Seawater intrusion processes, investigation and management: Recent advances and future challenges

Adrian D. Werner[a,b*], Mark Bakker[d], Vincent E.A. Post[a,b], Alexander Vandenbohede[c], Chunhui Lu[a.b], Behzad Ataie-Ashtiani[a,b], Craig T. Simmons[a,b], D. A. Barry[e]

[a]National Centre for Groundwater Research & Training, Flinders University, GPO Box 2100, Adelaide SA 5001, Australia

[b]School of the Environment, Flinders University, GPO Box 2100, Adelaide SA 5001, Australia

[c]University of Ghent, Department of Geology & Soil Science, Krijgslaan 281 S8, B-9000 Ghent, Belgium

[d]Delft University of Technology, Water Resources Section, Faculty of Civil Engineering & Applied Geosciences, Delft, Netherlands

[e]Laboratoire de technologie écologique, Institut d'ingénierie de l'environnement, Faculté de l'environnement naturel, architectural et construit (ENAC), Station 2, Ecole polytechnique fédérale de Lausanne (EPFL), 1015 Lausanne, Switzerland

*Corresponding author - Phone: +61-8-8201-2710 Fax: +61-8-8201-3399

Email addresses: A.D. Werner: adrian.werner@flinders.edu.au; A. Vandenbohede: alexander.vandenbohede@ugent.be; B. Ataie-Ashtiani: behzad.ataieashtiani@flinders.edu.au; C.T. Simmons: craig.simmons@flinders.edu.au; C. Lu: chunhui.lu@flinders.edu.au; D.A. Barry: andrew.barry@epfl.ch; M. Bakker: mark.bakker@tudelft.nl; V.E.A. Post: vincent.post@flinders.edu.au

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Abstract

Seawater intrusion (SI) is a global issue, exacerbated by increasing demands for freshwater in coastal zones and predisposed to the influences of rising sea levels and changing climates. This review presents the state of knowledge in SI research, compares classes of methods for assessing and managing SI, and suggests areas for future research. We subdivide SI research into categories relating to processes, measurement, prediction and management. Considerable research effort spanning more than 50 years has provided an extensive array of field, laboratory and computer-based techniques for SI investigation. Despite this, knowledge gaps exist in SI process understanding, in particular associated with transient SI processes and timeframes, and the characterization and prediction of freshwater-saltwater interfaces over regional scales and in highly heterogeneous and dynamic settings. Multidisciplinary research is warranted to evaluate interactions between SI and submarine groundwater discharge, ecosystem health and unsaturated zone processes. Recent advances in numerical simulation, calibration and optimization techniques require rigorous field-scale application to contemporary issues of climate change, sea-level rise, and socioeconomic and ecological factors that are inseparable elements of SI management. The number of well-characterized examples of SI is small, and this has impeded understanding of field-scale processes, such as those controlling mixing zones, saltwater upconing, heterogeneity effects and other factors. Current SI process understanding is based mainly on numerical simulation and laboratory sand-tank experimentation to unravel the combined effects of tides, surface water-groundwater interaction, heterogeneity, pumping and density contrasts. The research effort would benefit from intensive measurement campaigns to delineate accurately interfaces and their movement in response to real-world coastal aquifer stresses, encompassing a range of geological and hydrological settings.

1. Introduction

The management of freshwater reserves is an increasingly important imperative for the custodians of natural resources. Freshwater stored in coastal aquifers is particularly susceptible to degradation due to its proximity to seawater, in combination with the intensive water demands that accompany higher population densities of coastal zones. Seawater intrusion (SI, i.e., the landward incursion of seawater) is caused by prolonged changes (or in some cases severe episodic changes) in coastal groundwater levels due to pumping, land-use change, climate variations or sea-level fluctuations. The primary detrimental effects of SI are reduction in the available freshwater storage volume and contamination of production wells, whereby less than 1% of seawater (~250 mg/l chloride, WHO, 2011) renders freshwater unfit for drinking. The considerable threat of SI on the global scale is well documented [e.g., Kinzelbach et al., 2003; Post, 2005; Barlow and Reichard, 2010].

Here, SI refers to the subsurface movement of seawater, although surface water bodies (e.g., rivers, canals, wetlands) are impacted similarly by intruding seawater. Coastal aquifers are complex environments typified by transient water levels, variable salinity and water density distributions, and heterogeneous hydraulic properties. Climate variations, groundwater pumping and fluctuating sea levels impose dynamic hydrologic conditions, which are inter-related with the distribution of dissolved salts through water density-salinity relationships. These processes are often important at vastly different spatial and temporal scales, although cumulative small-scale factors (e.g., beach-scale dynamics) can combine to have wide ranging impacts on coastal hydrology and SI [e.g., Carey et al., 2009]. A simplified coastal aquifer representation showing a selection of hydrogeological processes of relevance to SI in a shallow unconfined aquifer is given in Fig. 1. Important aspects such as 3D effects, heterogeneity in aquifer properties and geometry, dispersion and diffusion, degree of aquifer confinement, hydrogeochemical processes, etc., are omitted from the figure, but their importance is recognized in the review that follows.

[FIGURE 1]

Fig. 1. Simplified diagram of a coastal unconfined aquifer setting, showing (a) seawater wedge toe, (b) density-driven circulation in the seawater zone, (c) seawater upconing due to well pumping, (d) coastal fringe processes, such as tidal seepage face and upper seawater recirculation zone, (e) head-controlled surface expression of groundwater.

Considerable research effort spanning more than 50 years has been devoted to understand better coastal aquifer flow and transport processes, to enhance coastal water security, and to avoid environmental degradation of coastal systems [Diersch and Kolditz, 2002; Post, 2005]. Indeed, the field of coastal hydrogeology, considered as a sub-discipline of hydrogeology, spans SI, submarine groundwater discharge (SGD), beach-scale hydrology, sub-seafloor hydrogeology and studies on geological timescales involving coastline geomorphology. Despite this, coastal aquifer hydrodynamics and SI remain challenging to measure and quantify, commonly used models and field data are difficult to reconcile, and predictions of future coastal aquifer functioning are relatively uncertain across both regional and local (individual well) scales [e.g., Sanford and Pope, 2010].

Here, we review the literature to outline recent progress in SI research, including both practical and theoretical elements of SI analysis and investigative tools. SI research encompasses a multi-disciplinary range of topics due to the complex nature of coastal aquifer flow and transport, which are influenced by unsaturated zone processes, interactions with surface water systems, shoreline geomorphology, microbiological and vegetation functioning, hydrogeochemical reactions, etc. It follows that SI research often involves linkages across traditionally disparate disciplines. Further, much of the SI literature focuses on the optimal use of coastal groundwater and issues of sustainability (i.e., management), including the uptake and application of new knowledge in understanding natural system functioning, and ultimately in the deployment of operational practices for regulating groundwater extraction and in mitigating SI. Given the inherent uniqueness of each real-world incidence of SI, well-documented case studies are an important aspect of SI research.

The review aims to summarize the evolution and current status of SI research. The following SI research categories are considered: processes (§2), measurement (§3), prediction (§4), and management (§5). Finally, we provide a prospective view of gaps in SI knowledge and investigative tools (§6). Fundamental aspects of SI theory and management are covered by Reilly and Goodman [1985], Custodio and Bruggeman [1987], Bear et al. [1999], Diersch and Kolditz [2002] and Cheng and Ouazar [2004], and are not repeated here.

2. SI processes

The processes and factors associated with SI are described qualitatively by Custodio [1987a, 1987b]. These include dispersive mixing, tidal effects, density effects including unstable convection, surface hydrology (e.g., recharge variability and surface-subsurface interactions), paleo-hydrogeological conditions (i.e., leading to trapped ancient seawater), anthropogenic influences, and geological characteristics that influence the degree of confinement as well as aquifer hydraulic and transport properties. The interactions between these and other processes (e.g., geochemical reactions, tsunamis and other episodic ocean events, beach morphological controls on shoreline watertable conditions, unsaturated zone flow and transport) provide for a seemingly infinite array of possible settings in which SI can occur. This poses a significant challenge for water resource managers in identifying the primary SI controlling factors and considering these in both the evaluation and optimization of groundwater use. In many cases, field observations reveal behavior that is unexpected or inexplicable if only a subset of coastal aquifer processes is considered [e.g., Turner and Acworth, 2004]. Laboratory and numerical modeling experimentation are typically the tools of choice in seeking to elucidate the influence of individual processes from multifaceted field measurements, and many of the studies reviewed below incorporate these types of approaches.

In all SI situations, controlling factors include buoyancy forces associated with density variations (controlled mainly through solute concentration but sometimes through temperature; Dausman et al. 2010b), advective forces resulting from freshwater discharge, dispersive mechanisms, and hydrological/geometric boundary condition controls (e.g., aquifer thickness and extent and characteristics of sources and sinks). Time lags in SI responses to both cyclic and prolonged fluctuations in aquifer stresses introduce further complexity, and could potentially confound the interpretation of SI from field-based measurements [e.g., Watson et al., 2010]. The situation of stable density stratification (freshwater overlying saltwater) is the most common salinity configuration, and is often explored through modified forms of the Henry [1964] problem. Dentz et al. [2006] defined dimensionless parameters for the Henry problem, representing the interplay between advective and diffusive mechanisms (i.e., the Péclet number; *Pe*), and the ratio of buoyancy to viscous forces (i.e., the Rayleigh number; *Ra*). Convective circulation of seawater, an important component of SI behavior, is demonstrated in the Henry problem. Abarca et al.

[2007b] considered dispersive forms of the Henry problem as more realistic representations of SI, and generated additional dimensionless parameters accounting for dispersive and anisotropic controls. Werner et al. [2012] considered confined and unconfined dimensionless parameters representing the mixed convection processes for steady-state situations involving recharge and a sharp freshwater-saltwater interface. The interplays between mechanisms associated with unstable situations (saltwater overlying freshwater), as might occur through transgression-regression cycles, tsunamis, etc., were discussed at length by Simmons et al. [2001], Diersch and Kolditz [2002] and Simmons et al. [2005].

2.1. Dispersive processes and freshwater-seawater mixing zones

The mixing zone between freshwater and intruding seawater is an important feature of coastal aquifers [e.g., Michael et al., 2005]. The salt concentration and fluid density vary across the mixing zone due to mechanical dispersion and molecular diffusion, which drive salt into the outflowing freshwater thereby contributing to the convective circulation within the wedge [e.g., Abarca et al., 2007a; Abarca et al., 2007b]. Effective management of coastal groundwater resources usually requires that the position and thickness of the mixing zone be reasonably well characterized.

Observed mixing-zone thicknesses vary widely between laboratory experiments, field situations and numerical simulations. Narrow mixing zones in homogeneous porous media were demonstrated in several laboratory experiments [Abarca and Clement, 2009; Goswami and Clement, 2007], and in numerical simulations using fine discretization and small dispersion coefficients [e.g., Jakovovic et al., 2011]. However, mixing zones in field situations vary considerably, ranging from a few meters to kilometers [Paster et al., 2006;

Price et al., 2003; Barlow, 2003]. In steady state, homogeneous systems, the thickness of the mixing zone is affected by mechanical dispersion (both longitudinal and transverse components), molecular diffusion, freshwater discharge, and the density contrast between seawater and the ambient groundwater [e.g., Volker and Rushton, 1982]. Volker and Rushton [1982] simulated wider mixing zones in systems characterized by higher dispersion coefficients, lower freshwater discharge (for a given velocity-independent dispersion coefficient) and higher density contrasts. Velocity-proportional dispersion will likely produce narrower mixing zones with lower freshwater discharge. Held et al. [2005] and Paster and Dagan [2007] suggested that local (or pore-scale) transverse dispersion is the primary mechanism responsible for the existence of steady-state mixing zones in homogeneous systems. In this case, narrow mixing zones are expected given typically small transverse dispersion coefficients obtained experimentally [Paster and Dagan, 2007]. It follows that wide mixing zones observed in field settings are probably not caused by local dispersion effects.

Analysis of an anisotropic dispersive adaptation of the Henry problem by Abarca et al. [2007b] showed that longitudinal and transversal dispersion are equally important in controlling the mixing-zone thickness, which is a function of the geometric mean of the two dispersivity coefficients. In reality, the mixing zones in coastal aquifers are the result of transport processes driven by density gradients, diffusion, dispersion, and kinetic mass transfer, which are influenced by many mechanisms and aquifer characteristics, such as spatial heterogeneities in the geologic structure, tidal and wave forcing, temporal variability in groundwater recharge, long-term changes in sea-level position, and pumping activities. Kinetic mass transfer refers to mass exchange processes between mobile and relatively immobile zones due to low-permeability zones, dead-end pores, etc. [e.g., Coats and Smith, 1964]. Other interplays between aquifer heterogeneities and mixing-zone thicknesses are discussed in the next section. The respective contributions of the various factors listed above to freshwater-saltwater interface thickness are not fully understood.

Fluctuations in both inland freshwater heads and seawater levels can play an important role in modifying mixing-zone thicknesses. Eeman et al. [2011] reported analyses of the mixing-zone thickness between a freshwater lens and the upwelling saline water, and showed that if significant fluctuations of the lens thickness occur, transverse dispersion might be less important than longitudinal dispersion. Ataie-Ashtiani et al. [1999] investigated the effects of tidal activities on SI in an unconfined aquifer and found that dispersion along the interface could be enhanced by tidal forces. Widening of the mixing zone due to tides is apparent despite only small net movements of the mixing zone through each tidal cycle [Ataie-Ashtiani et al., 1999; Chen and Hsu, 2004]. The mixing zone, in reality, seldom remains stationary due to seasonal variations (or less frequent events) in groundwater utilization and surface recharge [e.g., Dausman and Langevin, 2005; Michael et al., 2005]. Based on this assumption, Lu et al. [2009] demonstrated that kinetic mass transfer between the relatively mobile and immobile zones could significantly widen a moving mixing zone. Further studies by Lu and Luo [2010] indicated that mixing enhancement is mainly controlled by the unsynchronized transport behavior of salts in the mobile and immobile zones, and that the mixing-zone thickness within a seasonal fluctuation cycle can vary significantly.

When a partially penetrating well is located in an aquifer where freshwater and saltwater are stratified, pumping fresh groundwater can cause the movement of saltwater from the deeper saltwater zone upward into the fresh groundwater (i.e., saltwater upconing), influencing the mixing zone thickness. Laboratory visualizations and numerical modeling of such processes by Werner et al. [2009] and Jakovovic et al. [2011] demonstrated that longitudinal dispersion creates a dispersed plume during early stage upconing. Once the upconing interface has stabilized, the smaller transverse dispersion controls the interface thickness, which subsequently becomes narrower. The same controls on mixing zone dynamics likely apply to horizontal SI, by which longitudinal dispersion dominates the thickness of a laterally moving mixing zone, such as during active SI, in the same manner as any other mobile contaminant plume. The mixing zones of slower SI are probably influenced more strongly by transverse dispersivity.

