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Hydrogeology and management of freshwater lenses on atoll islands: Review of current knowledge and research needs

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ABSTRACT

On atoll islands, fresh groundwater occurs as a buoyant lens-shaped body surrounded by saltwater derived from the sea, forming the main freshwater source for many island communities. A review of the state of knowledge of atoll island groundwater is overdue given their susceptibility to adverse impacts, and the task to address water access and sanitation issues within the United Nations' Sustainable Development Goals framework before the year 2030. In this article, we review available literature to summarise the key processes, investigation techniques and management approaches of atoll island groundwater systems. Over fifty years of investigation has led to important advancements in the understanding of atoll hydrogeology, but a paucity of hydrogeological data persists on all but a small number of atoll islands. We find that the combined effects of buoyancy forces, complex geology, tides, episodic ocean events, strong climatic variability and human impacts create highly dynamic fresh groundwater lenses. Methods used to quantify freshwater availability range from simple empirical relationships to three-dimensional density-dependent models. Generic atoll island numerical models have proven popular in trying to unravel the individual factors controlling fresh groundwater lens behaviour. Major challenges face the inhabitants and custodians of atoll island aquifers, with rising anthropogenic stresses compounded by the threats of climate variability and change, sea-level rise, and some atolls already extracting freshwater at or above sustainability limits. We find that the study of atoll groundwater systems remains a critical area for further research effort to address persistent knowledge gaps, which lead to high uncertainties in water security issues for both island residents and surrounding environs.

1. Introduction

Atolls are low-lying reef-carbonate land areas that usually consist of a series of narrow islands surrounding a relatively shallow seawater lagoon. Amongst the 425 atolls of the world, most of them are located in the Pacific and Indian Oceans (Falkland, 1991; Falkland, 1992a). When precipitation recharge across an atoll island is sufficient, a fresh groundwater lens (FGL) will form, with freshwater floating above denser, saline groundwater derived from the sea. Rainfall and FGLs are the two main sources of freshwater for atoll island communities. As rainfall patterns can be irregular, FGLs form an indispensable natural buffer for the provision of freshwater during drought periods. Human existence and the health of groundwater-dependent ecosystems on atoll islands rely intrinsically on the state of groundwater resources, particularly given that atolls tend to be geographically isolated and highly exposed to natural disasters (Bruggeman and Custodio, 1987; Mimura et al., 2007; White et al., 2007a; Terry and Falkland, 2010).

In atoll islands, as in all small islands, the FGL is usually thin, and a mixing zone separates the FGL from the underlying seawater. The size and shape of atoll FGLs and the dynamics of the mixing zone are the result of density-dependent flow and transport processes that are strongly influenced by oceanic tidal forces, amongst other compounding factors (e.g., Falkland, 1991), of which heterogeneity of the subsurface sediments is perhaps the most important. Atoll FGLs are highly susceptible to modification from numerous natural and anthropogenic controlling factors, in terms of both their thickness and chemical composition. The most critical natural stresses are drought and ocean overtopping, whereas anthropogenic stresses include groundwater extraction, land-use and vegetation change, pollution, excavation for mining, land reclama-

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tion, and aquaculture (Falkland, 1991; Terry and Falkland, 2010; Chui and Terry, 2013). Small islands are also among the most vulnerable hydrological settings to climate change impacts, which include rainfall that is more variable, higher potential evaporation, more frequent and intense storms, and rising sea levels (Mimura et al., 2007; Abtew and Melesse, 2013).

The most comprehensive previous attempts to summarise the state of knowledge of atoll FGLs are the report by Falkland (1991) and the book by Vacher and Quinn (2004), first published in 1997. These works include an extensive overview of the geology, hydrology, management and investigation of atoll FGLs, and other small island types. Falkland (1991) found that numerical modelling of atoll FGLs was limited in the international literature at that time, particularly three-dimensional models. Recommendations for additional small island research included improved estimation of recharge and exploitable fresh groundwater, optimal design and use of pumping infrastructure, evaluation of freshwater discharge to the sea, and advanced application of numerical modelling. Underwood et al. (1992) used a variable-density numerical model to explore the effect of various hydrogeological parameters identified by Falkland (1991) on the size of the FGL, focussing particularly on the mixing zone and the propagation of tides through the subsurface. This study clearly showed the dominant role of tides in mixing-zone behaviour and the importance of including tidal processes in atoll groundwater modelling. Falkland (2002b) presented a summary of the main aspects of small island hydrology covered in detail by Falkland (1991), extending the discussion to review advances and future needs in hydrological management, research, education and training. Bailey et al. (2009) added to Falkland's (1991) study by applying numerical modelling to undertake a sensitivity analysis of atoll FGL thickness with respect to various hydrogeological parameters, using Laura Island (Marshall Islands) as the base case for simulations. Their review encountered ten prior modelling case studies, which they described. Bailey et al.'s (2009) sensitivity analysis results provide insights into some of the FGL controlling factors and rates of FGL recovery following El Niño drought events. White and Falkland (2010) also built on Falkland's (1991) report by summarising the characteristics of, and threats to, Pacific FGLs. They used Tarawa Atoll (Kiribati) to highlight critical elements of FGL behaviour and the implications for managing threats to FGL sustainability.

The motivation for the current review is a combination of the widely acknowledged threat of climate change impacts on atoll islands, the high vulnerability of atoll islands to adverse changes, and the increasing reliance by local communities on FGLs. The very high population densities of many atoll communities, which in many cases exceed sustainable limits, compound this last factor. Significant challenges for atoll inhabitants exist within the targets formulated under the United Nations' Sustainable Development Goal 6 on water access and sanitation, which prescribe that access to safe and affordable drinking water should be realised for all citizens by 2030. This review aims to define persistent knowledge gaps in atoll island hydrogeology through a summary of previous literature, thereby establishing guidance for future research direction and highlighting barriers to the effective management of atoll FGLs. Key focal points are FGL processes, methods of investigation, and management. The summary of prior research on atoll island hydrogeology seeks to provide background knowledge for FGL investigations that support the development of water and land management strategies. We add to the seminal works by Falkland (1991) and Vacher and Quinn (2004) by examining advances that have occurred during the last two decades. The numerical modelling studies by Underwood et al. (1992) and Bailey et al. (2009) are extended by considering a wider range of processes, methodologies and sustainability concepts of relevance to the management and investigation of atoll FGLs. Our analysis of the factors and processes specific to atoll island FGLs and our summary of modelling approaches adds to reviews of small islands by Falkland (1991, 1999, 2002b), and White and Falkland (2010).

2. Physiography and geology of atoll islands

Atolls are typically composed of a ring of small, low-relief coral islands surrounding a lagoon, enclosing it either partially or fully (Barnett and Adger, 2003). The maximum elevation of atoll islands is generally only several metres above sea level, with land areas commonly less than 1 km^2 (Wheatcraft and Buddemeier, 1981; Vacher, 2004; White and Falkland, 2010). The islands vary considerably in shape and size, ranging between 100 and 1500 m in width, and up to several kilometres in length (Bailey et al., 2008). They can be a continuous rim of connected or near-connected islands, or a number of islands separated by significant distances but all adjoining the same relatively shallow lagoon (Woodroffe, 2008). In some cases, atolls may also have islands in the middle of the lagoon, as observed in the Maldives (Woodroffe, 2008). The surface area of atoll lagoons is generally considerably larger than the land area, varying between 2 and 1000 km^2, and having depths of up to 100 m (Purdy and Winterer, 2001).

Atoll islands differ from other small islands in that they are generally composed of two overlying aquifer formations, comprising un- or poorly consolidated Holocene sediments deposited unconformably on Pleistocene limestone reef deposits (Ayers and Vacher, 1986; White and Falkland, 2010). The contact between the Holocene and underlying Pleistocene sediments, known as the 'Thurber discontinuity' or the 'Holocene-Pleistocene unconformity' (HPU), typically occurs between 15 and 25 m below sea level (Ayers and Vacher, 1986; Anthony, 2004; Buddemeier and Oberdorfer, 2004). The dual-aquifer configuration of atoll islands has been encountered in an extensive number of cases (e.g., Buddemeier and Holladay, 1977; Hunt and Peterson, 1980; Lloyd et al., 1980; Wheatcraft and Buddemeier, 1981; Oberdorfer and Buddemeier, 1983; Falkland, 1991). Below the dual-aquifer system are further carbonate sequences above a volcanic foundation, remnant of the atoll's formation origins (Vacher, 2004).

Darwin's evolutionary sequence of reefs reflects the geological development of atolls, whereby atoll structure is the result of gradual subsidence of volcanic foundations and upward growth of reefs (Falkland, 1991; Vacher, 2004; Woodroffe, 2008). Fig. 1 illustrates this sequence and provides modern examples of each phase. Following the formation of a volcanic island (Fig. 1a), the volcanic core begins to subside and a fringing reef develops that borders the island (Fig. 1b). The process continues until the volcanic rocks are buried by the reef (Fig. 1c); on top of which granular sediments form the low-lying islands, which typically surround a central lagoon in a ring structure (Fig. 1d). While the ring structure is characteristic of most atolls, there are some variations. For example, there may be a single island on a small reef, or the lagoon may be filled with sediment and occur as a residual feature (Woodroffe, 2008). This is the situation at Kiritimati (Kiribati), where the lagoon has partially filled up and the Pleistocene limestone is exposed (Vacher, 2004).

In geological terms, the atolls that exist today formed in relatively recent times. Changes in eustatic sea level played an essential role in the formation and current structure of atolls, as illustrated in Fig. 2 (Woodroffe, 2008). Atoll rims, similar to the structure of modern atolls, existed when the sea level was high during previous interglacial periods (Fig. 2a). Falling sea levels during the last glacial period exposed reef platforms and the emerged limestone underwent karstification (Fig. 2b). Reef growth re-established during the Holocene period when sea levels rose. Radiocarbon dating of coral samples indicates that prolific reef

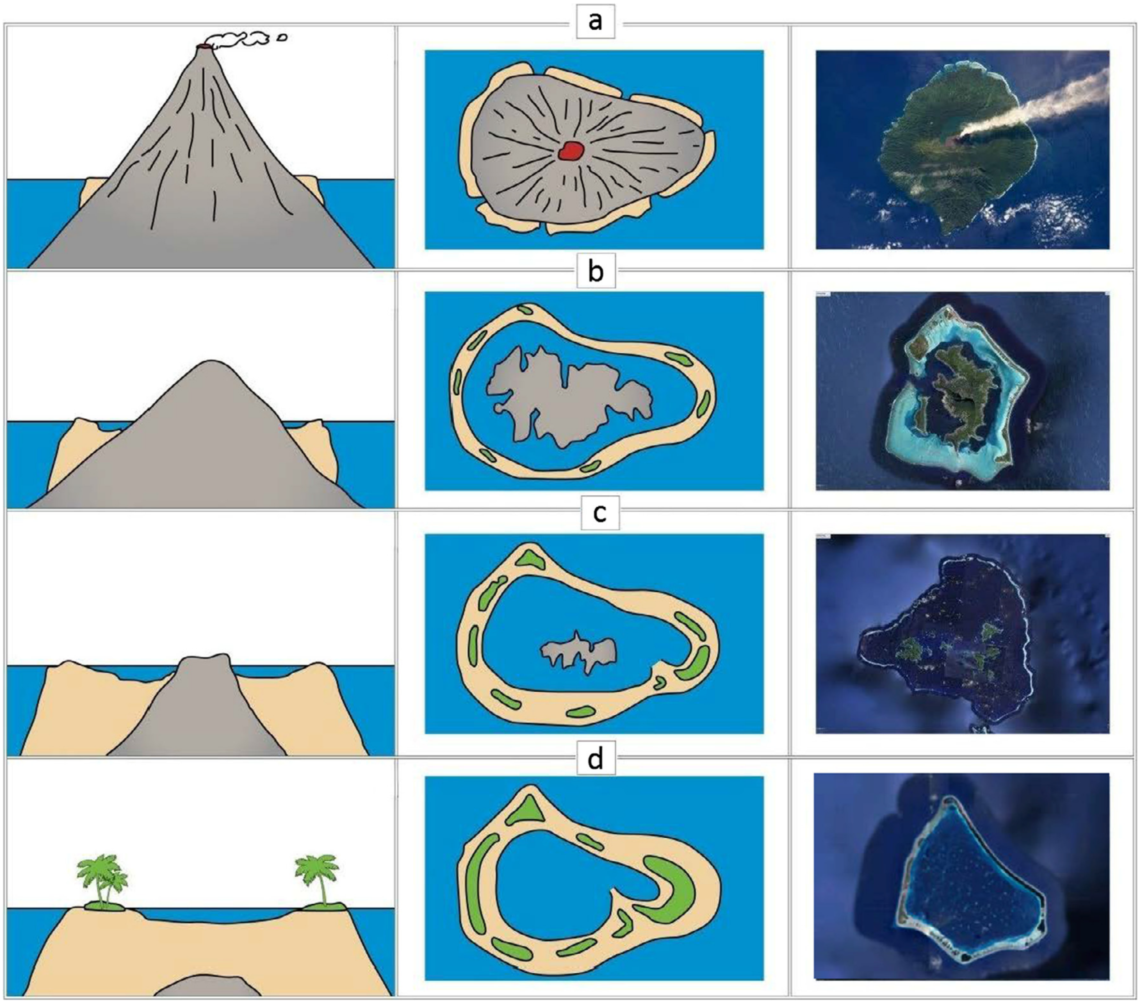

Fig. 1. The four stages in the geological history of an atoll: (a) Volcanic island formation, e.g., Gaua Island, Vanuatu (photo from NASA (2016)); (b) Subsidence of the volcanic core, e.g., Bora Bora, French Polynesia (Google Earth, 2016a); (c) Continued subsidence of the volcanic core and formation of the fringing reef, e.g., Chuuk Islands, Federated States of Micronesia (Google Earth, 2016b); (d) Final ring-like structure, e.g., Manihiki Atoll, Cook Islands (Google Earth, 2016c). Left and middle sub-figures are schematic illustrations in cross section and plan, respectively, and the right sub-figures are overhead photographs. Figure adapted from Ayers (1984) and Falkland (1991).

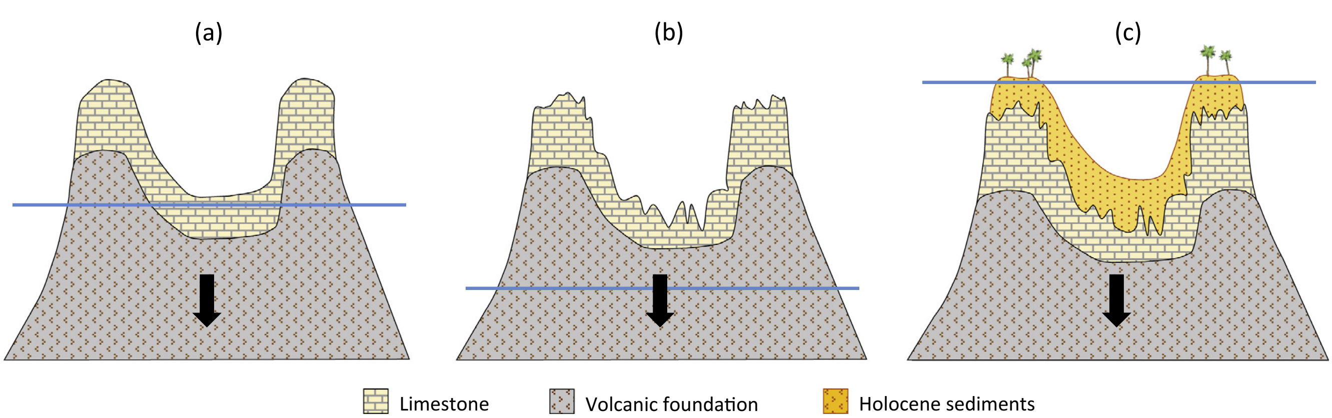

Fig. 2. Effects of eustatic sea-level (blue line) changes on the development and structure of atolls: (a) Occurrence of atoll rims in the final Pleistocene period that are similar to modern atoll rims; (b) Reef platform exposed with lower sea levels due to glaciation, leading to karstification of the exposed limestone; (c) Reef growth re-established with Holocene sediments deposited onto the underlying Pleistocene structure. The black arrow represents the subsiding volcanic foundation. Modified from Woodroffe (2008). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

growth occurred approximately 8000 y ago (Woodroffe and Falkland, 2004) (Fig. 2c). Thus, the HPU is remnant of these glacio-eustatic sea-level changes.

