 2019; Sims et al., 2020). To the best of our knowledge, there is only one prior numerical modelling study in an atoll island setting (unpublished report) that has performed variable-density and contaminant transport simulations investigating the subsurface transport of N in greywater to coastal waters (greywater) (Jazayeri et al., 2019). In this study the influence of greywater inputs was simulated in multiple islands of Tarawa Atoll and transient forcing including tides were not considered.

There are many knowledge gaps regarding fresh groundwater lens dynamics, groundwater contaminant transport and the effects of transient forcing such as tides and variable recharge patterns for the unique atoll island environment. This environment differs from continental coastal environments as atoll island aquifers have high connectivity with the ocean and a distinct multi-layered hydrogeological structure. Currently it is unclear the extent to which existing knowledge can be transferred from continental coastal aquifer settings to atoll island settings. This thesis focuses specifically on conceptualizing the fresh groundwater resources and nutrient subsurface transport pathways on Fongafale Islet, Funafuti, Tuvalu as there is considerable uncertainty on this atoll island regarding the contribution of groundwater to large macroalgal blooms observed in recent years in the coastal lagoon (Ceccarelli, 2018; Fujita et al., 2014, 2013; N'Yeurt & Iese, 2015). Due to the challenges and expense of conducting field investigations in remote locations such as Fongafale Islet, numerical modeling can provide important insights needed to target infrastructure programs such as upgrades to onsite wastewater systems, as well as to inform the design of future field groundwater investigations.

1.2 Research Objectives

To address the research gaps identified above the main objectives of this thesis are to:

1) Evaluate the influence of tides and variable (interannual) recharge patterns on fresh groundwater resources in an atoll island; and

2) Evaluate the subsurface transport of septic system-derived N in an atoll island groundwater system and its delivery to adjacent coastal waters including impact of tides, variable recharge patterns, wastewater loading rate and wastewater source locations.

These objectives were addressed by conducting variable-density groundwater flow and N transport simulations in SEAWAT-2005 (Guo & Langevin, 2002) that were based on conceptualization of the hydrogeological and hydrological conditions for the atoll island of Fongafale, Tuvalu. The knowledge generated is needed to inform water quality management and infrastructure programs aimed at protecting fresh groundwater resources as well as the coastal waters that surround atoll islands.

1.3 Thesis Outline

This thesis is written in “Integrated Article Format.” A description of each chapter is outlined below.

Chapter 1: Introduces background information on the topic and states the research objectives of the study.

Chapter 2: Describes the past work and knowledge of groundwater dynamics on atoll islands as well as N contamination and numerical modelling studies of both coastal and atoll island aquifers. A description of the study area (Fongafale Islet, Tuvalu) is also provided.

Chapter 3: Details the numerical model development, simulation results and provides discussion of the study findings focused on salt and N transport in atoll island groundwater systems.

Chapter 4: Summarizes the research findings and outlines recommendations for future work.

Appendix: Appendices are included to supplement the methods, and numerical model simulation results presented in Chapter 3.

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Chapter 2

2 Literature Review

2.1 Introduction

This Chapter provides a review of prior research of atoll island hydrogeology including freshwater-saltwater groundwater dynamics and the transport of contaminants including nitrogen (N) in atoll groundwater systems. Atoll islands typically have a unique hydrogeological structure compared to mainland coastal aquifer systems and this can result in distinct flow patterns and freshwater-saltwater dynamics. Current understanding of groundwater flows and freshwater-saltwater dynamics in atoll systems and how they may impact contaminant including N transport is explored. As limited studies have previously examined N groundwater contamination and transport in atoll islands, the review also includes prior studies that have examined N contamination and transport in mainland coastal aquifer systems. Finally, background information for the study area Funafuti, Tuvalu is provided.

2.2 Atoll Islands

Coral atolls are a type of carbonate island in which coral reef sediments grow upwards on a subsiding volcanic island or seamount (Falkland et al., 1991). They are often small, low-relief islands that surround a lagoon in a disconnected or fully connected ring-like shape. The geologic stages that are thought to produce atoll islands from volcanic islands are shown in Figure 2.1. These islands exist in a variety of shapes and sizes but are typically between 100-1500 m in width and many kilometres in length (Werner et al., 2017). As a result, the land area is often less than a square kilometre and the maximum elevation of an atoll island is generally only a few metres above sea level (Werner et al., 2017).

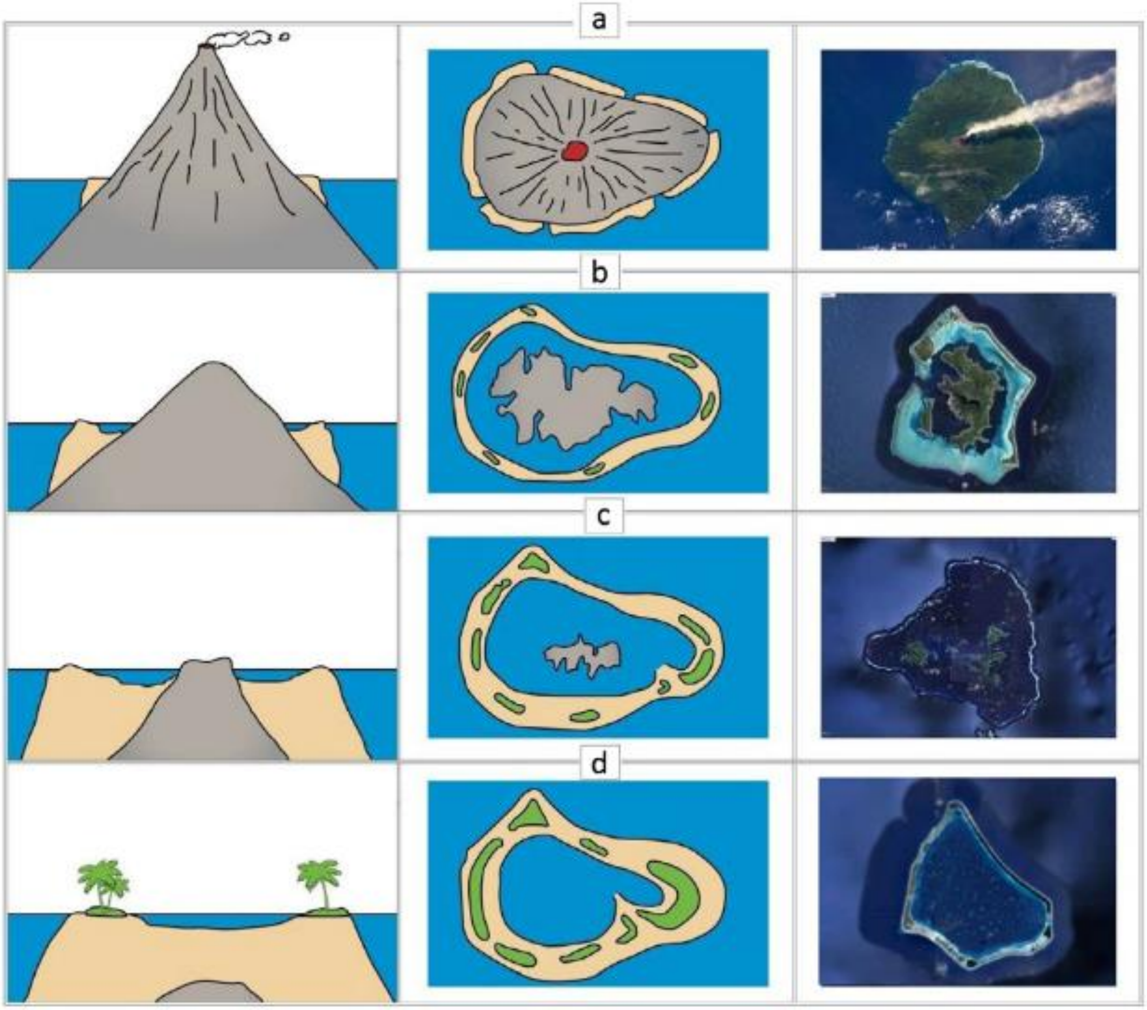

Figure 2.1. Coral atoll historical geologic stages: (a) Volcanic Island formation, (b) Settling of the volcanic core, (c) Further volcanic core settling with fringing reef growth, (d) Final ring-like structure. Schematic illustrations of the cross-section view are shown in the left figures, the plan view in the middle figures, and the overhead photographs on the right (Figure reproduced from Werner et al., 2017).

Atoll islands exist mainly in the Indian and Pacific Oceans and total over 400 worldwide (Bailey et al., 2010). These low-lying islands are often in remote settings and as such are particularly vulnerable to climate events including droughts and flooding, as well as over-use of fresh water supply (Bailey et al., 2010). The pressures of the changing climate are often exacerbated by high populations on these small islands with population densities reaching up to 12,000 people/km^2 (White & Falkland, 2010). Many atoll islands are recognized politically and economically in the subset of small island developing states (SIDS) which includes 38 UN member states within the relatively broad definition of SIDS (van der Velde et al., 2007). Many atoll island SIDs are highly vulnerable as the small physical and socio-economic size of atoll islands combined with their remote locations means that they often have limited capacity for adequate management both internally and imposed externally (van der Velde et al., 2007).

The socio-economic challenges faced by many atoll island communities together with environmental pressures imposed by climate change are resulting in increasing water quality challenges (Graves et al., 2021). Fresh groundwater reserves in atoll islands are often an important source of household water (drinking, bathing, washing) (Nakada, et al., 2012). Therefore securing and managing fresh groundwater lens is a necessity for many atoll island populations, their ecosystems and economies (Bailey et al., 2010; Graves et al., 2021; White & Falkland, 2010). Fresh groundwater reserves are being threatened by salinization due to climate change (e.g., sea level rise, seawater inundation events, droughts) as well as over-abstraction of fresh groundwater to meet the demands of increasing populations (Robins, 2013). The fresh groundwater reserves of densely populated islands are also being threatened by anthropogenic contaminants including those associated with waste disposal and wastewater discharge (M. Fujita et al., 2013). The role of groundwater in water management plans for atoll islands is an important topic that requires further attention to ensure management, programs and policy are effective at safeguarding fresh groundwater reserves (Werner et al., 2017).

The integrity of the lagoons and oceans that surround atoll islands is also critical for the livelihood of the communities that inhabit the islands with islanders often relying on coastal fisheries for their livelihoods (Gillett, 2016). Nutrients from sources including agricultural activities and wastewater are increasingly impairing the water quality and ecosystems of these marine waters (Graves et al., 2021). For example, fish numbers have been declining due to high nutrient inputs and consequent algal blooms along the Coral Coast, Fiji (Tamata, 2007) and in Fanga'uta Lagoon, Tonga (Morrison et al., 2013). Improved understanding of the relative contributions from the various anthropogenic nutrient sources as well as the pathways via which nutrients are being delivered to the coastal waters (e.g., overland or subsurface pathways) are needed to develop effective management programs and policies to protect the coastal waters. This understanding may be gained by continuous, long-term monitoring together with predictive modeling approaches (Borja et al, 2010).

2.2.1 Hydrogeology

The general hydrogeology of coral atoll islands is relatively similar between islands and is generally based on the time periods in which the geologic processes related to the formation of a given atoll occurred (Ayers & Vacher, 1986). Atoll islands typically consists of two primary aquifer layers that were formed during the Holocene and Pleistocene periods. This dual aquifer system comprises of poorly or unconsolidated Holocene sediments that overlie Pleistocene limestone reef deposits as shown in Figure 2.2 (Bailey, 2008). The hydraulic conductivity (K) of the Pleistocene limestone layer is typically between 1 to 2 orders of magnitude higher than that of the Holocene sediments (Bailey, 2008). Generally, the Holocene sediments have K values in the range of 5 to 20 m/d. In contrast, the K of the Pleistocene limestone generally increases with depth and is the range of 10 to 20 m/d near the Holocene sediments to greater than 100 m/d deeper in the aquifer (Post et al., 2018). Between the two layers there is an unconformity which is typically about 15-25 m below sea level and is known as the 'Thurber Discontinuity' (Werner et al., 2017). This dual aquifer formation has been seen in many atoll islands and is a widely recognized conceptual hydrogeological model commonly used for these islands (Post et al., 2018).

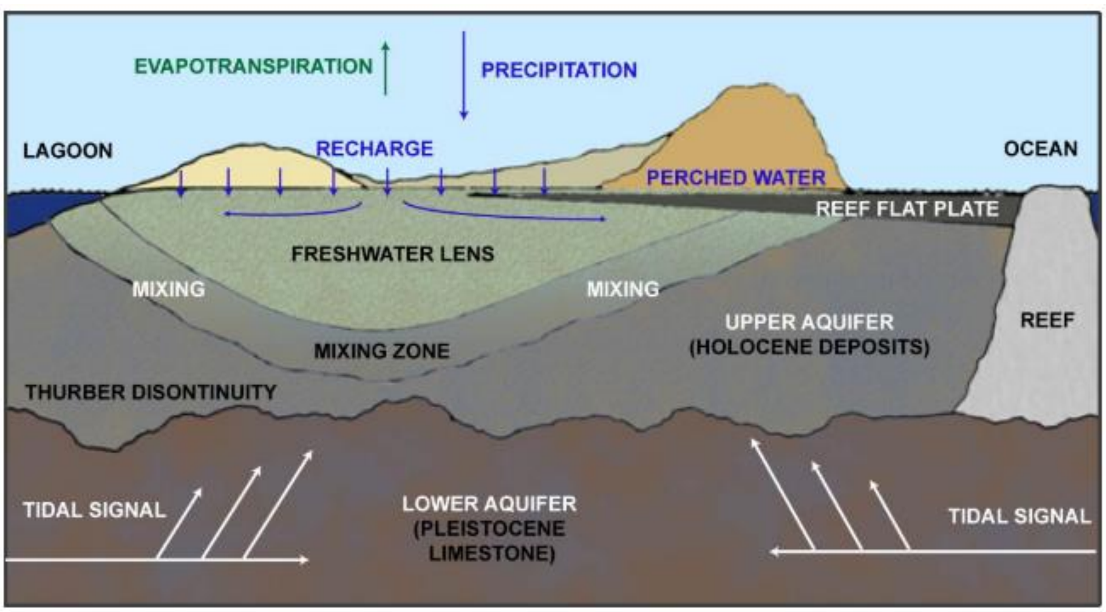

Figure 2.2. Conceptual model of atoll island hydrogeology (Figure reproduced from Ayers & Vacher, 1986).

Beyond this general dual aquifer conceptual hydrogeological model of atoll islands, the subsurface is often quite heterogeneous. This is because karstification processes and cementation of aging calcium carbonate can lead to decreased permeability in Holocene sediments with localized high hydraulic conductivity fissures (A.C. Falkland et al., 1991). The unconsolidated Holocene sediments can also be surrounded by horizontal sublayers with reduced porosity and permeability that may originate from cementation and buried beach rock or reef flat (Underwood, 1990). Some atolls can have moderately permeable reef-flat platforms near the ocean shoreline which have been exposed over time as the sea level has risen and they continue to expand laterally (Figure 2.2) (Woodroffe, 2008). The presence of these reef-flat platforms can in some cases lead to a confining of the Holocene aquifer which changes the groundwater dynamics in this aquifer unit (Bailey et al., 2009).

The surficial permeability across atoll islands can vary with lower permeability sediments often deposited on the lagoon side and more permeable sediments deposited on the ocean side of the atoll (Falkland et al., 1991; Underwood, 1990). With sediments building up on both the lagoon and ocean shoreline, the interior of an atoll island is often of lower elevation compared to closer to the shorelines with the ground surface in the interior of the island being near sea level (Chui & Terry, 2013). The surficial sediment in the lower-lying interior areas of atoll islands are often saturated and consist of dark, rich soils compared to the surrounding surficial sediment (Webb, 2007). These unique characteristics of the surficial permeability across coral atolls can influence the distribution of freshwater and seawater in the aquifer including creating a distinct freshwater lens shape (Figure 2.2). This is discussed further in the following sections.

2.2.2 Freshwater-saltwater groundwater dynamics

The size, shape and hydrogeology of an atoll island together with the amount of recharge (rainfall) are key factors in determining the presence and quantity of fresh groundwater resources on the island (Falkland et al., 1991). Another important factor is the density difference between the freshwater and seawater (Underwood et al., 1992). This is because a fresh groundwater lens is able to form when there is adequate volume of fresh recharge that is able to “float” above the more dense saline groundwater (Werner et al., 2017). Freshwater for household consumption is generally considered to be water that is less than 2.5% seawater concentration (R. W. Buddemeier & Oberdorfer, 1986).

The location of the freshwater-saltwater interface in a coastal aquifer can be approximated using the Ghyben-Herzberg relationship (Drabbe & Badon Ghijben, 1888; Herzberg, 1901) which assumes that the interface is sharp and that the freshwater and saltwater are immiscible (Underwood et al., 1992). However, use of this relationship is overly simplified particularly in atoll island aquifers as the width of the freshwater-saltwater mixing zone, which is commonly defined as between 2.5% to 95% seawater concentration, can often comprise much of the subsurface in atoll islands (White & Falkland, 2010). White & Falkland (2010) observed that the average thickness of the mixing zone reduces as the aquifer recharge and island width increase and the hydraulic conductivity and groundwater pumping decrease. The thickness of the fresh groundwater lens and mixing zone are also impacted by transient effects including variable recharge patterns (seasonal and longer-term), oceanic forcing including tides and storm surges, and longer-term changes including sea level rise (Werner et al., 2017; Nakada et al., 2012).

The common dual-aquifer configuration of atoll islands (discussed in Section 2.2.1) considerably influences the size of the fresh groundwater lens and has been shown to increase the thickness of the mixing zone compared to more homogeneous coastal aquifers (Bailey et al., 2010). For instance, the presence of the high permeability Pleistocene limestone layer can increase freshwater-saltwater mixing by allowing the tidal signal to penetrate further into the aquifer and this in turn can limit the size of the fresh groundwater lens (Werner et al., 2017). On the other hand, the surficial lower hydraulic conductivity Holocene sediments can retain more freshwater in the aquifer system and this can lead to a larger fresh groundwater lens (Bailey, 2008).