2.2. SI in heterogeneous systems

The inherent heterogeneities of geological systems create spatial variations in hydraulic properties that control the transport of dissolved constituents by perturbing fluid flow at various length scales [Diersch and Kolditz, 2002]. In the context of SI, heterogeneity effects are multifaceted and depend on the geomorphologic and geochemical characteristics of the porous medium. The scale of heterogeneity is important. Small-scale heterogeneities appear to have only a minor influence on mixing zone thicknesses, while macroscopic variations in aquifer properties have a stronger influence. Randomly distributed hydraulic conductivities (i.e. small-scale heterogeneities) can enhance macrodispersion and produce wider mixing zones [Kerrou and Renard, 2010], although Abarca [2006] found that the effects of moderate random heterogeneities on widening the steady-state mixing-zone thickness are small. However, the geological structures found in field situations (i.e., large-scale heterogeneities) can produce preferential flow paths by which rapid transport can occur [e.g., Calvache and Pulido-Bosch, 1997]. This follows density-independent analyses by Rahman et al. [2005], who demonstrate that heterogeneity has a negligible impact on transversal mixing, but produces significant plume meandering and distorted concentration profiles. Structured variations in the aquifer bottom boundary can similarly introduce preferential SI pathways [Abarca et al., 2007a; Mulligan et al., 2007]. Identifying and characterizing the controls of local geological features, and representing these appropriately in SI studies is one of the most challenging aspects of SI quantification and prediction, as it is for many problems that involve solute transport.

Few well-documented case studies characterize accurately the influence of aquifer heterogeneities on SI . Kim et al. [2006] highlighted complex interplays between aquifer heterogeneities and tidal effects in their field-based analysis of the freshwater-saltwater interface on Jeju Island (Korea). Oki et al. [1998] concluded that aquifer heterogeneities in the form of stratigraphic discontinuities controlled mixing zone widths in southern Oahu aquifers (Hawaii, USA). In their attempts to simulate a relatively small atoll-island aquifer, Ghassemi et al. [2000] found that aquifer heterogeneities were essential for numerical modeling results to match field observations. Similarly, Hodgkinson et al. [2007] highlighted the importance of heterogeneous stratigraphy on seawater-freshwater mixing in a barrier sand island. They concluded that low permeability sediments controlled the freshwater lens thickness. The review of coastal karst systems by Fleury et al. [2007] identifies conduit flows as the primary mechanism dominating solute transport (and therefore SI) in several coastal settings.

The variability of possible geological structures presents a significant barrier to the generalization of heterogeneity effects on SI. It is clear that such structures must be characterized if local geological controls on SI are important. Understanding is gained from analyses of SI in various simplified heterogeneous configurations. For example, Chang and

Yeh [2010] applied a spectral approach to analyze heterogeneity effects on the position of a steady-state interface, which was represented in 2D planar orientation. The correlation scale of the log-hydraulic conductivity field (i.e., controlling the continuity of preferential flow paths) was an important factor in influencing the position of the interface. Dagan and Zeitoun [1998a, 1998b] evaluated heterogeneity effects on the interface position within randomly stratified aquifers (i.e., with horizontal structures), and found that neglecting layered heterogeneities introduced large uncertainty in estimates of the toe position, but had relatively minor influence on calculations of upconing under axisymmetric conditions. Lu et al. [2011] demonstrated the mixing effects of stratified heterogeneities. They considered SI in a system with a low permeability aquifer layer overlying a higher-permeability layer, presenting results of laboratory experiments and numerical simulations. Slanting upward flow of diluted saltwater and circulated seawater flow was refracted at the interface between the two layers, resulting in streamline separation and mixing zone widening in the low-permeability layer. Their findings help to explain the existence of thick mixing zones, such as in the low-permeability caprock layer overlying a highly permeable volcanic aquifer in Oahu, Hawaii, USA [Oki et al., 1998]. Lu et al. [2011] also found that neglecting layered heterogeneity would lead to the overestimation of the toe penetration length, provided that the inland head boundary remained constant.

Effects of random heterogeneity on stationary mixing zones have been investigated by several authors through adaptations of the Henry problem. Held et al. [2005] employed homogenization theory to investigate the heterogeneous diffusive Henry problem, and found that it was not necessary to upscale the dispersivity coefficient to reproduce accurately the mean behavior of SI in a heterogeneous medium. However, a contrary result was obtained by Kerrou and Renard [2010] who, based on a more realistic dispersive form of the Henry problem, indicated that macrodispersion needs to be accounted for when upscaling SI in heterogeneous media. The contradiction in findings between the studies of Held et al. [2005] and Kerrou and Renard [2010] highlights important differences in SI upscaling that arises from diffusive and dispersive mixing zone representations. Kerrou and Renard [2010] also demonstrated that the impact of heterogeneity on SI was significantly different in 2D and 3D models, both in magnitude and in general trends. For example, as the degree of random heterogeneity increases, the toe penetration length reduces in 2D models, as suggested by Abarca [2006], while it can increase or decrease in 3D models depending on the degree of heterogeneity and anisotropy. This occurs because the addition of heterogeneity (i.e., in the third dimension) influences both the effective hydraulic conductivity and the effective dispersivity, which control the toe penetration in different ways.

Where SI induces unstable density configurations (e.g., due to SI along preferential flow paths), heterogeneity is a key factor controlling the development of saltwater fingers [Simmons et al. 2001; Diersch and Kolditz, 2002; Simmons, 2005]. Heterogeneities also influence the circulation of seawater within the coastal aquifer and the chemical composition and discharge location of SGD [Abraca, 2006; Fleury et al., 2007], while neglect of heterogeneities can result in a significant bias in predicting the influence of tides on shoreline watertable conditions [Carey et al., 2009].

2.3. SI implications of sea-level fluctuations

The influences of sea-level fluctuations on groundwater systems (and more specifically on SI) are complex and varied. Fluctuations induced by episodic events such as tsunamis and storm surges are irregular in magnitude and frequency, whereas tides are predictable, although their amplitudes vary substantially (up to 16.8 m in the Bay of Fundy) around the globe [Slooten et al., 2010]. Relationships between ocean stresses and SI involve many factors, including: aquifer hydraulic properties and configuration, shoreline morphology (e.g., beach slope), seepage face development, capillary fringe processes, and the characteristics of the ocean fluctuations themselves, amongst others [e.g., Cartwright et al., 2004b; Carey et al., 2009; Slooten et al., 2010; Bakhtyar et al., 2012]. Tidal inlets, creeks and rivers are also important in shaping both salinity and hydrodynamic conditions in many coastal aquifers [e.g., Lenkopane et al., 2009; Li et al., 2000a]. Here, we focus mainly on those near-shore aquifer processes with direct implications for SI.

Seawater inundation of land from climatic or geological events (e.g., tsunami, storm surge, transgression, etc.) differs from common forms of SI, such as depicted in the Henry problem, in that salinization occurs through downward advection, dispersion and diffusion, and in some cases as free convection in the form of lobe-shaped instabilities [e.g., Kooi et al., 2000]. Illangasekare et al. [2006] reported on salinization processes in Sri Lanka arising from the 2004 Indian Ocean tsunami, and concluded that free convection was an important mechanism driving rapid salinization of Sri Lankan freshwater lenses. They projected recovery rates of a few years from this rare event. Violette et al. [2009] calculated freshwater flushing timeframes of 3 to 7 years for an unconfined aquifer in India impacted by a tsunami from the same 2004 earthquake. Anderson and Lauer [2008] considered contributions from storm-induced over-wash to the aquifer salinity of a barrier island. Storm events were found to contribute substantially to the aquifer's persistent salinity conditions, imposing greater influence on mixing zone development than tidal oscillations and other factors. Even moderate storm effects can have significant influence on coastal hydrogeology [e.g., Wilson et al., 2011]. Freshwater lenses of islands are particularly vulnerable to storm-surge over-wash [Terry and Falkland, 2010].

Where sea-level fluctuations are retained within the intertidal zone (i.e., coastal barriers are not overtopped), their influence on SI is more ambiguous. Inouchi et al. [1990] found that interface movements in response to tides are small, although dispersive modeling produced significant tidal impact on mixing zone thicknesses for their simulated settings. Ataie-Ashtiani et al. [1999] concluded similarly that tides produce negligible impacts on the landward extent of the mixing zone, at least for tidal amplitudes much less than aquifer depth, and that tides changed substantially mixing zone configurations. A tidal tracer study by Acworth et al. [2007] revealed greater vertical fluctuations relative to horizontal movements in response to tides, which enhanced mixing zone thicknesses. In contrast, Morrow et al. [2010] concluded from geophysical surveys that mixing zone movements occurred in response to large tidal amplitudes (i.e., 3.8 m) without substantial widening of the mixing zone. Kim et al. [2006] monitored interface changes in response to tides in the Jeju Island aquifer (Korea), and found that geologic heterogeneities control tide-interface relationships. The main influence of tides was found to be enhanced mixing, consistent with the conclusions of Inouchi et al. [1990] and Ataie-Ashtiani et al. [1999]. The relative importance of different ocean signals is indicated by the characteristic attenuation length for head fluctuations: $L_c = \sqrt{PD/\pi}$, where *P* is the wave period and *D* is aquifer diffusivity ($D = K / S_s$, with *K* the hydraulic conductivity and S_S the specific storage coefficient) [e.g., Slooten et al., 2010]. The higher value of *D* for confined aquifers produces far more extensive tidal propagation. Also, high-frequency wind wave signals are dissipated at short distances from the beach face, while damping distances for tidal watertable fluctuations can reach several kilometers in confined aquifers. Storm surges lasting several days exert even more extensive controls [e.g., Lanyon et al., 1982; Horn, 2006]. Sloping beach faces affect the propagation of signals, with the production of weakly damped spring-neap fluctuations [Li et al., 2000]. The field-based investigation of Cartwright et al. [2004a], amongst other studies, found that tidal fluctuations (up to 2 m amplitude) produced minimal interface movements relative to the response caused by a modest storm event (i.e., 4.5 m significant wave height inducing interface movements of several meters).

Numerous factors complicate tidal propagation and salinity patterns in real-world coastal aquifers, and these are likely responsible for observed differences in the effects of tides on salinity distributions. For example, field monitoring and numerical modeling studies have identified an upper saline plume of recirculating seawater (see (d) in Fig. 1), generated by a combination of tides and waves [e.g., Li et al., 2004; Robinson et al., 2006, 2007; Werner and Lockington, 2006, Vandenbohede and Lebbe, 2006; Xin et al., 2010]. Beach morphology (i.e., profile and hydraulic properties) plays a significant role in this and other relevant tidal phenomena, such as seepage face formation [e.g., Nielsen, 1990; Li et al., 1997, 2002, 2008; Carey et al., 2009]. As mentioned above, aquifer heterogeneities are also important in both tidal propagation and salt transport [e.g., Calvache and Pulido-Bosch, 1997; Slooten et al., 2010]. In contrast, fluid density variability and capillarity effects have small influences on tidal propagation [Barry et al., 1996; Slooten et al., 2010]. The multitude of possible interactions between aquifers, the sea, and tidal surface water features such as estuaries and tidal wetlands further complicates coastal aquifer salinity responses to sea-level fluctuations, especially considering the tidal salinity dynamics of surface water features [e.g., Lenkopane et al., 2009; Wilson et al., 2011].

Tides not only create dynamic conditions in the near-shore aquifer, but they also influence time-averaged ocean boundary conditions, which regulate regional aquifer hydrology. Tidal watertable over-height refers to the super-elevation of head conditions at the coast arising from tidal effects [e.g. Nielsen, 1990; Ataie-Ashtiani et al., 2001; Song et al., 2006]. That is, tides imposed time-averaged head conditions at the coast that exceed mean sea level. Accurate representation of tidal over-height in the ocean boundary conditions of SI management models is essential to produce reasonable guidance on well-field operation protocols for avoiding SI, and relies on apposite knowledge of relevant coastal fringe flow and transport processes [e.g., Werner and Gallagher, 2006].

2.4. *Hydrochemical processes*

The composition of seawater is relatively constant around the world [Millero et al., 2008], whereas the composition of fresh groundwater in coastal aquifers can be quite variable. Despite this variability, some characteristic chemical water types often form in coastal aquifers. This is due to the chemical reactions between the host material and groundwater that occur when intruded seawater and fresh groundwater mix, or when one displaces the other. The geochemical properties of the rocks determine the dominant chemical processes. Due to the distinct difference in the chemical composition of fresh groundwater and seawater, various hydrochemical processes can accompany SI, which lead to important changes in water quality and can even alter the hydraulic properties of the subsurface. In this section, we discuss chemical processes that are typical during SI.

Classical studies on the effects of seawater and freshwater mixing in carbonate aquifers include Plummer [1975] and Wigley and Plummer [1976], who showed that carbonate dissolution can be an important process in the mixing zone. The dissolution process occurs even when both the seawater and freshwater end members in the mixture are at chemical equilibrium with the carbonate mineral. Due to the redistribution of carbonate species and the non-linear dependence of activity coefficients on ionic strength [Wigley and Plummer, 1976], the resultant mixture has the capacity to dissolve carbonate. Mixing-induced carbonate dissolution enhances porosity and can lead to cave formation [Smart et al., 1988], an important factor in coastline morphology development [Back et al., 1979].

Another well-documented group of chemical reactions in coastal groundwater systems is cation exchange, which is driven by the differences in cation dominance in seawater and fresh groundwater. Na and Mg are dominant in seawater, and freshwater typically is dominated by Ca, although this can vary depending on the mineralogy of the aquifer. Clay minerals and organic matter have the ability to adsorb cations. If there is a shift in the interface position, the equilibrium between groundwater and exchanger is disrupted, and the exchange process alters the cation concentrations of the solution. Well-documented case studies from the Netherlands [e.g., Appelo and Willemsen, 1987; Stuyfzand, 1999], Spain [Giménez Forcada, 2010] and the USA [Valocchi et al., 1981a] have shown that characteristic water types develop, and that chromatographic patterns can form along flow paths [Appelo, 1994, Van der Kemp et al., 2000; Martínez and Bocanegra, 2002], which can be used as an indicator of freshening or salinization of coastal aquifers. Faye et al. [2005] were able to distinguish between areas of freshening and salinization based on the sorption behavior of Boron (B) and, recently, Russak and Sivan [2010] used cation exchange patterns to infer a seasonal alternation between freshening and salinization phases.