The subsurface of atolls is highly heterogeneous (e.g., Weber and Woodhead, 1972). Much of the heterogeneity in the Pleistocene limestone arises from the karstification processes that occurred during the Pleistocene and some Holocene periods (Falkland, 1991), when the reef sediments were exposed (Fig. 2b). The formation of this significant secondary porosity resulted in a pronounced difference in permeability between the Holocene and Pleistocene sequences (Ayers and Vacher, 1986). Unaltered Holocene sediments are moderately permeable, but lose porosity and permeability with cementation. The cemented layers can be karstified with time, thereby enhancing the permeability and heterogeneity through the development of fissures (Falkland, 1991). Holocene sediments are also often characterised by sequences of horizontal sublayers, which impart strong anisotropy within stratigraphic units (Underwood, 1990; Underwood et al., 1992). In Enjebi Island (Enewetak Atoll, Marshall Islands), the heterogeneity of reef deposits occurs as short-range vertical and horizontal variations in lithology, and through the occurrence of several solution unconformities (Buddemeier and Oberdorfer, 1986). Specifically, there are significant variations in texture and lithification degree, with large voids and karst conduits that are irregularly spaced and highly variable in extent and dimension. In Tarawa Atoll (Kiribati), Marshall and Jacobson (1985) found that facies changes between corals and other unconsolidated sediments along the island margins, caused by unsynchronised deposition of these sediments, compounds Holocene sediment heterogeneity. Falkland (1991) also encountered horizontal variability in sediment deposition across Tarawa Atoll, with finer sediments deposited on the lagoon side, and coarser sediments more prevalent on the ocean side. This is a general pattern (e.g., Weber and Woodhead, 1972; Ayers and Vacher, 1986), and Bailey et al. (2010) reported a similar finding for leeward and windward oriented islands; the former composed of much finer sediments than the latter due to protection from wind and waves.

An additional geological feature of many atolls is the reef-flat plate, a semi-permeable reef rock that forms within the Holocene aquifer. According to Woodroffe (2008), the reef flats formed once reef growth caught up with post-glacial sea-level rise (SLR). As the sea level rose during the early Holocene period, ambient low tide levels exposed the growing reef, and lateral progradation of the reef occurred (Woodroffe, 2008). In Deke Island (Pingelap Atoll, Federal States of Micronesia), the reef-flat plate has been characterised as a dense rock extending from offshore on the ocean side to partway across the island, with low effective porosity and very low permeability caused by marine cementation (Ayers and Vacher, 1986). In Tarawa Atoll (Kiribati), the reef-flat plate extends to depths of 3 m below mean sea level, and outcrops on the edges of the islands forming terraces of up to 2 m above sea level (Marshall and Jacobson, 1985). Reef-flat plates also occur on Arno Atoll (Marshall Islands; Cox, 1951) and on islands of Davies Reef (Great Barrier Reef, Australia; Buddemeier and Oberdorfer, 1986), amongst others. Reef-flat plates can partially confine the Holocene aquifer, influencing groundwater occurrence and flow (Bailey et al., 2009).

Of particular importance in the development history of atoll islands is the sea-level high-stand of up to 3 m above the present-day sea level, during the mid-Holocene. It has been identified in sea-level reconstructions across the Indo-Pacific region, albeit with varying estimates of its timing and magnitude (Woodroffe and Horton, 2005). The cause of the high-stand is multi-faceted but it occurred when eustatic SLR slowed and other effects gained greater relative importance. These include the global redistribution of oceanic water as melting of the ice caps changed the earth's gravitational field, and the gradual isostatic adjustment of the ocean floor caused by the loading of the added ocean water mass. The fall in relative sea level after the mid-Holocene high-stand is believed to have been the main trigger for island emergence across the Indian and Pacific oceans. However, a competing hypothesis states that, at least in some cases, island formation already occurred when sea levels were still rising (Kench et al., 2014; Webb and Kench, 2010). This is an area of scientific debate that is highly relevant to the understanding of the future fate of atoll islands and their FGLs, particularly in the context of predicted SLR during the 21st century and beyond (IPCC, 2014).

3. Characteristics of atoll FGLs

Three key factors control the size and shape of FGLs within atoll islands: (1) the physiographic and hydraulic properties of the dual-aquifer system, (2) the hydrodynamic and dispersive processes accompanying freshwater-seawater interactions, and (3) external forces that modify inflows, discharge and boundary conditions. Studies that elucidate FGL physical processes and atoll island characteristics pertinent to FGL extent and behaviour are summarised in the following sub-sections.

3.1. Freshwater-seawater mixing zone

The mixing (or transition) zone that separates freshwater and seawater in atoll FGLs influences the depth to which potable water is available (Vacher, 2004). Unlike the sharp interface depicted by the Ghyben-Herzberg (GH) relationship (Drabbe and Badon Ghijben, 1888; Herzberg, 1901; see Section 4), the mixing zone contains a gradual transition in salinity from freshwater to seawater (Falkland, 1992b; White and Falkland, 2010). Buddemeier and Oberdorfer (1986) suggest that the mixing zone, which they defined by salinities between 2.5% and 95% of seawater concentration, may occupy a considerably greater volume of atoll island aquifers compared to the freshwater component (e.g., <2.5% of seawater concentration). For example, the mixing zone in some areas of the Kwajalein Atoll (Marshall Islands) is double the thickness of the FGL (Hunt et al., 1995).

Several factors influence the thickness of the mixing zone. From a theoretical standpoint, molecular diffusion and mechanical dispersion (known collectively as hydrodynamic dispersion) are the key drivers (Bruggeman and Custodio, 1987). Mechanical dispersion dominates molecular diffusion in atoll aquifer systems because permeabilities and groundwater velocities are high, notwithstanding relationships between dispersion and heterogeneity and associated challenges in selecting dispersion parameters for density-dependent problems (Underwood et al., 1992; Werner et al., 2013). White and Falkland (2010) used Volker et al.'s (1985) theory on the steady-state width of FGL mixing zones to demonstrate that the relative thickness of the mixing zone compared to the thickness of freshwater will increase as hydraulic conductivity (K) and the rate of pumping increase, and as recharge (W) and island width (L) reduce (see Section 4.3). Mixing processes may even eliminate freshwater altogether in some cases. For example, field observations from Enjebi Island (Marshall Islands) show that the mixing zone is so extensive that there is negligible usable freshwater in the lens (Buddemeier and Oberdorfer, 2004).

Field observations and numerical modelling of atoll islands reveal complex mixing-zone behaviour. For example, measured salinity profiles for Enjebi Island (Marshall Islands) between 1974 and 1976 appear to correlate with the location of solution unconformities, and are independent of tides and seasons (Buddemeier and Oberdorfer, 2004). Underwood et al. (1992) found that the component of longitudinal dispersivity in the vertical direction, in combination with vertical groundwater movement resulting from tidal fluctuations through the Pleistocene formation, most strongly influences the mixing-zone width and its rate of expansion within numerical experiments. An increase in the longitudinal dispersivity caused intensified mixing and a decrease in the extent of numerically predicted FGLs. This differs from numerical models of continental aquifers, where longitudinal and transverse dispersivity contribute equally to the mixing-zone width, at least for homogeneous aquifers (Abarca et al., 2007).

Other processes that affect mixing-zone extent include pumping, kinetic mass transfer (i.e., solute transfer between mobile and relatively immobile zones; Lu et al., 2009), and transient effects such as recharge seasonality and tides (Werner et al., 2013). Storm surge inundation and other episodic occurrences also modify mixing zones, potentially for significant periods after the event (see Section 5.1).

3.2. Hydraulic conductivity effects

Holocene sediments usually have much lower K values relative to those of the karstified Pleistocene deposits, typically by approximately one to two orders of magnitude (e.g., Oberdorfer and Buddemeier, 1983; Peterson, 2004; Vacher, 2004). This contrast in K has two key effects. Firstly, flow lines refract where meteoric water flowing from the interior of the island to the shoreline crosses the HPU (Vacher, 2004), as shown in Fig. 3. Refraction effects on solute distributions in K-stratified aquifers are complex where dispersion is accounted for (e.g., Sebben and Werner, 2016). By neglecting dispersion, Vacher (1988) was able to examine FGL behaviour in dual-aquifer settings using an analytical solution, which showed that the refraction of the lens at the HPU causes the lens to become thinner, compared to the case of a single-layer system. In essence, the higher K of the Pleistocene aquifer truncates the lens (Fig. 3). Dose et al. (2014) conducted physical sand tank experiments with layers of different K and simulated these using numerical models, which confirmed Vacher's (1988) analytical results. The degree of FGL truncation at the HPU is enhanced with increasing K contrast across the HPU, as demonstrated by Ketabchi et al. (2014) using numerical and analytical modelling.

Secondly, the higher K of the lower aquifer enables tides to penetrate landward with less resistance, resulting in enhanced mixing in the Pleistocene sediments (Buddemeier and Holladay, 1977; Hunt and Peterson, 1980; Vacher, 2004). Section 3.4 describes this process in more detail. This mixing adds to the truncating effect of the HPU on the FGL (Surface and Lau, 1988). FGL truncation at the HPU is evident at many sites, including Tarawa Atoll (Kiribati; Falkland and Woodroffe, 2004), Laura Island (Marshall Islands; Peterson, 2004), in atolls of the Cocos (Keeling) Islands (Woodroffe and Falkland, 2004), and at Diego Garcia Atoll (Chagos Archipelago, UK; Hunt, 2004).

Given that fresh groundwater is stored largely in the Holocene sediments, their properties rather than those of the Pleistocene limestone are dominant in controlling FGL characteristics (Ayers and Vacher, 1986). Holocene aquifers with lower K tend to accommodate thicker lenses (Bailey et al., 2008). Vacher and Rowe (2004) demonstrated the effect of K by comparing case studies in Bermuda, whereby the lower K (30–120 m/d) of the Langton aquifer creates a significantly thicker FGL than the adjacent Brighton aquifer, which has K values in the order of 1000 m/d. The location of the island with respect to the prevailing winds largely dictates the order of magnitude of the K of the Holocene sediments. Islands on the windward side of an atoll tend to comprise much coarser sediments and higher K than leeward islands, commonly resulting in thinner lenses (Anthony, 2004; Bailey et al., 2008, 2009).

The reef-flat plate creates a discontinuity of K in the Holocene aquifer, and thereby, like the HPU, its presence has important implications for the overall flow pattern of freshwater within atoll aquifers. Where the reef-flat plate exists, it can act as a confining unit for the Holocene aquifer, forcing groundwater from the FGL to flow below the seafloor and discharge through fractures in the reef-flat plate or at the reef margins (Cox, 1951; Buddemeier and Holladay, 1977; Bailey et al., 2008). Therefore, the discharge boundary for the FGL may not coincide with the shoreline of the atoll island if the island contains a reef-flat plate (Ayers and Vacher, 1986). Numerical modelling of FGLs by Bailey et al. (2009) indicates that the presence of the reef-flat plate slightly thickens FGLs. Field observations allude to a complex role of the reef-flat plate that acts to both promote and reduce FGL thickness. Ayers and Vacher (1986) found that in Deke Island, Pingelap Atoll (Federated States of Micronesia), the reef-flat plate acts as a barrier

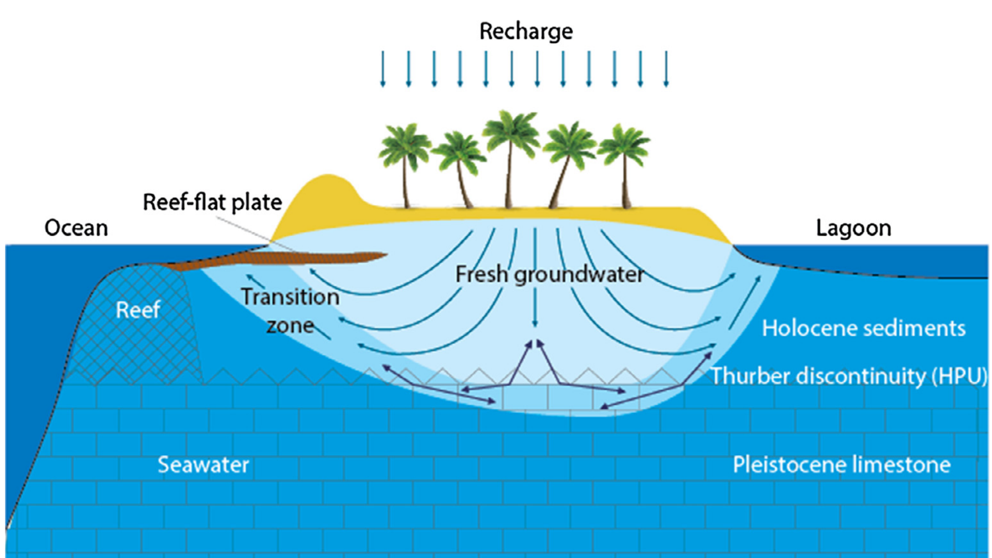

Fig. 3. Schematic of the refraction of groundwater flow lines across the HPU. Holocene and Pleistocene sediments occur above and below the HPU, respectively. The brown feature is the reef-flat plate, and the hatched area above the HPU on the ocean side designates reef deposits.

to recharge, given its low K. The accompanying water ponding above the plate and its subsequent discharge to the sea results in reduced recharge to the FGL. However, the reef-flat plate also acts as a barrier to deep root penetration and evapotranspiration (ET) from the lens, thereby reducing losses from the FGL where the reef-flat plate occurs. In some cases, atoll island communities penetrate the reef-flat plate to extract the confined freshwater beneath it or draw directly from the ponded water sitting on top (Bailey et al., 2008).

The Holocene sediments of atoll islands commonly exhibit cross-island gradation of K caused by the action of the winds and waves (Falkland, 1991; Hunt, 2004). Sediments on the lagoon side tend to be much finer and less permeable than on the ocean side (Cox, 1951). Field tests by Hunt (2004) on upper, unconsolidated units of Diego Garcia (Chagos Archipelago, UK) found K values spanning two orders of magnitude (3–300 m/d from the lagoon-side to the ocean-side, respectively) over a distance of less than 2 km. Peterson (2004) observed a similar range in K across Kwajalein Island (Marshall Islands). This cross-island variation in K commonly results in an asymmetrical FGL, with the lens thicker on the lagoon side of the island (due to lower K) and thinner on the ocean side. This form of asymmetric lens has been observed in numerous case studies (e.g., Anthony, 1991; Falkland, 1992a; Buddemeier and Oberdorfer, 2004; Falkland and Woodroffe, 2004; Peterson, 2004), and is discussed further in Section 3.5.

3.3. Recharge and evapotranspiration

A key factor in atoll FGL behaviour is the temporal and spatial distribution of recharge (W). It is primarily a function of rainfall and ET given that the low topographic gradient, permeable soil and small catchment area inhibit the generation of significant amounts of surface runoff on atoll islands (Underwood et al., 1992; Vacher and Rowe, 2004; White and Falkland, 2010). We use "ET" to refer to actual evapotranspiration, whereas "ET_P" is potential evapotranspiration, which we discuss in more detail below. The climate conditions experienced by atoll islands are wide ranging, leading to considerable variability in average-annual values of W. For example, in the Western Pacific, the mean rainfall ranges from less than 1000 mm/y on Kiritimati (Kiribati) to over 3500 mm/y near Funafuti Atoll (Tuvalu) (White and Falkland, 2010). The temporal variability in rainfall also differs between islands, and where it is high, it imparts significant fluctuations in the water levels and salinity distributions of atoll FGLs (Hunt and Peterson, 1980; Ayers and Vacher, 1986; Oberdorfer et al., 1990). The contrasts in climatic conditions lead to significant differences in FGL characteristics, as demonstrated by Vacher and Wallis (1992), who found that climate drivers were the cause of dissimilar FGLs on two low-lying carbonate islands in the Atlantic Ocean with distinctly different climates but otherwise similar properties.