The presence of a reef-flat plate can also influence the size and configuration of the fresh groundwater lens in atoll islands. This is because it can act as a confining unit in the Holocene sediments and as such may prevent freshwater from discharging near the ocean shoreline. This in turn can lead to a thickening of the fresh groundwater lens across the island (Bailey et al., 2009). However, in some cases the reef-flat plate with its low hydraulic conductivity can also impede aquifer recharge which in turn can limit the size of the fresh groundwater lens (Ayers & Vacher, 1986). The sediment size distribution across coral atolls with finer sediments generally found on the lagoon shore and coarser sediment near the ocean shore can also impact the configuration of the fresh groundwater lens (Werner et al., 2017). This distribution can result in a fresh groundwater lens that is asymmetrical with the lower hydraulic conductivity sediment near the lagoon side resulting in a thicker lens on this side of the island compared to the ocean side (Buddemeier & Oberdorfer, 2004; Falkland & Woodroffe, 2004; Falkland, 1992).

The magnitude and temporal variation (seasonal and interannual) of aquifer recharge is another important factor controlling the size of the fresh groundwater lens and the thickness of the mixing zone in atoll islands. In the absence of evapotranspiration and storage data, some prior studies have approximated recharge on atoll islands to be 50% of the mean annual rainfall (Ghassemi et al., 1998; Hamlin & Anthony, 1987; Jazayeri et al., 2019). Rainfall is often able to rapidly infiltrate into the atoll island subsurface since the soil cover is typically thin and open geologic features such as fissures may exist (Falkland et al., 1991). When adequate data is available, recharge can be more accurately determined using a water balance analysis which considers the rainfall, evapotranspiration and storage change in the unsaturated zone (Falkland & Woodroffe, 2004). Surface runoff is usually excluded in the water balance analysis for atoll islands as runoff is generally negligible due to the relatively flat topography and high infiltration capacity (Falkland, 1994; Lloyd et al., 1980).

2.2.3 Effect of tides on groundwater dynamics

Average annual recharge rates across atoll islands vary widely. For instance, mean rainfall ranges from over 3500 mm/year at Funafuti atoll, Tuvalu to less than 1000 mm/year on Kirimati, Kiribati (White & Falkland, 2010). Rainfall and thus recharge rates for atoll islands are also typically highly variable over seasonal and interannual scales (e.g., due to El Niño-Southern Oscillation (ENSO)) and this can influence the size of the fresh groundwater lens (Oberle et al., 2017). For instance, Oberle et al. (2017) showed using field monitoring how the El Niño-Southern Oscillation (ENSO) and seasonal and interannual rainfall variability including prolonged drought periods impacted the size of the fresh groundwater lens on Roi-Namur atoll.

Oceanic forcing including tides can considerably influence the water table elevations, groundwater flow patterns and salinity distribution in coastal aquifers including atoll island aquifers (Werner et al., 2013). The propagation of the tidal signal through aquifers has been well studied and it is well established that the tidal signal, which can be observed through fluctuating groundwater heads, becomes increasingly damped (smaller amplitude) and delayed (increasing time lag) as it propagates landward (Ferris, 1951). Tidal lag time refers to the difference in time between the peaks or troughs of the ocean tide compared to the groundwater level fluctuations observed at a distance inland from the shore (Werner et al., 2017). A variety of analytical solutions have been developed to describe tidal-induced fluctuating groundwater levels in homogeneous single layer aquifers (e.g., Jacob, 1950; Nielsen, 1990; Song et al., 2007; Turner et al., 1997). Additionally, solutions have been developed to describe tidal water table fluctuations in multilayer coastal aquifers (Jiao & Tang, 1999), submarine confined aquifers (Guo et al., 2007; Li & Chen, 1991) and submarine leaky confined-unconfined aquifer systems (Li & Jiao, 2001).

Tide-induced head fluctuations and thus hydraulic gradient fluctuations result in oscillatory landward and seaward directed flows in coastal aquifers (Robinson et al., 2007). This in turn influences the extent of freshwater-saltwater mixing as well as water exchange rates across the groundwater-ocean interface. Tidal effects can also influence transport pathways for groundwater contaminants in coastal aquifers and the concentrations of contaminants at the groundwater-ocean interface (i.e., exit concentrations) (Robinson et al., 2007). Finally, tides can also lead to the formation of a shallow seawater circulation zone close to the groundwater-ocean interface that can affect geochemical conditions and affect the ultimate discharge of groundwater contaminants to the ocean (Robinson et al., 2009; Robinson et al., 2007).

While most prior studies examining tidal effects on groundwater dynamics have focused on continental (mainland) coastal aquifers, some studies have investigated the tidal effects for atoll island aquifers. Due to the unique hydrogeology of atoll islands, tidal signal propagation is often non-uniform and complex (Oberdorfer, Hogan, & Buddemeier, 1990). The tidal signal can propagate rapidly through the high permeability Pleistocene aquifer and this in turn can lead to rapid propagation of tides upwards through the upper Holocene layer and oscillatory enhanced freshwater-saltwater mixing (Hunt & Peterson, 1980; Wheatcraft & Buddemeier, 1981; White & Falkland, 2010). Oberdorfer et al. (1990) reviewed field data from multiple studies and found that tidal fluctuations on atoll islands contribute to large a freshwater-saltwater mixing zone around the fresh groundwater lens. Further they found that the dampening and time delay of the tidal signal are essentially independent of distance from the shoreline but tend to increase with aquifer depth. Tidal effects have previously been accounted for in numerical groundwater models of atoll islands by using a transverse dispersivity twenty times that compared to if tides were explicitly simulated in the model (Werner et al., 2017). Few studies (Alam et al., 2002; R. T. Bailey et al., 2009; Ryan T. Bailey & Jenson, 2014) have explicated included tides in numerical groundwater simulations although their effects on freshwater-saltwater mixing and contaminant transport are likely to be considerable.

2.3 Nitrogen contamination

Fresh groundwater lenses on atoll islands are vulnerable to water quality degradation by anthropogenic activities and natural processes as there is minimal protection from infiltration due to a thin, permeable unsaturated zone (Robins, 2013). The high population density in many atoll islands also means that there is high potential for anthropogenic contaminants to impair groundwater quality (Werner et al., 2017). Both nitrogen (N) and phosphorus (P) are present in saltwater and freshwater environments and the optimal ratio for phytoplankton growth is known as the Redfield ratio at 16:1 (Slomp & Van Cappellen, 2004). Most coastal ecosystems have an N/P ratio lower than the Redfield ratio which indicates that marine waters are typically N-limited (Bowen et al., 2007). As such, high inputs of N are of key concern as they can impair fresh water supplies and lead to eutrophication of coastal waters. Eutrophication can degrade coastal marine ecosystems by promoting algal blooms, hypoxic conditions, and loss of habitats and reduced biodiversity (Anwar et al., 2014; Camargo & Alonso, 2006).

To safeguard fresh water supplies and prevent eutrophication of coastal waters, it is crucial to understand the sources, pathways and transformations of N in surface and groundwater systems (Rodellas et al., 2018). While a large number of studies have examined the sources, transport and transformations of N in continental coastal environments, few studies have examined N sources, transport and transformations in reef islands including atoll islands and subsequent N loading into coastal waters (Moosdorf et al., 2015). The following sections examine N sources, forms and transformation of N in groundwater, fate and transport of N in coastal aquifers, and delivery of N to coastal waters via submarine groundwater discharge (SGD).

2.3.1 Sources of nitrogen

Development of coastal areas for residential and agricultural activity is causing the high input of contaminants such as nutrients (N and P) into surface water and groundwaters (Slomp & Van Cappellen, 2004). Nitrogen in coastal waters is supplied primarily by “new” N production from both natural and anthropogenic sources (Paerl, 1997). Natural sources include weathering of minerals, decomposition, lightning, atmospheric deposition, and geothermal emissions. Anthropogenic N sources include wastewater and nonpoint sources such as fertilizers and animal waste (Paerl, 1997). Anthropogenic sources are often the main cause of N contamination of groundwater (Falkland et al., 1991; Fazal et al., 2003; Robins, 2013). Developing, rural and geographically remote communities such as those on many atoll islands often use decentralized on-site wastewater treatment systems including septic systems. While septic systems provide partial treatment of wastewater, N, commonly present as both ammonium (NH_4^+) and nitrate (NO_3^-), is released into the subsurface below a septic drain field (Robertson et al., 2021). In addition, onsite wastewater treatment systems including septic systems are known to fail in various ways and these systems can provide minimal treatment of wastewater with high N loads released into groundwater below the drain field (Withers et al., 2014).

2.3.2 Nitrogen forms and transformations in groundwater

The common forms of N in groundwater include ammonium (NH_4^+), nitrite (NO_2^-), nitrate (NO_3^-), soluble organic N, and N associated with sediment as exchangeable NH_4^+ or organic N (Camargo & Alonso, 2006). N cycles between its different forms with complex processes, mostly microbially mediated, controlling the form of N (Morrissy et al., 2021). Organic N can be transformed to NH_4^+ through a process called ammonification. Under oxic conditions, when dissolved oxygen levels are as low as 1 mg O_2/L, NH_4^+ can be oxidized to NO_3^- in a two-step nitrification process ($NH_4^+ \rightarrow NO_2^- \rightarrow NO_3^-$). NO_3^- can be transformed back into organic N by being taken up by plants including macrophytes, algae and bacteria (Camargo & Alonso, 2006). N can also be removed from groundwater by denitrification, where NO_3^- is transformed to N_2O (nitrous oxide gas) and N_2 (nitrogen gas). Finally, N can also be removed from groundwater by anammox where NH_4^+ reacts with NO_2^- to produce N_2. NO_3^- is typically the most abundant form of N in groundwater as it is highly soluble, mobile and does not adsorb to sediment (Brookfield et al., 2021). As such NO_3^- can also migrate large distances through aquifer and is often the dominant form of N delivered from groundwater to surface waters including oceans (Dubrovsky et al., 2010). Some factors that can control the degree of denitrification and concentrations of NO_3^- in groundwater include the hydrogeologic setting, the thickness of the unsaturated zone, the length of the flow paths, redox conditions and organic content (Tesoriero et al., 2015).

2.3.3 Nitrogen behavior in coastal aquifers

A large number of field and numerical modeling studies have evaluated the occurrence, transport and transformation of N in continental (mainland) coastal aquifers (Valiente et al., 2018; Wang et al., 2017; Xue et al., 2009). The highest N concentrations in groundwater are often found in shallow coastal aquifers with nearby anthropogenic N sources. Dissolved N concentrations in coastal aquifers vary depending on the source input rate, aquifer type and permeability, groundwater recharge rate, and the climate (Slomp & Van Cappellen, 2004). As N is transported through the aquifer towards the ocean various transformation may occur depending on the geochemical conditions in the aquifer. Important N transformation can occur in the zone where the discharging terrestrial (fresh) groundwater mixes with seawater (Oehler et al., 2021). This zone of mixing is commonly referred to as a subterranean estuary in recognition of the important influence it can have on chemical species concentrations including N species (Moore, 1999). The flow patterns, residence times, and redox conditions are important factors controlling the extent of N removal that may occur in a subterranean estuary (Slomp & Van Cappellen, 2004). Net N removal by denitrification has been observed to occur in some subterranean estuaries (Erler et al., 2014; Loveless & Oldham, 2010; Weinstein et al., 2011), whereas net N production (primarily due to mineralization of organic matter) has been observed to occur in other subterranean estuaries (Robinson et al., 2018). Montiel et al. (2019) applied a nutrient (N and P) mass balance, stable isotopes and other analyses to examine nutrient sources and biogeochemical transformations in a subterranean estuary adjacent to Mobile Bay, Alabama, USA. They found the processes that removed anthropogenic N were denitrification and reduction to ammonium, and hypothesized that similar biogeochemical transformations may occur in other shallow subterranean estuaries worldwide (Montiel et al., 2019). While many unconfined coastal aquifers may be relatively oxic and therefore may not have the necessary conditions for denitrification, there can still be partial NO_3^- removal particularly in aquifers where the NO_3^- residence time in the aquifer is long (Rivett et al., 2008).

2.3.4 Nitrogen contamination of atoll islands

Nitrogen contamination of groundwater and the transport of groundwater N to coastal waters in atoll island settings is not well understood (Werner et al., 2017). A few recent field studies have examined N concentrations in groundwater on atoll islands and other types of volcanic islands.

These studies are described in this section. Jones et al. (2011) conducted field monitoring on the coral atoll island of Bermuda and observed widespread N contamination of the fresh groundwater lens caused by on-site wastewater disposal systems. Their data indicate that high levels of dissolved inorganic N are discharging through the highly conductive limestone aquifer layer to coastal waters with limited residence time or suitable geochemical conditions for natural attenuation of N (e.g., denitrification) to occur. More recent sampling of groundwater wells in Bermuda conducted by Sims et al. (2020) observed NO_3^- concentrations to decrease with aquifer depth. The data suggest that the decrease in concentrations with depth may be due to dilution (mixing) rather than N removal processes (e.g., denitrification). Localized elevated NO_3^- concentrations (up to 60 µmol NO_3^-/L in groundwater and up to 44 µmol NO_3^-/L in submarine fresh groundwater discharge) have also been observed in the barrier reef islands of Tahiti and Moorea in low-elevation areas that have higher recharge (Haßler et al., 2019). Although these islands are geologically similar, it is thought that they have different processes that govern the transport of N to the adjacent coastal waters with data suggesting denitrification may be limited in the Tahiti groundwater system (Haßler et al., 2019). On Heron Island, Great Barrier Reef, Australia, a low N:P ratio was observed in the groundwater with radon-222 tracing of groundwater transport suggesting that the low N:P ratio may be due to high denitrification rates combined with dilution due to mixing (McMahon & Santos, 2017). From the available data there were no estimated groundwater denitrification rates, but some of the highest denitrification rates in coastal systems worldwide have been reported for the Heron Island lagoon carbonate sediments (McMahon & Santos, 2017).

A recent study by Graves et al. (2021) conducted a review of pressures on the ecosystem of the Pacific Atoll South Tarawa, Kiribati and found that septic systems and other waste sources continue to contaminate groundwater despite recent efforts to improve human waste disposal. They speculated that groundwater may be an important pathway delivering N to the lagoon as groundwater recharge for South Tarawa is high with rainfall amounts being considerably greater than evapotranspiration. Graves et al. (2021) indicated the need for more monitoring and process-focused studies of groundwater nutrient contamination to aid Kiribati in decision making regarding human waste disposal infrastructure and management. Overall, while these field studies described able provide some insight into N contamination and transport in atoll islands, information is generally limited. This is in part due to the challenges in conducting groundwater field studies on atoll islands due to their remote location and challenges in drilling groundwater wells without large drilling equipment. Given the challenges in obtaining field data, there is need for numerical groundwater modeling studies that are able to conceptualize groundwater N fate and transport in atoll islands (Jazayeri et al., 2019; Werner et al., 2017).

2.3.5 Importance of groundwater in delivering nitrogen to coastal waters

Over the past 30 years it has become well recognized that groundwater discharge can be an important source of nutrients to coastal waters (Moosdorf et al., 2015). Groundwater discharge to oceans is commonly termed submarine groundwater discharge (SGD). SGD includes terrestrial groundwater discharging to the ocean as well as seawater that is recirculating across the groundwater-ocean interface (Burnett et al., 2003). Estimates of N fluxes to coastal waters via SGD have varied over orders of magnitude, are generally highest for aquifers with anthropogenic N contamination, and have been shown to be regionally comparable to river inputs based on quantitative studies conducted for the U.S. and Australia (Slomp & Van Cappellen, 2004). The main N species in SGD were found to be NH_4^+ and dissolved organic nitrogen in a recent global review of over 200 study sites (Santos et al., 2021). In addition, SGD N fluxes were found to be higher than nearby river inputs in about 60% of study sites (Santos et al., 2021).

While a large number of field studies have measured N inputs to the ocean via SGD, few studies have been conducted in reef islands including atoll island environments. This is in spite of SGD from small islands being recognized to impact coastal ecosystem health including affecting the important fisheries of developing small-island nations (Santos et al., 2021; Werner et al., 2017). For instance, Paytan et al. (2006) suggested that SGD may be an important input of N to coral reef environments even when there is minimal fresh groundwater flow. It has been reported that the lagoon of Tongatapu (Tonga) is eutrophic where SGD occurs, but the source of N contamination was uncertain (van der Velde et al., 2007). McMahon & Santos (2017) showed SGD delivers high N inputs to the lagoon near Heron Island, at low tide and suggested further investigations are required to understand the impacts of these N inputs on the coastal ecosystem. More recently, SGD N inputs (at least partially sewage-derived) to Garapan Lagoon were found to be high year-round (around 60% of production) and high seasonally to Tanapag Lagoon in Saipan, Northern Mariana Islands (Knapp, Geeraert, Kim, & Knee, 2020).

2.4 Numerical modelling of salt and nutrient transport in coastal aquifers

Numerical groundwater flow and contaminant transport modeling has been used previously to investigate salt and N transport in coastal aquifers due to the challenges of field monitoring in coastal aquifer systems as well as to untangle the complex interacting flow, transport and reactive processes (e.g. Anwar et al., 2014; Kim et al., 2013; Robinson et al., 2009). Compared to analytical solutions, numerical models are able to include a wide variety of spatial and temporal boundary conditions and complex model domains (Werner et al., 2017).

Numerical models are commonly used to simulate variable-density flow and multi-species transport in coastal aquifers. Modeling codes commonly used include SUTRA (Provost & Voss, 2019) and SEAWAT (Guo & Langevin, 2002) for variable-density groundwater flow, which are coupled with MT3DMS for multi-species transport (Anwar et al., 2014; Kim et al., 2013; Robinson et al., 2009). SEAWAT, which is an extension of MODFLOW, has been widely used to investigate variable density flow including the effects of climate change, climate variably and well abstraction rates on the fresh groundwater lens in coastal aquifers including in atoll island settings (Jazayeri et al., 2019; Post et al., 2018, Werner et al., 2017). Prior models employed to simulate the fresh groundwater lens in atoll islands have used constant-head boundary conditions to represent the ocean and lagoon and do not explicitly simulate tidal forcing (Bosserelle et al., 2015; Jazayeri et al., 2019). In numerical groundwater models of atoll islands, tidal effects have typically been accounted for by using high dispersivity values that are able to reproduce the wide freshwater-seawater mixing zones (Werner et al., 2017). The few studies that explicitly studied tidal effects on atoll islands only used a generic conceptualization of the atoll island setting and provide limited sensitivity analyses and discussion of the impacts of tides (Bailey et al., 2009; Bailey & Jenson, 2014). Nevertheless, the effects of tides of freshwater-saltwater dynamics have been explicitly simulated in other coastal aquifer types (homogeneous, multiple aquifer layers etc.) and are found to considerably affect the groundwater flow patterns, mixing, and water exchange rates across the groundwater-ocean interface. Tidal forcing acting on coastal aquifers has been explicitly simulated within SEAWAT using the Periodic Boundary Condition (PBC) Package (Kim et al., 2017; Post, 2011).