Intruded seawater in coastal aquifers is often found to have a lower sulfate concentration than seawater. While Gomis-Yagües et al. [2000] argued that the depletion of sulfate can be attributable to gypsum precipitation, sulfate reduction is usually the result of the degradation of organic matter [e.g., Chapelle and McMahon, 1991; Grassi and Cortecci, 2005; Bratton et al., 2009]. This process has been studied extensively within the context of biogeochemical cycling and early diagenesis [Berner, 1980]. Examples of recent studies include de Montety et al. [2008], who showed that ongoing sulfate reduction could lead to a shift in carbonate equilibrium that can trigger the precipitation of dolomite and/or magnesium-bearing calcite. Beck et al. [2008] showed that the geologic heterogeneity of tidal flat sediments influenced organic matter remineralization by sulfate reduction, and also found a seasonal variation in the pore water chemistry due to organic matter supply and temperature variations. Riedel et al. [2011] found that rates of organic carbon mineralization and sulfate reduction in a shallow coastal aquifer are much higher in regions dominated by advection than in regions where diffusion is the dominant transport mechanism and thus where the supply of oxidants and nutrients is limited.

Over the past decade, many studies have focused on redox chemistry within the context of SGD, and the transformation of nitrogen (N) and phosphorous (P) in the mixing zone between fresh and saline groundwater in SGD settings has received considerable attention [Moore, 1999; Slomp and Van Capellen, 2004]. A comprehensive review on SGD and the associated fluxes of nutrients, metals and carbon was recently published by Moore [2010].

The exposure and inundation of coastal areas due to shifts in the location of the coastline can further trigger a variety of chemical reactions in the subsurface. Allen [2004] found evidence for mixing between meteoric recharge and either modern intruded seawater or Pleistocene age seawater below two islands in British Columbia (Canada) that have undergone postglacial isostatic uplift. Boman et al. [2010] described the development of acid sulfate soils in brackish-water sediments in Finland that became exposed due to the same geologic process. Exposure of the sediments to atmospheric oxygen, often promoted by drainage for agricultural purposes, causes oxidation of metastable iron sulfides and pyrite, which results in acidification and mobilization of trace metals. Similar conditions have developed in other parts of the world due to drainage and reclamation of sulfur-rich marine sediments. Johnston et al. [2009] studied the effects of seawater inundation of acidified soils and found that it led to considerable increases in soil pH, suggesting that marine inundation might be an effective remediation strategy.

2.5. Upconing below wells

Pumping from coastal aquifers can cause the vertical rise of saltwater and a reduction of the freshwater zone below pumping wells, a process called upconing. Under certain circumstances, equilibrium can be established in which a stable cone develops at some depth below the bottom of the well screen, such that upward forces caused by the well are balanced by downward forces caused by density effects [Bower et al, 1999]. The rate and extent of saltwater upconing depend on a number of factors, including hydraulic properties of aquifer systems, pumping rate and duration, initial position of the interface, density contrast between freshwater and saltwater, and other factors such as dispersion and sorption effects, groundwater recharge, regional flow rate, and the well and aquifer geometries [Wirojanagud and Charbeneau, 1985; Reilly and Goodman, 1987; Saeed et al., 2002]. Some analytical solutions and numerical models on upconing are reviewed in § 4.

3. SI measurement

The measurement of SI, considering the strictest definition in terms of a moving interface, requires temporal observations of salinity changes. Other SI indicators include hydraulic head trends and water chemistry characteristics that infer historical salt transport processes. Accurate delineation of the extent of saline groundwater in coastal aquifers is difficult due to the scarcity of water salinity measurements across scales of interest for the management of SI. Measurement of transient SI is difficult because the process is typically slow, and historic data are scarce or absent. This is reflected in the current lack of estimates of the scale of SI problems globally, although continental and national summaries are offered by various authors [e.g., Custodio, 2010; Barlow and Reichard, 2010; Werner, 2010]. Monitoring of coastal groundwater systems often entails a multi-disciplinary approach, as single techniques usually fail to provide unique answers. This review considers head measurements, geophysical methods, and applications of environmental tracers, as important elements of coastal aquifer measurements.

3.1. Head measurements

Head measurements are complicated in coastal aquifers because the density of the groundwater varies. Post et al. [2007] gave a detailed overview of the complications and their solutions. Three types of hydraulic heads can be distinguished: (1) Point water head, where the observation well is filled entirely with water of the same density as at the bottom of the well, (2) Freshwater head, where the observation well is filled entirely with freshwater, and (3) Environmental head, where the variation of the density of the water in the observation well is the same as in the aquifer. Post et al. [2007] suggested avoiding the environmental head. In the field, only point water head can be measured. For this purpose, water needs to be bailed from an observation well (equal to at least the volume of the observation well) before a measurement is taken. Contour plots of freshwater head in a horizontal plane at constant depth are useful, because flow in isotropic aquifers is normal to these contour lines. Contour plots of point water head in a vertical cross section are useful to compute the flow in the freshwater zone or the saltwater zone, but not in the mixing zone. In summary, head measurements in coastal aquifers are only useful when both the exact depth is recorded and when the salinity (and thus density) variation is known inside the well [Custodio, 1987a].

As in other aquifers, coastal boreholes can act as vertical short circuits between layers. Shalev et al. [2009] analyzed field data and conducted a numerical study to investigate the effects of boreholes with long screens on the measured position of the watertable and the interface. They found that the fluctuation of the interface within the well is higher by up to an order of magnitude than in the aquifer, and that the degree of mismatch between the two is dependent on the anisotropy of the aquifer and the borehole hydraulic properties. These outcomes illustrate the difficulties encountered in the monitoring of SI and the need for properly designed observation wells and other measurement infrastructure.

3.2. Geophysical methods

The large electrical resistivity contrast between seawater (0.2 Ωm) and freshwater (> 5 Ωm) makes it possible to map the subsurface groundwater salinity distribution using geophysical techniques. Direct current (DC)-resistivity and electromagnetic (EM) methods in particular have been applied successfully in coastal areas. The relatively recent development of multi-electrode resistivity arrays and airborne EM methods allow for mapping of the subsurface salinity distribution in unprecedented detail in both space and time. This section discusses the most commonly applied techniques, with an emphasis on non-standard applications and recent developments.

Resistivity methods have been applied in coastal aquifers for many decades. Swartz [1937] was one of the first to use the method to detect the interface depth on the islands of Hawaii. Well-documented cases from the subsequent decades include the coastal plains of northwest Europe [Hallenbach, 1953; Flathe, 1955; Van Dam and Meulenkamp, 1967; DeBreuk and DeMoor, 1969; Van Dam, 1976], Israel [Ginsberg and Levanon, 1976] and Florida [Fretwell and Stewart, 1981]. Data acquisition in these early studies was constrained to 1D surveys, either vertically with depth or laterally along a profile, but since the 1980s multi-electrode systems and computerized inversion algorithms enabled the development of 2D and 3D surveys [Dahlin, 2001].

Electrical resistivity tomography (ERT) is the visualization of the subsurface resistivity distribution in 2D or 3D. It has become a mainstream method in coastal hydrogeology over the past decade. Acworth and Dasey [2003] applied ERT to investigate the hyporheic zone below a tidal creek and identified an extensive mixing zone of seawater and infiltrated rainwater. They concluded that the use of borehole data in conjunction with surface-based geophysics was essential to interpret correctly the electrical images. Day-Lewis et al. [2006] and Henderson et al. [2010] discussed the applicability of ERT to delineate zones of SGD and found that the time-varying depth of seawater had a significant impact on the resolution and depth of the investigation. Offshore applications of DC resistivity methods are more challenging than land-based applications, but have been used successfully in a number of studies. Belaval et al. [2003] mapped the occurrence of fresh groundwater below coastal waters in South Carolina and Massachusetts. Manheim et al. [2004] found offshore extensions of fresh groundwater of more than 1 km below the bays of the Delmarva Peninsula (USA) based on a vessel-towed multi-electrode system, and confirmed the inferred presence of fresh pore waters by drilling. Underwater multi-electrode profiling was used by Andersen et al. [2007] to map the sub-seafloor distribution of freshwater of a sandy coastal aquifer, where concern existed over the discharge of nitrate-containing groundwater. The geophysical results appeared to be particularly useful in delineating the freshwater discharge at the local scale, where it is controlled by small-scale heterogeneities.

Time-lapse resistivity imaging using permanent electrode arrays is becoming more widespread. Poulsen et al. [2010] installed five vertical multi-electrode arrays in boreholes to monitor the temporal dynamics of the salinity distribution beneath a coastal dune in Denmark. Time-lapse electrical imaging was also used by de Franco et al. [2009] to monitor resistivity variations in an aquifer bordering the Venice Lagoon (Italy). This study showed that the resistivity was correlated to various environmental factors, such as rainfall, channel stage and tidal fluctuations at different depths and timescales. The proven feasibility of this technique for the detection of salinity changes has prompted the installation of a permanent telemetric system near Almeria (Spain) to monitor the effect of climate and land use change on groundwater salinity [Ogilvy et al., 2009].

Like resistivity methods, electromagnetic methods (EMs) have been used with considerable success to map groundwater salinity variations in coastal areas. Frequency-domain methods have been used for decades [Stewart, 1982] and, more recently, time-domain methods have become more popular [e.g., Fitterman and Stewart, 1986; Goldman et al., 1991; Marksamer et al., 2007]. EMs have been used in conjunction with resistivity methods. Examples include studies in which an EM method was used as a cheap and rapid screening tool that helped to identify locations for more detailed ERT measurements [e.g., Ong et al., 2010], and studies that have exploited the joint inversion of measurements to constrain the non-uniqueness of the interpretation [e.g., Albouy et al., 2001].

EMs do not require contact between the measurement device and the ground surface, which has enabled the development of airborne measurement systems [Paine, 2003]. These systems allow for cost-effective mapping of the subsurface resistivity over large areas, and in areas that are inaccessible by ground-based vehicles, such as the Everglades in Florida [Fitterman and Deszcz-Pan, 1998] or the Venice coastal lagoon [Viezzoli et al., 2010]. Kirkegaard et al. [2011] used airborne EM measurements to study the salinity of groundwater to depths of 200 m below a coastal lagoon and noted a strong correlation between the salinity and geological heterogeneity. A recent review of helicopter-borne EM methods was published by Siemon et al. [2009]. It contains further references to and an example of the application of this technique in coastal regions.

Airborne measurements need to be constrained and verified using ground-based and downhole resistivity measurements. Geophysical well-logging to characterize the formations surrounding the borehole is typically done after drilling and before completion of the borehole. Downhole measurements can sometimes be used to monitor temporal changes in groundwater salinity, although the presence of a metal casing (which acts as an electrical conductor) can be prohibitive. Vandenbohede and Lebbe [2003] used a frequency-domain probe to monitor the passage of a plume during a tracer test. Nienhuis et al. [2010] compared different methods for detecting the fresh-salt water interface in existing boreholes and found that fixed electrode pairs installed at fixed depths within the borehole provide the most accurate results. They also noted that borehole fluids used during drilling remain entrapped in confining units and bentonite seals for years, thereby limiting the resolving power of downhole methods.

There are two major causes of error in geophysical measurements. First, the observation model that relates the observation (the measured signal) to the desired state (salinity) is approximate. Second, the measured signal can be affected by other variations besides salinity, mostly notably changes in the geology such as the presence of clay layers. As a result, the salinity distribution obtained from an analysis of geophysical measurements is approximate.

3.3. Environmental tracers

There have been many studies that have used the chemical composition of coastal groundwater as a diagnostic tool to establish the origin of dissolved salts. Although intruded seawater is the most obvious candidate for explaining observed increases in groundwater salinity, other sources and processes can contribute as well [Stuyfzand and Stuurman, 1994; FAO, 1997; Werner and Gallagher, 2006]. A multi-tracer approach that combines hydrochemical and isotope data can be used successfully to identify salinity sources [Vengosh et al., 1999]. Some recent examples will be given here. Bouchaou et al. [2008] used Br/Cl ratios, $\delta^{18}O$, $\delta^{2}H$, ^{3}H, $^{87}Sr/^{86}Sr$ ratios, $\delta^{11}B$, and ^{14}C to discern between different sources of salinity in a coastal aquifer in Morocco. They found that rock salt dissolution (halite and gypsum in particular), connate saline groundwater and irrigation return flow were contributors to salinity besides SI. Based on a similar approach that also included the study of $\delta^{34}S$ data, Han et al. [2011] found that brines, with a TDS (total dissolved solids) > 100 g/l, in a coastal aquifer system in northern China constituted the main source of salt in the brackish water moving towards the pumping wells in the area, rather than intruded modern seawater. The contribution of brines was also recognized by Ghabayen et al. [2006] in the coastal aquifers of the Gaza Strip and by Yechieli and Sivan [2011] in Israel.

Boron concentrations and isotopes have been used by Giménez Forcada and Morell Evangelista [2008] to differentiate pollution sources in a coastal aquifer in Spain, which allowed seawater to be distinguished from industrial wastewater sources. The usefulness of B and its isotopes has been demonstrated in other coastal aquifer studies as well. Morell et al. [2008], for example, distinguished between salinity due to connate seawater from deep layers and modern SI based on $\delta^{11}B$ values. Panagopoulos [2009] found a depleted $\delta^{11}B$ signature and increased concentrations of Li and Sr in a karst aquifer in Greece, which were attributed to hydrothermal alteration of intruded seawater. Alcalá and Custodio [2008] demonstrated the use of the Cl/Br ratio to distinguish between natural salinity sources such as seawater and potassium halite, and anthropogenic sources such as wastewater, solid waste and pesticides. Contributions by irrigation return flow [e.g., Stigter et al., 1998; Kallioras et al., 2006; Lenehan and Bristow, 2010] and salt deposition by sea spray [Cruz and Silva, 2000] have been recognized in many areas around the world based on detailed hydrochemical characterization studies. Finally, it is noted that hydrochemical studies can be aided by statistical techniques such as factor analysis and principal component analysis. Such techniques have been used by several authors to distinguish water types and to classify water samples into groups [e.g., Aris et al., 2007; El Yaouti et al., 2009; Koh et al., 2009; Yidana, 2010; Yidana et al., 2010].

$^{87}Sr/^{86}Sr$ isotope ratios of groundwater are strongly controlled by the Sr isotopic ratio of the aquifer. This enabled Jørgensen and Heinemeier [2008] to identify a contribution of a third source of water to the mixing zone between freshwater and seawater in a shallow aquifer. During pumping, leakage from the deeper limestone aquifer into the shallow groundwater occurred, discernable by the $^{86}Sr/^{87}Sr$ ratio. Kim et al [2003] also used the $^{86}Sr/^{87}Sr$ ratio to identify the origin of saline groundwater on a volcanic island in Korea, whether from a modern SI event, or relict. Based on the similarity of the $^{87}Sr/^{86}Sr$ ratios of present-day seawater and the saline groundwater, they inferred that modern SI was occurring. Recently, Lin et al. [2010] concluded that SGD was occurring in deep canyons off the Taiwan coast, based on differences in $^{87}Sr/^{86}Sr$ isotope ratios between water in the canyon and the adjacent seawater.