Rainfall on atolls in the Pacific is strongly influenced by El Niño Southern Oscillation (ENSO) patterns, with periods of extreme wet or dry correlated to El Niño and La Niña episodes occurring every 2–7 y (White et al., 1999b; Bailey et al., 2008). The effect of ENSO events on rainfall in the Pacific depends on location. Atoll islands in the southern and northern Pacific, such as in Marshall Islands, commonly experience extended droughts during El Niño periods, because the low-pressure zone responsible for abundant rainfall shifts away from the region (Scott et al., 2003; van der Velde et al., 2006). Islands in Micronesia (Western Pacific) also experience drought during El Niño periods (Falkland, 1991; Bailey et al., 2009). For islands in the central Pacific, including Kiribati in particular, droughts occur during the intervening La Niña episodes (Falkland and Woodroffe, 2004; White et al., 2007b). For example, the El Niño event in 1983 resulted in a number of islands in Micronesia (Western Pacific) receiving just 13% of the average annual rainfall, whereas the same event resulted in higher rainfall rates across islands in the central Pacific (Falkland, 1991). The correlation between the Southern Oscillation Index (SOI) and annual rainfall has been quantified for a number of atoll islands, including Tarawa Atoll and Kiritimati (Kiribati; Falkland and Woodroffe, 2004), the Cocos (Keeling) Islands (Woodroffe and Falkland, 2004) and Micronesia (Bailey et al., 2008). Interestingly, despite this correlation, a case study on South Tarawa by White et al. (1999b) found that the SOI was not reliable in predicting extreme dry periods.

ET affects FGL thicknesses by reducing the net recharge of atoll islands (Bailey et al., 2008). Net recharge is the amount of rainfall that reaches the water table minus water lost from the saturated zone, predominantly by phreatophytes (Falkland, 1991). ET acts as an important freshwater sink during periods between recharge events, and while ET rates across the Pacific commonly account for a third to a half the annual rainfall on average, they usually exceed rainfall rates during the dry season (Falkland, 1993; White and Falkland, 2010). Robins (2013) tabulated average rainfall values for Malé (Maldives) and Kiritimati (Kiribati), and the raised atoll of Niue (847–2050 mm/y, respectively), and apportioned these into ET (56–75% of rainfall), W (25–42% of rainfall), and runoff (0–2% of rainfall). It should be noted that the annual rainfall given for Kiritimati has increased substantially from the figure provided by Robins (2013), which was based on analysis from the 1980s. Annual rainfall for this atoll island is now 1043 mm based on records from 1951 to 2016. Despite its importance, ET is the least quantified water-balance component of atoll islands (Falkland, 1991; White, 1996) because of the inherent difficulties in measuring it.

Estimates of atoll island ET include the mapping of Nullet (1987), who used the Priestley-Taylor equation to calculate ET_P rates for Pacific atolls. The resulting ET_P values were typically between 1600 and 1800 mm/y. An estimate of ET_P (1420 mm/y) for Tarawa Atoll (Kiribati), obtained using the Penman equation and climate station measurements (White et al., 2002), falls below this range. Woodroffe and Falkland's (2004) estimates of ET_P for atolls in the Cocos (Keeling) Islands, averaging 2000 mm/y, also fall outside of the range reported by Nullet (1987). Ayers and Vacher (1986) describe differences in ET processes across Deke Island (Federated States of Micronesia) arising from physiological features. They suggest that the central, low-lying region with shallow water-table conditions, characteristic of many atoll islands, may cause localised thinning of the lens due to direct ET of groundwater in the central part of the island. Vacher and Wallis (1992) observed the same processes in Great Exuma, a Bahamian island composed of calcarentic limestone, where ET from a central depression reduces FGL extent.

Aside from ET from waterlogged soils and shallow water tables, transpiration dominates ET losses from atoll islands. White et al. (2002) measured components of the water balance using various field measurements on Tarawa Atoll (Kiribati). Rain gauges were used to estimate daily precipitation as well as throughfall and interception by coconut trees, with tree transpiration estimated with sapflow sensors. These were coupled with daily measurements of soil moisture, climate drivers of ET and groundwater levels to estimate the net recharge. This study confirmed that coconut trees are significant water users. Quantifications based on field experiments indicate that they can transpire 60–160 L/d per tree (Bartle, 1987; Falkland and Brunel, 1989; White et al., 2002; Roupsard et al., 2006). Assuming 100% tree cover and tree spacing of at least 8 m (e.g., Falkland, 1994b), coconut trees can contribute 400–700 mm/y to ET. Falkland (1988) predicted a 20% reduction in W on the Cocos (Keeling) Islands as coconut tree coverage increased from zero to 100%. Falkland and Woodroffe (2004) calculated that W would increase by approximately 50% if all coconut trees were removed from Tarawa Atoll and Kiritimati (Kiribati). Gingerich (1992) used numerical modelling to demonstrate the significant effect of vegetation on FGL size on each half of Roi-Namur Island (Marshall Islands), which are identical apart from vegetation cover (coconut trees and dense vegetation compared to grassland). This study found that recharge on the densely vegetated side was 84% lower than the grass-covered half. These figures illustrate the importance of the role of vegetation and land-use management in the occurrence and sustainability of fresh groundwater. Water consumption by deep-rooted vegetation is a component of water-balance codes (see Section 4.5) that are routinely used to estimate net recharge (e.g., Griggs and Peterson, 1993; Comte et al., 2014), but explicit consideration of groundwater use by coconut trees in numerical models is not common practice (Bailey et al., 2008).

3.4. Tidal effects

In coastal aquifers, tides influence the groundwater flow pattern, salinity distribution and the characteristics of groundwater discharge to the sea (Werner et al., 2013). Tidal signals in aquifers generally have smaller amplitudes and larger time lags (relative to the sea) with increasing distance inland (Ferris, 1951). Amplitude reduction is quantified by the tidal efficiency (ε), which is the ratio of the groundwater tidal amplitude to that of the sea (Hunt and Peterson, 1980). The tidal time lag (t_τ) is the time difference between high or low tide in the ocean and the peak or trough of the groundwater fluctuation (Hunt and Peterson, 1980). Unlike continental coastal aquifers, where tidal signals diminish as they move inland from the coast, field measurements of FGLs in atoll islands indicate that ε and t_τ do not vary according to a systematic pattern (e.g., Buddemeier and Holladay, 1977; Oberdorfer et al., 1990; Peterson, 2004). Instead, tidal responses in atoll island aquifers are more so a function of depth (Hunt and Peterson, 1980). The cause of this is the rapid inland movement of the tidal pulse through the more conductive Pleistocene aquifer, which subsequently drives water-level behaviour in the Holocene sediments through upward transmission (Herman and Wheatcraft, 1984; Herman et al., 1986; Underwood et al., 1992; White and Falkland, 2010). This process results in oscillatory movement of the FGL and enhanced mixing of freshwater and seawater (Hunt and Peterson, 1980; Wheatcraft and Buddemeier, 1981).

In atoll islands, tidal responses (ε and t_τ) are most sensitive to the contrast in K between the upper Holocene (K_H) and lower Pleistocene (K_P) layers (Underwood et al., 1992). Hunt and Peterson (1980) suggest that cross-island asymmetry of K_H values may also influence tidal propagation, leading to interference between tidal signals originating from the lagoon and ocean sides of atoll islands. The storativity (S) is also a factor (Ferris, 1951) given that tidal propagation responds to the hydraulic diffusivity, which is the ratio of transmissivity (T) to S. The lower S of confined and semi-confined conditions within deeper sediments, relative to the specific yield that applies to the water table, adds to K_H-K_P contrast effects on atoll island tidal propagation (Hunt et al., 1995; Hunt, 2004).

Order-of-magnitude values for ε and t_τ in atoll islands are approximately 5% and 2.5 h, respectively, for water-table fluctuations in phreatic aquifers composed of relatively fine sediments. Islands with coarser sediments have water-table ε and t_τ values in the order of 45% and 2 h, respectively (Peterson, 2004; White et al., 2002; White and Falkland, 2011). Bosserelle et al. (2015) reported water-table values of ε generally between 4 and 17%, and t_τ between 0.5 and 3 h, in the Bonriki lens of Tarawa Atoll (Kiribati). Buddemeier and Oberdorfer (2004) found ε and t_τ ranges for Enewetak Atoll of 4–33%, and 1.7–3.7 h, respectively. A relationship between tidal response and depth on Diego Garcia (Chagos Archipelago, UK) obtained by Hunt (2004) showed that ε varied from 4 to 35% near the water table, increasing to 95% at a depth of 20 m below mean sea level (MSL). Similarly, t_τ decreased from 3 h at the water table to nearly zero at 20 m below MSL. This is consistent with the range of ε values (70–90%) at depth (i.e., near the HPU) within atoll islands reported by Falkland (2002b).

The degree to which tides affect the mixing zone is a function of the tidal amplitude and period, and aquifer properties, with the effects of dispersive mixing being greater for high values of the storativity (Underwood et al., 1992; Pool et al., 2015). Goter and Friedman (1988) identified the amplitude of diurnal tidal fluctuation as the principal factor preventing the establishment of permanent FGLs on Enewetak Atoll (Marshall Islands). On the other hand, Ayers and Vacher (1986) theorised that the lower permeability of the Holocene sediments found on parts of Deke (Federated States of Micronesia) would more strongly attenuate the tidal signal and result in a thinner mixing zone, allowing for a larger FGL to develop.

Several authors have identified that non-tidal models of atoll islands require modification to dispersivity parameters to compensate for the lack of tidal effects on mixing-zone width (Underwood et al., 1992; Ghassemi et al., 2000; Comte et al., 2014). For example, Alam et al. (2002) compared their tidal model of Bonriki Island (Kiribati) to the previous non-tidal model by Alam and Falkland (1997), finding that the tidal and non-tidal variants required calibrated α_T values of 0.05 and 1 m, respectively.

3.5. Lens asymmetry

Significant asymmetry has been observed in the shapes of many atoll FGLs. As mentioned above, cross-island variation in K in many cases may contribute to FGL asymmetry. Ayers and Vacher (1986) documented an asymmetric lens on Deke Island (Federated States of Micronesia), and related it to the presence of an aquitard at sea level on the ocean side of the island. Ayers (1990) reported a similar conclusion for Nukuoro Atoll (Federated States of Micronesia), where the presence of the reef-flat plate reduced recharge to the lens compared to the lagoon side.

Vacher (1988) studied the FGLs of strip-islands for cases with lateral variability in both W and K, and concluded that differences in K can cause stronger FGL asymmetry than the variability in W. However, asymmetric lenses have been observed in islands where no significant spatial variation in K is apparent (e.g., Bonriki lens, Kiribati), as reported by Falkland and Woodroffe (2004). They attribute the asymmetry at least partly to higher W on the lagoon side of the island. Other factors produce asymmetric FGLs, such as differences in the mean water levels of the lagoon and the ocean (Falkland, 1991). Differences in sea levels of up to 0.3 m occur across narrow atoll islands in the Pacific because of the prevailing winds (van der Velde et al., 2007a). It is likely that asymmetric FGLs may also arise from differences in tidal water-table overheight and wave run-up between the ocean and lagoon sides of islands (e.g., Nielsen, 1999). This aspect has received little attention in the context of atoll hydrogeology thus far.

4. Methods for investigating atoll FGLs

The sustainable development and preservation of atoll FGLs is predicated on knowledge of the lens extent and dynamics, to a level that allows reliable predictions of expected lens behaviour to be estimated for a variety of possible future conditions (Anthony, 1992; Bailey et al., 2010). To address this need, a number of field techniques have been developed and/or applied, including direct sampling approaches and indirect measurement techniques, such as geophysics and remote sensing (e.g., Hunt and Peterson, 1980; Ayers, 1981; Anthony, 1992; Falkland, 1993; White et al., 2002). These data are the foundation of empirical, analytical and numerical models to estimate and predict FGL behaviour (e.g., Oberdorfer and Buddemeier, 1988; Bailey et al., 2009). Analytical solutions rely on the assumptions inherent in the GH approximation and consider only simplified, steady-state conditions (see Section 4.3). Conversely, numerical modelling methods allow for the representation of the dispersive processes that lead to mixing-zone development, and are able to simulate both steady-state and transient conditions, as well as complex spatial variations in hydraulic properties. Empirical models effectively bypass the complex physical processes occurring on atoll islands to develop relationships between FGL characteristics and the conditions within which they occur. The following sub-sections summarise methods used in published case studies of atoll FGLs, including the estimation of recharge.

4.1. Methods for collecting FGL field data

Field investigations of atoll islands are resource intensive, and need to overcome the challenges of hydrogeological investigation in isolated settings (Falkland, 1991). Nevertheless, comprehensive assessments of FGLs have been achieved for a small number of cases, in which significant monitoring infrastructure have been installed, allowing for the application of both direct and indirect measurement techniques (e.g., Hunt and Peterson, 1980; Falkland, 1994a; Gingerich, 1996; Hunt, 1996; Sinclair et al., 2015a). Consequently, long-term monitoring data of pumping rates and salinities, and well-founded knowledge of the hydraulic and hydrochemical conditions are available for a selection of atoll islands, where they underpin the development of FGL management strategies (e.g., White et al., 2002; Falkland, 2004). In this section, we provide only a brief description of field-based methods applied in atoll settings because comprehensive reviews of this topic are available elsewhere (e.g., Dale et al., 1986; Falkland, 1991; IETC, 1998).

Monitoring wells are essential in any evaluation of FGL behaviour to obtain information about groundwater levels, aquifer parameters, geology, and hydrochemical characteristics (e.g., Falkland, 1994a). The Bonriki lens (Kiribati) has a network of 36 bores installed progressively since the 1980s, creating one of the longest running and dense groundwater monitoring datasets in the Pacific (Sinclair et al., 2015a). Monitoring boreholes contain bundled piezometers to obtain accurate data on salinity stratification (e.g., Surface and Lau, 1988; Falkland, 1994b; Sinclair et al., 2015a). This method avoids open boreholes and long screens, which is important for measuring salinity profiles without amplified mixing in the well from tidal fluctuations (Buddemeier and Holladay, 1977; Shalev et al., 2009). Falkland and Woodroffe (2004) argued for the installation of specific salinity monitoring bores in addition to standard water-level observation wells on the basis that field observations on Kiritimati (Kiribati) showed that water-table elevations alone were poor indicators of lens behaviour. Measurements of salinity profiles within observation wells have also proven essential in investigating the consequences of FGL overtopping events in a number of studies (e.g., Buddemeier and Oberdorfer, 2004; Terry and Falkland, 2010).

The FGL thickness on atoll islands is often determined using geophysical surveys, which are less expensive and time-consuming than direct sampling methods (Falkland, 1993). Electrical resistivity (ER) and electromagnetic induction methods have traditionally been the most popular approaches for extending direct measurements for the development of conceptual models. This is demonstrated by positive contributions of geophysical surveys to the understanding of FGLs in the Cocos (Keeling) Islands (Woodroffe and Falkland, 2004), Micronesia (Anthony, 1992), and Kiribati (e.g., Mather, 1973; Lloyd et al., 1980; Daniell, 1983; Sinclair et al., 2015a). Seismic surveys have also been conducted on a number of atolls to characterise features of the subsurface, including geology, depth to the water table, and location of the HPU (e.g., Ayers and Vacher, 1986; Ayers, 1990). Geophysical survey techniques generate results that commonly suffer from non-uniqueness (when more than one model can explain the same set of data), and must be verified with salinity measurements and lithological data (Anthony, 1992; Falkland, 1993). Falkland (1994a) describes a prominent example of atoll FGL investigation using drilling and geophysical methods, leading to characterisation of the FGL on South Keeling Atoll (Cocos (Keeling) Islands). He used boreholes to obtain accurate measurements of the lens at discrete points, with an ER survey providing reasonable estimates of the FGL thickness at intermediate locations. Further reading on the use and limitations of geophysical methods in relation to small islands is available within the detailed descriptions provided by Kauahikaua (1987) and Stewart (1988).

Remote sensing has been employed primarily to assess long-term shifts in the extents of atoll islands, aimed at identifying the controls on coastline dynamics, largely in response to the anticipated future effects of climate change (e.g., Webb, 2006; Webb and Kench, 2010; Rankey, 2011; Mann and Westphal, 2014; Purkis et al., 2016). One of the earliest applications of remote sensing to atoll island characterisation was the preliminary mapping of the remote and uninhabited Temoe Atoll (Gambier Islands) as part of a general geomorphological survey (Pirazzoli, 1987). Falkland and Brunel (1993) suggest that remote sensing techniques in combination with surface measurements could improve the extrapolation of ground-based ET measurements to island-wide estimates. Although some studies have used remote sensing to evaluate changes in vegetation cover across atoll islands, thereby supporting analyses of recharge and ET changes (e.g., Kuwahara et al., 2005; Jost and Andréfouët, 2006), further efforts to estimate atoll island ET from remote sensing are needed.