To our knowledge, only one study (unpublished report) has previously simulated contaminant multi-species transport in an atoll island setting. This study by Jazayeri et al. (2019) performed variable density and multi-species transport simulations of N (greywater) at multiple islands of Tarawa atoll using SEAWAT. However, the effects of oceanic forcing including tides were not considered in this modelling study but were recommended as an important factor to consider in future studies (Jazayeri et al., 2019) A number of studies have simulated nutrient (N and P) transport in other coastal aquifer types, often considering the aquifer to be homogeneous (Meile et al., 2010; Okuhata et al., 2021; Spiteri et al., 2008). For instance, Spiteri et al. (2008) conducted density-dependent reactive nutrient transport simulations in a homogeneous coastal aquifer and showed that biogeochemical reactions occurring in coastal aquifers, particularly in the freshwater-saltwater mixing zone (e.g., denitrification and phosphate adsorption) had a large impact on the ultimate delivery of terrestrial groundwater nutrients to the ocean. Anwar et al. (2014) extended this numerical study to investigate the effect of tides on nutrient transport in coastal aquifers and showed that tides significantly altered N transformations in the coastal aquifer due to the enhanced mixing between the freshwater and seawater. Tides were also shown to increase the residence times for terrestrial nutrients discharging through the coastal aquifer and considerably reduced the N concentrations at the groundwater-ocean interface (e.g., exit concentrations) (Anwar et al., 2014). More recently, a density-dependent groundwater model was developed for the Keauhou basal aquifer, Hawaii, USA to assess groundwater flow and nutrient transport (Okuhata et al., 2021). In this study, it was found that nutrient inputs from urban areas did not considerably change groundwater N and P concentrations, but when water fluxes were decreased and saltwater intrusion increased, N and P concentrations increased across the aquifer system (Okuhata et al., 2021).

2.5 Study area description: Funafuti, Tuvalu

Tuvalu is a small atoll island nation located in the South Pacific Ocean between Australia and Hawai'i that consists of three reef islands and six atoll islands which all total to approximately 26 km^2 (Figure 2.3a and b) (Ceccarelli, 2018). The area of interest for this study is the capital of Tuvalu, Fongafale Islet, located on the eastern side of Funafuti Atoll (Figure 2.3c). The population of Fongafale Islet is 6,716 with the total population of Tuvalu being only 10,645 (Central Statistics Division of Tuvalu, 2017). Fongafale Islet and the roughly 40 other small islands around Funafuti atoll surround a central lagoon that reaches about 50 m deep (Fujita et al., 2014). The atoll is very low lying with the highest point only 4.6 m above sea level. This means that the population around the atoll are highly vulnerable to climate change effects, particularly sea level rise and storm surges. Substantial landform changes were engineered across Fongafale Islet around 1942-1943 by the U.S. Corps of Engineers during the WWII Pacific Campaign (Webb, 2007). The effects of these engineered landform changes continue to impact the Fongafale Islet environment, hydrology and coastal instability, with the open borrow pits and the airport runway occupying considerable space on the small island (Webb, 2007). Landform changes including reclamation of swamp areas in the center of the Island was completed using permeable coral blocks (Fujita et al., 2014). Shells of various species of large benthic foraminifers make up roughly 40% of the land and beach sediments on Fongafale Islet while most carbonate sediment gets deposited on the reef flats (Collen & Garton, 2004). Rainfall and groundwater are the only sources of freshwater on Fongafale Islet and the rainfall is not reliable and fresh groundwater resources are often brackish and depend on recharge from rainfall (Thaman et al., 2016). The environment, including freshwater resources, are highly stressed on Fongafale Islet due increasing pressures from new development, high population density, and the legacy of historical landform and coastal engineering (Webb, 2007).

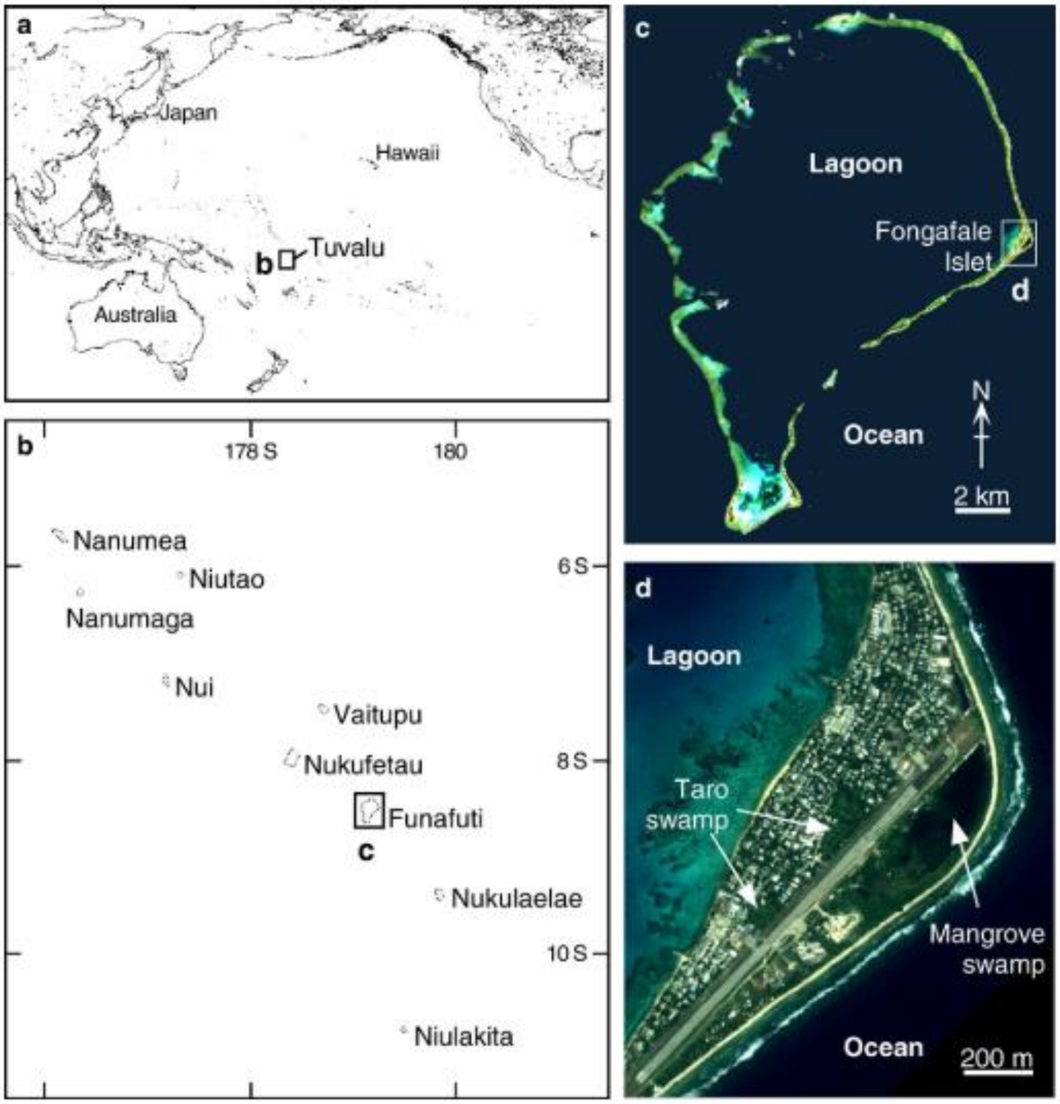

Figure 2.3. **(a) Location of Tuvalu in the Pacific Ocean, (b) Location of Funafuti Atoll within Tuvalu, (c) Location of Fongafale Islet on Funafuti Atoll, (d) Fongafale Islet with locations of Taro Swamp and Mangrove swamp (Figure reproduced from Yamano et al., 2007).**

Tuvalu exists in a tropical climate where the El Niño/Southern Oscillation Index (ENSO) plays an important role in the occurrence of the dry season (May-October) and the rainy season (November-April) which is also cyclone season (Ceccarelli, 2018). Funafuti Atoll has a low temperature variability with the mean daily temperature ranging from 28 to 31 °C and the average lagoon water temperature is 29.4 °C ± 1 °C (Fujita et al., 2014). The precipitation is approximately 3000 mm/year with 60% of the rain occurring during the rainy season (Fujita et al., 2014). Fongafale Islet experiences surface water flooding caused in part by rising sea levels but also by saline groundwater breaking out and flooding at the ground surface during spring tides, known as "King Tides" (Nakada et al., 2012). Flooding has also been reported in the low-elevation areas of the Island that were reclaimed for construction of the airfield and taro swamps (Figure 2.3d) (Nakada et al., 2012).

These geographical and climate conditions of Tuvalu together with the high population density and low-socio economic conditions create a vulnerable environment for the contamination of the groundwater and coastal waters. Firstly, Fongafale Islet has a limited fresh groundwater lens in part due to the narrow width of the island, but also because of the natural hydrogeological conditions and due to the use of highly permeable, porous coral gravel in the reclamation of the taro swamps (Nakada et al., 2012). Further, most households in Tuvalu use poorly constructed or maintained septic tanks as their wastewater systems and many of these are thought to be leaking or experience overflow whereby the wastewater constituents including N may have contaminated the groundwater (Ceccarelli, 2018). Since 2011, macroalgal blooms have been observed in the Funafuti lagoon with their occurrence positively correlated with high NO_3^- concentrations in the lagoon (Nakamura et al., 2020). While human wastewater is thought to be the major source of N to the lagoon, another potential source is upwelling of nutrient-rich deep water as seen at the Great Barrier Reef (Fujita et al., 2014). Regardless, human wastewater is thought to be the primary source of other pollutants such as high coliform bacteria, and low redox conditions have been reported in the lagoon at ebb and low tides (Fujita et al., 2013). These water quality issues in the lagoon threaten fish and invertebrate populations (Ceccarelli, 2018) which Tuvalu relies on as its main source of foreign income as tuna fishery access is sold to foreign fishing vessels (Andréfouët et al., 2017). Multiple studies that have monitored the macroalgal blooms at Fongafale Islet have suggested that groundwater may be an important pathway delivering pollutants including N to the lagoon (Ceccarelli, 2018; M. Fujita et al., 2014, 2013; N'Yeurt & Iese, 2015). However, conducting field investigations to determine the importance of groundwater as a pathway is difficult as installing groundwater monitoring wells is challenging due to the remoteness of the island and complex hydrogeological conditions (large drill rig required) combined with high population density (housing) across the island.

2.6 Summary of knowledge gaps

This review of the literature has revealed key knowledge gaps that need to be addressed to improve understanding of freshwater-saltwater groundwater dynamics and groundwater N transport in atoll island settings including our study site of Fongafale Islet. These include:

- Variable-density groundwater flow models of atoll islands have focused on studying impacts of climate change and climate variability on the fresh groundwater lens as well as estimations of sustainable yield (Werner et al., 2017). No prior studies have explicitly examined the effect of tides with sensitivity analyses on the fresh groundwater lens including the thickness of the freshwater-saltwater mixing zone. Further no prior studies have examined the interacting effect of tides and variable (interannual) recharge patterns on the fresh groundwater lens.
- Only one prior study has simulated contaminant groundwater transport in an atoll island setting (Jazayeri et al., 2019). This study did not consider the effect of tides which are known to considerably influence transport pathways, residence times and exit concentrations for contaminants in coastal aquifers. Due to the complex hydrogeological conditions of atoll islands combined with the tidal forcing from the ocean and lagoon sides of the islands the subsurface transport of N and its ultimate discharge to coastal waters is expected to differ in atoll island compared to continental environments.

A numerical study examining salt and nutrient transport across Fongafale Islet will provide new conceptual understanding that can be applied to guide groundwater and nutrient management scenarios on Fongafale Islet as well as to other atoll islands globally. This study will focus on the groundwater N contamination and transport across Fongafale Islet and the findings in the following chapter will contribute to filling the knowledge gaps identified here.

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Chapter 3

3 Evaluation of freshwater groundwater resources and nitrogen discharge to the coastal lagoon of an atoll island

3.1 Introduction

Atoll islands are small, low-relief islands surrounding a lagoon that typically have a land area less than a square kilometer and maximum elevation a few meters above sea level (Werner et al., 2017). These low-lying islands are often in remote tropical settings and the communities are particularly vulnerable to climate events including droughts and flooding, as well as over-use of fresh water supply (Bailey et al., 2010). Fresh groundwater is often the main potable water source for the communities that inhabit these small islands, where population densities can reach up to 12,000 people/km^2 (Bailey et al., 2010). Increasing populations and associated development on many atoll islands globally is depleting fresh groundwater reserves contributing to water insecurity. In addition, anthropogenic activities are causing high input of contaminants such as nutrients into groundwaters and adjacent coastal waters (eg. Klassen & Allen, 2017; McMahon & Santos, 2017; Osawa et al., 2010). Impairment of coastal water quality is a major challenge as the integrity of the lagoon and ocean reef habitats that surround atoll islands often support important coastal fisheries that are critical for livelihoods (Gillett, 2016).

Potable groundwater on atoll islands typically exists as a fresh groundwater lens that floats overtop of more-dense seawater. The configuration and thickness of the freshwater lens is determined by various hydrogeologic and hydrologic factors including the distinct hydrogeological structure of an atoll island as well as precipitation patterns(Ketabchi et al., 2014; Werner et al., 2017). The shallow fresh groundwater reserves are vulnerable to contamination by anthropogenic activities (e.g. waste disposal, wastewater discharge, agricultural activities) and natural processes (e.g. sea level rise, seawater inundation) as there is often minimal protection from infiltration due to a thin, permeable unsaturated zone (Robins, 2013; Werner et al., 2017). The thickness of the fresh groundwater lens is dynamic and varies in response to transient forcing including variable precipitation-driven recharge patterns (seasonal and inter-annual), oceanic fluctuations including tides and storm surges, and longer-term changes such as sea level rise (Werner et al., 2017; Nakada et al., 2012). While it is well established that oceanic fluctuations including tides considerably influence groundwater flow patterns and the salinity distribution in continental (mainland) coastal aquifers, their effect on the fresh groundwater lens and thus fresh groundwater reserves on atoll islands are not well understood (Werner et al., 2013). This is in spite of the high connectivity between the aquifer and coastal waters in atoll island settings. Prior numerical models used to simulate freshwater lenses in atoll islands have typically applied constant-head conditions (i.e. constant mean sea level) along the aquifer-ocean and aquifer-lagoon boundaries (Bosserelle et al., 2015; Jazayeri et al., 2019). Of the few studies that have explicitly simulated tidal fluctuations, they have only used generic conceptualized atoll island setting and provide limited sensitivity analyses and discussion of the impacts of tides (Bailey et al., 2009; Bailey & Jenson, 2014).

High inputs of nitrogen (N) to coastal lagoons are of key concern as they can cause eutrophication which can lead to algal blooms, hypoxic conditions, and loss of habitats and reduced biodiversity (Anwar et al., 2014; Camargo & Alonso, 2006). Anthropogenic sources such as wastewater and fertilizers are often major sources of N in atoll island settings (Falkland et al., 1991; Fazal et al., 2003; Robins, 2013). While discharge of N-rich groundwater may be contributing high N loads to coastal lagoons surrounding atoll islands, the relative importance of groundwater inputs is poorly understood (Moosdorf et al., 2015). The flux of N from groundwater to a coastal lagoon is governed by the N contamination source, subsurface transport pathway, travel (residence) time, and geochemical processes along the discharge pathway (Anwar et al., 2014; Slomp & Van Cappellen, 2004). Based on studies focused on continental coastal aquifers it is expected that oceanic forcing including tides may affect N discharge from groundwater to a coastal lagoon by modifying the groundwater transport pathway, travel time, and geochemical conditions (Robinson et al., 2009; Robinson et al., 2007), but this has not previously been explored for atoll island settings. For instance, Anwar et al. (2014) numerically investigated the effect of tides on N transport in a continental coastal aquifer and showed that tides significantly altered N transformations in the coastal aquifer due to the enhanced mixing between fresh groundwater and seawater recirculating through the aquifer. Tides were also shown to increase the travel times for N discharging through the coastal aquifer and considerably reduce N concentrations at the groundwater-ocean interface (also known as exit concentrations) (Anwar et al., 2014).

While several studies have examined the sources, transport, and transformations of N in continental coastal aquifer systems (eg. Valiente et al., 2018; Wang et al., 2017; Xue et al., 2009), few studies have examined N sources, transport and transformations in reef island aquifers including atoll island aquifers and subsequent N discharge to coastal lagoons (eg. Graves et al., 2021; Haßler et al., 2019; Sims et al., 2020). Developing, rural and geographically remote communities such as those on many atoll islands often use decentralized onsite wastewater treatment systems including septic systems (Robertson et al., 2021). These systems can fail in various ways, and failing systems can lead to high amounts of wastewater contaminants including N released into the groundwater (Withers et al., 2014). N groundwater contamination on atoll islands has been observed previously. For instance, Haßler et al. (2019) observed elevated N concentrations in groundwater with limited denitrification in barrier reef islands of Tahiti and Moorea. More recently, Graves et al. (2021) found that septic systems and other waste sources may be causing N groundwater contamination in the atoll island of South Tarawa, Kiribati and N-rich groundwater may be discharging to the lagoon. The complex geological heterogeneity of atoll islands as well as the high connectivity between the aquifer, lagoon and ocean means that subsurface N transport and discharge in atoll island environments may considerably differ from continental coastal environments. Therefore studies are required that are specific to atoll island settings (Werner et al., 2017). While prior field studies provide some knowledge of the occurrence of high N in groundwater in atoll islands (Osawa et al., 2010; Redding et al., 2013), the number of field studies remains limited as it is often challenging to conduct groundwater field investigations on atoll islands due to their remote location and constraints in using large drilling equipment.