Coastal areas have often been subjected to multiple phases of marine inundations. Distinguishing between modern versus relict SI is an important objective of many studies [Darling et al., 1997; Petalas and Diamantis, 1999]. One particular example of an aquifer system where ^{14}C has been used successfully to distinguish between modern and former SI is the Mediterranean coastal aquifer system of Israel. Sivan et al. [2005] used ^{3}H and ^{14}C to show that seawater had moved up to 100 m inland over a few decades. Yechieli et al. [2009] were able to discriminate between modern SI and older seawater based on ^{3}H and ^{14}C measurements, and found that the age of the old saline groundwater decreased with decreasing depth, which they attributed to rising sea levels during the Holocene.

4. SI prediction

The distinguishing feature of SI models, relative to groundwater flow models, is the variation of density caused by the variation in salinity. Although the seawater density is only 2.5% larger than that of freshwater, the difference has a major impact. There are two types of flow models for the simulation of SI: interface models and variable density models. In interface models, the freshwater and saltwater are treated as two immiscible fluids separated by an interface, along which freshwater and saltwater pressures are continuous. In variable density models, the transition zone between freshwater and saltwater has a finite thickness and the density of the water varies continuously. SI analytical solutions are predominantly based on the interface assumption, whereas variable density models are mainly resolved using numerical solutions. Only variable density models provide salinity predictions that can then be compared to field salinity measurements.

4.1. Analytical solutions

The interface flow formulation requires the solution of two coupled systems: flow in the freshwater zone and flow in the saltwater zone, which need to be solved simultaneously such that pressure and normal flux are continuous across the interface. There is a shear flow along the interface (i.e., velocities tangent to the interface differ between the freshwater and the saltwater) that is a function of the density difference and the angle of the interface [e.g., Bear, 1972]. The solution of interface flow problems is simplified significantly when only the steady-state position of the interface is of interest. For such cases, there is flow in the freshwater zone only and the saltwater is at rest. Solution can be simplified further through adoption of the Dupuit approximation, which means that the resistance to flow in the vertical direction is neglected within an aquifer so that the vertical pressure distribution is hydrostatic. For Dupuit interface flow, the head in the freshwater zone is a function of the horizontal coordinates only and the thickness of the freshwater zone can be computed with the Ghyben-Herzberg formula [e.g., Bear, 1972; Strack, 1989; Fitts, 2002], which is a linear relationship between the depth of the interface and the head in the aquifer.

There are many exact solutions for steady interface problems; Reilly and Goodman [1985] and Cheng and Ouazar [1999] provide partial overviews, while Bruggeman [1999] gives exact solutions to 36 interface flow problems. Two main solution techniques can be distinguished. Exact solutions for steady 2D interface flow in the vertical plane can be obtained with the hodograph method in combination with conformal mapping [e.g., Verruijt, 1970; Bear, 1972; Strack, 1989; Zhang et al., 1997, 1999; Bakker, 2000]; recent applications include Kacimov [2001], Kacimov and Obnosov [2001] and Kacimov et al. [2006]. Alternatively, 2D interface flow may be solved with a Green's function approach [e.g., Hocking et al., 2011], although this requires numerical solution of an integral equation. Exact solutions for Dupuit interface flow can be obtained with the Strack potential [Strack, 1976, 1989]. The power of the Strack potential lies in the fact that the solution can be written as one expression for the potential as a function of the horizontal coordinates regardless of the position of the interface, i.e., $Q_x = -\partial\Phi/\partial x$ for 1D flow. Here, Q_x is groundwater discharge towards the sea (L^2T^{-1}), Φ is the Strack potential (L^3T^{-1}), and x is the direction perpendicular to the coastline (L). The Strack potential is valid for piecewise homogeneous aquifers. When the potential fulfills the Laplace or Poisson equations, solutions may be obtained through superposition. Analytic element solutions of steady interface flow can be obtained with the analytic element code GFLOW [http://haitjema.com/, last accessed 6 February 2012] and AnAqSim [http://www.fittsgeosolutions.com/, last accessed 27 February, 2012]. A popular application of the Strack potential is the optimization of groundwater withdrawals in coastal aquifers (§4.4). The maximum (or critical) pumping rate is reached when the interface becomes unstable either from below or from the side [Strack, 1989].

Interface flow solutions become more difficult when the Strack potential fulfills a non-linear differential equation. Sikkema and van Dam [1982] present a solution technique for semi-confined interface flow. Bakker [2006] applied this technique to obtain exact solutions for the outflow zone along ocean bottoms with a leaky bed. Dagan and Zeitoun [1998a, 1998b] developed an exact solution for an interface in a stratified coastal aquifer with a random permeability distribution.

Few exact solutions exist for interface flow where both the freshwater and the saltwater are moving. Bakker [1998] presented a method to compute the instantaneous flow field where both the fresh and salt water are moving by extending the Strack potential. Maas [2007] presented a solution for a freshwater lens in a 2D vertical cross-section affected by brackish flow from below. This solution is valuable to assess the sustainability of freshwater lenses in low-lying areas affected by an increased upward leakage of brackish water, for example caused by sea-level rise. Zhang et al. [2009] presented a boundary integral solution for a drain that extracts both fresh and salt water.

The only (semi) analytic solution for the evolution of the dispersed interface is a small perturbation solution for upconing below a well [Dagan and Bear, 1968]. Werner et al. [2009] and Jakovovic et al. [2011] showed that this solution gives good results when compared to results of sandbox experiments for lower density contrasts and higher pumping rates, but that the analytical solution over-predicted the interface rise for lower pumping rates. Zhang et al. [2012] showed that the upconing was well predicted using a sharp-interface model.

The main approximation of interface flow is that the freshwater and saltwater are immiscible fluids. The consequence of this approximation is that SI is overestimated [Dausman et al., 2010a; Pool and Carrera, 2011] and thus maximum pumping rates are underestimated. Recently, several attempts have been made to modify interface solutions to take mixing into account. Paster and Dagan [2008a, 2008b] included mixing in an interface solution through application of a boundary layer approach. Pool and Carrera [2011] took a simpler approach by modifying the saltwater density in the Strack potential according to the transverse dispersivity using an empirical formula obtained through comparison of interface flow solutions with numerical results of variable density models.

Henry [1964] presented a steady solution for a problem of 2D variable density flow in a vertical cross-section; alternative solutions to the same problem were published later by Segol [1993] and Dentz et al. [2006]. The Henry problem is unique as it is the only analytic solution that exists for steady SI with variable density flow, but it has little utility for simulating SI in real aquifers as only diffusion and no dispersion is simulated. A few (semi) analytic solutions exist for steady convective flow, but these are also limited to diffusion only [Voss et al., 2010].

4.2. *Numerical modeling and benchmarking*

The limited number of analytical solutions for variable density flow problems creates a heavy dependence on numerical codes for simulating SI. Such computer models have become irreplaceable tools to gain insight in real-world SI issues ranging from system understanding on local or regional scales to future projections of SI for management purposes. The combination of the improved understanding of physical and chemical processes with advancements in numerical methods, computer languages, and raw computational power has led to a number of SI codes that are capable of simulating SI in real-world coastal aquifers.

Physical processes are captured to varying degrees, and challenges persist in the local-scale simulation of real-world conditions, such as the representation of near-shore hydrodynamics, including seepage face development and tidal/wave fluctuations.

Most codes solve a coupled system of variable density flow and solute transport equations, which is numerically complicated and computationally demanding. The main difficulty lies in the accurate solution of the solute transport part, which requires a fine discretization of the aquifer. Common techniques such as the finite element or finite difference methods perform well for the solution of the flow equations, but they result in unwanted numerical dispersion when applied to solve the solute transport equations. Solution methods for the transport equations such as the total variation diminishing method or the method of characteristics commonly yield better results but require larger computational times for the same grid resolution. The computational effort associated with regional-scale SI problems usually leads to trade-offs between model accuracy and run-times that are important in interpreting model-predicted salinities and in model calibration.

Many codes exist for the simulation of SI; a list of popular codes is given in Table 1. The most widely used codes are SEAWAT [Langevin et al., 2007] and SUTRA [Voss and Provost, 2002]. SEAWAT was specifically designed for the simulation of SI, although it has many other applications as well, notably the combined simulation of groundwater flow and heat transfer. SEAWAT uses MODFLOW (finite differences) to solve the flow system and MT3D with its many solution techniques to solve the solute transport equations. Most MODFLOW packages are available in SEAWAT to simulate different stresses and boundary conditions. SUTRA is a general-purpose code that applies a finite-element and integrated-finite-difference hybrid method to solve both the flow system (including the unsaturated zone) and the transport equations. Both SEAWAT and SUTRA are well documented and freely available (http://water.usgs.gov/software/lists/groundwater/, last accessed 11 October 2011).

All but the last code in Table 1 solve the combined flow and transport equations and can be applied to simulate freshwater-saltwater mixing, the evolution of transition zones, and complex phenomena such as density inversions (where saltier water lies above fresher water). The final entry in the table, the SWI package for MODFLOW, was especially designed to simulate transient regional seawater intrusion. SWI does not require vertical discretization of the aquifer. The salinity distribution in an aquifer is specified through one or more interfaces that separate water of uniform or linearly varying densities. SWI is based on the Dupuit approximation within an aquifer and does not simulate mixing beyond the initial salinity distribution or inversions within an aquifer, but its computational performance is commonly three orders of magnitude better than the other models listed in Table 1 [Bakker et al., 2004; Dausman et al., 2010a]. SWI is freely available from http://modflowswi.googlecode.com/, last accessed 27 February, 2012.

Numerical codes can be verified through comparison with analytic solutions for variable density flow (§ 4.1). Further testing of codes can be carried out through benchmarking. Benchmarking refers to the process of comparing output of models for well-defined standard problems [Simpson and Clement, 2003]. A number of benchmark problems have been proposed for SI problems. Diersch and Kolditz [2002] and Voss et al. [2010] provide a discussion on the applicability of different tests to SI codes. The Henry [1964] and Elder [1967a, 1967b] problems prevail as the most popular test cases. As mentioned, the Henry problem is the only known exact solution to a variable density flow problem. In Henry's original formulation, a transition zone develops due to high molecular diffusion, which to a certain extent obscures the density-dependence of the flow. Simpson and Clement [2004] suggested reducing the freshwater recharge, which increases the influence of density-dependent effects. Abarca et al. [2007b] developed a dispersive rather than a diffusive Henry problem but did not present an analytic solution. The Elder problem concerns an unstable transient natural convection problem where saltier water moves downwards in fresher water [Voss and Souza, 1987]. There were early indications that there is no unique solution to the Elder problem because instabilities and fingers are influenced by numerical factors [Voss et al., 2010]. The recent study by van Reeuwijk et al. [2009] confirmed the existence of multiple (three) physically plausible solutions of the classic Elder problem using the pseudospectral method to avoid discretization errors. This important result has major consequences for model benchmarking in that a single solution does not exist. While the Elder problem is a good benchmark problem for variable density codes, it is not very representative of SI problems. Even if an Elder-like instability occurs in a SI problem, the grid resolution is rarely adequate to simulate the evolving salinity distribution accurately.

Other popular benchmark problems include the HYDROCOIN salt dome problem [Konikow et al., 1997], salt lake problem [Simmons et al., 1999], salt pool problem [Johannsen et al., 2002; Oswald and Kinzelbach, 2004], rotating fluids problem [Bakker et al., 2004], and the Henry and Hilleke problem [Henry and Hilleke, 1972, Langevin et al., 2010]. Existing benchmark problems have two main shortcomings. First, only two problems represent 3D flow (the salt pool problem and the 3D stability test in an inclined box by Voss et al. [2010]), while SI is commonly a 3D process. Second, many of the listed benchmark problems are developed from laboratory experiments; boundary conditions are not very representative of real world SI problems and it is often difficult to set up the code to replicate the laboratory experiment exactly.

Parameter estimation and uncertainty analysis of SI models is only in its infancy. Model independent tools such as PEST [Doherty, 2004] or UCODE [Poeter et al., 2005] are applied regularly to groundwater models, but rarely to SI models. Carrera et al. [2010] reviewed the computational and conceptual challenges of automatic calibration of SI modeling. Their review highlighted the extensive simplifications adopted by the application of automatic calibration to SI models, with the exception of the studies by Dausman et al. [2009a, 2010b]. Carrera et al. [2010] list a number of obstacles to the automatic calibration of SI models including the density-dependence of head measurements, the sensitivity of salinity concentration data to flow, the sensitivity of SI models to aquifer bottom topography, and difficulties with defining initial conditions. The uncertainty of calibrated SI models is rarely quantified, and often omitted entirely from SI modeling studies. One of the few exceptions is the analysis of Herckenrath et al. [2011], who developed a robust methodology for predictive uncertainty analyses of SI, based on variable-density flow and transport modeling of a hypothetical aquifer resembling the Henry problem.

Table 1. Popular SI codes

[Table 1]

The aforementioned models all focus on the near-shore aquifer, often considering tides, but with crude representations of waves and beach morphology. Bakhtyar et al. [2012] combined SEAWAT and a detailed wave motion model that solves the Reynolds-Averaged Navier Stokes equations with k-ε turbulence closure. This model updated that of Bakhtyar et al.

[2011], which was validated using data from a laboratory flume experiment but did not account for density dependence. Various wave conditions were modeled, as were changes in the beach morphology. While the latter was affected markedly by wave characteristics, the SI was essentially independent of them. A similar modeling study was reported by Xin et al. [2010], who used a simpler wave model (shallow water equations) but included tides. They coupled their wave/tide model with SUTRA, which simulates saturated-unsaturated flow with variable density. They showed that the circulation within the beach face is significant, and that tide and waves, considered separately, have less impact on such circulations than with both acting in tandem. These models are capable of describing in considerable detail the physical processes accompanying SI, as well as associated phenomena such as transport of pollutants into coastal seas.

4.3. Reactive transport modeling

Reactive transport modeling of hydrogeochemical processes accompanying SI and SGD (§5.3) has been attempted since the 1980s but remains challenging to date. The earliest applications focused on modeling cation exchange and the development of chromatographic patterns [e.g., Valocchi, 1981b; Appelo and Willemsen, 1987; Beekman and Appelo, 1991]. Mao et al. [2006a] created PHWAT, the first general-purpose density-dependent flow and reactive transport model, coupling SEAWAT (version 2) and PHREEQC-2 [Parkhurst and Appelo, 1999]. Besides permitting the inclusion of arbitrary complex reaction networks, their model included the effect of density changes due to reaction-induced concentration changes, as given by the VOPO model (Monnin, 1994). Mao et al. [2006a] applied their model to the experimental data of Panteleit [2004], which described a careful laboratory experiment combining density-dependent flow with cation exchange in a three-layer aquifer. PHWAT successfully simulated the data, including the cation peak ("snow-plough" effect, e.g., Barry et al., 1983) that is produced when a high-concentration solution flushes resident cations from the soil. Post and Prommer [2007] used an updated version of the PHWAT code based on SEAWAT-2000 to develop a multi-component reactive transport variant of the Elder problem and to test the impact of density effects due to chemical reactions on flow patterns. They found that those were significant only for small density contrasts and high permeabilities, whereas they were negligible at density contrasts of seawater. Boluda-Botella et al. [2008] investigated the sensitivity of various transport and geochemical parameters on the results of a reactive transport model of cation exchange and gypsum dissolution. These models cannot account for waves or variable saturation porous media, and so are not suitable for describing near-shore aquifer processes.