4.2. Empirical models of FGLs

The complex nature of atoll island hydrogeology has led to the development of empirical relationships for determining the characteristics of FGLs, thereby bypassing the high data needs and computational expense of physically-based modelling. For example, various researchers have developed empirical equations to relate recharge and island width to the occurrence of usable FGLs (e.g., Oberdorfer and Buddemeier, 1988; Underwood et al., 1992). While empirical approaches provide rough estimates of FGL features, they fail to account for many other factors known to significantly influence FGL occurrence, including the K contrast of the two aquifers, HPU effects, and the role of vegetation on net recharge (Falkland, 1994a; Bailey et al., 2008). Bailey et al. (2010) demonstrated significant limitations in Oberdorfer and Buddemeier's (1988) empirical model, as did Woodroffe and Falkland (2004) who showed that Oberdorfer and Buddemeier's (1988) empirical model underpredicts FGL thickness by up to 7 m, relative to field measurements. Bailey et al. (2010) used numerical modelling to derive an empirical model for estimating FGL thickness that included a larger number of island characteristics relative to previous attempts. Bailey et al.'s (2010) empirical model was used by Bailey et al. (2016) to predict future changes in atoll FGLs in response to climate change, including projected changes in rainfall and SLR.

An empirical approach to obtaining the sustainable yield of atoll FGLs is described by Hunt and Peterson (1980) and Peterson (2004). They suggest pumping the FGL at a set rate that is below the assumed sustainable yield while observing salinities and water levels over time. Incremental increases in pumping are imposed until deleterious effects are observed, at which point the pumping rate is reduced to avoid long-term damage to the FGL.

4.3. FGL analytical solutions

Analytical solutions describing the shape of atoll FGLs primarily adopt the GH principle, which relates the position of the freshwater-saltwater interface to the elevation of the water table as a function of the freshwater-seawater density difference (Drabbe and Badon Ghijben, 1888; Herzberg, 1901). The GH approximation assumes that the system is in hydrostatic equilibrium and that seawater and freshwater are immiscible fluids, separated by a sharp, continuous interface of equivalent pressure. The Dupuit (or Dupuit-Forchheimer) assumption, in which the aquifer's resistance to vertical flow is taken to be negligible, usually accompanies the GH approximation in developing analytical solutions (Bruggeman and Custodio, 1987; Strack, 1989). The GH principle leads to the function (White and Falkland, 2010):

$$H_u = h_0\left(1+\frac{1}{\beta}\right) \approx 41h_0 \quad (1)$$

Here, h_0 the highest FGL water-table elevation above MSL [L], H_u is the maximum FGL thickness [L] and β is the density ratio: $\beta = (\rho_s - \rho_f)/\rho_f$, where ρ_s and ρ_f are seawater and freshwater densities, respectively. Henry (1964) used this relationship to obtain an analytical solution for the elevation of the water table (h) above MSL for infinite strip islands as a function of recharge, island width, and homogeneous and isotropic K:

$$h^2 = \frac{\beta W x}{K(1+\beta)}(L-x) \quad (2)$$

Here, x is the distance from the coast [L]. Fetter (1972) provides the corresponding solution for circular islands in terms of radial coordinates.

The single-layer lens theory described above was used by Chesnaux and Allen (2008) and Greskowiak et al. (2013) to develop analytical solutions for the age distributions within FGLs. Morgan and Werner (2014) used single-aquifer lens theory to develop indicators of the vulnerability of strip island FGLs to recharge change and SLR (including land surface inundation effects). They showed that SLR impacts are significantly worse in islands where the lens cannot rise due to topographic controls, amongst other observations regarding lens vulnerability.

Bailey et al. (2010) reviewed single-layer analytical solutions using comparisons with field observations, and concluded that they provide reasonable initial estimates of the FGL where the lens sits completely in the Holocene aquifer. The lens is otherwise overestimated. Truncation of the lens at the HPU also means that the FGL thickness is not discernible by application of the GH relationship to a single water-level measurement (Bailey et al., 2010). Recognising the importance of the stored freshwater volume in strip island FGLs, Vacher (1988) produced an analytical solution for the FGL volume per unit length of shoreline (V) by integrating Eq. (2), as:

$$V = \left(\frac{\pi L^2}{8}\right)\sqrt{\frac{W(1+\beta)}{K\beta}} \quad (3)$$

Extensions to single-aquifer analytical solutions to account for the dual-aquifer configuration of atoll islands include the formula developed by Fetter (1972), who used the average of the Holocene and Pleistocene K values as an effective K. This neglects HPU refraction and FGL truncation, and is therefore only appropriate where the lens is completely contained within the Holocene sediments (Vacher, 1988). Vacher (1988) developed several analytical solutions for infinite strip islands composed of layers of different K to explore lens truncation at the HPU. Ketabchi et al. (2014) extended Vacher's (1988) results by deriving explicit analytical solutions for strip and circular dual-layered islands. For a strip, dual-aquifer island, h at some horizontal distance x from the shoreline is (Ketabchi et al., 2014):

$$h = \Delta\rho\left[\frac{(K_P-K_H)d}{\Delta\rho K_H + K_P\rho_f} + \sqrt{\frac{(L-x)Wx}{\Delta\rho(\Delta\rho K_H + K_P\rho_f)} + \frac{(K_P-K_H)\rho_s K_H d^2}{\rho_f(\Delta\rho K_H + K_P\rho_f)^2}}\right] \quad (4)$$

where $\Delta\rho$ is the difference between ρ_s and ρ_f [M L^{-3}], and d is the thickness of the Holocene layer [L]. This equation applies to the region where the freshwater-saltwater interface lies below the HPU. The analytical solution by Fetter (1972) can be used to calculate h in the zone where the interface occurs above the HPU. For circular islands, Ketabchi et al. (2014) modified Eq. (4) by replacing the Cartesian coordinates with radial coordinates. Ketabchi et al. (2014) tested the performance of the circular island analytical solution against numerical modelling for a number of scenarios, including instantaneous SLR and land-surface inundation, which they represented simply by increasing d and reducing the island's radius based on the land-surface slope.

The main limitation of FGL analytical solutions is the exclusion of aquifer heterogeneity, and dispersive and transient effects. Neglecting dispersion results in FGL overestimation (Oberdorfer et al., 1990). Hunt (1979) appended an analytical dispersion model to a steady-state solution for the shape of the FGL on Tongatapu Island (Tonga). This allowed for the calculation of the vertical variation in salinity as a function of W. Volker et al. (1985) produced an analytical solution to estimate the mixing-zone thickness at the centre of a lens within a strip island subjected to pumping and spatially variable recharge. The solution relies on the GH and Dupuit assumptions, and treats the mixing layer as a boundary of finite width that acts as a laminar boundary layer between two fluids moving at different velocities. The solution assumes that both K and dispersion are homogeneous, potentially limiting its applicability, particularly near the shoreline (Volker et al., 1985). Based on Volker et al.'s (1985) theory, White and Falkland (2010) presented a formula for the average mixing-zone thickness (ϖ_u) in the absence of pumping:

$$\varpi_u = \frac{H_u}{W}\left(\frac{DK_H}{\beta L}\right)^{1/2} \quad (5)$$

where D is the dispersion coefficient, and H_u is obtained from sharp-interface analysis. White and Falkland (2010) assumed that useable FGLs exist when $\varpi_u < 2H_u$ or $W > \sqrt{(DK_H)/(4\beta L)}$.

Pumping increases ϖ_u according the following relationship (White and Falkland, 2010):

$$\varpi_p = \frac{\varpi_u}{\sqrt{1-\frac{Q_p}{AW}}} \quad (6)$$

Here, A is the island's surface area [L^2]. According to Eq. (6), the modified mixing-zone thickness (ϖ_p) increases with larger values of the total rate of pumping (Q_p) and with lower recharge. Eqs. (5) and (6) provide for initial estimates of FGL changes during drought and under the effects of pumping, although only where the assumption of homogeneity in island sediments is reasonable (White and Falkland, 2010).

Analytical solutions based on Dupuit and GH assumptions typically do not allow for the outflow face through which fresh groundwater discharges to the sea (Fetter, 1972; Falkland, 1991). Glover (1959) provided the first solution for the outflow face geometry using a conceptual model of a confined, continental aquifer. van der Veer (1977) developed a phreatic aquifer analytical model to calculate the depth to the interface at the shoreline. However, this solution requires knowledge of the horizontal length of the freshwater outflow face on the seabed. Vacher (1988) overcame this limitation by adopting simplifying approximations for the outflow boundary conditions, and showed that the revised formulations for the outflow face matched closely the results of van der Veer (1977). Vacher (1988) suggested that the effect of the outflow face when calculating the depth of the interface of FGLs is largely negligible, except near the shoreline to a distance of about 1 to 5% of the island's width.

4.4. Numerical methods

Numerical solutions are able to overcome the limitations of analytical solutions, allowing transient problems involving dispersive, density-dependent flow and transport to be determined without requiring assumptions of steady-state, sharp-interface conditions, or homogeneity. Numerical techniques also apply to a greater array of boundary conditions, and therefore more diverse island settings can be represented (Underwood et al., 1992). They do come, however, at the cost of a large data need and additional computational effort. Numerical models have been used to undertake a number of island-specific investigations, as well as to analyse particular processes that affect FGLs (e.g., Bailey et al., 2009; Terry and Chui, 2012). Previous reviews of numerical modelling studies of atoll FGLs were published by Underwood et al. (1992), Griggs and Peterson (1993), and Bailey et al. (2009). We extend their summaries by including more recent atoll island modelling investigations, as shown in Table 1, which lists the study site, model characteristics and modelling software for 24 modelling case studies. While our review of the literature encountered over 50 groundwater modelling analyses of atoll FGLs, only the more prominent and well-documented examples are included in Table 1.

The study by Griggs and Peterson (1989) represents one of the earliest attempts to include dispersive, density-dependent flow and transport in a numerical model of an atoll FGL, overcoming the limitations reported in previous numerical modelling investigations that neglected density and/or invoked GH and Dupuit assumptions (e.g., Lloyd et al., 1980; Herman et al., 1986). Density-dependent numerical models are now commonplace in investigations of atoll FGLs, with SUTRA and SEAWAT emerging as the preferred models over the past 27 y (Table 1). Based on Table 1, SEAWAT is the favoured code for three-dimensional (3D) simulations, but SUTRA has the advantage of being able to simulate variably saturated processes. The majority of studies listed in Table 1 adopt the dual-aquifer conceptual model and include anisotropic K values, thus more closely representing typical atoll island settings. Holding and Allen (2015b) are an exception to this, considering only a single hydrostratigraphic unit (i.e., above the HPU) to reduce computational demand.

The majority of models listed in Table 1 represent two-dimensional (2D) vertical cross sections, reflecting the elongated shape of most atoll islands (Falkland, 2002a; Chui and Terry, 2015). Vacher (1988) suggested that this assumption is valid when the island's length is at least 3 times greater than its width. A small number of studies have assumed horizontal symmetry along the midline of the island, making it possible to model only half the island width (e.g., Terry and Chui, 2012; Chui and Terry, 2015). This assumption is appropriate only where the factors that cause lens asymmetry can be neglected (see Section 3.5). Circular islands have been modelled using 2D vertical cross-sections in radial coordinates (e.g., Ketabchi et al., 2014). The use of 3D simulations appears to be rising, likely due to increased computing power and the recognition that 2D models may be inadequate for simulating key atoll island processes (Ghassemi et al., 2000; Bosserelle et al., 2015). 2D models are unable to represent the spatial variability (in plan) of aquifer parameters, recharge and the sea boundary, in addition to the effects of pumping and other processes causing divergent or convergent flow (Bosserelle et al., 2015). However, 3D models have larger data and computational requirements. This effectively precludes their application to the field-scale simulation of data-scarce atoll islands, and/or where fine discretization of space or time is required to capture the flow and salinity conditions near pumping wells, or tidal effects and other high-frequency processes (Ghassemi et al., 2000).

Tidal boundary conditions have been incorporated in few numerical models since 1992, despite the importance of including tidal effects being demonstrated in a number of early studies (e.g., Oberdorfer et al., 1990; Underwood, 1990; Gingerich, 1992; Underwood et al., 1992). This is largely due to the accompanying significant increase in computational demands (e.g., Comte et al., 2014; Bosserelle et al., 2015) that arises from the high temporal discretisation requirement. Ghassemi et al. (2000) adopted a tidal boundary condition in 3D simulations of the FGL of Home Island (Cocos (Keeling) Islands) to assess tidal propagation, resolved at 36 min time steps. A longer-term (1000 d) tidal simulation using 6 h time steps (i.e., presumably a triangular tidal signal), run to quasi-equilibrium conditions, demonstrated that dispersion parameters used in non-tidal models primarily represent tide-induced mixing. Tides were excluded from predictive transient simulations because higher dispersion parameters were reported as adequate surrogates in non-tidal models, and this avoided the large computation effort involved in tidal simulations. Bailey et al. (2009) and Bailey and Jenson (2014) have accounted for tidal fluctuations in generic atoll island modelling, with the latter study finding that this boundary resulted in differences of up to 0.75 m in lens thicknesses obtained by non-tidal models. Of the studies included in Table 1, only Alam et al. (2002) included tidal effects in predictions of future atoll FGL conditions. Non-tidal models of atoll islands generally adopt MSL as the coastal boundary, and thereby neglect the potentially significant tidal effects on shoreline heads discussed in Section 3.4.

Collectively, the case studies in Table 1 highlight the wide range of numerical modelling objectives. While earlier studies published in the scientific literature were largely concerned with sustainable yield estimation, more recent modelling efforts demonstrate a shift towards investigating the response of FGLs to various aspects of climate variability and change, in particular SLR, inundation periodicity and magnitude, and recharge variability. Additionally, an increasing number of atoll FGL studies attempt to adopt typical island parameters, with the aim of drawing generalizable conclusions.

The parameter values adopted in each of the Table 1 studies are summarised in Table 2, and the range in values reflects the variability in conditions encountered on atoll islands. The depth of Holocene sediments ranges between 5 and 50 m, with most studies specifying a thickness of 12–15 m. K_H is generally one to two orders of magnitude lower than K_P (see Section 3.2). $K_{H,x}$ typically ranges between 15 and 80 m/d, although Bailey and Jenson (2014) adopted higher values to examine the response of FGLs to changes in this parameter. Gingerich (1992) also applied a higher value of 140 m/d, based on local aquifer pumping tests. The author noted that due to the shallowness of the pumping and observation wells used, this value was likely to be invalid for deeper parts of the aquifer.

Table 2 shows that several studies divided the Holocene sediments into two sublayers, reflecting the zones of differing K_H encountered in geological field investigations (e.g., Bailey et al., 2009; Bailey and Jenson, 2014). Values for $K_{P,x}$ vary between 173 and 5000 m/d, with most studies adopting values between 500 and 1000 m/d. Alam and Falkland (1997), Alam et al. (2002), and Bosserelle et al. (2015) specified two sublayers for the Pleistocene

Table 1
Examples of numerical modelling case studies of atoll islands.