Numerical modeling may be a valuable approach for conceptualizing the groundwater flow patterns and N transport pathways in atoll islands. Numerical models have been widely used to simulate variable-density flow and multi-species transport in continental coastal aquifers including evaluating the subsurface transport and transformation of N (Valiente et al., 2018; Wang et al., 2017; Xue et al., 2009). To our knowledge only one study (unpublished report) has previously simulated subsurface contaminant transport in an atoll island setting. This study by Jazayeri et al. (2019) performed variable-density and contaminant transport simulations of N (greywater) at multiple islands of Tarawa Atoll using SEAWAT.

While some of the knowledge from continental coastal aquifer systems regarding the transport and transformations of N including the effects of transient forcing such as tides and variable recharge may be transferrable to atoll island settings, there are many gaps in knowledge for the unique atoll island environment. This study aims to evaluate 1) the effects of tides and variable (interannual) recharge patterns on the configuration of the fresh groundwater lens, including the thickness of the freshwater-saltwater mixing zone in an atoll island; and 2) transport and discharge of septic system-derived N in an atoll island groundwater system including impact of tides, variable recharge patterns, wastewater loading rate and wastewater source locations. To address these objectives a variable-density groundwater flow and N transport model was developed in SEAWAT-2005 (Guo & Langevin, 2002) based on conceptualization of the hydrogeological and hydrological conditions for the atoll island of Fongafale, Tuvalu. Salinity distributions, N transport pathways and travel times and N exit concentrations are calculated for models that consider no tides, tides, interannual recharge variability, and different N wastewater source locations and loading rates. The findings of this research are needed to improve prediction of atoll island freshwater lens extent, and for evaluating the subsurface transport of N from septic systems and its discharge to coastal lagoons. While this study focuses on N, the findings are relevant for other groundwater contaminants and may be valuable in providing new conceptual understanding of subsurface contaminant transport processes to be applied to inform atoll island water quality management programs.

3.2 Methodology

3.2.1 Study Site

This study focuses on the fresh groundwater resources and wastewater N contamination on Fongafale Islet, located on the eastern side of Funafuti Atoll, Tuvalu (Figure 3.1c). Tuvalu is a small atoll island nation located in the South Pacific Ocean that consists of three reef islands and six atoll islands, which all total to approximately 26 km^2 (Figure 3.1a and b) (Ceccarelli, 2018). Coral reefs in Tuvalu extend over 710 km^2 but reefs have been declining in recent decades due to storms, fishing practices, coral predators, and bleaching (Lovell. et al., 2004). Fongafale (Figure 3.1d) is the largest population centre in Tuvalu with 63% of the total population of Tuvalu which is approximately 10,650 (Central Statistics Division of Tuvalu, 2017). About 70% of the Fongafale population lives in the central part of the Islet towards the lagoon side (Fujita et al., 2014). Funafuti atoll is highly vulnerable to climate change effects including sea level rise and has recorded rates of sea level rise at three times the global average rate (Becker et al., 2012).

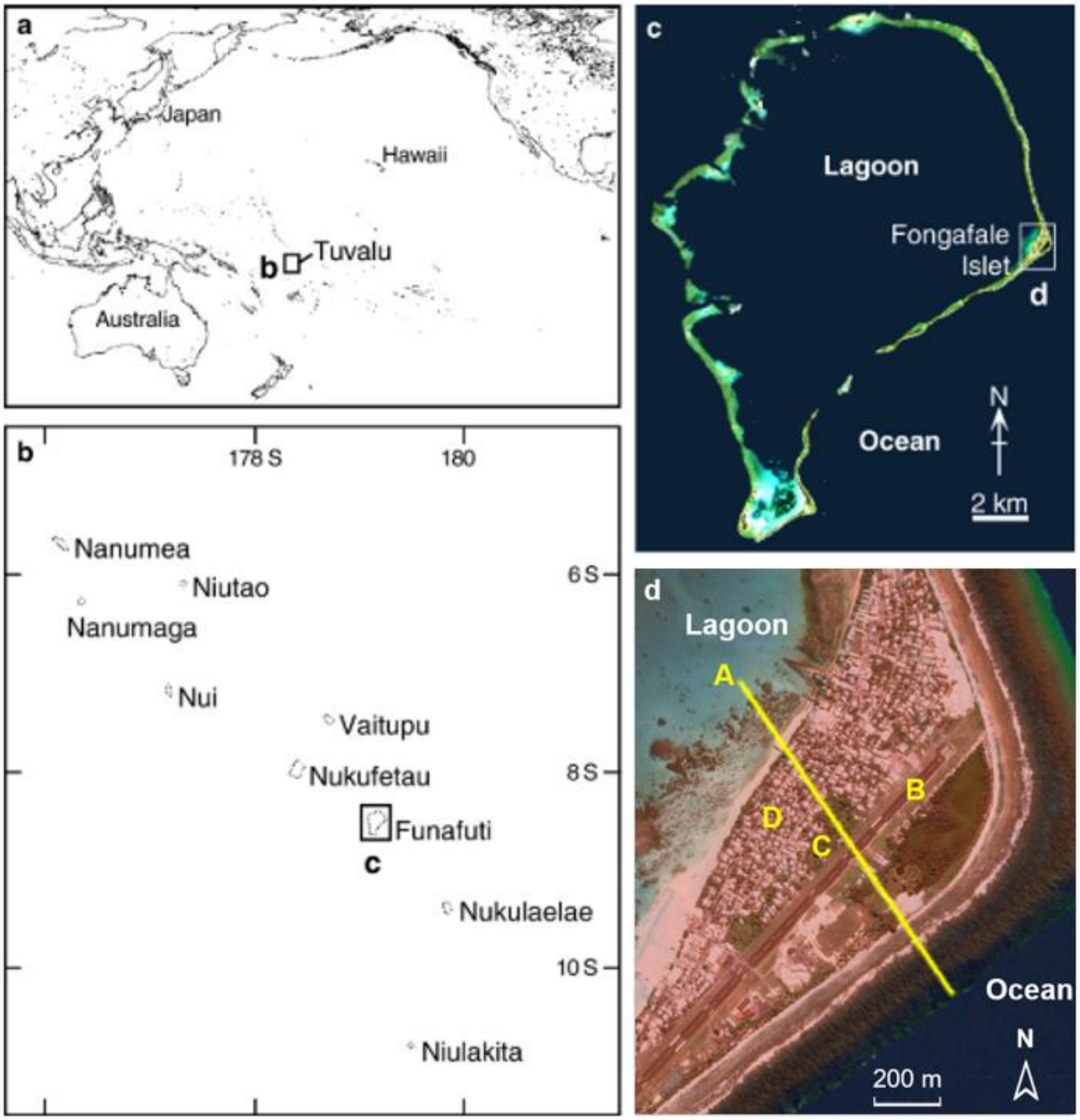


Figure 3.1. (a) Location of Tuvalu in the Pacific Ocean, (b) Location of Funafuti Atoll within Tuvalu, (c) Location of Fongafale Islet on Funafuti Atoll, (d) Fongafale Islet with areas of interest indicated including A. Cross-islet transect simulated in this study, B. Airstrip, C. Taro pits, D. Main population centre and infrastructure (Figure reproduced from Yamano et al., 2007).

Similar to many other atoll islands, the geomorphology of Fongafale Islet is characterized by a lagoonward beach ridge about 2.5 m above mean sea level (MSL), a central depression containing swampland, and an oceanward storm ridge constructed of coral rubble (Figure 3.2). The oceanward storm ridge is the highest point on Fongafale Islet with a maximum elevation of ~5 m above MSL (Yamano et al., 2007). The central part of Fongafale Islet was once an extensive mangrove swampland, but this area was reclaimed around 1942-1943 by the U.S. Corps of Engineers using permeable coral blocks as fill material (Fujita et al., 2014; Webb, 2007). Fongafale Islet experiences surface water flooding around the airstrip and other low-elevation central parts of the island caused in part by rising sea levels and storm surges but also by groundwater breaking out and flooding at the ground surface during king spring tides (Nakada et al., 2012; Yamano et al., 2007). The shallow, saturated soils of the taro pits (Figure 3.1d) in the central, low elevation area of the island have been found to salinize annually potentially as a result of seawater intrusion and flooding (Webb, 2007).

The environment, including freshwater resources, are highly stressed on Fongafale Islet due to increasing pressures from new development, high population density, and the legacy of historical landform and coastal engineering (Webb, 2007). Much of the Fongafale population uses rainwater harvesting systems for freshwater supply, but the reliability of this water source is poor due to periodic droughts (Thaman et al., 2016). The fresh groundwater reserves are limited compared to other atoll islands with the shallow groundwater often brackish and the thickness of the freshwater lens dependent on recharge from rainfall (Thaman et al., 2016). Similar to other atoll islands, the hydrogeology of Fongafale Islet is characterized by unconsolidated sediment (Holocene sediments) overlying cemented coral rubble or phosphatic limestone (Pleistocene limestone) (Dickinson, 1999). The limited fresh groundwater reserves on the island compared to other atoll islands are thought to be due to the narrow width of the island, the hydrogeological structure, and the highly permeable, porous coral gravel used in the reclamation of the taro pits (Nakada et al., 2012). Pollution of the Fongafale Lagoon by various anthropogenic sources including human wastewater has led to the degradation of the marine environment that is critical for livelihoods and conservation (Thaman et al., 2016). Investigation of factors affecting the configuration of the fresh groundwater lens and the potential role of groundwater in delivering N from domestic septic systems to the Fongafale Lagoon is needed to inform policy and infrastructure decision making.

3.2.2 Two-Dimensional Groundwater Flow and Transport Model

A two-dimensional numerical model was developed to simulate variable-density groundwater flow and N transport for a transect extending across Fongafale Islet. The location of this transect (shown in Figure 3.1d) corresponds to the location of geoelectric surveys conducted by Nakada et al. (2012). Numerical transient variable-density groundwater flow was simulated using SEAWAT-2005 and conservative N transport was simulated using MT3DMS (Guo & Langevin, 2002). The governing equation used by SEAWAT-2005 for density-dependent groundwater flow is provided in Appendix A. The model domain used to simulate groundwater flow and transport across the transect on Fongafale Islet is shown in Figure 3.2. The coordinate system is defined such that x = 0 m and z = 0 m are at the mean shoreline position on the lagoon side of the transect. The simulated domain extends 415 m into Fongafale Lagoon, 220 m into the ocean, and 50 m below MSL. The topography across the transect was specified using 2019 LIDAR elevation data (Resture & Ewekia, 2019). The LIDAR data was corrected from GRS80 ellipsoid to the Local Mean Sea Level (LMSL) using Earth Gravitational Model 2008 (Resture & Ewekia, 2019). The model domain was discretized with Δx = 5 m and Δz = 1 m resulting in a model with 55 layers and 260 columns and a cross-shore width of 1 m. Preliminary simulations were conducted to ensure the model solutions were converged and independent of the grid discretization.

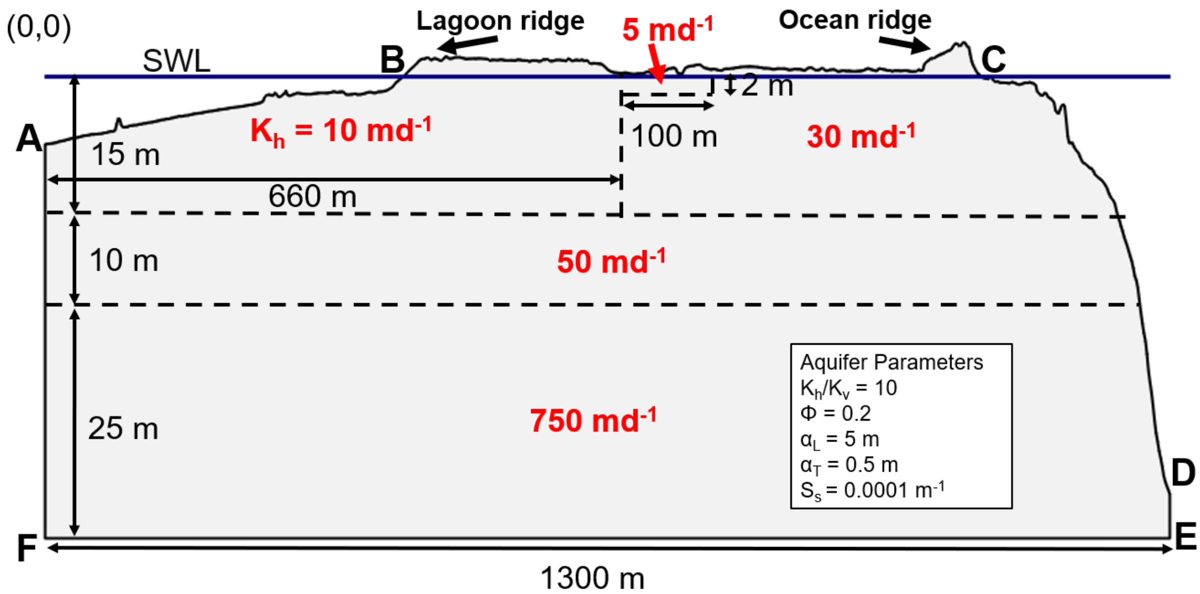


Figure 3.2. Numerical groundwater model domain and parameters: horizontal hydraulic conductivity (K_H) values used in different zones are identified in red, horizontal to vertical conductivity ratio (K_h/K_v), porosity (Φ), longitudinal dispersivity (α_L), transverse dispersivity (α_T), and specific storage (S_s). Fongafale Islet landform features include the lagoon ridge near point B, taro pits at the central low-elevation area, and the ocean ridge near point C.

The availability of hydrogeological data for Fongafale Islet including the simulated transect location are limited. The hydrogeology across the transect was conceptualized by reviewing knowledge of typical atoll island hydrogeology (Ayers & Vacher, 1986; Bailey et al., 2009; Hamlin & Anthony, 1987), information on Fongafale Islet from Nakada et al. (2012) and Ohde et al. (2002), and summarizing approaches used to simulate similar atoll islands (Bailey et al., 2009; Bosserelle et al., 2015; Jazayeri et al., 2019). For our model, five hydraulic conductivity (K) zones were adopted as shown in Figure 3.2. The horizontal K (K_h) of the central, near-surface K zone representing the taro pit was set to 5 md^{-1}, surficial Holocene sediments of the lagoon side were set to 10 md^{-1}, unconsolidated Holocene sediments across the central and ocean sides of the model were set to 30 md^{-1}, the upper Pleistocene layer was set to 50 md^{-1}, and lower Pleistocene layer was set to 750 md^{-1}. Detailed analyses examining the sensitivity of the simulation results to the location of the K zones and K_H values adopted were conducted. A summary of these analyses is provided in Appendix B. The values for other aquifer parameters including porosity (ϕ), specific storage, longitudinal dispersivity (α_L), transverse dispersivity (α_T), and the ratio of horizontal to vertical K (K_h/K_v,)are provided in Figure 3.2 and were based on values commonly used in groundwater models of atoll islands (Bailey et al., 2009; Bailey & Jenson, 2014; Jazayeri et al., 2019; Post et al., 2018). Sensitivity analysis simulations were performed to examine the influence of dispersivity values on the simulations results and these results are provided in Appendix B.

Simulations were performed with and without tides, and with and without variable recharge to identify the impact of tides and variable recharge conditions on the freshwater lens extent and N subsurface transport. For all simulations no flow boundary conditions were specified for the bottom boundary of the model (EF) and the vertical boundaries on the lagoon (AF) and ocean (DE) sides. The upper boundary BC was specified as a Type 2 boundary condition with the water flux based on an estimated recharge rate. For the constant recharge simulations, the recharge rate was based on average monthly precipitation for Tuvalu for the years of 1901 to 2016 (The World Bank Group, 2017). Based on prior studies (Hamlin & Anthony, 1987; Hunt & Peterson, 1980; Jazayeri et al., 2019), the average recharge was calculated to be 4.1 mm/d by assuming it was 50% of average precipitation over this period (Appendix C). For the variable recharge simulations, average monthly precipitation over a period of 20 years from 1991 to 2010 were used to calculate average monthly recharge rates that were applied as a time-varying flux along the boundary BC (Figure 3.3). The average recharge rate over this period was 4.2 mm/d which is similar to the recharge rate used for the constant recharge simulations. While it is expected that the percentage of precipitation that recharges the aquifer may be higher during dry periods compared to wet periods this was not considered in calculating the variable recharge amounts. For the non-tidal simulations Type 1 constant head boundary conditions were applied along the lagoon-aquifer interface (AB) and ocean-aquifer interface (CD). For the tidal simulations, tidal fluctuations were simulated along the aquifer-lagoon (AB) and aquifer-ocean (CD) boundaries by using the Periodic Boundary Condition (PBC) Package to specify time-varying heads (Post, 2011). The tidal amplitude and period were specified as 0.6 m and 12 hours, respectively, based on the 2020 Funafuti tidal calendar and hourly sea level data at the Fongafale station (Commonwealth of Australia, 2021).

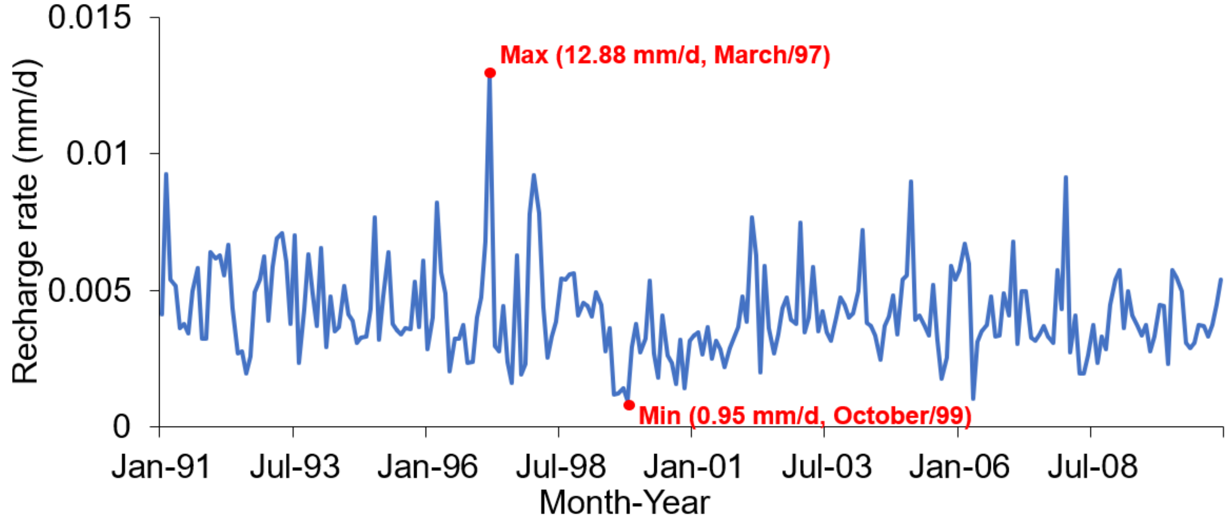


Figure 3.3. Monthly average recharge rate (50% of rainfall) over period from 1991-2010 for Tuvalu. This data was used in simulations considering variable recharge patterns.