Sanford and Konikow [1989] used a reactive transport model to study the feedback between groundwater circulation in the transition zone and the development of porosity and permeability due to carbonate dissolution. They found a positive feedback in which porosity development in the mixing zone led to stronger circulation, which led to further carbonate dissolution and increased porosity. The validity of their results was later confirmed by Rezaei et al. [2005]. Romanov and Dreybrodt [2006] further refined previous approaches by decoupling the geochemical and the solute transport equations, which allowed them to model dissolution in narrow mixing zones.

Modeling of the degradation of organic chemicals in tidally controlled groundwater systems has been attempted by Mao et al. [2006b], Brovelli et al. [2007], Robinson et al. [2009] and Li and Boufadel [2010]. These studies showed that flow patterns induced by tidal oscillation in combination with variable-density effects result in complex spreading of contaminant plumes. Mixing is enhanced by the tidal oscillations, which tend to promote biodegradation through constant supply of oxygenated water below the beach face [Robinson et al., 2009].

The importance of geochemical zonation on the fate of pollutants in SGD systems has been studied using reactive transport models by several authors. Jung et al. [2009] simulated the transport and sorption of arsenic (As) to sediments of the shallow aquifer of Waquoit Bay (Massachusetts, USA). The mixing of oxidized and reduced groundwater triggers the precipitation of iron oxides, to which As is sorbed. Based on modeling, Spiteri et al. [2006] found that pH, and not oxygen, exerts the strongest control on iron precipitation. In their model, the authors also considered the effect of iron precipitation on phosphate removal. Subsequently, Spiteri et al. [2008] examined the role of iron oxidation by denitrification and found that this process enhances the formation of iron oxides, and the removal of phosphate from discharging groundwater.

4.4. *Pumping optimization*

Optimal management of groundwater in coastal aquifers is a critical task due to the threat and complex nature of SI. Optimization methods are applied to address various practical questions relating to pumping operation, well placement, design of artificial recharge schemes and other mitigation measures, trade-offs between environmental and socioeconomic factors, and interdependencies between surface and subsurface systems. Optimization problems involve minimizing or maximizing an objective function, e.g., the total safe pumping rate, subject to a set of constraints that define technical or physical limitations (e.g., pump capacity). Decision variables are used to define and differentiate alternative decisions (e.g., pumping or injection rates), and state variables represent system response to stresses (e.g., hydraulic head). Wagner [1995] and Qin et al. [2009] provide reviews on optimization problems for groundwater flow and contamination problems, and Dhar and Datta [2009] offer a useful introduction to SI optimization methods. Cheng et al. [2004] use second-order stochastic analysis to determine variances of interface toe location. Recent seminal contributions to SI optimization are reviewed here.

Approaches to SI optimization can be grouped according to the method of SI simulation within the optimization strategy, as the embedding approach and the simulation-optimization approach. Qahman et al. [2005] consider the response matrix approach as a third category. The embedding approach incorporates the discretized governing equations into the constraint set of the optimization model, which can become excessively large and complicated when density variations and transience of real-world SI applications are considered. Only a few hypothetical cases of SI optimization using the embedding approach and miscible flow and transport equations have been demonstrated [e.g., Das and Datta, 1999]. The simulation-optimization approach involves repetitive simulation of SI to determine state variable values (e.g., head and concentration) to assess perturbations in decision variables. SI simulation is achieved using analytical solutions, numerical solutions or surrogate models (otherwise known as "meta-models") [Sreekanth and Datta, 2010; Dhar and Datta, 2009]. Surrogate models can be described in simple terms as computationally efficient, empirical replacements of physically based simulators, and these are in widespread use in SI optimization [e.g., Sreekanth and Datta, 2010]. Different non-linear statistical data modeling tools such as artificial neural networks have been used as surrogate models for SI management studies [e.g., Rao et al., 2004; Bhattacharjya and Datta, 2009; Kourakos and Mantoglou, 2009; Dhar and Datta, 2009]. The response matrix approach is essentially a primitive form of surrogate model, and is intended for linear or mildly non-linear problems where superposition assumptions apply [Dhar and Datta, 2009]. Response matrix approach applications usually consider only hydraulic aspects of drawdown and discharge, and usually avoid salinity state variables due to the nonlinearity restrictions [e.g., Reichard and Johnson, 2005; Abarca et al., 2006; Karterakis et al., 2007]. Due to the limitations of SI analytical solutions and the computational burden of SI numerical simulation, the simulation-optimization approach by use of surrogate models appears as the most popular approach in recent research of SI optimization problems.

A range of mathematical techniques has been used in optimizing groundwater extraction in coastal aquifers, including: linear programming (LP) [e.g., Mantoglou, 2003], non-linear programming (NLP) [e.g., Mantoglou and Papantoniou, 2008] and evolutionary algorithms (EA) [e.g., Kourakos and Mantoglou, 2009; Dhar and Datta, 2009; Ataie-Ashtiani and Ketabchi, 2011]. Genetic algorithms (GA) are the most popular EA approach [Nicklow et al., 2010]. SI optimization is inherently a nonlinear problem that is often subject to highly irregular derivatives in decision variable-objective function relationships, potentially comprising many locally optimal solutions. Classical techniques such as NLP often resolve only local optima due their reliance on functional gradients. In addition, these methods are often computationally inefficient in obtaining whole Pareto-optimal solutions to multiple objective problems (i.e., considering trade-offs between different objectives), and can suffer from convergence problems and other numerical difficulties [e.g., Bhattacharjya and Datta, 2009]. As a result, recent SI simulation-optimization studies are based mostly on non-traditional algorithms, such as EA, because of their effectiveness in converging on the global optimum for highly nonlinear or irregular problems [Dhar and Datta, 2009; Mantoglou and Papantonioua, 2008]. EA-based approaches include GA [Bhattacharjya and Datta, 2005; Qahman et al., 2005; Park and Aral, 2004], simulated annealing [Rao et al., 2004], differential evolution [Karterakis et al., 2007] and elitist continuous ant colony optimization [Ataie-Ashtiani and Ketabchi, 2011]. Table 2 provides a categorization of optimization technique for SI management problems with some example references.

Recent SI optimization research efforts have considered more advanced artificial neural network-based simulation and GA optimization methods, aimed at reducing the computational burden required to explore SI management solutions [e.g., Bhattacharjya and Datta, 2009; Kourakos and Mantoglou, 2009]. For example, Dhar and Datta [2009] proposed and applied an optimization methodology based on a numerical simulation model, meta-model, and multiple objective GA (i.e., non-dominated sorting genetic algorithm) for multiple objective management of coastal aquifers. Recent applications of genetic programming as potential surrogate models in multi-objective problems have brought promising results in reductions of computation time in assessing optimal coastal aquifer pumping strategies [e.g., Sreekanth and Datta, 2010; Sreekanth and Dattaa, 2011a].

Table 2. Categorization of optimization technique in SI management

[Table 2]

5. SI management

Much of the SI literature focuses on the quest to manage coastal groundwater sustainability, thereby balancing the social, economic and environmental benefits derived from coastal groundwater reserves. Strategic SI sampling regimes are precursors to avoiding groundwater quality deterioration, however, coastal aquifers are complicated and resource intensive to assess. Hence, a key component of SI research relates to practices in coastal aquifer management, including the operation and control of groundwater extraction, and the associated links between pumping regulation and aquifer condition and trend as determined through monitoring and assessment. Coastal aquifer management often involves SI remediation and mitigation measures (discussed below). The potential impacts of SI on ecosystem health are also reviewed, given the importance of environmental assets in triple bottom line evaluation and management. We consider linkages between SI and SGD. The fresh component of SGD (SFGD) is the net recharge to the coastal aquifer, and SGD chemistry is closely tied to mixing processes and movements in the freshwater-saltwater interface. Finally, studies exploring the impact of future climate change and sea-level rise on SI are highlighted, given that these are key areas of SI research and integral to prospective SI management.

5.1. *Practical SI management considerations*

The challenges of SI management are multi-faceted. For example, precursors to the effective regulation of pumping for the purposes of SI control include: 1. sufficient knowledge of hydrogeological processes relevant to SI, 2. monitoring and interpretation of water level and water quality conditions and trends, 3. consideration of relevant socio-economic and environmental factors, and 4. retrospection and prospection across the three aforementioned categories. The latter includes consideration of future climate change and sea-level rise, in the context of historical climatic and sea-level variability of the region. In addition, government policies, legislative frameworks and regulatory authority are necessary to invoke constraints on land use and on the operation of well fields.

Where controls over groundwater pumping are enabled, SI management usually involves invocation of sanctions and measures to regulate groundwater extraction. Werner et al. [2011] summarized recently the approaches to well field operation and groundwater regulation (for the purposes of SI management), and demonstrated practical and scientific challenges that are faced by coastal aquifer custodians. They categorized SI operational strategies into trigger-level and fluxed-based approaches, and recommended that elements of both approaches are optimal in managing SI. Devising hydrological and hydrochemical triggers (i.e., characteristics of the system that infer whether a management response is needed) is challenging. For example, SI can produce freshwater storage losses with very small reductions in groundwater levels, and therefore water level temporal trends may not be suitable measures of coastal aquifer conditions.

Efforts to manage coastal aquifers efficaciously have led to the construction and application of an extensive array of highly complex numerical models and innovative field sampling techniques. However, in many cases, management decision-making requires rapid assessment of coastal aquifer vulnerability (or other performance-based measures such as resilience and reliability [Hashimoto et al., 1982]) to SI, or only a highly simplified approach is affordable (e.g., if national or continental scales are considered). In response to this, a number of SI vulnerability assessment approaches have evolved in the last decade. For example, Lobo-Ferreira et al. [2007] developed the GALDIT method, an indexing approach to SI vulnerability assessment that is comparable in principle to the DRASTIC methodology [Aller et al., 1987] whereby causative factors are weighted and summed to produce an indexed evaluation of groundwater contamination vulnerability. Other indexing methods that assess more broadly coastal aquifer vulnerability to sea-level rise include those of Thieler and Hammar-Klose [1999] and Ozyurt and Ergin [2010]. Werner et al. [2012] asserted that existing indexing approaches to SI vulnerability assessment fail to capture the key physical elements that control SI, and suggested application of methods that have theoretical underpinnings. An example of this is given by Wriedt and Bouraoui [2009], who undertook large-scale screening of SI risk along the Spanish Mediterranean coast. Their two-tiered approach first evaluated the balance of recharge versus pumping, and subsequently adopted simple SI analytical solutions to infer steady-state SI responses to pumping stresses.

5.2. *Engineering measures for SI remediation and mitigation*

Efforts to address incidences of SI usually involve, in the first instance, attempts to regulate groundwater pumping, taking into account the interplay between the conditions of the aquifer and economic and social factors (and in some cases environmental impacts). An example of this is the modeling study by Qureshi et al [2008], who included surface water use, groundwater use, SI, agricultural productivity, water-logging and economic factors in exploring policy options for managing the Burdekin aquifers, Australia. Conjunctive use scenarios explored variations in irrigation usage from surface water and groundwater reserves to demonstrate economic implications of modifying water use patterns. Unfortunately, pumping regulation is either ineffective or untenable in many cases, and subsequently it often fails to successfully mitigate or remediate SI. In these cases, the deployment of engineering measures, such as enhanced recharge or physical modification of the aquifer can provide alternative and economically viable solutions, despite that these are invariably costly options. Reichard et al. [2010] and Koussis et al. [2010] demonstrated the economic benefits of including various recharge augmentation measures in assessing water-supply options using examples of the Los Angeles Basin (USA) and the Akrotiri basin (Cyprus), respectively.

Any human intervention to control or ameliorate SI needs to modify the aquifer's water and/or salt balances. This can be achieved through numerous means, such as reducing SGD, reducing coastal aquifer evapotranspiration, reducing discharge to (or enhancing recharge from) terrestrial surface systems containing freshwater, enhancing aquifer recharge (e.g., through wastewater injection or artificial recharge ponds), accessing a proportion of the aquifer's saline water (e.g., for the purposes of desalination), impeding seawater inflow, and/or by inducing inflows from neighboring aquifers (e.g., enhanced influxes from overlying, underlying or inland aquifers). A comprehensive introduction to this topic was presented by Custodio [1987c]. Oude Essink [2001a] provides a summary of SI control measures. Here, we review a selection of these approaches and focus on prominent case studies and recent developments.

Perhaps the least intrusive engineering measure for addressing SI is the application of horizontal wells, skimming wells and surface drains, i.e., based on the premise that shallow withdrawal reduces saltwater ingress (due to upconing) at the point of extraction [Custodio, 1987c]. The reader is directed to § 2.5 and § 4.4 for examples of upconing modeling studies and approaches for optimizing freshwater extraction.

A variety of measures to enhance aquifer recharge can be instigated where changes in well field operation and design fail to remedy SI issues. Vandenbohede et al. [2009] described sustainable water extraction practices in the western Belgian coastal plane, where SI-induced reductions in pumping within one system were offset by artificial recharge and subsequent enhanced extractions from a neighboring coastal aquifer. They describe long-term and regional perspectives to coastal aquifer management that have led to enhanced water supply security and sustainability. Abarca et al. [2006] applied both linear and nonlinear optimization methods to design alternative remedial measures for the Llobregat Delta aquifer (Spain), and found that the development of a hydraulic barrier allowed for gains in freshwater extraction that exceeded rates of injection. The most extensively studied examples of freshwater injection to mitigate SI are those of the Los Angeles coastal basins, where engineering modifications have been implemented since about 1960. Schroeder et al. [1989] described early attempts to simulate hydraulic barrier operation to allow enhanced groundwater extraction and protect against SI. More recently, Reichard and Johnson [2005] and Bray and Yeh [2008] detailed simulation-optimization studies aimed at improving the efficiency and effectiveness of SI controls in those systems. Studies that demonstrate optimal practices in hydraulic barrier application include that of Luyun et al. [2011], who used laboratory experiments and numerical simulation to show that freshwater injection was most effective if applied at the toe of the saltwater wedge, and that therefore well injection is probably more effective than surface recharge in controlling SI.

Cases where freshwater is injected into saline groundwater for the purposes of aquifer storage and recovery (ASR) were examined by Ward et al. [2009] and Bakker [2010], amongst others. They offered guidance on the performance of ASR (i.e., recovery efficiency of injectant) in variable salinity systems, including critical combinations of aquifer hydraulic and transport properties, fluid density differences and ASR operational conditions.

Aliewi et al. [2001] suggested that lenses less than 90 cm in thickness are more economically utilized by applying “scavenger wells” to pump saltwater concurrently and from below the point of freshwater extraction. Their numerical modeling study of the Gaza and Jericho aquifers (Palestine) demonstrated relationships between aquifer parameters, recharge, and well construction and operation on the performance of skimming-scavenger well systems.