References	Island	Model type	Study objective	Transient flow	Dual aquifer	Tidal boundary	Anisotropic K	Pumping
Griggs and Peterson (1989)	Laura Island (Marshall Islands)	SUTRA (2D)	Sustainable yield and buoyancy effects	X	X	X	X	X
Oberdorfer et al. (1990)	Enjebi Island (Marshall Islands)	SUTRA (2D)	Hydrogeological controls on flow patterns	X	X	X		
Underwood (1990)	Generic	SUTRA (2D)	Flow processes and mechanisms, and transition zone characteristics	X	X	X	X	
Gingerich (1992)	Roi-Namur (Marshall Islands)	SUTRA (2D)	Sustainable yield	X	X	X	X	X
Underwood et al. (1992)	Generic	SUTRA (2D)	Transition zone width and propagation of tides	X	X	X	X	
Griggs and Peterson (1993)	Laura Island (Marshall Islands)	SUTRA (2D)	Sustainable yield	X	X	X	X	X
Alam and Falkland (1997)	Bonriki lens (Kiribati)	SUTRA (2D)	Climate change impacts on lens thickness	X	X		X	X
Ghassemi et al. (1999)	Home Island, (Cocos (Keeling) Islands)	SALTFLOW (3D)	Sustainable yield	X	X		X	X
Ghassemi et al. (2000)	Home Island, (Cocos (Keeling) Islands)	SALTFLOW (3D)	Development of a 3D model to overcome limitations of 2D model	X	X	X	X	X
Alam et al. (2002)	Bonriki and Buota lenses (Kiribati)	SUTRA (2D)	Sustainable yield	X	X	X	X	X
Lee (2003)	Generic	TOUGH2 (3D)	Influence of geometric aspect ratio and fractures on lens thickness and mixing zone	X				
Bailey et al. (2009)	Generic	SUTRA (2D)	Influence of climatic and geologic factors on lens thickness	X	X	X	X	
Comte et al. (2010)	Mba Island (New Caledonia)	SEAWAT (3D) SUTRA (2D)	Influence of recharge on lens geometry		X		X	
Guha (2010)	Generic	SEAWAT (3D)	Seawater intrusion in response to SLR		X		X	X
Chui and Terry (2012)	Generic	SUTRA (2D)	Response and recovery of FGL to salinisation by storm wave overwash	X	X		X	
Terry and Chui (2012)	Generic	SUTRA (2D)	Assess response of FGLs to storm-generated inundation in combination with SLR	X	X		X	
Chui and Terry (2013)	Generic	SUTRA (2D)	Extend Terry and Chui (2012) by assessing FGL response to over-wash for various island widths	X	X		X	
Bailey and Jenson (2014)	Generic	SUTRA (2D)	Influence of environmental and human factors on response and recovery of FGL after storm surge	X	X	X	X	
Bailey et al. (2014)	Maldives	SUTRA (2D)	Changes in lens thickness due to seasonal and long-term changes in rainfall	X	X		X	
Comte et al. (2014)	Grande Glorieuse	SEAWAT (3D)	Lens response to climate and vegetation changes	X	X		X	
Bosserelle et al. (2015)	Bonriki Island (Kiribati)	SEAWAT (3D)	SLR and inundation effects on lens	X	X		X	X
Holding and Allen (2015b)	Generic	HydroGeoSphere	FGL response to overwash events for various small island types	X				
Chui and Terry (2015)	Generic	SUTRA (2D)	Influence of central topographic depressions on saline contamination of FGLs	X	X		X	
Gulley et al. (2016)	Generic	SEAWAT (3D)	Impact of lake inundation and evaporation on loss of fresh groundwater					

aquifer, and assigned two corresponding values of $K_{P,x}$. The intention was to better simulate the transition zone, which was located at or just above the HPU in the two-layer model. The geological justification for this subdivision is the higher degree of karstification of the top part of the Pleistocene limestone relative to deeper sections. Anisotropy ratios (horizontal to vertical) of the Holocene

Table 2
Parameter values used in atoll island numerical modelling studies.

Reference	Parameter										
	n [–]	d [m]	$K_{H,x}$ [m/d]	$K_{H,z}$ [m/d]	$K_{P,x}$ [m/d]	$K_{P,z}$ [m/d]	$\alpha_{L,x}$ [m]	$\alpha_{L,z}$ [m]	α_T [m]	S_y [–]	W [mm/y]
Griggs and Peterson (1989)	0.2 (H) 0.3 (P)	6–12	17.3–60	n/a	173–600	1.73–6	0.4–15	n/a	0.06–0.75	0.18	1780
Oberdorfer et al. (1990)	0.3	12	10	n/a	1000	n/a	0.02	n/a	0	n/a	500
Underwood (1990)	0.2–0.25	15	50	10	500	100	6	0.01–0.3	0.001–0.01	0.25–0.3	200–1000
Gingerich (1992)	0.3	11.5	140 (U) 9 (L)	30 (U) 5 (L)	140 (U) 275 (L)	12 (U) 60 (L)	0.02–3.0	n/a	0.001	n/a	312–576
Underwood et al. (1992)	0.25	15	50	10	500	100	6–12	0.01–0.05	0.01	0.25	500–2000
Griggs and Peterson (1993)	0.2 (H) 0.3 (P)	5–28	60.5	n/a	605	n/a	0.4–8	n/a	0.05	0.18	1780
Alam and Falkland (1997)	0.2	15	6–12	1.2–2.4	15–20(U) 500 (L)	3.5–4.5 (U) 100 (L)	0.6–8	n/a	0.3–0.5	n/a	926–991
Ghassemi et al. (1999)	0.3	12	10	2	500	100	5	0.01	1	n/a	855
Ghassemi et al. (2000)	0.25	12	15	15	1500	300	4	0.05	1	n/a	855
Alam et al. (2002)	0.2	15	8–30	3–5	30–50 (U) 1400 (L)	7–10 (U) 200 (L)	0.05–8	n/a	0.05	n/a	926–980
Lee (2003)	0.25–0.33	n/a	1	0.2	n/a	n/a	n/a	n/a	n/a	n/a	500
Bailey et al. (2009)	0.2 (H) 0.3 (P)	5–10 (U) 8 (L)	50 (U) 60 (L)	10 (U) 12 (L)	5000	1000	n/a	n/a	n/a	0.18	1250–2750
Comte et al. (2010)	0.2 (H) 0.3 (P)	n/a	10	7	900	180	0.7	0.1	0.02	n/a	50–300
Guha (2010)	0.2 (H) 0.3 (P)	15	70	7	700	70	3	0.01	0.2	0.15	730–2920
Chui and Terry (2012)	0.42	7	85^{*}	8.5^{*}	848^{*}	85^{*}	10	1	0.1	n/a	730–1460
Terry and Chui (2012)	0.42	7	85^{*}	8.5^{*}	848^{*}	85^{*}	10	1	0.1	n/a	1460
Chui and Terry (2013)	0.3	12	85^{*}	8.5^{*}	848^{*}	85^{*}	10	1	0.1	n/a	1095
Bailey and Jenson (2014)	0.2 (H) 0.3 (P)	5–10 (U) 8 (L)	50–400 (U) 60–480 (L)	10 (U) 12 (L)	5000	1000	6	0.5	0.05	0.2	2000
Bailey et al. (2014)	0.2 (H) 0.3 (P)	13–18	75	15	5000	1000	6	n/a	0.05	0.2	458–1631
Comte et al. (2014)	0.25 (H, U) 0.2 (H, L) 0.01–0.15 (P)	20 (U) 30 (L)	86 (U) 9 (L)	69 (U) 1.7 (L)	0.1–173	0.01–173	3–50	n/a	0.03–0.5	n/a	n/a
Bosserelle et al. (2015)	0.3	14–23	1.2–17.9	0.02–3.6	10–20.6 (U) 456 (L)	3.5–5.8 (U) 91 (L)	1	0.01	0.05	n/a	102–1993
Holding and Allen (2015b)	0.2	20	60	n/a	n/a	n/a	1	n/a	0.01–0.1	<0.25	953
Chui and Terry (2015)	0.3	12	85^{*}	8.5^{*}	848^{*}	85^{*}	10	1	0.1	n/a	1095
Gulley et al. (2016)	0.3	n/a	50	n/a	n/a	n/a	1	n/a	n/a	n/a	200

Note: (U) and (L) apply, respectively, to upper and lower sublayers within Holocene/Pleistocene layers; subscripts x and z refer to horizontal and vertical components of K and α; α_L and α_T refer otherwise to longitudinal and transverse dispersivities, respectively; S_y is specific yield; n is porosity; $^{*}K$ has been approximated from permeability; n/a – not applicable or unreported.

and Pleistocene formations range from 1:1 to 10:1 and 1:1 to 100:1, respectively, reflecting the layered lithology of these sequences. Fifteen of the 24 studies did not specify a value for S_y. Where this parameter was given, it ranged between 0.15 and 0.3. Recharge rates range between 200 and 2920 mm/y, which is of course due to the variability of atoll island climatic conditions. Dispersion parameters depend on tidal representation, with tidal simulations having lower dispersivities than non-tidal simulations (see Section 3.4). Longitudinal dispersivity values vary between 0.05 and 50 m in the horizontal direction, with lower values between 0.01 and 1 m adopted in the vertical direction. Transverse dispersivity values range from 0.001 to 0.5 m.

4.5. Recharge estimation methods

Along with determining the size and shape of the FGL, quantifying recharge is critical for estimating sustainable extraction rates. The most common method is water-budget analysis of the vegetation canopy and the unsaturated zone (Chapman, 1985; Falkland, 1991). The unsaturated zone water balance can be written in terms of the total rainfall recharge as (e.g., Falkland and Woodroffe, 2004):

$$W = P - \mathrm{ET} - \Delta V \tag{7}$$

where, for a given time step, P is rainfall [L], ET is actual evapotranspiration [L], and ΔV is the increase in storage within the unsaturated zone [L]. All quantities in Eq. (7) represent water volumes per unit surface area per unit time, and are expressed as water column heights, usually in mm. Where the water budget is calculated using long time steps ($\geq$1 month), ΔV is usually neglected (e.g., Hunt and Peterson, 1980; Falkland, 1991). Surface runoff is also often excluded due to the high infiltration capacity of soils and the flat topography of most atoll islands (Lloyd et al., 1980; Falkland, 1994b). This assumption may not hold for very high rainfall events or where impermeable surfaces (e.g., roads, compacted surfaces, airstrips, etc.) exist (Lloyd et al., 1980; Peterson, 2004). Eq. (7) sometimes omits groundwater contributions (i.e., via capillary rise) to both ET and ΔV, because these fluxes are difficult to measure and/or quantify.

Falkland (1991) and Falkland and Woodroffe (2004) subdivided ET into three components: evaporation from interception storage (E_I), evapotranspiration from the unsaturated (soil) zone (E_S), and the transpiration of groundwater (i.e., from the saturated zone) by deep-rooted vegetation (E_G). Subtracting the water consumption of deep-rooted vegetation from the total rainfall recharge gives the so-called net recharge (W_{net}), as (Falkland, 1991):

$$W_{net} = W - E_G = P - E_S - E_I - E_G - \Delta V \tag{8}$$

White (1996) includes the contribution of capillary rise to the soil-store within the E_G term, which allows the model to represent direct evaporation of groundwater from the unsaturated zone where the water table is shallow. The capillary fringe in atoll islands is probably rather thin given the coarse nature of Holocene sediments, but evaporation from shallow water tables could nonetheless be important under particular circumstances. Changes in the soil store also occurs due to water-table fluctuations, but including this effect on the unsaturated zone water budget requires either a coupled unsaturated-saturated model or a priori knowledge of temporal water-table behaviour.

In water-balance calculations, estimates of actual E_S and E_G values are based generally on available measurements of ET_P (Falkland, 2002b). For example, atoll island ET_P is most often estimated using the Penman or Penman-Monteith equation and/or measurements of pan evaporation (Falkland, 1991, 2002b). The de facto reference for the calculation of ET_P from meteorological observations is the report by Allen et al. (1998), which recommends specific parameter values for islands. ET may then be derived from ET_P via simple crop factors (e.g., Doorenbos and Pruitt, 1977; Allen et al., 1998). However, crop factors are poorly constrained for typical island vegetation types, and represent an important source of model uncertainty (White, 1996). McMahon et al. (2013) provided a recent overview of the various ET_P and ET estimation formulae. Whilst their review does not specifically consider islands, it nonetheless offers useful guidance on the application of, and pitfalls associated with, the various models.

Various methods to obtain ET have been adopted in island studies. In some cases, no conversion was attempted and ET_P estimates were used directly in water-balance calculations (e.g., Ghassemi et al., 2000). Hunt and Peterson (1980) took ET to be equal to ET_P for all cases except where soil moisture was at wilting point. Comte et al. (2014) used an equation for estimating ET from ET_P derived by Roupsard et al. (2006) that is based on field measurements of ET. Some studies have used measurements of solar radiation, air temperature, wind speed and relative humidity from weather stations to calculate ET_P using a combination formula such as the Penman equation (e.g., Falkland, 2002b; White et al., 2007). As a solution to the challenge of obtaining ET from atoll islands, several studies recommend applying measurement techniques used in continental settings (e.g., Falkland, 1993; Falkland and Brunel, 1993). These include the Bowen ratio method, the eddy covariance technique and lysimeters; however, White (1996) describes these particular methods and concludes that few appear suitable.

Early recharge estimates used relatively simplified water-balance equations in obtaining atoll island recharge. For example, Hunt and Peterson (1980) neglected E_G, E_I and ΔV but included surface runoff to obtain a first approximation of W for Kwajalein Island (Marshall Islands), testing the effect of using both daily and monthly time steps. Hamlin and Anthony (1987) considered only annual rates of P, E_S and E_I in calculating that the average annual W was 50% of P on Laura Island (Marshall Islands). Comte et al. (2014) adopted a monthly time step in developing a distributed water-budget model of the unsaturated zone, similar to Eq. (8), to calculate W_{net} over Grande Glorieuse Island. They neglected changes in soil moisture under the assumption that the coarseness of the soil at the study site prevents water retention. The effect of deep-rooted vegetation was taken into account, producing W as a function of tree density.

While long-term approximations of W are attainable using Eq. (7), Chapman (1985) suggests that because of the short residence time of water in atoll island unsaturated zones, only a daily time step leads to meaningful results. Hunt and Peterson (1980) and Falkland (1988) showed that the use of monthly rather than daily rainfall data resulted in W being under-estimated by 6–10%. However, Falkland (1994a) and Falkland and Woodroffe (2004) recommend that mean monthly ET data are adequate for recharge modelling given the small variations in daily ET, at least in tropical climates. They adopted daily time steps in calculating W_{net} using a lumped-parameter representation of the unsaturated zone. Their model partitions rainfall into storage and outflow mechanisms (Eq. (7)) using a series of water stores. Rainfall first fills the interception store, and all the intercepted water (E_I) is assumed to evaporate. Once this store fills, additional rainfall spills to the soil-moisture store, from which E_S is taken at a rate that varies linearly with the soil-moisture content. The potential transpiration of various vegetation communities is determined by multiplying ET_P by a crop factor (e.g., Doorenbos and Pruitt, 1977). Any surplus water in the soil-moisture zone drains to the water table as total recharge (i.e., W), and W_{net} is finally determined by removing E_G due to deep-rooted vegetation. This model is incorporated into the WATBAL computer program and has been applied to determine W_{net} on Home Island (Cocos (Keeling) Islands) (Ghassemi et al., 2000) and Bonriki Island (Kiribati) (Bosserelle et al., 2015).

Other methods of recharge estimation for atoll islands include the measurement of drainage fluxes beneath root zones (van der Velde et al., 2005), and analysis of long-term water-level records adjusted to account for tidal and barometric pressure fluctuations (Furness and Gingerich, 1993). Preliminary approximations of W have also been obtained using the chloride mass-balance approach (e.g., Hunt, 1979; Ayers, 1981) but this approach may be complicated by dry deposition from salt spray (Chapman, 1985). Where there are insufficient data to obtain an accurate estimate of W, regional approximations of rainfall-recharge relationships are often used. Examples include the rainfall-recharge curve produced on the basis of a number of atoll island studies by Chapman (1985) and subsequently updated by Falkland (1991), or the map developed by Nullet (1987) for the Pacific region. Comte et al. (2010) sought to characterise W_{net} in a small coral reef island using a combination of numerical modelling and electrical resistivity tomography, thereby attempting to overcome the non-uniqueness issues known to inhibit the estimation of recharge through model calibration (e.g., Knowling and Werner, 2016). Their analysis was able to eliminate hypotheses regarding relationships between W_{net} and topography on the island, and order-of-magnitude recharge rates were obtained.

5. Management and sustainability of atoll FGLs

The FGLs of atoll islands are particularly vulnerable to degradation in water quality via both natural processes and anthropogenic pathways (Falkland, 1991; Robins, 2013). For example, the thin, permeable unsaturated zone provides little protection against the infiltration of biological and chemical surface contaminants, and the high population densities of some atoll islands has led to unsustainable rates of extraction (Falkland, 1991; Robins, 2013). Thus, efficient and effective management of FGLs in atoll islands is critical to secure adequate freshwater supplies for island residents (Bailey et al., 2009). The sub-sections that follow summarise available literature on atoll FGL management, in particular in relation to FGL threats, sustainability, development and regulation. We consider climate change and SLR as anthropogenic (rather than naturally occurring) threats to FGLs, and hence they are included in Section 5.2.

5.1. Naturally occurring threats to atoll FGLs

The viability of atoll FGLs is highly dependent on their resilience during drought periods (Scott et al., 2003). Despite many atoll islands occurring in wet tropical climates, periods of drought arise due to the often large inter-annual variability of natural rainfall cycles, threatening the reliability of water supplies drawn from FGLs (White et al., 1999b; van der Velde et al., 2007b). For example, rainfall records for Tarawa Atoll (Kiribati) show annual totals varying from 398 to 4356 mm/y during 1947–2013 (Bosserelle et al., 2015). White et al. (1999b) suggest that drought periods occur at this location approximately every 6–10 y.