The groundwater flow and salt transport across the model domain without tides and with constant recharge was first simulated with the models run until the groundwater flows and salt concentrations reached steady state. The final heads and salt concentrations from this model were used as the initial conditions for simulations considering tides, variable recharge, and N transport (Table 3.1). These models were then run until the groundwater flows and salt and N concentrations reached quasi-steady state. For the variable-density groundwater flow model with no tides and constant recharge, the initial salt concentrations in the model domain were 35 g/L. For all simulations salt concentrations along the aquifer-lagoon (AB) and aquifer-ocean (CD) boundaries were set to 35 g/L, and the salt concentration of the recharge (through boundary BC) was 1 g/L.

Table 3.1. Base model simulation cases conducted. The wastewater N concentration, wastewater loading rate, and wastewater source location were constant for these simulations.

Simulation	Sea level	Recharge	N transport simulated
1	Constant	Constant	no
2	Tidal	Constant	no
3	Constant	Variable	no
4	Tidal	Variable	no
5	Constant	Constant	yes
6	Tidal	Constant	yes
7	Constant	Variable	yes
8	Tidal	Variable	yes

The N transport models were set up to simulate the subsurface transport of N originating from household locations (and thus septic systems) along the transect. To estimate the wastewater loading rate and N concentration in the wastewater from an individual septic system, data was collated on the average household water consumption, the daily average N content in the residents' diets, and the average number of people living in a household in Fongafale. The N concentration in the wastewater was estimated to be 0.26 g/L and the wastewater loading rate from an individual septic system was estimated to be 70 L/d (see Appendix C for calculations). The N concentration in the wastewater assumes the worst case where the wastewater is released directly from a household septic system into the subsurface with no treatment and attenuation of N (Fujita et al., 2013). For each septic system a wastewater plume containing N was injected into the top of the model domain along a length of 10 m (two model cells) to represent the length of an average septic system drainage field. Multiple simulations were conducted to explore the sensitivity of the N transport to the wastewater loading rate, and to the location and number of septic systems across the model transect (Table 3.2). For the base model, a single septic system was simulated with this system located 150 m from the lagoon shore which is in the middle of the populated area of Fongafale. The initial N concentration in the model domain was 0 g/L for the simulation without tides and constant recharge, and the final N concentrations across the domain from this model were used as the initial conditions for the simulations that considered tides and variable recharge. The

N transport models were run until quasi-steady state was reached with respect to the N concentrations across the model domain.

Table 3.2. Sensitivity simulations run with and without tides and interannual recharge variability. The model was the same as the base model (Table 3.1) except for the variable for which the sensitivity was conducted. The base model values are marked by bold.

Sensitivity variable	Values simulated
Wastewater loading rate (L/d)	35, **70**, 350
Wastewater source location (m from the lagoon)	25, 50, 100, **150**, 175, 200
Multiple wastewater sources (distances from lagoon)	25, 50, 70, 100, 120, 135, **150**, 175, 200 (m from the lagoon)

3.3 Results & Discussion

3.3.1 Salinity distribution including effect of tides, interannual recharge and wastewater inputs

The quasi-steady state salinity distributions for the simulations that consider no tides, tides, and variable recharge are shown in Figure 3.4. For all simulations, the results indicate that there is no distinct freshwater lens across the transect. The fresh groundwater lens is typically defined as groundwater with salt concentration less than 2.5% relative seawater salinity (herein referred to as % seawater). The absence of a distinct freshwater lens is consistent with prior studies that have reported minimal availability of fresh groundwater on Fongafale Islet (Nakada et al., 2012). For all simulations the groundwater salt concentrations are lower in the shallow aquifer on the lagoon side of the transect compared to the ocean side. This is because the lower permeability sediment (lower K) associated with the taro swamp retains more freshwater compared to the surrounding higher permeability zones where higher freshwater-saltwater mixing occurs. It is important to note that the salinity distributions shown in Figure 3.4 vary with aquifer parameters including dispersivity ($\alpha_{L,}$ α_T) and K. The results of sensitivity analyses conducted to examine the influence of these parameters on the salinity distribution are provided in Appendix B.

The simulation results show that tidal fluctuations increase the width of the freshwater-saltwater mixing zone. This in turn considerably reduces the amount of groundwater along the transect with salt concentration less than 10% seawater (e.g., compare Figure 3.4a and 3.4c). For instance, the 10% seawater contour extends to 20 m below the land surface for the simulation with no tides (Figure 3.4a) compared with only 2 m below the land surface for the simulation with tides (Figure 3.4c). The wider freshwater-saltwater mixing zone for the tidal simulations is caused by oscillatory tide-induced flows that enhance dispersion. This thicker mixing zone has previously been observed in numerical simulations of atoll island aquifers by Bailey et al. (2009) and Oberdorfer et al. (1990) that considered tidal effects. Our results are also consistent with those of Bailey & Jenson (2014) who demonstrated the freshwater lens thickness was reduced by up to 0.75 m when tidal fluctuations were explicitly simulated rather than a constant head condition adopted along the coastal boundaries. These results highlight the importance of including tidal fluctuations in numerical models simulating fresh groundwater availability in atoll islands as the extent of the freshwater lens may be substantially overestimated when tidal fluctuations are neglected.

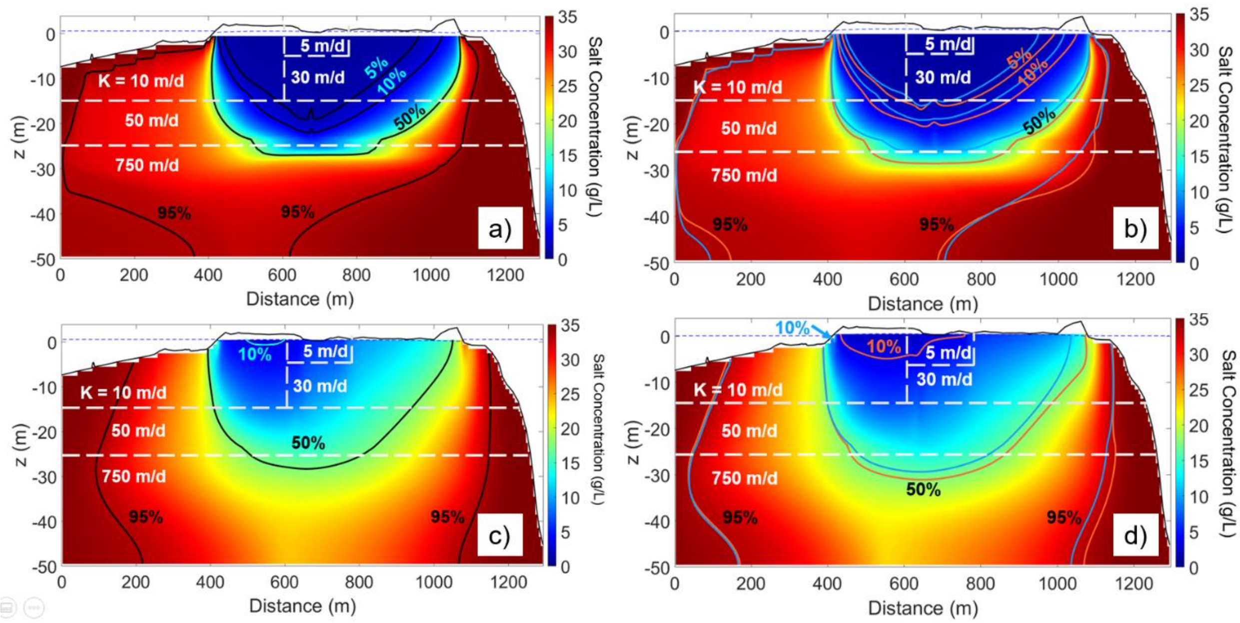


Figure 3.4. Salinity distribution along the simulated transect of Funafuti Islet with colored contours indicating the salt concentration (g/L) and line contours indicated the isolines for percentage of relative seawater salinity. Four simulation cases are shown and include, a) Constant recharge without tides, b) variable recharge without tides, c) constant recharge with tides, and d) variable recharge with tides. The white dashed lines show the different *K* zones with the white text indicating the corresponding K_H values. In Figure 3.4b and d the blue and red line contours show the isolines at the times of minimum and maximum recharge, respectively.

Consideration of variable recharge patterns rather than constant recharge only had a small effect on the simulated salinity distribution in the aquifer (Figure 3.4). The orange and blue contour lines in Figures 3.4b and d show the % seawater isolines at times of maximum and minimum recharge, respectively. As expected, they show that for both the non-tidal and tidal simulations the zones of low salinity groundwater slightly expand at times when the recharge is greater. For example, for the non-tidal simulation the 10% seawater isoline reaches a depth of 20 m below the land surface at the time of maximum recharge compared to only 18 m below the land surface at the time of minimum recharge. The effect of considering variable recharge on the salinity distribution is greater when tides are also considered compared to the non-tidal simulations as evident by the greater difference in the depth of the 10% seawater isoline between the simulations with and without variable recharge considered (compare Figures 3.4c and d). Overall, although variable recharge does influence the salinity distribution, the simulations suggest that, for the conditions on Funafuti Islet, tides have a larger effect on the salinity distribution than variable recharge patterns.

Wastewater inputs from septic systems are an additional source of freshwater to the aquifer in addition to recharge from precipitation. The simulations which consider wastewater input into the subsurface illustrate the considerable influence of these inputs on the salinity distribution and freshwater lens extent (compared % seawater isolines in Figure 3.5 compared to Figure 3.4). For the non-tidal and tidal simulations with wastewater input from only one septic system considered, the size of the zone with low salinity groundwater (<10% seawater) increased and a small area with < 2.5% seawater is present (Figure 3.5). For or the non-tidal simulations with input from one septic system (constant and variable recharge), the zone with < 2.5% seawater (e.g. freshwater lens) reached 10 m deep and approximately 30 m wide (Figure 3.5a and b), whereas there was no groundwater with <2.5% seawater when the wastewater source was not considered (Figure 3.4a and b). As expected, the extent of low salinity groundwater (<10% seawater) zone increased further when wastewater inputs from nine septic systems across the transect were simulated (Figure 3.6). For the non-tidal simulations with input from nine septic systems (constant and variable recharge), the region with salt content less than 2.5% seawater reached approximately 15 m deep and 200 m wide (Figure 3.6a and b). Jazayeri et al. (2019) similarly observed that the injection of greywater into the aquifer led to a greater extent of low salinity groundwater in the aquifer and thus thicker freshwater lens. These results highlight the need to consider wastewater inputs into the aquifer in addition to precipitation-driven recharge in simulating the salinity distribution and extent of the freshwater lens in atoll islands.

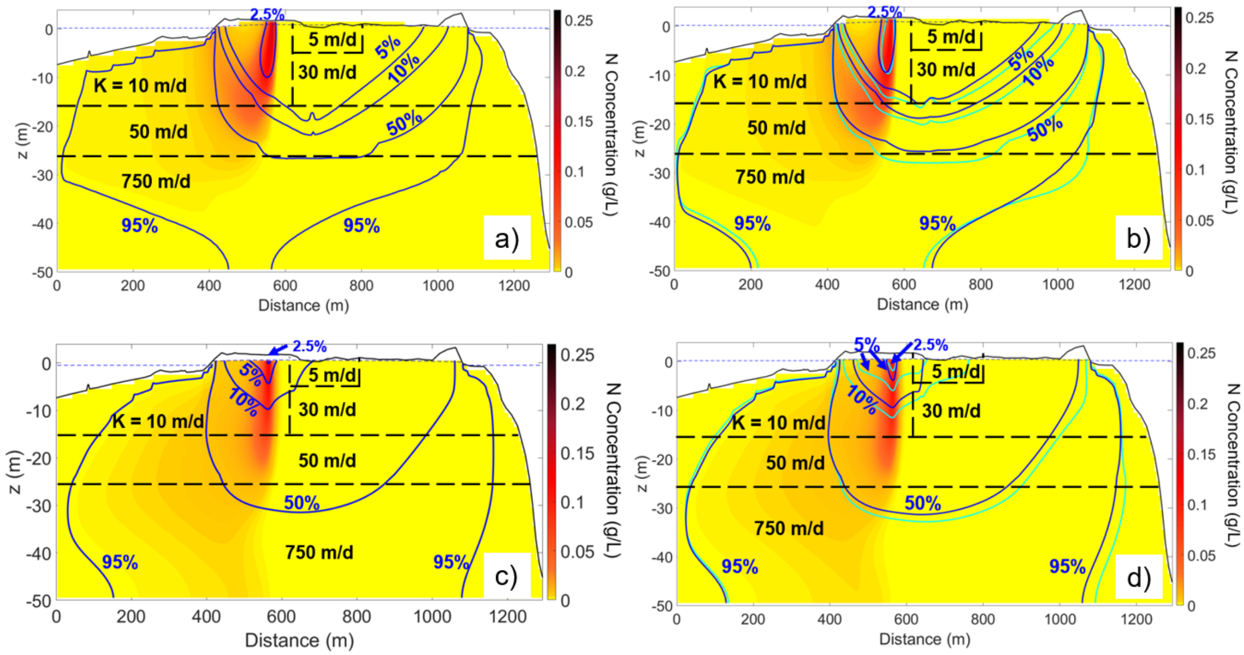


Figure 3.5. Quasi-steady state N concentrations (shaded contours) and salt concentrations (% relative seawater salinity; line contours) for simulations with one wastewater source located at 150 m from inland from the lagoon shoreline and considering a) constant recharge without tides, b) variable recharge without tides, c) constant recharge with tides, and d) variable recharge with tides. The hydraulic conductivity zones are indicated by the black dashed lines and assigned K_h values for each zone are shown. In Figure 3.5b and d the blue and green line contours show the isolines at the times of minimum and maximum recharge, respectively.

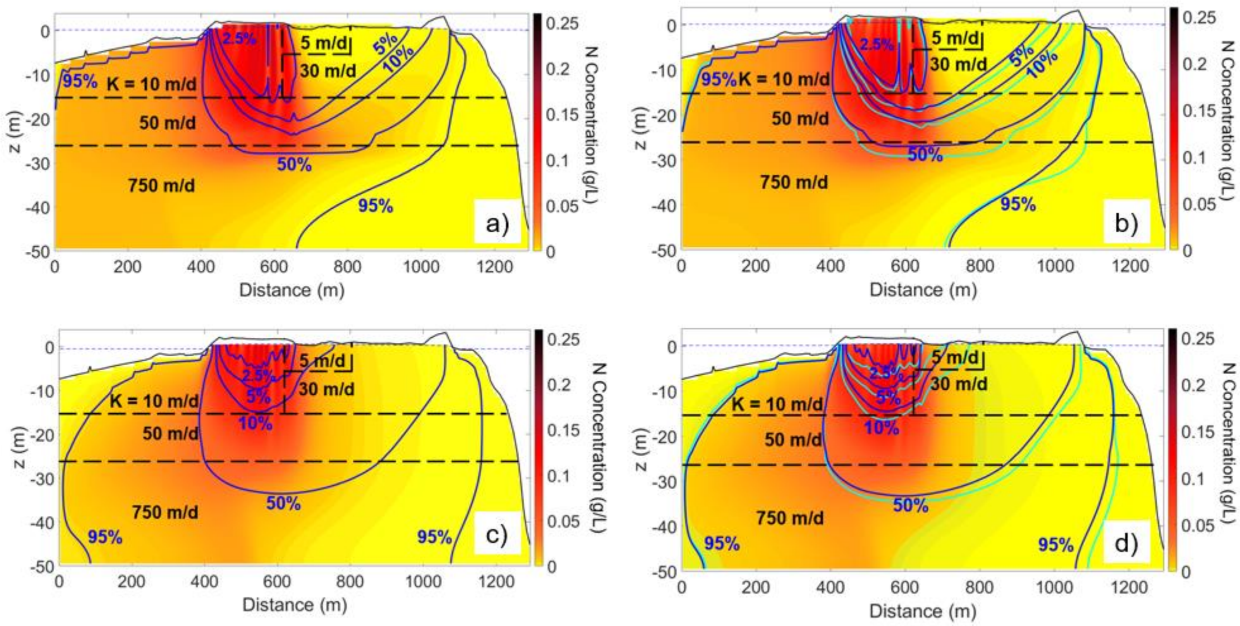


Figure 3.6. Quasi-steady state N concentrations (shaded contours) and salt concentrations (% relative seawater salinity; line contours) for simulations with multiple (nine) wastewater sources considering a) constant recharge without tides, b) variable recharge without tides, c) constant recharge with tides, and d) variable recharge with tides. The hydraulic conductivity zones are indicated by the black dashed lines and assigned K_h values for each zone are shown. In Figure 3.6b and d the blue and green line contours show the isolines at the times of minimum and maximum recharge, respectively.