Pool and Carrera [2010] used hypothetical numerical simulations to examine the effectiveness of negative hydraulic barriers, in which seawater is pumped from a well located between the freshwater well and the coast. Critical pumping rates producing minimum salinity in the freshwater well were influenced by seawater and freshwater pumping rates, depth of well penetration, well locations and aquifer properties, as expected. The system was controlled by 3D effects, by which seawater bypassed the negative hydraulic barrier to eventually reach the freshwater well. This caused a reduction in system effectiveness when the rates of seawater pumping were too high.

The most invasive measures to control SI are physical subsurface barriers, which can be installed to retain groundwater and/or inhibit SI, with notable applications in Japan [Luyun et al., 2009]. Recently, Luyun et al. [2011] showed that flow barriers are more effective when situated deeper in the aquifer and at locations closer to the coast, through laboratory experimentation and numerical simulation of simplified situations. Luyun et al. [2009] examined the transience of trapped saltwater following subsurface barrier installation, and found that the subsurface wall height was the key factor in controlling flushing rates and the relative mix of seawater and freshwater on the seaward side of the barrier. While such materials as bentonite clay, concrete grout, bituminous substances or sheet piles are suggested as barrier wall materials [Custodio, 1987c; Luyun et al., 2011], the concept of injecting air to reduce aquifer permeability has also been evaluated. For example, Dror et al. [2004] undertook air injection laboratory experiments and found that the formation of a low-conductivity barrier, which was a combination of trapped air and pore-filling precipitate, reduced discharge by an order of magnitude or more, at least temporarily. From extension of their laboratory results, they concluded that the practice of air injection to create temporary groundwater flow barriers was potentially viable at field-application scales, although they conceded that further analysis was needed to assess the performance of such approaches in field-scale SI applications.

5.3. *Submarine groundwater discharge and SI*

The discharge of freshwater to the sea, referred to here as submarine fresh groundwater discharge (SFGD), mixes in the aquifer with recirculated seawater producing SGD [Li et al., 1999]. SFGD and coastal aquifer geochemical processes are vital components of coastal ecosystems, and are sensitive to anthropogenic effects [Moore, 1999; Moore 2010]. The risk of SI, i.e., the propensity for freshwater-saltwater interface movements, is closely related to SFGD through constitutive relationships between interface location and groundwater flow [e.g., Strack, 1976; Werner et al., 2012]. For example, in the case of the shrinking Aral Sea, Shibuo et al. [2006] demonstrated that a reduced risk of SI was accompanied by enhanced SFGD.

SGD and SI are complementary and inter-dependent processes [Taniguchi et al., 2002]. For example, the salinity of SGD has been linked to SI through both field observation [e.g., Dimova et al., 2011] and numerical modeling by Kaleris [2006], who demonstrated that the salinity of SGD increases when the extent of the interface increases. Price et al. [2006] proposed that nutrient discharge to the Everglades (Florida, USA) is influenced by SI through geochemical processes occurring within the freshwater-saltwater interface. Further, the presence/position of the seawater interface is known to influence groundwater flow pathways, and therefore SI potentially impinges on the fate of coastal fringe contaminants [e.g., Kaleris et al., 2002].

From a water resources perspective, maximizing the capture of SFGD while avoiding excessive SI is an elemental principle of coastal aquifer management strategies in cases where SFGD provides little benefit to receiving marine environments [e.g., Babu et al, 2009], notwithstanding groundwater dependencies of coastal terrestrial ecosystems. Depending on its chemical constituents, SFGD can impose negative influences on marine water bodies (e.g., through excessive nutrient loads [Tse and Jiao, 2008]), or alternatively, marine ecosystems can be SFGD-dependent [e.g., Duarte et al, 2010]. These hazards depend on the mixing of the receiving waters. Ideally, conjunctive goal setting, accounting for SI and marine ecosystem health is required in considering the management of SFGD [McCoy and Corbett, 2009]. To achieve this, knowledge of the rate and quality of SFGD is required. However, complex relationships between SFGD, SGD and SI exist in natural settings. For example, wave and tidal action can reduce the SFGD portion of SGD to only 4% [Li et al., 1999]. This presents significant challenges in reconciling model- and field-based methods for estimating SFGD.

Geological variability also plays a major role in SFGD rates and distributions. For example, Lin et al. [2011] reported significant SFGD from a multiple aquifer system in which substantial pumping-induced overdraft might otherwise be expected to minimize SFGD, at least from a simple water balance perspective. Mulligan et al. [2007] highlighted the importance of paleochannels both as SFGD conduits and providing preferential pathways for SI. Marine and terrestrial hydrodynamics are also key factors in SGD, SFGD and SI relationships. Taniguchi et al. [2006] found that terrestrial groundwater connections impose controls on SGD on the landward side of the freshwater-saltwater interface, where SGD seaward of the interface was mostly controlled by oceanic dynamics, for their study area on Kyushu Island, Japan.

Estimation of SFGD through the summation of direct measurements is virtually impossible in some cases due to the extensive distances over which SFGD occurs through the ocean floor. Bakker [2006] reported that a significant portion of SFGD from the southern part of the Floridan aquifer likely occurs at the edge of the continental shelf, some 120 km from the shore. In these cases, the application of conceptual and numerical models to extend hydrogeological and submarine measurements is essential to approximate the SFGD component of coastal aquifer water balances. In cases where fresh groundwater resources extend significant distances below the sea [e.g., Kooi and Groen, 2001], groundwater extraction can produce SI in offshore aquifers, whereby sub-sea freshwater reserves are depleted, representing a situation of non-renewable groundwater mining [e.g., Cohen et al., 2010]. Evidence of SI in these circumstances may involve considerable lag-times. In a review of methods for SGD quantification by Burnett et al. [2006], it was recommended that multiple methods be applied, including geophysics, seepage meters, natural tracers and various hydraulic and water balance approaches, amongst others, given the significant uncertainty in most measurement methods and the difficulties in reconciling SGD predictions from models with the complexities of field situations.

5.4. SI impacts on ecosystems

SI impacts are associated primarily with losses of freshwater resources and contamination of water supply wells, and only a few studies consider adverse ecological impacts directly linked to SI. Environmental degradation arising from SI is commonly linked to the application of high salinity groundwater in agriculture, resulting in modified soil chemistry and reduced soil fertility [e.g., Darwish et al., 2005; Qi and Qiu, 2011]. The hydraulic aspects associated with SI, such as watertable decline leading to reductions in both SFGD and groundwater availability for terrestrial groundwater-dependent ecosystems, have also been recognized [e.g., Duarte et al., 2010; Kaplan and Munoz-Carpena, 2011]. Additionally, the discharge of saline groundwater into terrestrial freshwater surface features has been well studied [e.g., Simpson et al., 2011; Oude Essink et al., 2010], in particular associated with polders of the Netherlands, although subsequent ecological implications are not documented. With these exceptions, SI ecosystem impacts are otherwise seldom acknowledged in coastal aquifer management studies, even where the focus is ecological and water supply sustainability. However, research in the last decade has developed an enhanced appreciation of the broader environmental implications of SI, although these have not been summarized previously. Here, we provide a brief overview of potential (and actual) ecosystem impacts arising from SI.

As discussed in § 2.4, SI induces various hydrochemical and geochemical reactions depending on the characteristics of the subsurface environment. The most direct biological impacts of changes in groundwater chemistry are likely those of groundwater systems themselves, e.g., stygofauna (i.e., groundwater fauna) and subsurface microbial communities. These ecosystems are in closest proximity to the interface, and most directly affected by SI. For example, Santoro [2010] suggests that SI can significantly alter microbial functioning including the catalysis of important nutrient transformations. Edmonds et al. [2009] found that while SI in laboratory flow-through columns induced considerable changes in biogeochemical cycling, the microbial communities exhibited extraordinary physiological flexibility, retaining their community structure over the five-week duration of the experiments.

There are no studies on potential or actual linkages between stygofauna diversity and SI, although Humphreys [2008] indicates that stygofauna distributions are dependent on salinity and on other physico-chemical attributes. The Yucatan Peninsula, where open cenotes (sinkholes) host a great variety of food webs that include endemic species [Schmitter-Soto et al., 2002], serves as a prominent example of potential SI-ecosystem interactions. Freshwater occurs as a thin lens overlying saltwater, which is found more than 110 km from the coast [Steinich and Marin, 1996]. SI has influenced large parts of the aquifer, reaching tens of kilometers inland [Bauer-Gottwein et al., 2011]. The aquifer includes the world's most extensive underwater cave system, of which the cenotes are surface expressions [Bauer-Gottwein et al., 2011], and hence the potential for SI to cause ecosystem degradation is especially high.

Many of the anthropogenic stresses that induce SI, such as groundwater pumping or the construction of surface drainage networks in coastal settings, also affect both the rate and water chemistry of SFGD [e.g., Moore, 1999]. Further, SI potentially modifies the chemistry of SGD directly, following the discussion of links between SI and SGD identified above [e.g., Taniguchi et al., 2006]. Price et al. [2010], in their study of phosphate adsorption and desorption, concluded that processes inducing SI such as sea-level rise will release phosphate from freshwater coastal carbonate aquifers into the sea. SI is also known to release and transport other toxic elements, such as arsenic [Nath et al., 2008], and it is possible that these and other SI-derived heavy metals might degrade ecosystems via SFGD delivery mechanisms. In order to assess in more detail human stresses on the many marine ecosystems that depend on SFGD [e.g., Johannes, 1980], further understanding of the hydrogeochemical relationships that govern linkages between SI and SFGD is required.

In non-tidal cases, SI most likely occurs predominantly within the lower aquifer domain due to the higher density of seawater relative to ambient freshwater, e.g., following salinity patterns similar to the dispersive Henry problem [Abarca et al., 2007b]. However, Robinson et al. [2006] and Werner and Lockington [2006] demonstrated that tidal fluctuations can induce saltwater recirculation in the upper aquifer, producing a saline watertable in the coastal fringe. Werner and Lockington [2006] extend the quasi-steady situation of a stable landward boundary condition to consider the transient situation of SI (i.e., interface movement in response to a watertable drop) in 2D. Their modeling results show that watertable salinization can arise from SI where a tidal boundary condition is considered. It follows that secondary capillary-driven salt rise into the unsaturated zone, following watertable salinization, is a plausible outcome of SI [e.g., Werner and Lockington, 2004].

Werner and Lockington's [2006] results indicate that SI in tidal systems can impart more considerable ecosystem stress than in the non-tidal case, particularly where vegetation (e.g., phreatophytes) and other biota in coastal and estuarine fringes are dependent on fresh groundwater conditions. There are implications here also for urban infrastructure that has susceptibility to rising damp and salt-decay [e.g., Lopez-Arce et al., 2009]. Field evidence for SI-induced soil salinization in the manner proposed here is limited. Buscaroli and Zannoni [2010] examined salt rise from saline watertables in a Mediterranean pinewood forest, and concluded that surface soluble salt accumulation was independent of watertable salinity at least for deep vadose zones. Where ecosystem stress in coastal settings has been at least partly attributed to salinity changes [e.g., Kaplan et al., 2010], there is usually an element of both subsurface and surface SI, which has confounded the diagnosis of soil salinization processes. Perhaps the strongest link between ecosystem decline and SI is reported by Antonellini and Mollema [2010], who found that symptoms of vegetation distress and regression in species richness of the coastal forests and wetlands of Ravenna (Italy) were associated with watertable depth and groundwater salinity. In particular, tree-rooting patterns were influenced by watertable salinities, and wetland vegetation species richness was controlled by water salinities more generally. They attributed high watertable salinities to saltwater upconing potentially driven by evapotranspiration in combination with mechanical drainage. While tides of the region are small, these might also impart an influence on the nature of SI, by influencing watertable salinities in a similar manner to that reported by Werner and Lockington [2006].

The results of ecological impact predictions due to sea-level rise by Saha et al. [2011] indicate that SI will contribute significantly to the degradation of coastal forests. Their analysis neglected the role of tides on watertable salinization, and hence might have under-estimated the ecological degradation arising from SI. Greaver and Sternberg [2010] examined relationships between rainfall patterns, SI and coastal dune ecosystems in open ocean islands of the Bahamas. They concluded that soil salinization occurred predominantly from SI and accompanying salt rise, both of which occurred seasonally. While Greaver and Sternberg [2010] ruled out tides as the primary factor controlling SI and soil salinization, they provided no indication or analysis of the relative contribution of tides to watertable salinization.

5.5. SI, climate change and sea-level rise

Anticipated future variations in the hydrologic stresses of coastal aquifers, such as changes in climatic characteristics (e.g., rainfall and evapotranspiration) and sea levels need to be considered in devising coastal aquifer management strategies. In particular, sea-level rise and changes in both recharge and evaporation due to global climate change are expected to exacerbate SI [e.g., Oude Essink et al., 2010], either directly or through secondary effects such as increased pumping in drying climates, enhanced frequency and severity of storm surge inundation, marine transgression, and rising salinities in surface water bodies (i.e., due to seawater incursion into surface water features). Here, we focus on the direct influence of projected climate change and sea-level rise on SI in coastal aquifers, acknowledging that the secondary effects listed above can also contribute significantly to the threat of SI.

There is a growing body of literature describing case studies of anticipated SI arising from future sea-level rise. A wide range of SI responses to sea-level rise is reported. Sherif and Singh [1999] predicted that the extent of SI induced by a sea-level rise of 50 cm was 9 km in the semi-confined Nile Delta, but only 0.4 km in the confined Madras aquifer. Oude Essink [2001b] concluded from numerical modeling of a Dutch SI case study that sea-level rise would have a significant impact on rates of aquifer salinization in low-lying coastal aquifers of the Netherlands, albeit the time lag for some aquifers was expected to exceed 100 years. Dausman and Langevin [2005] predicted that SI would invade an additional 1.5 km inland if sea levels rise by 48 cm, thereby threatening water supply wells in a surficial aquifer in Florida. Nishikawa et al. [2009] evaluated SI under future sea-level rise considering two different management approaches for the coastal Los Angeles aquifers (USA) and concluded that sea-level rise would cause significant SI under the business-as-usual scenario, but that raising inland groundwater levels mitigated SI caused by sea-level rise. Shrivastava [1998] used numerical simulation to show that a confined alluvial aquifer in Jamaica would incur no significant SI under either sea-level rise or storm surge, albeit interface salinities were modified. Yechieli et al. [2010] report that sea-level rise, neglecting inland shifts in the shoreline, is likely to produce only minor SI in the phreatic and confined aquifers of the Israeli Mediterranean coast. From these studies, it is difficult to draw general conclusions about the potential deleterious effects of SI arising from sea-level rise given the wide range of SI responses, ranging from no impact to several kilometers of interface movement.