When droughts occur, FGLs can contract considerably. For example, White et al. (2007b) measured a reduction in thickness of 50% of the Bonriki Lens (Kiribati) during the 1998–2001 drought event. Similar reductions in thickness were observed in some parts of Laura Island (Marshall Islands) (Presley, 2005). FGL reductions during drought occur partly because storage losses are the result of both rising salinities and falling water tables, as expected given the GH relationship. Bailey et al. (2009) used numerical modelling to investigate the response of FGLs to drought, including the depletion of groundwater resources as a function of time and general island characteristics. They found that some islands would be unlikely to support communities during drought, depending partly on island location in relation to wind direction. This is consistent with the conclusions of White et al. (1999b) and White and Falkland (2010), who also suggest that some drought events could cause total loss of FGLs on smaller islands.

The low land-surface elevation of atoll islands also makes them vulnerable to inundation (i.e., overtopping) by seawater, resulting from storm surges, wave setup, extremely high tides and tsunamis (Terry and Falkland, 2010; Bailey and Jenson, 2014). In general, FGLs exposed to seawater inundation may be unfit for human consumption, or at least remain inaccessible to local extraction wells due to shallow saline plumes, for periods ranging from a few months to several years after the event (Scott et al., 2003; Chui and Terry, 2012; Bailey and Jenson, 2014). The recovery of FGLs from overtopping events depends on several factors, including recharge, hydraulic properties, the degree of contamination, and density effects (Terry and Falkland, 2010; Holding and Allen, 2015b).

Overtopping causes seawater to enter the aquifer through vadose zone infiltration, direct inflow through boreholes, and initially rapid and continuing infiltration of ponded seawater in trenches and surface depressions (Terry and Falkland, 2010; Chui and Terry, 2013; Bailey and Jenson, 2014; Holding and Allen, 2015a). Inundated trenches and wells allow seawater to bypass the unsaturated zone, creating larger saline plumes (e.g., Holding and Allen, 2015a). Holding and Allen (2015a) found that recovery times under inundated trenches on Andros Island (Bahamas) were longer than on other parts of the island. Terry and Chui (2012) drew similar conclusions from numerical experimentation. Holding and Allen (2015a) suggest that draining of inundated trenches shortens recovery times and is a worthwhile remedial action to restore the lens to potable concentrations sooner. These forms of direct inflow occur rapidly, although delayed infiltration through the vadose zone usually has a much larger spatial extent (Bailey et al., 2015; Holding and Allen, 2015b).

The downward movement of the seawater plumes created by surface inundation occurs due to both density and hydraulic gradients (Holding and Allen, 2015b). Rainfall recharge that occurs subsequent to inundation serves to dilute the saltwater plume and flush it towards the sea (Terry and Falkland, 2010; Holding and Allen, 2015a; Holding and Allen, 2015b). The diluting effect of rainfall recharge also depletes the density gradient driving seawater downwards into the FGL. Buddemeier and Oberdorfer (2004) described the effects of inundation on Enewetak Atoll (Marshall Islands) due to a hurricane in 1979. Salinity values of up to two-thirds of that of seawater were found in shallow wells one week after the storm. Salinities at shallow depths recovered to nearly pre-storm values within about one year, and the authors suggested that the higher density of seawater promoted the lens remediation as it leads to increased rates of vertical downward flow.

The geology and physiography of the lens also play a role (Bailey and Jenson, 2014; Holding and Allen, 2015b). For example, thicker unsaturated zones allow a greater volume of seawater to enter the subsurface and subsequently the FGL, and lower soil K inhibits seawater infiltration (Holding and Allen, 2015b). In some cases, in particular below reef-flat plates, a narrow body of freshwater persists between an upper saline plume created by infiltrating seawater and the saltwater below the FGL base (Bailey and Jenson, 2014). Terry and Falkland (2010) observed this in their field investigation of the three islets of Pukapuka Atoll (Northern Cook Islands) after a category 5 cyclone resulted in a storm surge in February 2005. While the resulting salinization of the relatively thin FGL on Motu Kotawa islet was dispersed throughout the FGL's depth ($\sim$3.2 m), a trapped body of freshwater occurred within the thicker FGLs of Motu Ko ($\sim$10 m) and Wale ($\sim$5.5 m). Saline plumes persisted for several months and remnants were still observable after 26 months (Terry and Falkland, 2010). Numerical modelling by Bailey and Jenson (2014) showed that wells penetrating the reef-flat plate could significantly reduce the volume of the freshwater body during an overwash event.

Central depressions in the topography of atoll islands may also influence the salinization of FGLs during overtopping events. Chui and Terry (2015) demonstrated that the presence of surface depressions reduces the rate of infiltration of seawater during overtopping events because the soil zone below is already fully saturated. However, a more important finding was that ponding of seawater in the depression acted as a significant source for ongoing seawater infiltration, extending the period of contamination of the FGL. Numerical simulations using a generic island model by Terry and Chui (2012) indicate that for some overtopping events, this topographical feature may cause complete loss of freshwater in the middle portion of the FGL. One of the most advanced studies of seawater overtopping effects was completed by Bosserelle et al. (2015), who used a calibrated 3D variable-density model to assess the impact of future overtopping on the Bonriki lens in Tarawa (Kiribati). The inundation extent and depth were based on detailed bathymetry and topography data, and predictions based on oceanographic models under various assumptions of future sea level, as well as wave and wind conditions. The study highlighted the importance of the timing of the inundation, with effects being more persistent when overwash occurs during a dry period rather than a wet phase.

5.2. Anthropogenic impacts on atoll FGLs

Extraction of water from FGLs has a pronounced effect on the thickness of the lens. Pumping causes drawdown in hydraulic heads and decreases the freshwater discharge to the sea, resulting in thinning of the FGL and widening of the mixing zone (Hunt and Peterson, 1980; Volker et al., 1985). Using the idealised, steady-state analysis of Volker et al. (1985), White and Falkland (2010) demonstrated that pumping at a rate equal to 50% of average recharge will reduce the maximum thickness of FGLs by almost

30%. Mixing causes approximately an additional 20% loss of FGL thickness. By combining the two pumping impacts, White and Falkland (2010) concluded that FGLs will persist provided:

$$\varpi_u < 2(1-q)H_u \tag{9}$$

where q is the ratio of pumping to recharge, given by $q = Q/AW$, where Q is the total pumping rate [$L^3\,T^{-1}$] (see Section 4.3 for an explanation of other parameters).

Because of the demographic pressures on many atoll islands, over-extraction of groundwater has become a common problem. For example, Ibrahim et al. (2002) report that over-pumping has severely affected islands in the Maldives, causing two- to fivefold reductions in FGL thickness on Male' between 1983 and 1989 under the stress of rapid increases in population. During that time, the consumption of water exceeded recharge (Falkland, 1991). Demand for groundwater in many other locations has also increased due to rising populations and growing expectations of the community, with consumption increasing with wealth (White et al., 2007a; Robins, 2013). Historically, water-supply systems design relied on consumption rates of 30–50 L/person/d, whereas 100–150 L/person/d may be more realistic (Falkland, 2002a). These values exclude the water requirements of sectors like tourism and agriculture, which further enhance demand (Robins, 2013). Demand can also spike during periods of low rainfall, as groundwater becomes the sole source of freshwater (Presley, 2005; White et al., 2007a; Bailey et al., 2010). This issue may become more significant in the future if droughts increase in severity, as expected from more intense El Niño/La Niña episodes in the coming decades predicted by climate models (Ali et al., 2001; Scott et al., 2003; Abtew and Melesse, 2013).

Over-pumping from FGLs may cause salinization of individual wells due to the localised rise of the underlying saltwater, a process known as 'upconing' (White et al., 1999b). Griggs and Peterson (1993) simulated different extraction scenarios for Laura Island (Marshall Islands), showing that a pumping rate of 40% of mean annual recharge caused severe upconing, while pumping at a rate of 60% of recharge resulted in complete loss of the FGL. Falkland and Woodroffe (2004) reported regular incidences of upconing due to high pumping rates on Kiritimati (Kiribati), prior to the installation of horizontal collector wells, otherwise known as infiltration galleries. Aside from its impact on water supply, pumping-induced salinization of FGLs can also impact yields from traditional staple crops such as coconut trees and taro (White et al., 1999a; White et al., 2007b).

Land-use changes and contamination from human settlements can also significantly affect FGL systems (White and Falkland, 2010). In some instances, human activities can have a positive impact on the FGL, for example on Tarawa Atoll (Kiribati) where clearing of coconut trees enabled significantly more rainfall to reach the lens as recharge, which thickened parts of the FGL (Falkland and Woodroffe, 2004). However, the impacts on FGLs by anthropogenic activity are more often negative. Atoll FGLs are highly susceptible to contamination from sources such as sanitation systems, agriculture, and landfill, because the water table is typically less than 2 m below the surface (Dillon, 1997; White et al., 2007a). In some instances, surface pollutants can reach the water table in less than 2 h (Falkland, 2002b). The high permeability of the Holocene sediments of Tarawa Atoll (Kiribati) reduce this figure to less than 1 h (White et al., 2006). Mining activities, such as the excavation of sand and gravel for construction, sometimes expose the water table, posing a threat of direct contamination.

Detay et al. (1989) documented significant bacterial contamination of wells across the Federated States of Micronesia with sewerage disposal and landfill often the cause. Poorly functioning sanitation systems are also responsible for much of the widespread groundwater contamination across the Maldives (World Bank, 2013). Large areas of the lenses on Tarawa Atoll (Kiribati) are now unfit for potable use as a direct result of pollution (White and Falkland, 2011). In an attempt to reduce faecal contamination, some islands including Majuro (Marshall Islands) and Tarawa (Kiribati) have installed saltwater sewerage systems, although leaks may contribute to salinization of the lens (Robins, 2013). Falkland (1991) suggested that standard design criteria (such as minimum distances) for the placement of infrastructure are not appropriate for atoll island conditions because of the high vulnerability of FGLs to contamination. White et al. (2005) explore the impacts of land-use within groundwater source areas on Tarawa Atoll (Kiribati) in detail.

The response of atoll islands to climate-induced SLR is the subject of debate in the literature (Webb and Kench, 2010; Terry and Chui, 2012; Weiss, 2015). Robins (2013) dismisses SLR as a critical threat to FGLs, because already-occurring storm surges can significantly exceed predicted sea-level change in many regions. However, Hoeke et al. (2013) show that modern SLR increased appreciably the severity of inundation events across the Western Pacific and conclude that such events will become more frequent. Atoll islands in the tropical Western Pacific will become especially vulnerable to these impacts because SLR in this area is over three times the global average (Keener et al., 2012). Aside from the impacts on freshwater supplies, this has important implications for the cultivation of salt-sensitive crops like taro, with evidence of salinity impacts of this type already reported in the literature (Ayers, 1990; Hoeke et al., 2013; Chui and Terry, 2015).

While Woodroffe (2008) also concludes that SLR will likely exacerbate inundation, he cautions against attributing erosion (and subsequent loss of island area) to climate change without detailed consideration of the island geomorphology and the accretion that may accompany SLR. Connell (2015) reiterates this, and suggests that reports of SLR-induced erosion in urban areas may partially explain the widespread perception that SLR will similarly erode the sand mass of atoll islands. On the contrary, Webb and Kench (2010) and Kench et al. (2014) show that atoll islands often remain stable or even increase in area during rising sea levels, a finding that contradicts the assumptions of several studies on SLR (e.g., Ketabchi et al., 2014). Webb and Kench (2010) further demonstrate that despite maintaining the same land area during SLR, atoll islands have undergone other large morphological changes such as lagoon migration and lateral extension of shorelines, with accelerated SLR likely to increase the pace of these changes. Kench et al. (2009) highlights that along with these changes, the degree of disruption to key geomorphological processes must also be considered when predicting the consequences of climate change for coral reef geomorphology.

Studies on Tarawa Atoll (Kiribati) and Enjebi Island (Marshall Islands) have shown that SLR below 1 m is unlikely to cause adverse impacts on FGLs if land around the margins is not lost (Oberdorfer and Buddemeier, 1988; Alam and Falkland, 1997; World Bank, 2000). Some studies indicate that SLR may increase the thickness and volume of FGLs, as more of the lens would sit within the upper, less-permeable Holocene aquifer (Alam and Falkland, 1997; White and Falkland, 2010). However, whether this occurs depends on increases in ET arising from SLR-induced higher water tables. Preliminary modelling by Gulley et al. (2016) further shows that rising water tables from SLR may lead to lake formation in low-lying areas of atoll islands, creating greater losses of groundwater through ET. Analytical modelling by Ketabchi et al. (2014) indicates that although land-surface inundation from SLR significantly affects FGLs, the freshwater volume is more sensitive to changes in recharge. Connell (2015) highlights that a wider range of factors than SLR and climate change, including anthropogenic factors, lead to adverse impacts to atoll islands.

5.3. Sustainable yield estimation

For atoll islands where hydrogeologic data are lacking, simple rules have been devised to predict whether the island is likely to host usable freshwater. For example, for a given W, larger islands are more likely to sustain usable FGLs than smaller islands (Bailey et al., 2008), primarily because wider islands capture more rainfall, and subsequently greater total inputs of recharge. This leads to higher rates of fresh groundwater discharge to the sea, which invoke steeper head gradients at the shoreline and thicker FGLs (Falkland, 1991). Using the Dupuit approximation and assuming a steady-state sharp interface, the maximum thickness (H_u) of the FGL is proportional to the square root of W, and to L, or in the case of circular islands, the radius (see Section 4.3) (Visher, 1960; Henry, 1964). Oberdorfer and Buddemeier (1988) developed an empirical relationship between lens thickness, annual rainfall and the island width (L). Their analysis indicates that no FGL will form on islands less than 118 m wide, regardless of rainfall. Field measurements on the Cocos (Keeling) Islands show that the smallest island with a FGL was 270 m wide, under a W of 1938 mm/y (Woodroffe and Falkland, 2004). Underwood et al. (1992) presented a series of curves relating L to potable water depth for different rates of W. They found that usable freshwater occurs where W is 2000 mm/y or greater and the island is at least 250 m wide, whereas usable FGLs develop where W is 250 mm/y and L is 750 m or greater. This was based on the assumption that a minimum FGL thickness of 2–3 m is required for a lens to be viable. Bailey et al. (2009) drew a similar conclusion from numerical simulations, and found that as the thickness approaches the depth of the HPU, deepening of the FGL diminishes with increasing W. This trend is pronounced in islands with widths greater than 600 m. In addition to recharge, the minimum L that allows for a usable FGL is dependent on island characteristics that include aquifer properties, vegetation density and freshwater-seawater mixing processes, amongst others (Falkland, 1994a).

FGL studies usually aim to estimate or at least inform the sustainable yield, which is the rate at which groundwater can be extracted without causing adverse effects. Sustainable yields of atoll islands have traditionally been based on percentages of recharge, commonly 25–50% (Hunt and Peterson, 1980; White and Falkland, 2010). For example, Ibrahim et al. (2002) used 30% of rainfall as an approximate estimate of sustainable yield to determine that Maldivian FGLs can support 69 persons/ha, whereas ten islands in the Maldives have population densities ranging from 107 to 602 persons/ha. This approach recognises that a large amount of average annual recharge is required to maintain FGLs in a particular year (White and Falkland, 2011). However, sustainable yield estimates should ideally take into account a greater range of factors, including recharge rates, mixing processes, drought periodicity and the availability of freshwater from alternative sources. Without considering these processes, it is difficult to predict the degree to which extraction affects the FGL thickness, width of the transition zone, and resilience of the FGL to drought and overtopping (e.g., Hunt and Peterson, 1980; Falkland, 1993). Aside from hydrogeological elements, selecting FGL sustainable yields needs to account for societal, environmental and economic aspects (e.g., Rudestam and Langridge, 2014). For example, the acceptable salinity limit of drinking water, which has a significant bearing on calculations of sustainable yield, may differ between drought and non-drought periods depending on community expectations and economic factors.