3.3.2 N transport pathways

3.3.2.1 Effect of tides and variable recharge on N transport from a single septic system

The quasi-steady state N distributions across the transect for simulations considering a single septic system wastewater source located 150 m inland from the lagoon shoreline are provided in Figure 3.5. For all simulations (non-tidal, tidal, constant recharge, variable recharge), the N plume is transported towards the lagoon. The plume is initially transported downwards through the lower *K* zone (10 m/d) where it then migrates horizontally through the higher *K* layer (50 m/d) before being transported upwards to discharge to the lagoon. For the non-tidal simulations (constant and variable recharge) the N plume discharges close to the shoreline (Figure 3.5a,b). The simulated N transport pathway is similar to that observed by Jazayeri et al. (2019) who showed that a greywater contaminant plume released towards the middle of an atoll island initially move vertically downwards to depths of 20-30 m below the land surface before being transport horizontally towards the lagoon or ocean (depending on location of source along the transect). Importantly, the general transport pathway observed in our study and by Jazayeri et al. (2019) differs from that simulated previously by Anwar et al., (2014) for a continental coastal homogeneous non-tidal aquifer. In their simulations Anwar et al., (2014) showed that the N plume migrated horizontally through the upper aquifer before discharging close to the shoreline. The difference in the transport pathway is thought to be due to the distinct hydrogeological structure of atoll islands (high K zones at depth) as well as the larger influence of recharge to the land surface on the groundwater flow patterns for atoll islands. Our simulations show that tidal fluctuations and subsequent tide-induced oscillatory flows modify the specific N transport pathway and increase the mixing and dispersion of the N plume before it reaches the lagoon. As shown in Figure 3.5c,d, tidal fluctuations caused the N plume to be transported deeper into the aquifer with the N becoming well-mixed with the ambient groundwater before discharging to the lagoon.

In addition to affecting the specific N transport pathway between the source and lagoon, tidal fluctuations impact the N exit concentrations at the lagoon (groundwater N concentration at the aquifer-lagoon interface). For the non-tidal simulation with constant recharge, the N exit concentration was highest close to the shoreline (1.6×10^{-2} g/L) with the exit concentration decreasing to 1×10^{-4} g/L at 350 m offshore. For the tidal simulation with constant recharge the maximum exit concentration was 3.8×10^{-3} g/L N at the lagoon shore and decreased to 2.7×10^{-5} g/L at 350 m offshore. Therefore, the maximum exit concentration was about four times lower than that observed for the non-tidal constant recharge simulation highlighting the large effect of tides on dilution of N as it is transported through the subsurface to the lagoon. Nitrate (NO_3^-) levels in Fongafale lagoon have been reported to reach up to 3×10^{-4} g/L near the shore in the heavily populated areas (N'Yeurt & Iese, 2015). The N exit concentrations for the models with and without tides were at least an order of magnitude higher than this lagoon NO_3^- concentration indicating the potential role of groundwater delivering high N concentration water to the lagoon. However, it is important to note that the simulated N exit concentrations represent an upper conservative value as no attenuation of N was considered in our model.

Average travel times for N to be transported from the wastewater source to the lagoon provide insight into the potential for N to transform (e.g. denitrification) along its discharge pathway and therefore to better predict the ultimate flux of N to the lagoon. Generally it is expected that longer travel times will correspond to higher attenuation of N before it reaches the lagoon. For the non-tidal constant recharge simulation the average travel time for N to be transported from the wastewater source to the lagoon was approximately 1700 days. Tides increased this average travel time to approximately 1900 days. Numerical simulations by Anwar et al. (2014) for conservative N transport in a continental homogeneous coastal aquifer similarly showed that tides increase the average travel times for N to be transported from a land source to the ocean. In our study, the long N travel times calculated for both the non-tidal and tidal simulations suggest the potential for natural N attenuation (denitrification) to occur in the subsurface. For instance, Couturier et al. (2017) considered a travel time of 166 days through a 50 m distance in a shallow continental coastal aquifer to be sufficient for denitrification reactions to occur. In addition, O'Driscoll et al. (2014) reported attenuation of NO_3^- derived from an onsite wastewater system in a coastal aquifer with NO_3^- concentrations observed to decrease from 8.3×10^{-3} g/L in the septic system drainfield to 1.5×10^{-4} g/L adjacent to the estuary (40-50 m from the drainfield).

For both the non-tidal and tidal simulations, the N exit concentrations changed in response to variable recharge patterns. For instance for the non-tidal simulations the N exit concentrations ranged from 1.4×10^{-2} to 1.9×10^{-2} g/L in response to the varying recharge. There was a delay between the times of maximum (2280 d) and minimum (3224 d) recharge and the times of maximum and minimum exit concentrations. The time delay between the recharge and exit concentration corresponded to the average N travel time (approximately 1000 day time lag). Variable recharge had a greater influence on the N exit concentrations under non-tidal conditions compared to tidal conditions (see Appendix D for detailed results). For instance, during a 2.6 year period during which the recharge decreased by 93%, a 20% decrease in N exit concentration was observed in the non-tidal simulation results, but a considerably lower change was observed for the simulation with tides although more output data points are necessary to fully observe the variation in N in response to changing recharge. Similar to our simulation results, Colman & Masterson (2008) found in their simulations of N transport in a coastal aquifer a small, but minimal response of the N exit concentrations to a change in recharge.

3.3.2.2 Effect of wastewater loading rate of N transport from a single septic system

Sensitivity analyses were performed to evaluate the influence of the septic system wastewater loading rate from a single wastewater source on the N flow pathways, N travel times and N exit concentrations. Wastewater loading rates varying from 35 L/day to 350 L/day were simulated. For all simulations the N concentration in the wastewater was 0.26 g/L. The N plume migrated towards the lagoon for all wastewater loading rates simulated but the N plume was transported deeper into the aquifer when the wastewater loading rate was higher (Appendix E). Jazayeri et al. (2019) similarly showed in their simulations of subsurface transport of greywater in different islands in South Tarawa, Kiribati that a greywater plume migrated deeper into the aquifer when the loading rate was higher.

For the non-tidal simulations, the maximum N exit concentrations at the aquifer-lagoon interface increased from 8×10^{-3} to 7×10^{-2} g/L as the wastewater loading rate increased from 35 to 350 L/d (Figure 3.7b). For the tidal simulations, the maximum N exit concentrations also increased with increasing wastewater loading rate but to a lesser extent compared to the non-tidal simulations (2×10^{-3} and 2×10^{-2} g/L for wastewater loading rate of 35 and 350 L/d respectively; Figure 3.7b). Higher wastewater loading rate resulted in a lower average travel time for N to be transported from the wastewater source to the lagoon. For the non-tidal simulations the average travel time was approximately 1790 days for a wastewater loading rate of 35 L/day compared to 1350 days for a wastewater loading rate of 350 L/day. (Figure 3.7a). The average N travel time similarly decreased with increasing wastewater loading rate for the tidal simulations (1926 days for 35 L/day compared with 1543 days for 350 L/day; Figure 3.7a). The average simulated travel times found in this study were larger than that calculated using a steady state Darcy's law based analytical solution from Chesnaux & Allen (2008) for horizontal transport through an ocean strip island (calculated time = 734 days; see Appendix F for calculation). This is likely because in our study the N plume migrates downwards before being transported horizontally towards the lagoon. The decrease in average travel time as the wastewater loading rate increases is because the higher wastewater loading rate leads to a higher hydraulic gradient and thus higher lagoon-directed groundwater flows between the wastewater source and the lagoon. In addition, as the N plume was transported deeper into the aquifer as the wastewater loading rate increased, the plume reached the higher K zone (K = 50 m/d), which facilitated faster transport of the N plume towards the lagoon and thus lowered travel times. Based on the observed impact of the wastewater loading rate on N transport to the lagoon it is recommended that household surveys are conducted on atoll islands to better characterize wastewater loading rates from septic systems and thus be able to better evaluate the discharge of N wastewater plumes to adjacent coastal waters.

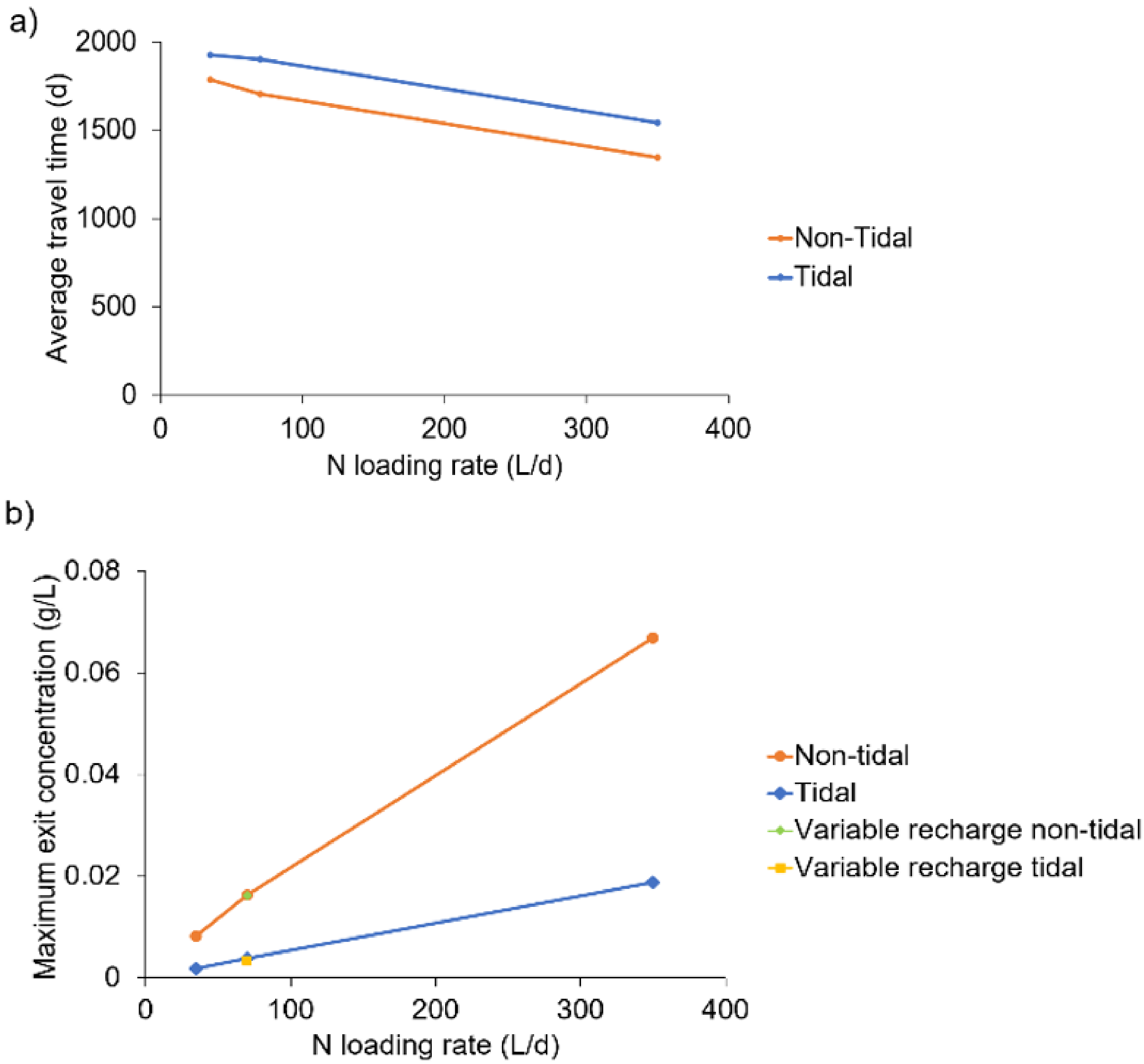


Figure 3.7. Effect of wastewater loading rate on a) average N travel times, and b) maximum N exit concentrations at aquifer-lagoon interface.

3.3.2.3 Effect of N wastewater source location

Sensitivity analyses were conducted to examine how the N transport pathways varied depending on the location of the wastewater source along the transect. Simulations were run with an individual wastewater source located at distances ranging from 25 m to 200 m from the lagoon with these locations spanning the area of high housing density along the transect (Figure 3.1d, point D). The location of the wastewater source had a large impact on the N flow pathways as well as on the average travel times and the maximum exit concentrations at the lagoon (Figures 3.8 and 3.9). For the simulations with and without tides, the N flow path expanded horizontally and vertically as the wastewater source was located farther from the lagoon (Figure 3.8). When the wastewater source was located at 50 m from the lagoon, the N plume was transported horizontally through the upper Holocene (lower permeability) sediment (Figure 3.8a and d). In contrast, the N plume was transported downwards to reach the top of the lower high permeability Pleistocene layer (K_H = 750 m/d) when the wastewater source was located 200 m from the lagoon (farthest location; Figure 3.8c and f). These results are consistent with Jazayeri et al. (2019) who also showed that the downwards transport of the contaminant flowpath was greatest for point sources farthest from the shore.

For the non-tidal simulations, the average travel times for the N plume to be transported from the wastewater source to the lagoon varied from approximately 75 to 2880 days as the source location increased from 25 m to 200 m from the lagoon. The average travel times were generally longer when tides were considered with the average travel times ranging from approximately 120 to 2630 days as the source location increased from 25 m to 200 m from the lagoon. Although in most cases the average travel times were slightly larger for the tidal simulations compared to the non-tidal simulations (Figure 3.9a), the tide-induced oscillatory flows caused the N plume originating 200 m from the lagoon to reach the lower high K layer (K_H = 750 m/d) whereby the plume was then more rapidly transported horizontally causing a lower travel time compared to the non-tidal simulation. Overall, the larger travel distances and longer travel times for the N plumes originating farther from the lagoon are expected to lead to higher N attenuation in the aquifer (e.g. denitrification) thus lowering N flux discharging to the lagoon. In their numerical modeling study examining the impact of septic system-derived N loading to coastal waters Meile et al., (2010) found that denitrification increased from 15% to 50% as the distance between the N source and coastal water increased from 25 m to 60 m.

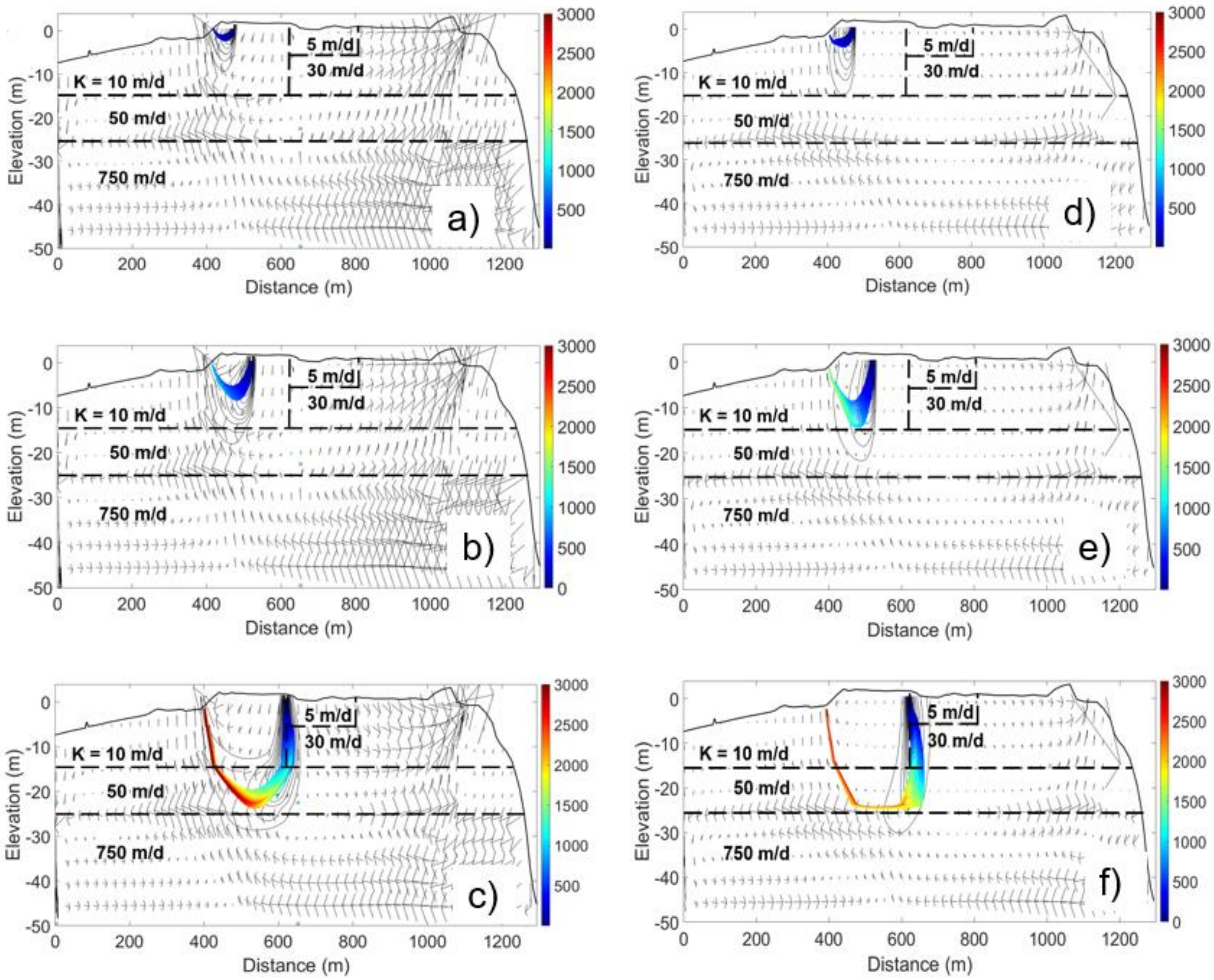
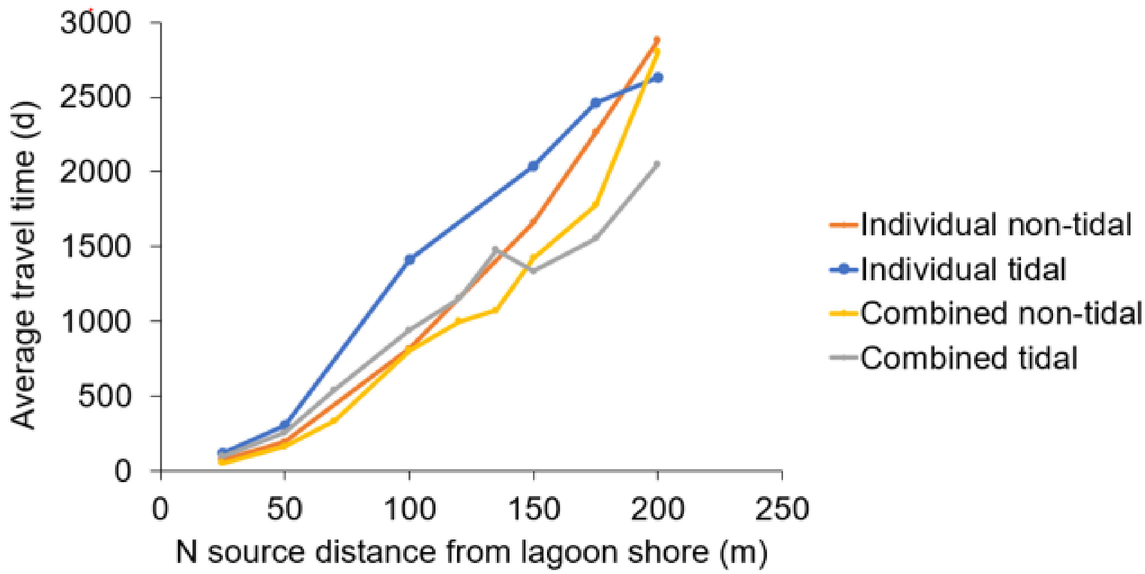