Werner and Simmons [2009] used simple steady-state changes in the position of an interface to assess sea-level rise in unconfined aquifers. They found that the conceptualization of the inland boundary conditions was a primary factor in the estimation of ensuing SI. Specifically, flux-controlled conditions (i.e., the seaward discharge persists despite sea-level rise) produced SI in the order of tens of meters, whereas head-controlled conditions (i.e., watertable rise is restricted by surface controls such as lakes, rivers, wetlands or head-dependent pumping or drainage at some point inland; Fig. 1) produced SI up to several kilometers. Flux-controlled conditions are more prevalent, although examples of SI in head-controlled systems subjected to sea-level rise are given by Feseker [2007] and Kundzewicz and Doll [2009]. The important implications, regardless of inland boundary condition, are that coastal aquifer management strategies need to allow the piezometric surface to rise commensurate with sea-level rise to avoid significant SI (neglecting secondary effects as listed above). Critical values of parameter combinations that were associated with severe SI (i.e., asymptotic SI-parameter relationships) were identified by Werner and Simmons [2009]. For example, coastal discharge (i.e., SFGD) needs to exceed a minimum value ($q_{\min}$) to avoid substantial SI (i.e., "active SI" or salt transport driven by landward freshwater hydraulic gradients), given as:

$$q_{\min} = \sqrt{\frac{WK\,(1+\alpha)z_0^2}{\alpha^2}} \tag{1}$$

Here, W is net recharge (LT^{-1}), K aquifer hydraulic conductivity (LT^{-1}), z_0 is the sea level above aquifer basement (L), and α is the dimensionless density ratio $\rho_f/(\rho_s - \rho_f)$, where ρ_f and ρ_s are the freshwater and seawater densities (ML^{-3}), respectively.

Werner et al. [2012] extended the sea-level rise-SI analysis of Werner and Simmons [2009] to include confined aquifers, and used the equations of Strack [1989] to assess relationships between SI and aquifer stresses, again for steady interface conditions. They found that sea-level rise was essentially inconsequential in flux-controlled confined aquifers, but that head-controlled confined aquifers (likely a rarer situation) were susceptible to SI from sea-level rise. The conclusions regarding SI and sea-level rise by Werner et al. [2012] are consistent with site-specific observations from the individual case studies as presented above.

In most cases, it is expected that sea-level rise will be accompanied by altered recharge regimes due to climate change, producing joint impacts on SI [e.g., Oude Essink et al., 2010]. Kourakos and Mantoglou [2011] demonstrated that climate-driven reductions in recharge potentially require substantial mitigation efforts to avoid freshwater decline, even in the absence of sea-level rise. Kundzewicz and Doll [2009] asserted that, with the exception of low-lying coastal regimes, recharge decline would induce a larger reduction in groundwater availability than that caused by sea-level rise. The simulations of Yechieli et al. [2010] were consistent with this view, with recharge decline producing significantly larger SI relative to sea-level rise impacts in their case study of a Mediterranean unconfined aquifer. However, the investigation of unsaturated-saturated zone flow and salt transport behavior on Bahaman islands by Greaver and Sternberg [2010] revealed important interplays between sea-level variation and precipitation. This demonstrated that, in some cases, there are interdependencies between climate and sea-level variations that need to be considered. Nonetheless, separating the relative contributions to SI imposed by the two individual stresses (sea-level rise and recharge change) is important for planning purposes, and to account separately for the considerable uncertainties associated with predicting both factors.

Feseker [2007] considered both recharge reductions (i.e., due to climate change) and sea-level rise, and found that sea-level rise was expected to produce more extensive salinization compared to recharge changes, which were abated by the head-controlling conditions imposed by drainage networks. Carneiro et al. [2010] predicted only minor salinization for future sea-level rise and recharge declines in a shallow unconfined aquifer on the Mediterranean coast of Morocco. Vandenbohede et al. [2008c] examined the situation in Belgium under expected increased recharge and sea-level rise, and concluded that while higher groundwater levels produced positive outcomes in the form of seaward interface movements, negative implications included higher rates of dewatering of low lying areas and environmental stress in near-coastal dune ecosystems.

Werner et al. [2012] compared sea-level rise impacts to recharge reductions through manipulation of simple analytical solutions to the steady-state position of an interface. They found that sea-level rise impacts were more substantial in head-controlled systems, whereas recharge change impacts were more extensive in flux-controlled systems. Parameter combinations leading to critical vulnerability conditions were given for a range of aquifer stress perturbations, which included changes in sea-level, recharge and freshwater discharge to the sea.

A limitation of the analyses by Werner and Simmons [2009] and Werner et al. [2012] is the assumption of steady-state conditions. This precluded consideration of SI time-scales, which are clearly important aspects considering that Oude Essink et al. [2010], Feseker [2007] and others reported extensive SI time lags, in contrast to more rapid SI responses determined by Yechieli et al. [2010]. Investigations by Watson et al. [2010] and Webb and Howard [2011] extended the analysis of Werner and Simmons [2009] through systematic numerical experiments of transient sea-level rise in hypothetical unconfined aquifers of varying parameter combinations. Both studies assessed the time-scales and disequilibrium of sea-level rise-induced SI. Watson et al. [2010] considered primarily flux-controlled cases and adopted instantaneous sea-level rise, whereas Webb and Howard [2011] assumed head-controlled settings and examined a 90-year sea-level rise event. The time-scales (and inland distances) of SI were broadly consistent between the studies, ranging from decades to centuries. Webb and Howard [2011] highlight that systems with high K-to-W ratios and high effective porosities were associated with the longest lag times and most extensive disequilibriums after the 90-year sea-level rise events. Watson et al. [2010] noted that different aspects of the saltwater wedge (i.e. center of mass, toe position, etc.) approached steady state at different rates, and they observed a transient over-shoot phenomenon whereby the landward extent of the interface temporarily exceeded the steady-state condition. Chang et al. [2011] further explored SI time-scales under sea-level rise, considering confined aquifer conditions. Their conclusions repeat those of Watson et al. [2010] and Werner et al. [2012] in observing temporary SI overshoot and noting essentially inconsequential SI in confined aquifers. The gradual sea-level rise simulations of Chang et al. [2011] demonstrated that the overshoot phenomena discovered by Watson et al. [2010] also occurs in confined aquifers and under gradual-rise conditions.

6. Prospecting scientific challenges in SI

Despite nearly 50 years of research, numerous SI research questions persist. Many of the remaining scientific challenges are fundamental, and have serious implications for managing SI and the associated threats to freshwater resources and ecosystems. SI research remains an active field, underpinned by complicated ocean-aquifer interactions and, in dependent of climate change, growing human stresses on coastal aquifers. Our review of SI literature indicates that the greatest shortfall in SI research is the lack of intensive monitoring studies of SI in which mixing zone changes are precisely mapped and reconciled with methods of prediction. Long-term, high-density monitoring of field-scale SI is needed to address this, and to facilitate the transfer of knowledge and methods from hypothetical and/or small-scale experiments to real-world situations. Multiple case studies are needed to gain insight into SI in different geological and hydrological settings. Specific scientific challenges are described below in the following categories: (a) process understanding, (b) measurement and modeling, and (c) application to field-scale problems.

6.1 Process understanding

The mechanisms affecting the thickness of freshwater-seawater mixing zones in field conditions are multiple and cumulative. No agreement has been reached so far in terms of the controlling mechanisms. Most previous studies on the dispersive mixing zone are based on either laboratory-scale experiments or various forms of the Henry problem. Those investigations provide invaluable insights into mixing-zone development, but the application to field conditions requires further consideration of scaling and field-scale heterogeneity. Various forms of the Henry problem continue to be used as a surrogate for SI process understanding, and while this affords advantages of simplicity, the representativeness of the Henry problem to real-world SI remains questionable. A key issue here is that dispersion coefficients derived from laboratory or field experiments cannot lead to the thick mixing zones found in some coastal aquifers. The strategy of using larger dispersion coefficients to reproduce thick mixing zones is likely due to under-representation of both aquifer heterogeneities and temporal wedge dynamics and, therefore, the employment of large dispersion coefficients in numerical simulations is a simplification that does not explain the underlying mechanisms. Other questions regarding mixing zone evolution remain; e.g., under what conditions can we use equivalent homogeneous conditions to reproduce SI behavior and when do we need to use explicit heterogeneity? Universal scaling relationships for SI have not been derived, which contributes to the challenge of physically based SI simulation in which dispersive processes are accurately captured. Future analyses of dispersive processes in field-scale SI problems need to consider alternatives to single-porosity, constant-dispersivity approaches, amongst various options.

Studies of SI mixing zones and heterogeneity effects have thus far focused almost solely on steady-state conditions. However, the effects of heterogeneities on transient SI (in particular active SI cases) and the associated temporal evolution of mixing zones is expected to be significant given observations of mixing zone dynamics (widening and narrowing) in response to transient stresses. Transient SI in heterogeneous environments, incorporating pumping effects, is an area clearly worthy of further investigation. In particular, quantitative analyses of SI in fractured and fissured rock systems are lacking. SI controlling factors such as geological structure and three-dimensionality should be considered more frequently in future SI research. There is a potential for an increased role in linking sedimentological facies data into such analyses.

There are few field-based studies involving detailed measurements of upconing processes and the associatedsalinization of pumping wells. Rather, knowledge in this area is based almost entirely on laboratory and mathematical experimentation, and involving only simple forms of saltwater upconing as might be expected under ideal conditions. However, it is more likely that the interplay between density effects, heterogeneities and dispersion/diffusion produce complex well salinization mechanisms that are unexplored, and that current intuition on well salinization mechanisms is over-simplified. In particular, situations where undiluted seawater intercepts well screens and mixes with freshwater in the well have not been adequately evaluated and could be more widespread than previously thought. Additional investigation is warranted to characterize better the 3D nature of saltwater upconing in real-world systems.

Recent research into coastal fringe hydrodynamics has demonstrated that local scale processes influence both the time-averaged head conditions of the ocean boundary and the salinity profile near the coast. The implications for SI (and its investigation) are potentially significant for several reasons. For example, pumping in tidal settings potentially causes watertable salinization near the shoreline. This could have consequences in terms of soil salinization and ecosystem degradation, albeit the conditions under which this can occur have not been assessed. Also, observations of tidal over-height far exceeding theoretical values (caused by site-specific aquifer heterogeneities and beach morphology) demonstrate the need to characterize coastal fringe processes to avoid under-estimating the hydraulic head imposed by the ocean (and subsequently under-estimating the threat of SI). It is clear that all areas of SI process research would benefit from future investigations that combine theoretical analyses with field experimentation to provide guidance on the transfer of small-scale experimental results to real-world simulations.

6.2 Measurement and modeling

Comprehensive SI measurement campaigns integrated with modeling efforts that adopt formal calibration and predictive uncertainty analyses are a key area for future research in SI characterization. We found limited cases where water level, hydrochemistry, environmental tracer and geophysical information have been intensively collected and compared with geological controls and hydrological stresses in a quantitative framework. Also, better integration of terrestrial aspects with marine and submarine facets of coastal aquifer functioning and SI is warranted, because SFGD forms a fundamental component of coastal aquifer water balances, and many coastal aquifers discharge into sensitive marine environments. To achieve this, a greater focus on multi-disciplinary approaches is needed. In addition, links between data capture and modeling need to be enhanced. For example, there are potentially significant gains to be made from evaluating the contribution of various types of measurements to reducing the predictive uncertainty of SI models, thereby providing much-needed guidance on cost-effective deployment of SI characterization methods. Equally, the seamless integration of data collection and storage networks into SI modeling platforms could assist in guiding adaptations of sampling strategies and facilitate more frequent SI model redevelopment. The techniques and computational power needed for these types of analyses are available.

There is a host of future opportunities to develop further the various SI measurement approaches, building on recent advances. For example, airborne geophysical and associated ground-truthing approaches continue to evolve and offer regional-scale depictions of coastal aquifer salinity distributions. The need to monitor temporal evolution in salinity distributions requires new methods and applications of time-lapse geophysical measurement systems. Also, monitoring of diffuse and localized SFGD, and upscaling of SFGD measurements to obtain regional-scale aquifer discharge remain significant challenges. Environmental (and artificial) tracer techniques, including various combinations of chemical species, isotopic signatures and heat could partly provide solutions. These methods have not been exhaustively explored for SI problems, in particular to quantify SI trends and residence times, and to distinguish transport paths and groundwater and saltwater origins.

There is also an important scope for reactive transport modeling to aid in and strengthen the interpretation of environmental tracer data. The combination of complicated flow fields due to variable density effects and the potential chemical transformation pathways of the chemical and isotopic species, results in such complexity that these models are indispensable tools to guide our intuition and improve our conceptual understanding. The applicability of radium isotopes in detecting and quantifying SFGD is an example here, as the fate of radium in the mixing zone between fresh and saline groundwater is controlled by redox conditions and salinity-dependent sorption. The implications of the interplay of spatiotemporal salinity changes and geochemical processes for the interpretation of radium data have not been explored fully, for which reactive transport models would be valuable tools.

Displacement of the transition zone, for example due to pumping, sea-level rise, coastal erosion or changes in recharge, induces hydrochemical reactions. The latter are catalyzed by microbes, and in turn influence microbial functioning. Aquifer hydraulic properties can be affected also, depending on hydrogeochemical and ecological conditions, but the degree of these effects is not well constrained. These factors are likely to have major impact on the water quality of SFGD under certain conditions, e.g., where the transport and degradation of contaminants are mediated by microbial organisms and/or geochemistry. The importance of these and other factors in assessing SI through laboratory experimentation and numerical simulation requires additional testing.

There is little quantitative guidance in current SI literature on what level of complexity or simplicity is warranted in SI models. For example, how and under what conditions should a fully detailed geological characterization be employed in a numerical model, and when is it appropriate to simplify to an effective homogeneous K value? The same sorts of simplification questions apply to the hydrochemical aspects of SI. The nature of how simple or complex a modeling approach and associated data should be relative to intended model purpose remains a major challenge for this field.

A further challenge specific to SI models is the salinity distribution in the aquifer at the beginning of the simulation. Two main goals of SI models are the prediction of the impact of SI on well fields and the prediction of the effect of human or natural changes in hydrological conditions (land use, climate change). The starting salinity distribution in the aquifer has a major effect on the accuracy of such predictions. Calibration data can be obtained from water quality sampling or geophysical measurements, coupled with geostatistical methods to model the 3D salinity distribution at the beginning of the simulation period. When measurements are insufficient, the starting salinity distribution is estimated by modeling the historical evolution of the salinity distribution provided that historic influences (e.g., pumping rates, recharge data, river stages, etc.) are obtainable. This approach partly moves the difficulty to the estimation of the salinity distribution at the start of the historical evolution. Common choices for the start of the historic evolution are either a steady-state situation (also referred to as pre-development conditions) or an entirely saline or fresh aquifer. A greater understanding of the role of geological evolution (e.g., Holocene transgression and accompanying changes to coastal morphology) on coastal aquifer salinity conditions is needed for these purposes. Either way, the specific choice of the starting point of historic simulations is ideally designed to have only a minor effect on the outcome, but guidance on this is lacking.