Management of atoll FGLs requires measurement, interpretation and prediction of the freshwater extent. FGL thickness is usually taken as the depth below the water table to a particular salinity value. Studies by Todd and Meyer (1971), Vacher (1978), and Ritzi et al. (2001) adopted 50% relative seawater salinity, representing the centre of the mixing zone, as the limit of the FGL. This enables comparison of field data to GH-type analyses. In many other studies, the depth to potable water within an FGL has been based on a salinity of 2.5% relative to seawater, or approximately 500 mg/L chloride (e.g., Lloyd et al., 1980; Underwood et al., 1992; Bailey et al., 2010). In the case of Lloyd et al. (1980), this limit was derived by applying a safety margin to the 600 mg/L limit recommended in the World Health Organisation (WHO) drinking water guidelines that were current at the time (WHO, 1971). This concentration has since been commonly adopted to calculate potable water depths for atolls in the Pacific (Falkland, 1988; Falkland and Woodroffe, 2004). The 600 mg/L chloride limit is significantly higher than the current concentration recommended by WHO (2011) for drinking water quality (250 mg/L), which is based on aesthetic rather than health criteria. However, it reflects the salinities of atoll FGLs, and is necessary to obtain sufficient quantities of water for local communities (Falkland, 1991; Woodroffe and Falkland, 2004; Alam et al., 2002). WHO (2011) suggest that palatability is increasingly compromised for total dissolved solids (TDS) levels >1000 mg/L.

Since the electrical conductivity (EC) is often the parameter that is measured in the field as a proxy for salinity, the lens geometry and potable limits are sometimes expressed as EC values. For Bonriki (Kiribati), White and Falkland (2004) defined the upper limit for fresh water as EC = 2700 μS/cm and adopted a conversion relationship of TDS = 0.74 × EC; however coefficients other than 0.74 have been used elsewhere. For example, van der Velde et al. (2006) used 0.64 for Tongatapu (Tonga), whereas Jacobson (1976) used 0.54 for Cocos (Keeling) Islands. The divergence of these values reflects the non-uniqueness of the TDS-EC relationship arising from processes other than mixing that affect it. These include carbonate dissolution and ET. The uncertainty that this imparts is problematic when converting model simulation outcomes, which are usually in terms of TDS, to EC values, which are required for model calibration purposes. Unfortunately, the largest relative uncertainty occurs within the range of potable water guideline values, and is resolvable only through the attainment of water chemistry data.

The challenge for atoll island communities is to determine and regulate long-term sustainable extraction rates. Meaningful estimates of sustainable yield require significant field measurements, including historical observations of FGL performance (Hunt and Peterson, 1980; White et al., 2002). When sufficient data are available, the use of numerical models provides opportunities to project into the future plausible outcomes from alternative management approaches (Table 1). Bosserelle et al. (2015) reports on multiple revisions of the sustainable yield for Bonriki Island (Kiribati), starting from <170 m^3/d (Mather, 1973) to the current value of 1660 m^3/d, based on historical and recent modelling.

There is concern for some atoll islands where pumping has occurred at or above the estimated sustainable yield, given the approximate nature of recharge and sustainable yield estimates, and the inherent uncertainties in predicting future threats, such as climate change, catastrophic events, and population rise (Roumasset and Wada, 2010; Rudestam and Langridge, 2014; Bailey et al., 2015; Bosserelle et al., 2015). Rates of extraction are approximately equal to the sustainable yield on Home Island (Cocos (Keeling) Islands, Woodroffe and Falkland, 2004), as they are on Tarawa Atoll (Kiribati; Bosserelle et al., 2015), and pumping has often exceeded sustainable yield in the Maldives (Ibrahim et al., 2002). Total use is projected to exceed the sustainable yield in the Maldives by 2033 (World Bank, 2013). Over-extraction from FGLs is usually linked to inadequate understanding or quantification of the resource, poor regulation and monitoring of extraction rates, and/or increasing demand pressures and suboptimal pumping systems (Falkland, 1991; White and Falkland, 2010). Massive water losses from leaking supply systems only aggravate the problem, with losses of up to 40% typical for island systems (Johnson, 2012). Supplementing the groundwater supplies of atoll islands is generally challenging and expensive, although a growing number of islands have turned to desalination plants or water importation in the face of water shortages (e.g., Falkland, 1999; Ibrahim et al., 2002; Presley, 2005). Attempts to enhance artificially the sustainable yield of atoll FGLs through anthropogenic modification of the hydrogeology are rare. Only White and Falkland (2010) suggest selective clearing of vegetation as a means of increasing recharge and, subsequently, atoll island sustainable yield; however, there are concerns that this measure may affect the livelihoods of local residents relying on these naturally occurring resources (White et al., 2006).

5.4. Groundwater development

Groundwater development is achieved in various ways on atoll islands. Domestic water supplies on atoll islands are most often obtained from hand-dug wells, which take advantage of shallow water-table depths (Falkland, 1991). Water is often extracted by hand bailers, hand pumps or small electric pumps, which provide adequate yield for household supply (White and Falkland, 2010). However, increased demand and a need for centralized water governance has led to large-scale, municipal water-supply systems on highly populated atolls (Falkland, 1994b). A brief summary of the main approaches used for groundwater development is given below, with a more extensive overview provided by Falkland (1991).

Traditional groundwater development on atoll islands was achieved through the installation of vertical wells (Falkland, 2002b). However, salinization of vertical wells due to upconing, especially in thin FGLs, led to the need for alternative designs (Griggs and Peterson, 1993; White and Falkland, 2010). Where the lens is sufficiently thick, low-pumping wells spaced widely apart have drawn target groundwater volumes of adequate water quality from FGLs (Whitaker and Smart, 2004).

Large-scale water supply from groundwater is now more commonly achieved using horizontal extraction systems such as infiltration galleries or skimming wells (Falkland, 2002b). Positioned at or just below sea level, skimming well systems comprise of a series of horizontal, permeable conduits connected to a central pumping well (Falkland, 1994b). Infiltration galleries are able to skim water from the upper part of the FGL, minimizing excessive drawdown and thus the risk of upconing (Woodroffe and Falkland, 2004). As demonstrated numerically by Peterson (2004) and Griggs and Peterson (1993), this method is the most efficient means of extracting groundwater on atoll islands, resulting in sustainable extraction rates that are approximately double those possible from individual boreholes. White et al. (2007b) further showed that the drawdown from skimming wells is relatively small, finding that the drawdown in the water table on Tarawa Atoll (Kiribati) was less than diurnal tidal variations. Comparisons of salinity data collected before and after the installation of the skimming wells by Falkland (1993, 1994b) showed a reduction in the salinity of the extracted water attributable to the shallower depth of abstraction. The use of infiltration galleries is now widespread, being used to extract fresh groundwater from Tarawa and Kiritimati (Kiribati), Home Island (Cocos (Keeling) Islands), Aitutaki Island (Cook Islands) and Majuro and Kwajalein Atolls (Marshall Islands) (Hunt and Peterson, 1980; Falkland, 1994a; Peterson, 2004; White and Falkland, 2010).

5.5. Management and regulatory water policy

Planning and management controls, in the form of policy and regulation, are critical to maintain FGLs and for the long-term sustainability of atoll island communities. These should cover not only assessment and abstraction of the resource, but include provisions for ongoing management and monitoring of the system and protection of the catchment (Falkland, 2002b). We summarise the key elements of water management and regulatory policy below; additional discussions are provided by Falkland (1991, 2002a), and White et al. (2006, 2007a).

The importance of comprehensively assessing available groundwater resources prior to development is discussed above. However, ongoing monitoring and analysis of FGLs is also an integral part of water management (White and Falkland, 2010). This accords with Werner et al.'s (2011) recommendation that coastal groundwater should be managed through a combination of flux-based (estimates of pumping versus recharge) and trigger-level (minimum allowable groundwater levels, or maximum allowable salinity levels) strategies. In spite of this, long-term monitoring remains a significant problem for many island communities, compounded by insufficient trained personnel, resources and equipment (Falkland, 2002a). Regional organisations have begun to assist in addressing this issue, but ongoing training and mentoring of staff are still required (van der Velde et al., 2007a, White et al., 2007a).

Management of supply and reticulation systems is also necessary for sustaining atoll FGLs. Pumps must be effectively monitored and metered to ensure extraction rates remain below the sustainable yield (Falkland, 1991). Regular maintenance is also needed to detect and repair damage or leaks (Falkland, 2002b), which can result in the loss of up to 70% of the total volume of water extracted (White and Falkland, 2010). Appropriate management of groundwater-supply systems suffers from a lack of institutional capacity, cohesive national policy, and government and community support, with often no clear definition in the roles and responsibilities of the various agencies and departments involved (White et al., 2007a; Moglia et al., 2008). There is also widespread failure to value water resources at appropriate levels. This can lead to a lack of coordination between separate agencies, inefficient use of resources, reluctance to share information, and conflicts with local communities (Crennan, 2002; Falkland, 2002b; White and Falkland, 2010).

Numerous studies highlight stakeholder participation and behaviour change as critical elements of successful water management on atoll islands, but note that they are rarely addressed in practice (White et al., 2006; Moglia et al., 2008). This is often due to a preoccupation with technical solutions or a reluctance from both government and donor agencies to invest the required time and money into such long-term programs (Crennan, 2002; White et al., 2006). Failure to understand traditional values and practices or to address the cultural and social barriers to technical and managerial approaches will continue to impede effective water supply and governance (White et al., 2006). For example, Moglia et al. (2008) cites a lack of social acceptance of current and proposed infiltration galleries on South Tarawa (Kiribati), due to the need for land acquisition, as a major obstacle to improving supply. Further, despite demonstrating the low impacts of pumping on crop production compared to natural stressors, White and Falkland (2004) acknowledge that until the community is included in the research, the findings will do little to change the negative perceptions of current extraction practices.

Demand management is becoming a prominent feature of water management programs (White et al., 2007a). Metering and charging for water has been successful in some locations (e.g., Marshall Islands, Tonga, Federated States of Micronesia), often resulting in a significant reduction in demand (Falkland, 2002a). In Kiribati, single-dwelling storage tanks supplied by a continuous trickle feed (suitable for meeting daily demands) have been trialled as a demand management approach, transferring the responsibility for conserving water to households (Metutera, 2002); however this approach failed to generate significant changes in demand and is consequently not used in new connections. In some cases where supply is insufficient, demand is more or less managed through intermittent supply. However, such an approach leads to significant wastage as people leave taps on permanently to intercept supply when it resumes (Robins, 2013). Community education and awareness regarding water conservation is recognised as a necessary component of any demand management program (Falkland, 2002a).

Effective catchment management across atoll islands is needed to protect FGLs from contamination. A number of atoll island governments have declared certain areas as public groundwater reserves, which restrict the level and nature of development that can occur (Falkland, 2002a). However, competing demands for limited land, increasing population densities and traditional land-use rights have made the establishment and management of these reserves problematic and in some cases they have been abolished (White et al., 2007a). White et al. (1999a) and Moglia et al. (2008) recommend the involvement of landowners in the management of groundwater reserves as a potential solution; however, they remain excluded from the process. White et al. (1999a) gives a detailed description of the issues involved in the management of water reserves on atoll islands along with potential solutions, using the case of Bonriki (Kiribati).

6. Research needs

6.1. FGL physical characteristics and processes

The key factors that control freshwater-seawater mixing in atoll islands have been evaluated for idealised conditions (see Section 3.1). However, mixing is influenced by important interrelationships between the effects of K_H-K_P contrasts, the HPU, tides, pumping, heterogeneity and density effects. Therefore, a systematic evaluation of mixing on atoll islands is warranted whereby more of the key controlling factors are combined. This analysis should draw on new field measurements of mixing zones, including to depths that extend well below the HPU and considering a range of different atoll island settings. Future research should attempt to reconcile mixing zones obtained from field measurement with 3D numerical simulation, and explore further the primary controlling factors using such techniques as geophysical surveys and physical experimentation.

Many of the same factors that drive mixing will contribute to freshwater and seawater flow patterns in atoll islands. However, key features of groundwater-flow patterns on atoll islands have not been characterised. For example, while seawater recirculation in continental aquifers has been examined under both non-tidal (e.g., Smith, 2004) and tidal (e.g., Lenkopane et al., 2009) conditions, seawater recirculation in atoll islands has not been explored. Even the three-dimensional nature of freshwater flow in atoll islands is rarely known, despite that knowledge of these flow fields is essential for determining the fate of contaminants.

The heterogeneous nature of atoll sediments imparts significant influence on FGLs (see Section 3.2). However, the majority of previous efforts to quantify FGL behaviour treat the Holocene and Pleistocene sequences as homogeneous, with the exception of the reef-flat plate. Additionally, the HPU is often included in modelling studies using a highly simplified geometry, often treated as a horizontal layer, whereas geophysical surveys of the HPU show a complex geometry that likely modifies the FGL characteristics. Further research into the influence of heterogeneities on groundwater flow will enhance future attempts to predict contaminant pathways and mixing processes within atoll islands, particularly if tidal effects, pumping and recharge variability are integrated into the analysis. This research should include the role of the reef-flat plate.

While ET from topographic depressions can have a strong influence on atoll island FGLs, previous studies to estimate ET and recharge neglect many of the topographical features that dominate atoll landscapes, such as taro pits, sand mining excavations, and natural land-surface undulations. The difference in ET between shallow water-table conditions and exposed water are expected to be significant where the capillary fringe is narrow (e.g., within coarse Holocene sediments). Future research to account for topographic variability will probably require high-resolution surveys (e.g., lidar; Shih et al., 2011), and analysis of the contrasts between open water and shallow water-table ET. These data will assist also in the analysis of the impacts of seawater inundation events.

The effects of tides on coastal aquifers are multifaceted. While the role of tides in modifying mixing-zone dynamics on atoll islands has been studied for simple situations (see Section 3.4), other tidal controls have been overlooked. For example, tides are known to create a seawater recirculation plume (e.g., Werner and Lockington, 2006; Robinson et al., 2006) that has not been evaluated in atoll settings. The development of seepage faces during falling tides influences submarine groundwater discharge; however this has not been considered in previous atoll island investigations. Tides also create water-table over-height, which raises the time-averaged head of the sea at the shoreline (e.g., Nielsen, 1990), and potentially influences groundwater flow (e.g., Carey et al., 2009). There is a need to calculate differences in water-table over-height between the lagoon and ocean sides of atoll islands, to assess whether this contributes to the FGL asymmetry that is common in atoll islands (see Section 3.5). An investigation to determine the time-averaged effect of the various processes accompanying tidal fluctuations on atoll shorelines is required to inform the choice of boundary conditions in non-tidal modelling studies of atoll islands. This will need to incorporate morphological and tidal signal differences between the lagoon and ocean sides of islands.

Hunt et al. (1995) and Hunt (2004) recommend to investigate the relative influences of K and aquifer storage on atoll island tidal propagation, which to date have only been explored theoretically. The effect of reef-flat plates on groundwater tides is also worthy of additional investigation, which should consider the field observations of tides in atoll islands containing reef-flat plates by Ayers and Vacher (1986) and Ayers (1990). Additionally, tidal pumping through fractures and wells, including within the reef-flat plate, may be an important process, but this has been neglected in prior atoll island studies (e.g., Guha, 2010; Bailey and Jenson, 2014). Given the small amplitude of the water-table fluctuations in response to tidal forcing, there is a risk of them being mistaken for atmospheric tides, i.e., periodic air pressure variations driven by the daily cycle of heating and cooling of the atmosphere. These are known to contribute measurably to water levels in continental coastal aquifers (Merritt, 2004), and can presumably be equally important in atoll islands. More studies of the water-level variations on atoll islands that decompose the signal into the various contributing components are therefore required.

The enhanced mixing created by tides means that higher dispersion parameters are required in non-tidal models of atoll islands. Future atoll island modelling would benefit from a systematic analysis of relationships between common dispersion parameters (for use in non-tidal simulation), tidal characteristics and hydrogeological parameters. This would inform the selection of dispersion parameters for use in non-tidal numerical models.

Studies that investigate the effects of SLR and overtopping on atoll FGLs neglect tidal effects. SLR is likely to modify the beach morphology; however, the change in shoreline conditions due to the combination of SLR and tidal effects has not been investigated.

This would need to consider sediment accretion and erosion effects to incorporate properly the role of beach slope on changes to tidal controls on FGLs. Such studies require multi-disciplinary collaborations between groundwater experts, coastal geomorphologists and oceanographers to comprehensively characterise the changes to the coastal zone that will impact on FGLs.

6.2. FGL investigation methods

While some atoll islands are relatively well-characterised, significant uncertainty in the subsurface and groundwater properties remain despite prolonged and multifaceted field monitoring programs. Emphasis should be placed on installing dedicated salinity monitoring boreholes, as these have been shown to be the most appropriate and comprehensive method for long-term monitoring of FGLs. Obtaining agreement between alternative methods is often difficult (e.g., Sinclair et al., 2015a). Therefore, efforts to apply and adapt field measurement techniques, such as drilling and geological sampling, geophysics, direct and automated measurement of groundwater heads, hydrochemical and tracer sampling and interpretation, and remote sensing, to the specific conditions of atoll settings should continue. In particular, comparisons between multiple approaches to subsurface characterisation are required to improve the current understanding of the uncertainty in the various techniques.