Figure 3.8. Groundwater velocities and N flow paths from an individual wastewater source (septic system) located at varying distances from the lagoon. Results are shown for simulations conducted with wastewater source located at a) 50 m, b) 100 m, and c) 200 m from the lagoon without tides, and wastewater source located at d) 50 m, e) 100 m, and f) 200 m from the lagoon with tides. The flow pathways as determined by particle tracking and average time travelled in days are shown by the colored pathlines. Groundwater velocity vectors, N concentration contours (solid black lines) and *K* zones (thick dashed black lines) are also shown.

a)



b)

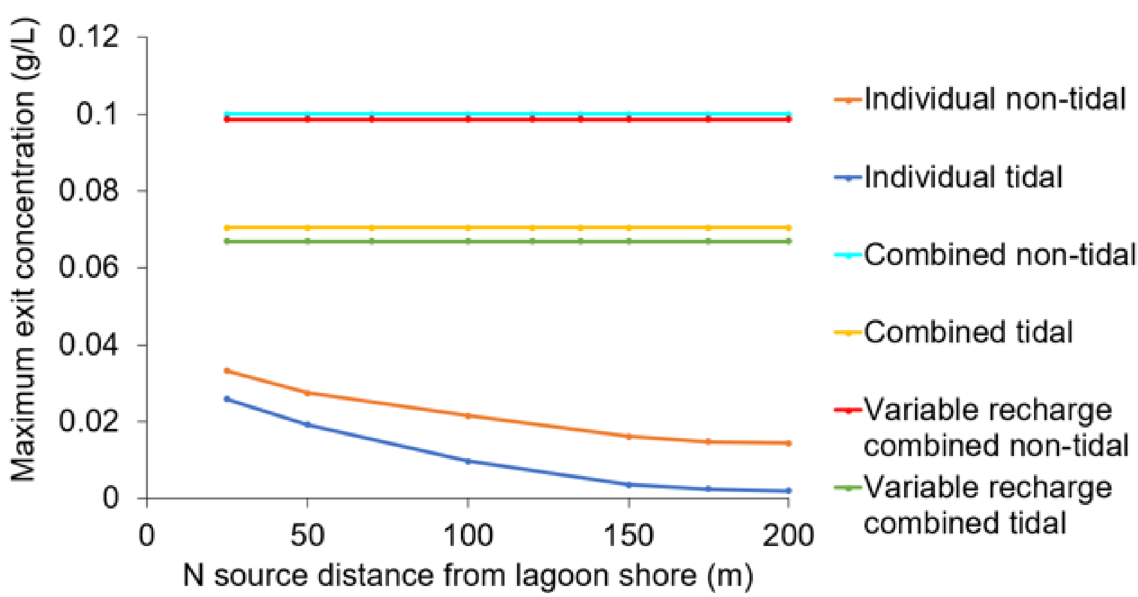


Figure 3.9. Effect of wastewater source location on a) average N travel times, and b) maximum N exit concentrations at the lagoon. Results are shown for simulations with and without tides, and with an individual wastewater source and multiple (combined) wastewater sources. All simulations used a wastewater loading rate per septic system of 70 L/d.

For the non-tidal and tidal simulations, the N exit concentrations decreased as the distance between the wastewater source location and lagoon increased (Figure 3.9b). This is expected as the amount of dispersion of the plume increases with increasing travel distance. For the non-tidal simulations, the maximum N exit concentrations varied from 3.3×10^{-2} to 1.4×10^{-2} g/L as the wastewater source moved from 25 m to 200 m from the lagoon shore. The maximum N exit concentrations were consistently lower for the tidal simulations with the concentrations for these simulations varying from 2.6×10^{-2} to 2×10^{-3} g/L as the wastewater source moved from 25 m to 200 m from the lagoon shore location. Importantly for the range of distances considered for the wastewater source location all maximum exit concentrations observed are higher than the maximum NO_3^- concentration of 3×10^{-4} g/L observed in the Fongafale lagoon (N'Yeurt & Iese, 2015) and higher than the NO_3^- level required to support a healthy coral reef system ($< 1.4 \times 10^{-5}$ g/L) (Mosley & Aalbersberg, 2005). While more research is needed to evaluate potential N transformations as the wastewater is transported from a septic system to the lagoon, the results highlight the need for wastewater treatment infrastructure programs focused on improving lagoon water quality to prioritize upgrades for household septic systems located closest to the lagoon.

3.3.2.3 Multiple wastewater N sources

The quasi-steady state N distributions for simulations considering nine wastewater sources (septic systems) located along the transect are shown in Figure 3.6. These simulation results are thought to best represent the conditions along the transect with the nine wastewater sources corresponding to locations of households along the transect. For both the non-tidal and tidal simulations, wastewater inputs from multiple wastewater sources led to a large region of high N groundwater concentrations across the lagoon side of the transect extending down to the lower high permeability Pleistocene layer. The particle tracking results show that the general flow paths for N originating from each source are similar when all nine sources are included (Figure 3.10) compared to when each source was simulated individually (Figure 3.8). Consistent with when the sources were simulated individually, the results show that tides cause the N plumes to be transported deeper in the aquifer and there is greater mixing of N with the ambient groundwater as it is transported towards the lagoon (Figure 3.6 a,b compared with Figure 3.6 c,d).. Simulations also show that consideration of variable recharge rather than constant recharge does not considerably affect the overall distribution and magnitude of N in the aquifer (Figure 3.6 a,c compared to Figure 3.6 b,d).

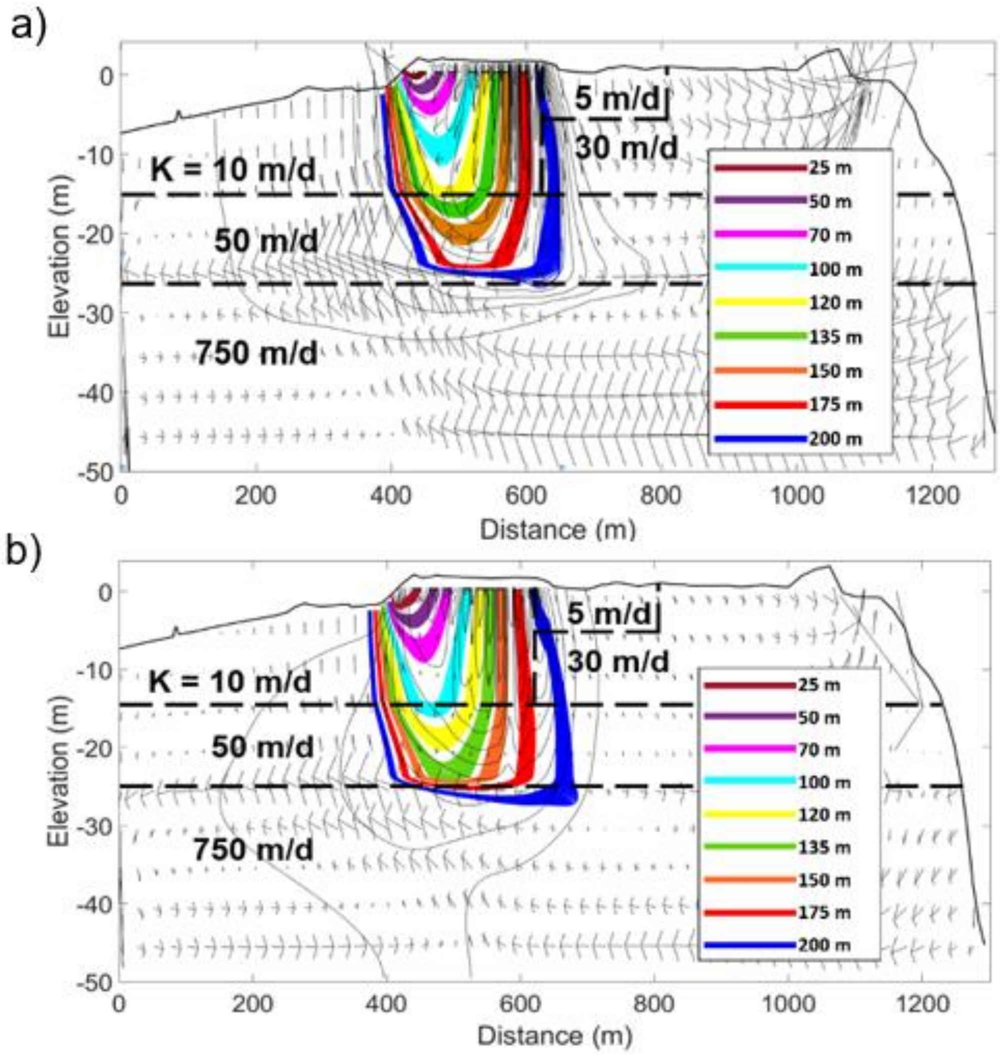


Figure 3.10. N transport pathways from multiple wastewater sources towards the lagoon when nine wastewater sources are simultaneously simulated. The sources are located at 25, 50, 70, 100, 120, 135, 150, 175, and 200 m from the lagoon and each have a wastewater loading rate of 70 L/d. The particle tracking flow paths originating at each source are shown in different colors. Results are shown for simulations with constant recharge and a) without tides, and b) with tides.

Based on the particle tracking results the average travel times for N to be transported from each wastewater source to the lagoon decreased when multiple sources were simulated rather than simulating the sources individually (Figure 3.9a). This is likely due to the higher combined wastewater input from the nine sources resulting in a higher hydraulic gradient across the lagoon side of the transect and therefore faster flow velocities between the sources and lagoon. The average travel times are lower for the tidal simulations compared to the non-tidal simulations for source locations greater than 150 m from the lagoon (Figure 3.9a). This is thought to be because, for the tidal simulations, N originating from the sources greater than 150 m from the lagoon is transported downwards and reaches the lower high permeability Pleistocene layer (K = 750 m/d) where N is then transported rapidly horizontally towards the lagoon (Figure 3.10b). For the non-tidal simulations N flow paths do not reach the lower high permeability Pleistocene layer even for the more inland wastewater sources (Figure 3.10a). Generally, although the travel times for the simulations considering multiple sources are lower than those for the simulations considering the sources individually, the travel times are still sufficiently long for N attenuation (denitrification) to reduce the N flux to the lagoon particularly for wastewater sources farther inland (Meile et al., 2010).

As expected, the N exit concentrations at the lagoon are considerably higher when all nine wastewater sources are simulated rather than only one source. For the non-tidal simulations, the maximum N exit concentration when the multiple wastewater sources were considered was 0.1 g/L whereas it ranged from 1.4×10^{-2} - 3.3×10^{-2} g/L when the sources were simulated individually (Figure 12b). For the tidal simulations, the maximum N exit concentration was 7×10^{-2} g/L when the multiple wastewater sources were considered whereas it ranged from 2×10^{-3} to 2.6×10^{-2} g/L when the sources were simulated individually (Figure 12b). Consistent with the simulations when the wastewater sources were simulated individually, variable recharge only had a minor effect on the N exit concentrations compared to tides (Figure 12b).

3.3.3 Conclusions

The results from this study primarily highlight the importance of including tidal fluctuations in modelling the saltwater-freshwater dynamics in atoll islands and in modelling groundwater contaminant transport from inland sources to an atoll lagoon or the ocean. Future modelling studies of Fongafale Islet and other atoll islands should consider tidal fluctuations (and possibly other coastal forcing including storm surges), while variable interannual recharge patterns were found to be a minor factor for determining salinity and contaminant transport compared to tides. The results also show that the salinity distribution and N transport pathways are a strong function of the distinct hydrogeology of atoll islands including the configuration of high and low permeability zones across the island. The simulations showed that the maximum exit concentrations of N in the groundwater discharging to Fongafale Lagoon may be higher than concentrations previously measured in the coastal lagoon (N'Yeurt & Iese, 2015) suggesting that groundwater discharge may be delivering N-rich waters into the lagoon. However, it is important to note that the simulated N exit concentrations represent an upper conservative value as no attenuation of N was considered in the model. It is recommended that household surveys on atoll islands be conducted to better characterize wastewater loading rates from septic systems and be able to better evaluate the discharge of N wastewater plumes to adjacent coastal waters. Further, to improve the accuracy of modelling studies for Fongafale Islet and other atoll islands, future numerical studies should also include reactive transport modelling that considers N attenuation. While the main findings of this study are based on Fongafale, Tuvalu, the conceptual results are also relevant for other atoll island settings that have similar hydrogeological and hydrological conditions.

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Chapter 4

4 Summary & Recommendations

4.1 Summary

The saltwater-freshwater dynamics and contaminant transport pathways in atoll island aquifers differ from continental coastal aquifers due to the unique hydrogeology of atoll islands and the high connectivity between the aquifer and ocean. While many studies have investigated factors that influence the fresh groundwater lens and thus fresh groundwater resources on atoll islands, there is little knowledge of the interacting impacts from transient effects such as tides and variable recharge patterns on the fresh groundwater resources and limited understanding of contaminant transport pathways in atoll islands. This study aimed to address the following two objectives: 1) Evaluate the influence of tides and variable (interannual) recharge patterns on fresh groundwater resources in an atoll island; 2) Evaluate the subsurface transport of septic system-derived N in an atoll island groundwater system and its delivery to adjacent coastal waters including impact of tides, variable recharge patterns, wastewater loading rate and wastewater source locations. These objectives were addressed by developing a numerical variable-density flow and conservative N transport model based on available hydrogeological knowledge and data for Fongafale Islet, Tuvalu.

Previous variable-density groundwater flow models of the atoll islands have typically focused on estimations of fresh groundwater sustainable yield and the impacts of climate variability and climate change on the fresh groundwater lens (Werner et al., 2017). The impacts of transient forcing including tides and variable recharge patterns and the impacts of water inputs from anthropogenic sources including wastewater on the availability of fresh groundwater resources have not previously been examined in detail. This study demonstrated how tidal fluctuations facilitate considerable freshwater-saltwater mixing in the aquifer which increases the size of the mixing zone and reduces the zone of low salinity groundwater in the surficial aquifer. The inclusion of wastewater injection in simulations demonstrated its large influence on atoll island freshwater-saltwater dynamics with the surficial low salinity region considerably expanding with an increase in the wastewater loading rate and when more wastewater sources were simulated.

The potential importance of groundwater in delivering contaminants including nitrogen (N) into coastal lagoons adjacent to atoll islands is poorly understood. This study showed how the specific subsurface N transport pathway is different in an atoll island compared to a more homogeneous continental coastal aquifer due to the distinct hydrogeological structure of atoll islands. Further, the N transport pathway between a land-based wastewater source and the adjacent coastal lagoon is modified by tidal fluctuations and tide-induced oscillatory flows increasing mixing and dispersion of the N plume before it reaches the lagoon. The N flow path extended deeper into the aquifer when the wastewater loading rate was higher and expanded both vertically and horizontally for wastewater sources located farther from the lagoon. Maximum N exit concentrations in groundwater discharging to the lagoon were about four times lower when tides were considered for simulations with constant recharge and a single wastewater source. Aside from the impact of tides, N exit concentrations at the lagoon increased when the wastewater source was closer to the lagoon and when multiple wastewater sources were simulated. The maximum exit concentrations for all simulations were considerably higher than NO_3^- concentrations previously measured in the Fongafale lagoon (N'Yeurt & Iese, 2015) suggesting the potential importance of groundwater in delivering N-rich water to the lagoon. However, only conservative N transport simulations were conducted and therefore the simulated N exit concentrations represent an upper limit on the N concentrations in groundwater discharging to the lagoon. The average travel times for N to travel from a wastewater source to the lagoon increased as the wastewater source moved further inland, as the wastewater loading rate decreased and when tides were considered. The results highlight the need to prioritize upgrades for household septic systems located closest to the lagoon and the need to consider tidal fluctuations when evaluating N transport pathways in atoll island aquifers. As discussed by Meile et al. (2010), higher travel times are typically expected to lead to higher N attenuation in the aquifer (e.g. denitrification) that can lower the groundwater-derived N loads to the lagoon. Finally, consideration of variable recharge patterns rather than constant recharge did not considerably affect the overall fate and transport of N in the aquifer although maximum N exit concentrations showed a delayed response to the most extreme shifts in recharge over the simulation period.

The findings of this study will contribute to informing future numerical modelling, field studies, and planning and management strategies for wastewater infrastructure and pollution control on

Fongafale Islet and on other atoll islands dealing with eutrophication of their marine ecosystems. Water quality managers and decision makers will be better informed on the potential role of groundwater in delivering N from septic systems to adjacent lagoons as needed to prioritize future testing of groundwater quality and to improve wastewater treatment infrastructure programs and thus water quality in coastal lagoons adjacent to atoll islands.

4.2 Recommendations

The model simulations presented in this study provide a base conceptual understanding of processes that affect the freshwater-saltwater dynamics in an atoll island aquifer and N transport and discharge to an atoll island lagoon. It is recommended that future work aiming to better predict fresh groundwater availability in atoll islands and to evaluate the contribution of groundwater discharge to N loading to coastal lagoons consider the following:

- The simulations in this study only consider conservative N transport while reactive transport simulations should be performed in future studies to improve prediction of the fate and transport of N in the aquifer and its ultimate discharge to the coastal lagoon.
- Field sampling in lagoons and groundwater monitoring should be performed to better characterize the hydrogeology of atoll islands and to better understand groundwater salinity and quality including N concentrations and geochemical conditions. Modelling studies would be better informed and validated with this data to improve prediction of the transport and discharge of contaminants in wastewater plumes to adjacent coastal waters.
- Household surveys should be performed to better understand septic system sources in atoll island communities including wastewater loading rates and contaminant concentrations in wastewater entering the subsurface. This will inform future predictions of contaminant pathways within atoll islands in addition to directly informing the decision-making for prioritizing the most effective solutions for improving lagoon water quality
- Additional transient forcing should be included in simulations including variable wave conditions, variations in tidal amplitude, long-term sea level rise and coastal flooding events. It is unclear the impacts of these transient forcing on the freshwater-saltwater dynamics and N transport pathways in atoll island aquifers.