The computational demand of SI models is large even though most SI codes use relatively large cell sizes that are not adequate to simulate all SI processes. Research is needed to improve the computational performance of SI codes and to take advantage of, for example, code parallelization. This will allow finer mesh discretization and/or similar calibration techniques to be applied as those used in flow-only groundwater modeling. Limited attempts and restricted capacity to automatically calibrate SI models is compounded by shortcomings in field-scale SI characterization. The use of SI models in designing field monitoring strategies should be explored further, although it is often difficult to reconcile uncertain salinity measurements with the predictions of regional-scale SI models. Formal uncertainty analyses of SI models are also lacking, and hence field measurement campaigns and model calibration efforts are rarely interdependent in the strictest sense. The combination of inverse modeling with predictive uncertainty analyses of SI models is needed for management applications, including the estimation of risks associated with such effects as climate change and other uncertain aquifer stresses. Predictions regarding the effects of climate change are even more uncertain because climate change scenarios themselves are highly uncertain. That is, the uncertainties inherent in climate scenarios need to be coupled to SI prediction uncertainties to decipher the results of models and how they might influence management decisions, such as the optimization of groundwater use.

It is noteworthy that most SI optimization research considers only hypothetical or highly simplified coastal aquifers. There are no examples of highly complex, well calibrated SI models that are applied to rigorous pumping optimization evaluation, despite such methods being applied to other hydrogeology problems. Indeed, despite that numerous optimization methods have been applied to hypothetical and simple SI problems, optimization research is not in mainstream use, and is still largely quarantined to the research domain. This typifies a significant time lag between research findings and the techniques for SI investigation adopted by hydrogeology practitioners. Future research in this field needs to develop, evaluate and implement optimization methodologies in the context of real-world systems, which are usually characterized by large scales, scant data, an inordinate number of pumping-related decision variables and constraints, and SI processes occurring over a variety of scales (e.g., upconing, SI from tidal waterways, relic seawater mobilization, etc.). Added to this, water resource custodians require information pertinent to prediction uncertainty and salinization risk, and to address these issues it is often necessary to invoke stochastic approaches to SI simulation. Only hypothetical examples presently demonstrate such an approach. SOA problems are amenable to parallel computing, which provides opportunities to capitalize on expanding computation resources. Further efforts to enhance the efficiency and accuracy of density-dependent flow and transport simulation, either directly or by surrogation with simpler models, will also assist in bringing SI optimization methods into mainstream use.

6.3 Application to field-scale problems and management

Practical SI problems involve managing a complex and in some cases slow-moving phenomenon with limited resources and scant field measurements. In many practical applications, there is a strong case to be made for using more of the available tools to better map and define the saltwater wedge, to develop enhanced SI conceptual models, and to obtain larger datasets of parameters. More data are required in both the spatial and time domains, although the 3D nature of SI leads to often costly and point-based or snapshot methods of measurement. Integrated measurements and analyses using modeling, head and salinity data, geophysics and tracer information will allow better problem conceptualization and model constraint. For example, only limited field observations of saltwater wedge movement or upconing are available, and instances of accurate plume mapping near pumping wells is rare. In many SI cases, the paucity of data and measurement leads to a heavy reliance on models to make estimates. However, few models are calibrated rigorously and verified with respect to accurate monitoring and characterization of the saltwater distribution.

Critical SI management questions remain unresolved. For example, while sea-level rise is likely to impact on coastal aquifer hydrological processes interdependently with recharge changes, modified recharge and sea-level rise have been evaluated for only a handful of cases and/or considering highly simplified situations. Recent evaluations of the temporal aspects of SI in response to sea-level rise neglect situations involving transgression and associated land surface over-topping. The free convection processes associated with transgression under current rates of sea-level rise require further analysis to assess their importance relative to lateral SI. A systematic study of both sea-level rise and recharge drought, considering a tidal boundary condition, is also needed to demonstrate the magnitude of tidal influences on SI processes and subsequent ecological impacts under future scenarios, taking into account various hydrogeological, climatic and tidal settings.

The resilience of coastal aquifer systems to SI is also not well understood. Indeed, transient effects remain poorly studied, and there are important implications here in terms of the reversibility of SI, and the remediation of aquifers in combination with artificial recharge and ASR schemes. SI management requires substantial research effort to develop improved practices for SI control. Application of desalinization techniques, including in situ (down-hole) methods, is growing rapidly in coastal aquifers. The byproduct of desalinized salt or brackish water is a concentrated saline stream (a brine), which is usually injected in deeper (saltier) layers or in the ocean. The (long-term) effect of either on the freshwater resources in coastal aquifers is an important point of research.

Engineering and pumping control measures need to be incorporated into decision support systems, linking biophysical models with social constraints, as applied in other areas of water resources management. Here, an enhanced appreciation of the links between SI and environmental concerns, such as linkages between watertable salinization and interactions with ecosystems, is required. The hydro-ecological aspects of SI have been largely neglected. Major challenges in coastal subsurface hydro-ecology relate to the difference in temporal and spatial scales of interest between water resource managers and ecologists, and the need for quantified ecological responses to environmental stress changes, in particular associated with SI and taking into account saturated-unsaturated and surface-subsurface effects. There are also no reported case studies of measured declines in SFGD or SGD where these are linked to accompanying SI in a quantitative manner. As such, the use of SFGD as a SI management indicator remains elusive, partly due to challenges in SFGD estimation across large scales. As pressure on coastal systems increases, SI processes associated with engineering interventions (e.g., hydraulic, gas-based and physical barrier systems, artificial recharge, scavenger wells, etc.) are likely to come under increased scrutiny. However, the performance of engineering measures requires more extensive analysis to evaluate the response time-scales and the resilience once mitigation measures are installed. A holistic comparative analysis of the different engineering options for SI remediation and mitigation, to guide decision-makers on engineering approach and design, is lacking.

The current worldwide SI status is not entirely clear. There are some baseline data, but in the absence of quantitative national and continental summaries, it is not possible to project reliably into the future under climate change or population growth scenarios. Extraction from coastal aquifers is sure to change in time, but the impact of SI globally is unknown. Linkages between population growth, groundwater extraction, contamination, SFGD and climate change effects are needed to properly distinguish between the various coastal impacts and the different SI causal factors. A SI typological approach, drawing from other typological studies, could also provide the basis for useful national and continental summaries, i.e., to at least illustrate qualitatively the propensity of SI to occur and to rank SI hotspots, considering large lengths of coastal fringe.

Perhaps the most important scientific challenge in SI research relates to implementation of SI knowledge, regarding hydrogeochemical processes, investigative methods and management approaches, into practice. Strategies are needed to reduce substantial gaps between SI knowledge and management practice. More studies that retrospectively and critically evaluate the effectiveness of management practices, and in particular initiatives arising from SI research findings, are needed to demonstrate the benefits or otherwise of methods and knowledge gained from SI research. Post-audits of this nature are by-and-large absent from the literature, but would undoubtedly lead to substantial knowledge gains regarding SI processes, investigation methods and management practices.

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Table 1. Popular SI codes

SI Code	Basic Model Features[1]	Model testing[2]	Practical SI applications
2D/3DFEMFAT	FE, SU	H	Zhang et al. [2004]
FEFLOW	FE, SU, GUI	H, E, C	Gossel et al. [2010], Watson et al. [2010], Yechieli et al. [2010]
FEMWATER	FE, SU, GUI		Datta et al. [2009], Carneiro et al. [2010]
HYDROGEOSPHERE	FE, SU	E, C	Thompson et al. [2007], Graf and Therrien [2005]
MARUN	FE, SU	H	Li and Boufadel [2011], Abdollahi-Nasab et al. [2010], Boufadel et al. [2000], Boufadel et al. [2011]
MOCDENS3D	FD, S, GUI	H, E, R	Bakker et al. [2004], Oude Essink [2001a, 2001b, 2001c], Oude Essink et al. [2010], Giambastiani et al. [2007], Vandenbohede and Lebbe [2006, 2007], Vandenbohede et al. [2008a, 2008b, 2008c, 2009, 2010]
MODHMS	FD, SU, GUI	H	Werner and Gallagher [2006]
SEAWAT	FD, S, GUI	H, E, C, R	Cherubini and Pastore [2011], Bakker et al. [2004], Langevin et al. [2010], Vandenbohede and Lebbe [2011], Webb and Howard [2011], El-Bihery [2009], Lin et al. [2009], Mao et al.[2006], Kourakos and Mantoglou [2009, 2011], Abdullah et al. [2010], Praveena et al. [2010], Praveena and

			Aris [2010], Luyun et al. [2009], Goswami and Clement [2007], Dausman et al. [2009b]
SUTRA	FE/FD, SU, GUI	H, E, C	Nishikawa et al. [2009], Pool and Carrera [2010]
SWI	FD, S, IF	R	Bakker et al. [2004]

[1] FD - finite difference; FE - finite elements; IF – interface flow interfaces; S - saturated flow only; SU - saturated-unsaturated flow; GUI - dedicated graphical user interface available

[2] E - Elder problem; H - Henry problem; C - HYDROCOIN salt-dome problem; R - rotating fluids problem

Table 2. Categorization of optimization technique in SI management

Optimization category and simulation approaches[1]		Mixing Zone[2]	Optimization technique[3]	Example References
EMA		IF	NLP	Shamir et al. [1984]
		VD	NLP	Das and Datta [1999]
SOA	Numerical Solution	IF	GA, NLP	Willis and Finney [1988]; Finney et al. [1992]; Emch and Yeh [1998]; Mantoglou et al. [2004]; Mantoglou and Papantoniou [2008]
		VD	SA, GA	Qahman et al. [2005], Abarca et al. [2006]
	Analytical Solution	IF	LP, NLP, GA, ACO	Cheng et al. [2000]; Mantoglou [2003]; Cheng et al. [2004]; Park and Aral [2004]; Ataie-Ashtiani and Ketabchi [2011]

	Surrogate Models	RMA		LP, NLP, GA	Hallaji and Yazicigil [1996]; Abarca et al. [2006]; Karterakis et al. [2007]
		Neural Networks and Genetic Programming	IF	GA	-
			VD	GA	Rao et al. [2004]; Bhattacharjya and Datta [2005, 2009]; Dhar and Datta [2009]; Kourakos and Mantoglou [2009]; Sreekanth and Datta [2010]; Sreekanth and Datta [2011a]

[1]EMA: Embedding approach; SOA: Simulation-optimization approach; RMA: Response matrix approach

[2]IF: Interface flow; VD: Variable density flow

[3]LP: Linear programming; NLP: Non-linear programming; EA: Evolutionary algorithm; GA: Genetic algorithm; SA: Simulated annealing, ACO: Ant-colony optimization

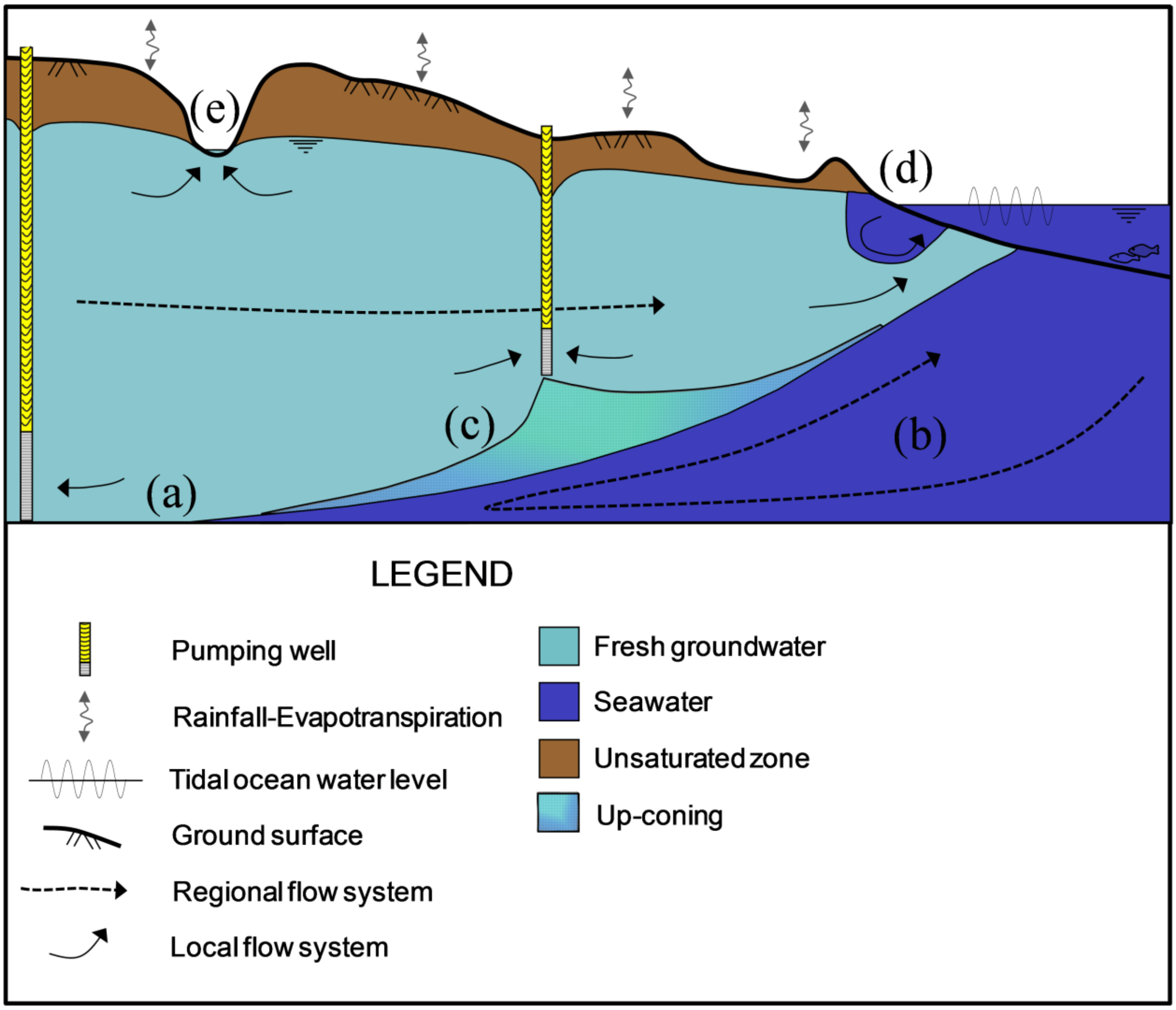

Fig. 1. Simplified diagram of a coastal unconfined aquifer setting, showing (a) seawater wedge toe, (b) density-driven circulation in the seawater zone, (c) seawater upconing due to well pumping, (d) coastal fringe processes, such as tidal seepage face and upper seawater recirculation zone, (e) head-controlled surface expression of groundwater.