Despite the fact that tidal fluctuations are routinely used to obtain aquifer parameters for continental coastal aquifers, this approach has so far had limited application within atoll island literature. While tidal response data are used to infer the degree of hydraulic connection between the aquifer and ocean, this information is largely in specific project reports rather than published papers and therefore is often not widely available. Banerjee et al. (2008) applied the method described by Ferris (1951) to estimate T and S for Lakshadweep Atoll (India), with resulting parameters of the same order as those obtained through pumping tests. A subsequent study by Chattopadhyay et al. (2014) using the same method suggested that the difference in results was due in part to spatial variation in aquifer parameters and geology. Rotzoll et al. (2013) also concluded that the classical Ferris (1951) interpretation model functioned well in the limestone of the Northern Guam Lens Aquifer, which shares many characteristics with atoll aquifers. These findings indicate that there is merit in wider application of tidal methods to infer the aquifer properties of atoll islands, although guidance is presently lacking on the specific conditions on atoll islands within which these techniques can be used.

Remote sensing techniques are used routinely to obtain water-balance components in continental settings, although the scale of resolution is typically rather coarse (e.g., Zhang et al., in press). Application of these approaches to atoll island characterisation needs to overcome several challenges. Canoy (1983) investigated the potential for remote sensing to assess soil moisture and therefore to identify possible aquifers in the U.S. Virgin Islands, and found the approach to be unsuitable. Other attempts to apply remote sensing to investigate small islands have encountered problems associated with cloud cover, vegetation density, and the small spatial scales of atoll island hydrogeological features, which in combination have inhibited efforts to obtain meaningful hydrogeological information (Burke et al., 1988; Falkland, 1991). Investigations targeting atoll FGLs rarely apply remote sensing, despite the successful application of airborne methods to map subsurface salinity distributions in continental coastal settings (e.g., Jørgensen et al., 2012). To a large degree, this is due to their relatively small size, but this may change with current advances in unmanned aerial vehicles, which are used for an increasing range of environmental, agricultural and land management remote sensing applications (e.g., Colomina and Molina, 2014).

Time-series analysis in general appears to be a promising field of research for atoll islands because it exploits the predictive capability of correlations between indicators of the lens state and rainfall. For example, van der Velde et al. (2006) were able to model the variability of the salinity of pumped water based on input time series of rainfall, and found that it was possible to predict freshwater salinity based on the SOI with a lag time of 10 months. The advantage of such models is that they require input time series of parameters that are relatively easy to obtain, such as rainfall, without the need for a complete characterisation of the subsurface in terms of salinity and hydraulic property distributions. However, site-specific model parameters are needed, which can only be determined if measurements, spanning a sufficiently long period of time, are available. Moreover, van der Velde et al. (2007b) encountered difficulties with obtaining realistic parameter distributions, and the robustness of time-series models when various processes change simultaneously is presently unknown. Being able to predict salinity several months into the future without the need for a complex numerical model is appealing and therefore further research is warranted into time-series applications.

The sharp-interface assumption of the majority of FGL analytical solutions inhibits their utility in many atoll settings, where mixing zones are usually relatively wide. A correction factor to account for the lack of dispersion in sharp-interface solutions was developed by Pool and Carrera (2011), and later modified by Lu and Werner (2013). While the correction has been applied to continental settings, there have been no attempts to use it under island situations. Research to combine the analytical solution of Ketabchi et al. (2014) and the Pool and Carrera (2011) correction factor, with comparison to the mixing-zone solution of Volker et al. (1985), may allow for wider application of existing sharp-interface analytical solutions. Alternatively, direct coupling of Ketabchi et al.'s (2014) analytical solution with Volker et al.'s (1985) mixing-zone theory may provide similar utility. Analytical solutions of groundwater age in dual-layer systems, to extend existing single-layer solutions by Chesnaux and Allen (2008) and Greskowiak et al. (2012), would add further to the available tools for assessing atoll FGLs.

Numerous studies point to the lack of suitable methods for the direct and indirect measurement of evapotranspiration on small islands. Inaccurate estimates of this component of the water balance lead to errors in quantifying recharge and sustainable yield. More research is required into adapting techniques used on larger landmasses for the specific characteristics of atoll islands. Comte et al. (2014) highlights in particular the need to understand better the regression of transpiration that occurs as plants reach their salt tolerance under seawater intrusion or drought scenarios.

6.3. FGL management

Previous studies of atoll island inundation have adopted simplified subsurface conditions in analysing the subsequent FGL salinization, including its persistence. The role of the unsaturated zone is commonly ignored. Factors like air entrapment, antecedent moisture conditions and multi-domain solute transport seem highly relevant but are effectively neglected in overtopping studies to date, possibly due to the rapid infiltration rates of saltwater into the FGL. The assignment of appropriate land-surface boundary conditions for overtopping in numerical models seems to be without guidance, and the representation of density-driven flow in the form of salt fingers in numerical models remains problematic.

The response of atoll FGLs to climate variability and change is the subject of considerable uncertainty. While the effects of storm-induced inundation on groundwater resources are increasingly being assessed in the literature, no consensus has been reached on the likely impacts of SLR. Previous work has shed light on the geomorphic response of atoll islands to historical changes in sea level but further work is required to understand the pace and nature of these changes under future accelerated SLR. Such research would enable SLR to be better represented in numerical modelling studies. Surprisingly, it seems that no studies to date have exploited naturally occurring sea-level variations of up to 20–30 cm, such as those driven by ENSO cycles, to investigate the impacts of future SLR. The effects of those on groundwater levels have recently been identified on Rottnest Island (Bryan et al., 2016), which is a carbonate island that has many similar characteristics to atoll islands. The relevance of such sea-level fluctuations for atoll FGLs has not been assessed quantitatively, and research on this topic seems overdue.

Increases in ET due to SLR-induced water-table rise also remains unquantified, an issue that must be addressed in order to determine the accuracy of studies that suggest SLR may increase FGL volume. Studies of FGLs under future climate conditions have so far tended to examine the influence of single climate change elements such as recharge variability or inundation and have commonly excluded pumping. However, sustainable management of groundwater resources requires an understanding of the combined effect of predicted impacts including extraction regimes. Geographic differences in morphological changes and projected climate scenarios also indicate the need for region or island-specific investigations.

Several knowledge gaps exist in the understanding of contamination from surface processes and related guidance on water-quality protection. For example, the setback distance between points of abstraction of potential contamination sources has not been well studied in terms of large-scale field investigations, and is likely influenced by the physical and chemical properties of the carbonate sands. There is some debate in the literature on the mobility and persistence of pathogens in the subsurface of atoll islands. While a number of studies indicate that the thin soils of atoll islands makes them highly vulnerable to contamination (e.g., Dillon, 1997; Crennan, 2001), other laboratory experiments indicate that the carbonate sands may inhibit the subsurface movement of some pathogens (e.g., Burbery et al., 2015). Further research is needed into the efficacy of carbonate sands for pathogen removal. Atoll-specific knowledge of travel times, subsurface attenuation rates and pathogen decay is also needed to improve planning and management strategies for waste and pollution. The development of appropriate and cost-effective treatment solutions also requires improved knowledge of the rates of pollutant loading into FGLs.

The role of groundwater infrastructure in water-quality management warrants further investigation. Following an investigation on Lifuka (Tonga), Crennan (2001) suggested that because of the population density, there is no safe distance for siting wells away from sanitation facilities, making it necessary to evaluate alternative solutions such as source control and treatment. Various reports (e.g., Loco et al., 2015; Sinclair et al., 2015b) and well surveys across 36 villages and eight outer islands of Kiribati (Beru, Nonuti, Butaritari, Makin, Maiana, Abaiang, Nikunau and Marakei) indicate that well construction is an important factor controlling water-supply well contamination, rather than just the type of, or distance to, contamination sources. Preliminary data show that closed wells appear to host lower levels of E. coli than open wells. Properly constructed wells have casing (e.g., concrete rings) and above-ground parapets, with concrete aprons to shed surface water away from the wells, and effective well covers to reduce the ingress of surface water. The lack of dedicated monitoring wells in the majority of populated atoll islands brings into the question whether contaminated wells are a reflection of aquifer contamination or not, given the ease with which hand-dug wells are accessed from the surface (Metutera et al., 2002).

Studies on submarine groundwater discharge from small islands indicate the importance of this pathway for the health of coastal ecosystems, and highlight that it has received little attention in the literature to date. For example, the contribution of groundwater and its management to the nutrient and pathogen loads into lagoons requires consideration (Moosdorf et al., 2015), given reports of contamination in near-shore environments of atoll islands. In the lagoon of Tarawa Atoll (Kiribati), both seawater and edible shellfish were found to contain faecal bacteria at unsafe levels (Johannes et al., 1979). van der Velde et al., (2007a) report significant and rapid eutrophication of the lagoon of Tongatapu (Tonga) at points of groundwater discharge, but note uncertainty surrounding the source of the contamination. Deterioration of near-shore water quality may also adversely affect the health and functioning of vital reef ecosystems. Ebrahim (2000) and Osawa et al. (2010) found that eutrophication of the lagoon around dense human settlements reduced the production of foraminifera, decreasing the availability of sediments used in the formation and maintenance of coral atolls. The effects of water-quality deterioration on the lagoon and reef ecosystems of atoll islands also remain largely unquantified (Fujita et al., 2014), as does the contribution of different nutrients and pathogens to ecosystem degradation.

The sustainability of FGLs relies heavily on the accurate estimation of sustainable yield (see Section 5.3). While historical investigations have aimed to identify a single, fixed volume of water that can be safely abstracted, the dynamic environment of atoll islands makes this approach potentially problematic. Current research into sustainable management is beginning to suggest that the focus should be on maintaining water quality rather than quantity, with abstraction limited to a salinity level rather than a specific volume. Further research is required into the feasibility of such an approach, especially in regards to the volumetric reductions that may be required as salinity limits are reached. This dynamic approach to volumetric assignment of water in atolls calls for the development of well-informed and prescribed approaches for dealing with regulators, operators and the community, which presents a significant challenge. Hydro-economic models of water management (e.g., Harou et al., 2009), which adopt a dynamic approach to water allocation, may also be applicable to atoll islands, but this requires investigation. Gaps also exist in the development of surrogate warning indicators for FGL contraction. For example, preliminary investigations by White et al. (1999b) showed that while accumulated rainfall percentiles may be appropriate for the prediction of dry periods, they do not necessarily indicate significant reductions in FGL volume.

In the face of growing demand and increased climatic vulnerability, an exploration of the strategies to increase and improve water supply on atoll islands is needed. In many cases, rainwater tanks are used to supplement domestic water supply, with their increased utilisation often cited as a means to improve supply and reduce vulnerability, especially during drought. A comprehensive assessment covering both rainwater and groundwater is needed to optimise the storage and use of freshwater on atoll islands, and to understand interrelationships between aquifer recharge and surface water capture and redistribution. Apart from some discussion of selective land clearing, an examination of engineering measures to increase the volume of fresh groundwater is also largely missing from the literature. For example, the infilling of historical borrow pits to improve groundwater recharge was suggested in the 2010 Tarawa water master plan (White, 2010), and is currently the subject of a feasibility study of the western end of Bonriki (Kiribati). A similar proposal involves the construction of an island in the lagoon nearby North Tarawa for fresh groundwater production. Gingerich (1996) proposed using captured rainfall to artificially recharge the lens and reduce the effects of washover on Roi-Namur (Marshall Islands), with initial trials proving successful. Methods used in other coastal settings to enhance sustainable abstraction (e.g., Lu et al., 2013) should be evaluated for their applicability to atoll islands. Sustainable expansion of groundwater-supply systems would benefit from improved quantification of domestic and commercial rates of consumption as well as system losses such as leakage.

Unlike terrestrial systems, the importance and health of groundwater-dependent ecosystems has not been considered for atoll islands, with surveys of freshwater fish or other island species absent from the literature (e.g., Donaldson and Myers, 2002). In many cases, economic interests are in direct opposition to the needs of the environment (van der Velde et al., 2007a). Exclusion of these ecosystems during groundwater development threatens both environmental sustainability and the economic development of atoll island communities (van der Velde et al., 2007a).

Apart from the study by White and Falkland (2004), the impacts of changes in the groundwater system on cropping (e.g., taro cultivation) and other important land-uses have not been properly investigated (Rao et al., 2013). Salinisation of the FGL through inundation or drought is already causing significant damage to local crops such as taro, with anticipated rates of SLR potentially exacerbating the issue. Given that initial findings indicate taro and similar species may be able to tolerate short spikes in salinity but not long-term increases, particular focus should be placed on identifying salt-tolerant varieties or alternative methods of cultivation (Webb, 2007; Sinclair et al., 2011; Rao et al., 2013). It should be noted that in some cases, such as South Tarawa and parts of North Tarawa (Kiribati), damage to taro crops is primarily due to the taro beetle rather than salinization, with the majority of pits now abandoned (Thomas, 2002).

Unsurprisingly, atoll islands suffer from data shortcomings in all but a handful of cases. This continues to hamper the assessment and management of atoll island FGLs. While concerted effort is needed to improve data collection and monitoring strategies across all atoll islands, continued use of data from established and long-running sites such as Bonriki (Kiribati) or Cocos (Keeling) Islands is encouraged. In addition, while a host of studies provide generic advice on atoll island hydrogeology and FGL processes, investigation methods, and management, conditions differ from site to site. There appears to be few studies that translate and critically assess general guidelines in terms of the specific conditions of individual atoll island settings, where cost-effective advice to freshwater custodians is crucially important.

7. Concluding remarks

The prospect of rising sea levels has put low-lying coral atoll islands at the centre of international attention as they are among the most vulnerable environments where human-induced climate change may have impending and catastrophic impacts (Weiss, 2015). Notwithstanding the relevance of climate change threats, another cause for concern is that population growth and development, particularly through inward urban migration from outer islands to the main islands (e.g., UNFPA, 2014), have led to rising demands for freshwater and expanded land areas of human activities in these atolls. The permeable soils and shallow water tables of atolls provide a limited barrier to biological and chemical inputs to aquifers, and thus, increasingly precious fresh groundwater resources are at risk to contamination. Given this context, the present literature review aimed at synthesising the current knowledge, and recent advances of atoll island hydrogeology.

Indeed, this review showed that the scientific attention has shifted from a focus on water supply in the studies up until the mid-1990s, to the impacts of climate change effects, such as seawater overtopping and drought, on freshwater availability in more recent studies. Regardless of the focus, the scientific endeavour to investigate FGLs has led to significant advancement in the understanding of the features and processes that control their occurrence. It has contributed to the scientific specialisation of coastal hydrogeology more broadly in that some discoveries from research into atoll islands are transferrable to coastal zones on the continents as well. For example, the geometry of the mixing zone between freshwater and seawater and the dynamics thereof, as investigated on several islands, provide prime field examples that can rival well-executed studies of transition zones on continental aquifers in both number and level of detail. Moreover, management practices that are commonplace on atoll islands, such as the use of skimming wells, are important examples of SWI mitigation strategies that could be applied elsewhere. On the other hand, Ketabchi et al. (2016) recently noted that published studies of SLR impacts on coastal aquifers are biased towards the conditions typical of continental settings, and highlighted the need for more studies of small islands. We support this notion, as our review shows that the resilience of atoll FGLs to changing natural and demographic factors has only recently emerged as an area of active research.

Quantification of the surface and subsurface hydrological components of atoll islands remains highly challenging and requires multifaceted investigations given interrelationships between human activities, climate stresses, ocean forces, and the complex hydrogeology of atoll islands. Difficulties in reconciling field measurements with modelling results have highlighted the importance of processes extrinsic to modelling assumptions. The results of scientific exploration and analysis require authentic engagement with island communities before management practices and policies can adapt to new knowledge of freshwater resources. However, as long as quantitative prediction of the dynamic response of FGLs to a combination of stresses remains associated with a high degree of uncertainty, adequate management and policy responses cannot be formulated. There are justifiable fears that SLR will destroy the FGL before islands become submerged, but such fears may lead to unfounded claims that misinform stakeholders if the true responses of FGLs are not properly understood. Hydrogeologists must therefore continue their key role in developing the science that will help island communities face the challenges of the future.

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