- Three dimensional models of the atoll island subsurface should be developed to observe N transport and discharge from multiple wastewater sources across densely populated areas such as Fongafale Islet. This more advanced modeling approach will provide further insights into the transport, and discharge of contaminants such as N to the adjacent coastal waters.

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Appendices

Appendix A. SEAWAT and MT3DMS governing equations

The governing equation used by SEAWAT-2005 for variable-density flow in terms of freshwater head is:

$$\nabla.\left[\rho K_f\left(\nabla.h_f + \frac{\rho-\rho_f}{\rho_f}.\nabla z\right)\right] = \rho S_f\frac{\partial h_f}{\partial t} + n_e\frac{\partial\rho}{\partial t} - \rho_s q_s$$

Where:

ρ is the fluid density [ML^{-3}]

K_f is the equivalent freshwater hydraulic conductivity [LT^{-1}]

h_f is the equivalent freshwater head [L]

ρ_f is the freshwater density [ML^{-3}]

z is the vertical coordinate in the upward direction [L]

S_f is the equivalent freshwater storage coefficient [L^{-1}]

t is time [T]

n_e is the effective porosity

ρ_s [ML^{-3}] and q_s [T^{-1}] are the density and flow rate per unit volume of aquifer of a source or sink

(Langevin, 2003)

Solute transport in SEAWAT and MT3DMS are simulated using the following partial differential equation (Guo & Langevin, 2002):

$$\frac{\partial C}{\partial t} = \nabla \cdot (D \cdot \nabla \mathrm{C}) - \nabla \cdot (\vec{v}\mathrm{C}) - \frac{q_s}{\theta} C_s + \sum_{k=1}^{N} R_k$$

where:

C is solute concentration [ML^{-3}]

t is time [T]

D is the hydrodynamic dispersion coefficient [L^2T^{-1}],

$\vec{v}$ is the fluid velocity [LT^{-1}],

q_s is the volumetric flow rate per unit volume of aquifer representing sources and sinks [T^{-1}]

θ is porosity

C_s is the solute concentration of water entering from sources or sinks [ML^{-3}], and

R_k (k=1, ..., N) is the rate of solute production or decay in reaction k of N different reactions [$ML^{-3}T^{-1}$]

Appendix B. Sensitivity of salinity distribution to hydraulic conductivity (*K*) values and dispersivity ($\alpha_{L,}$ α_T)

The size of the freshwater lens increases as *K* decreases for both tidal and non-tidal case, even though freshwater lens is minimal for the tidal case (Figure B1). With lower *K* values more freshwater is retained close to the surface and saltwater is prevented from mixing with it as easily. Dispersivity has similar impact on the size of freshwater lens, where lower dispersivity resulted in a larger freshwater lens due to limited saltwater-freshwater mixing (Figure B2). Both *K* and dispersivity control the saltwater-freshwater mixing, where mixing is limited in low *K* and low dispersivity cases minimizing the saltwater infiltration and spread in the aquifer. These observations were consistent with those made by Bailey & Jenson (2014) in sensitivity simulations of lens recovery from an overwash event where lower *K* and lower dispersivities resulted in a thicker freshwater lens. It is worth noting that for both cases of *K* and dispersivity, freshwater lens was reduced when tides were considered, which was consistent with observations in Figures 3.4, 3.5, and 3.6. The highest *K* values used in Figures B1c,f and the lower α_L and α_T values in Figure B2a,c were chosen to be used as final model parameters. These results showed reasonable agreement with how the Fongafale Islet hydrogeology was described by Nakada et al. (2012) and the range of *K*, $\alpha_{L,}$ and α_T used in atoll island groundwater modelling (Bailey et al., 2009; Bailey & Jenson, 2014; Jazayeri et al., 2019; Post et al., 2018).

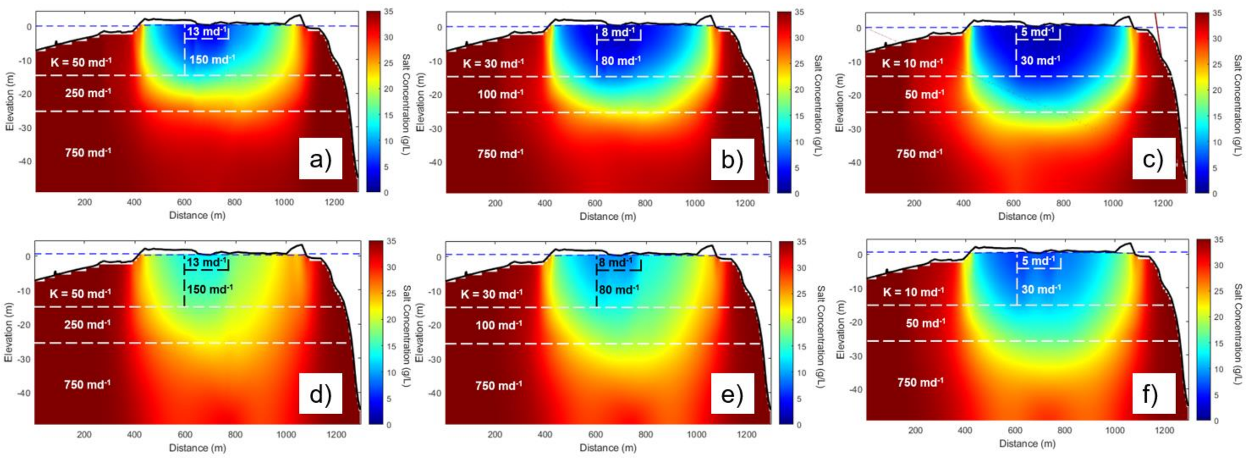

Figure B1. Salt concentration profiles at different K levels a) high K. b) mid-range K. and c) low K without tides, and d) high K e) mid-range K f) low K with tides. The white and black dashed lines show the different K zones with the white text indicating the corresponding K_H values.

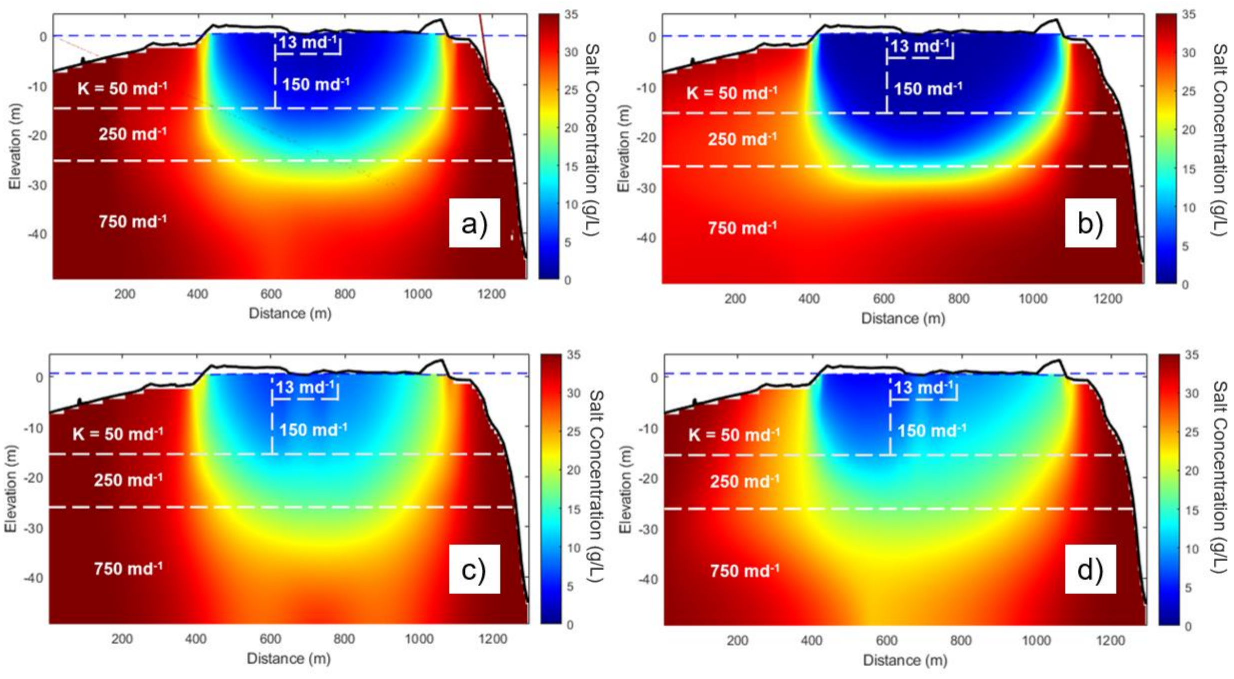

Figure B2. Salt concentration profiles at different dispersivities a) high dispersivity ($\alpha_L = 12$ and $\alpha_T = 1.2$), and b) low dispersivity ($\alpha_L = 5$ and $\alpha_T = 0.5$) without tides c) high dispersivity, and d) low dispersivity with tides

Appendix C. Data calculations for numerical model

Average daily recharge

Average annual rainfall was calculated to be 3150 mm/yr from 1901 to 2016 (The World Bank Group, 2017). Based on this the average daily rainfall was calculated with average daily recharge taken as 50% of this value at 4.1 mm/d.

Average wastewater loading rate

The average water consumption for a small household was taken as 350 L/d from a Water Security report by the Department of Environment in Tuvalu (Kinrade et al., 2021). This total water consumption is assumed to all be released directly to the subsurface from the on-site sewage system at a Fongafale Islet household (Fujita et al., 2013). The numerical model only simulates a unit width of the subsurface and therefore of the septic system as well, thus the water consumption rate to be injected into the aquifer model was considered as a fifth of the total rate (70 L/d). Sensitivity analysis of the wastewater loading rates shown in Appendix E and Figure 3.7 demonstrate that there is a difference is the average travel time, exit concentration and, to a lesser extent, flow paths, between 70 L/d and 350 L/d wastewater loading rates. A wastewater loading rate of 70 L/d is chosen for the model simulations to accurately represent a unit width of the total septic system loading of 350 L/d.

N concentration in effluent

The N concentration in Tuvalu wastewater was determined from the N content in the Tuvaluan diet, the average household size on Fongafale Islet and the average household water usage. The quantity of protein in the Tuvaluan diet was found to be 95 g/person/day (Tuvalu Food Security Profile, 2020). The N content in food is linearly correlated with the protein content thus N content in excreta is commonly calculated from the protein content in food supplies (Jönsson & Vinnerås, 2003). From all the countries examined by Jönsson & Vinnerås (2003), the protein content of the Swedish diet (98 g/person/day) was closest to that of Tuvaluans thus the protein to nutrient ratio from Sweden was used to calculate the N content for Tuvalu. The average number of residents per household was calculated to be 7.4 from the Tuvalu Population and Housing Census (2012) data of the total population and number of households in the Senala and Alapi villages of Fongafale. The N concentration in Tuvalu wastewater was calculated as follows:

$$wastewater\ N\ concentration = \frac{12.31\,\frac{\mathrm{g\ N}}{\mathrm{person} * \mathrm{d}}}{7.4\ people/household \times 350\mathrm{L/d}} = 0.26\ g/L$$

Where:

$12.31\,\frac{\mathrm{g\ N}}{\mathrm{cap}*\mathrm{d}}$ is the daily N content per person in the Tuvaluan diet

$7.4\,\frac{people}{household}$ is the average Fongafale household size

350 $\mathrm{L/d}$ is average daily household water usage

Appendix D. N exit concentrations at lagoon over time for variable recharge simulations

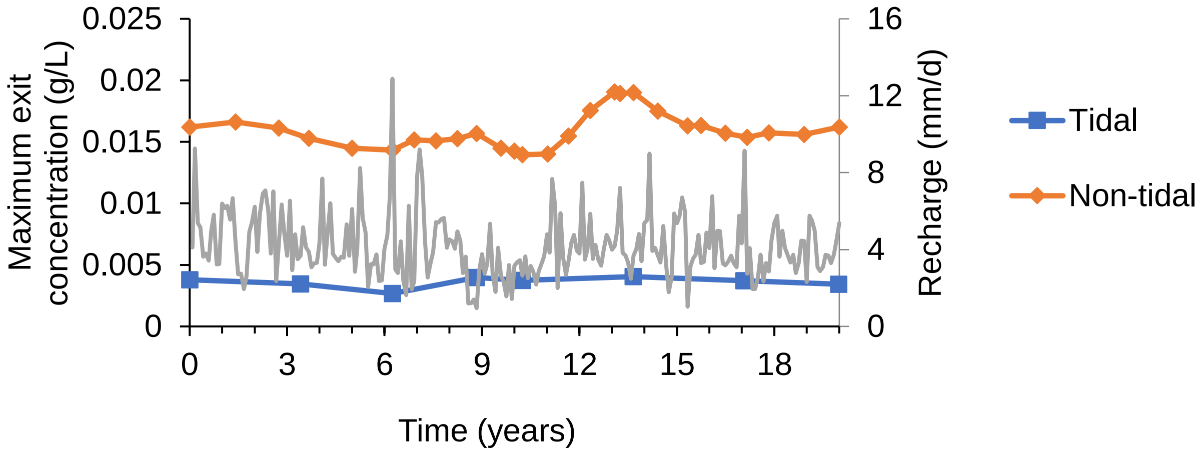

Figure D1. Maximum N exit concentrations over time in response to interannual variable recharge. Exit concentrations are presented for non-tidal and tidal simulations.

Appendix E. Sensitivity of N pathways to wastewater loading rate

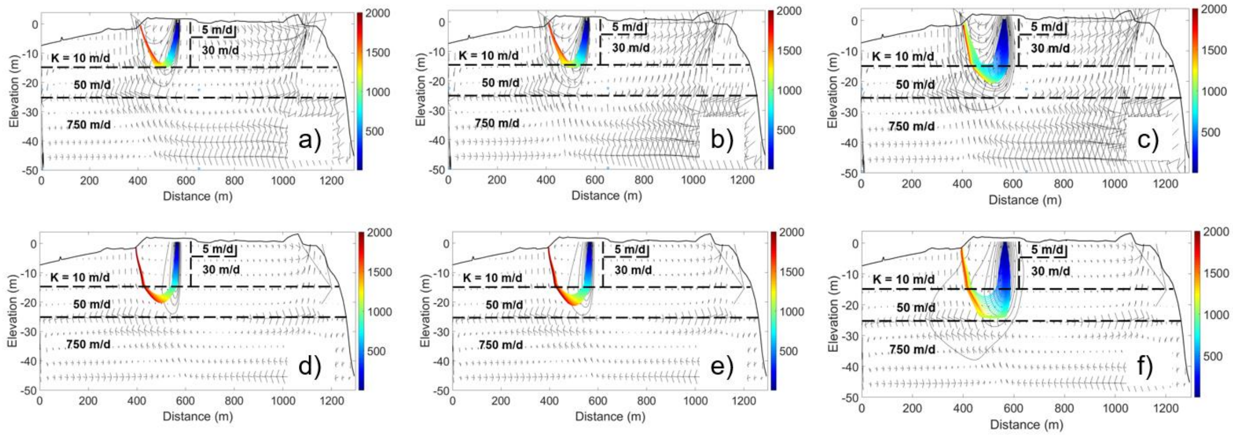

Figure E1. Groundwater velocities and N flow paths at different wastewater loading rates. Results are shown for simulations conducted with one septic system located at 150 m from the lagoon shoreline and with wastewater loading rates of a) 35 L/d, b) 70 L/d, and c) 350 L/d without tides, d) 35 L/d, e) 70 L/d, and f) 350 L/d with tides

Appendix F. Simplified analytical solution for N travel time through an oceanic strip island

Calculated travel times are based on Darcy's Law describing groundwater discharge through the unconfined aquifer of an strip oceanic island The solutions apply for a homogenous aquifer with surface recharge infiltration and discharged by a down-gradient, fixed-head boundary, under steady-state conditions (Chesnaux & Allen, 2008). This analytical solution was used to calculate the travel time through the Fongafale Islet groundwater with variables to match the simulated domain as closely as possible. The equation used is the general closed-form solution of the travel time of water ($t_{SOI}(x)$) between the initial position x_i and a position x in a strip oceanic island aquifer that is unconfined, horizontal, and recharged by infiltration (Chesnaux & Allen, 2008).

$$t_{SOI}(x) = n_e\sqrt{\frac{\rho_f + \Delta\rho}{WK\Delta\rho}} \times \left[\sqrt{L^2 - x^2} - \sqrt{L^2 - x_i^2} - L\ln\left(\frac{L + \sqrt{L^2 - x^2}}{L + \sqrt{L^2 - x_i^2}}\frac{x_i}{x}\right)\right]$$

Where:

n_e is the effective porosity = 0.2

ρ_f is the freshwater density [ML^{-3}] = 1000 g/L

$\Delta\rho$ is the difference in fluid density between freshwater and saltwater [ML^{-3}] = 25 g/L

K is the hydraulic conductivity [LT^{-1}] = 10 m/d

W is the surface recharge rate [LT^{-1}] = 0.0041 m/d

L is the half-width of the oceanic strip island [L] = 332.5 m

x is the final position of the water [L] = 332.5 m

x_i is the initial position of the water (starting from the centre of the island) [L] = varying

$$t_{SOI}(x_i) = 0.2\sqrt{\frac{1000 + 25}{0.0041 \times 10 \times 25}}$$

$$\times \left[\sqrt{332.5^2 - 332.5^2} - \sqrt{332.5^2 - x_i^2} \right.$$

$$\left. - 332.5 \times ln\left(\frac{332.5 + \sqrt{332.5 - 332.5^2}}{332.5 + \sqrt{332.5^2 - x_i^2}} \frac{x_i}{332.5} \right) \right]$$

Distance from lagoon	x_i	t
25	307.5	42.33455
50	282.5	124.3498
70	262.5	212.7863
100	232.5	383.1787
120	212.5	523.7471
135	197.5	644.9544
150	182.5	781.2783
175	157.5	1047.59
200	132.5	1374.754

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