CHAPTER 8

Effects of land-use changes and groundwater pumping on saltwater intrusion in coastal watersheds

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Abstract

This chapter describes the occurrence of saltwater/freshwater interface in coastal aquifers. It has a brief description of the physics, hydraulics, and mathematics behind the theory of saltwater intrusion and the saltwater/freshwater interface. It also includes a section on numerical modeling techniques and the available computer models for saltwater-intrusion problems in coastal aquifers. Anthropogenic effects on saltwater intrusion such as changes in land use and landscape vegetation cover, groundwater pumping patterns, and the amount of pumping are also discussed. Finally, remedies, control and management of saltwater intrusion such as implementation of recharge wells, creation of artificial recharge basins, and construction of barriers are discussed. This chapter is expected to be useful to agricultural communities of coastal plains whose irrigation water is solely dependent on groundwater resources. It may also be of interest to individuals working in coastal-watershed and land-management businesses and governmental or local decision makers by addressing the problem of saltwater intrusion into coastal aquifers and its possible remediation and control techniques.

1 Introduction

Coastal aquifers are the main freshwater water supply sources for most urbanized coastal areas around the world. These aquifers are highly sensitive to disturbances such as land-use changes and groundwater pumping. Inappropriate management
of these aquifers may limit their use as sources of freshwater due to saltwater intrusion. Saltwater intrusion and water-quality degradation have become two of the major constraints in groundwater management problems of coastal aquifers. Saltwater intrusion degrades water quality of production wells and consequently the wells have to be abandoned. To avoid saltwater-intrusion problems, adequate natural recharge of these aquifers should be maintained. Pumping rates and production-well locations are the two key parameters that can be optimized for adequate groundwater management.

Under a natural condition, pressure balance exists on both sides of the saltwater/freshwater interface. This natural condition turns into an environmental problem if this pressure balance is changed in favor of saltwater. This pressure imbalance is mainly the results of two major human activities: excessive groundwater pumping; and reduction of groundwater recharge as a result of land-use/land-cover changes. Long-term droughts and the subsequent reduction of recharge will also cause saltwater intrusion. During groundwater pumping from a coastal aquifer, freshwater levels drop; consequently the saltwater intrusion can move further inland toward the pumping wells. Eventually, the saltwater degrades the water quality of drinking and irrigation waters. To prevent saltwater intrusion, more and more countries adopt extensive monitoring schemes and numerical models to assess sustainable groundwater use that will protect their coastal groundwater resources.

This chapter presents our current knowledge of the saltwater-intrusion process in the coastal aquifers. The basic physical theory behind the saltwater intrusion, various conceptual modeling approaches, and preventive measures against the saltwater-intrusion problems have been discussed by focusing on the environmental and ecological concerns in coastal aquifers. Several numerical models and their application studies have also been discussed. Numerous studies have shown the effectiveness of numerical and optimization models in simulating the saltwater intrusion and the measures taken against saltwater intrusion before field implementation of costly measures. This chapter focuses on anthropogenic effects such as changes in land use and groundwater pumping and natural effects such as tides on saltwater intrusion in a cause-and-effect relationship context.

2 Concept of saltwater intrusion in coastal aquifers

All coastal aquifers are subject to saltwater intrusion, which is defined as flowing of saltwater from the sea toward inland freshwater aquifers by pushing the saltwater/freshwater interface landward and/or upward. The density difference between freshwater and saltwater causes this inland flow from high-density seawater to low-density freshwater. If a denser fluid column coincides with less-dense fluid column, the former starts flowing towards the latter column because of the pressure difference between the two fluid columns until a new pressure balance is achieved. Pressure at the bottom of a water column is the product of specific weight and water height. If these water columns were connected at the bottom, then the saltwater column starts flowing towards the freshwater column because the pressure beneath the saltwater column is larger than that of the freshwater column.
of the same height. This phenomenon precisely explains the physics behind the natural saltwater intrusion in coastal aquifers. Naturally farther inland, the higher freshwater levels trigger flow of freshwater seaward, due to its higher potential. These two reverse flow types, i.e. from higher density to lower density and from higher potential to lower potential, complete the picture of the saltwater-intrusion process. At a coastal boundary, freshwater flows towards sea in the upper part of the coastal aquifer while saltwater flows toward inland in the lower part of the coastal aquifer. As a result, a freshwater lens forms on the top of the saltwater zone in the form of cone shape or a wedge towards inland (Fig. 1).

In coastal aquifers, the freshwater body overlies the saltwater body because the unit weight of freshwater is less than that of saltwater. A boundary surface exists between fresh and saltwater bodies known as the “saltwater/freshwater interface.” The thickness of this interface is relatively constant along which freshwater and saltwater are mixed by hydrodynamic dispersion, which results from molecular diffusion and mechanical dispersion. This boundary is a transitional zone of varying salinity (Fig. 1). In aquifers, groundwater flow is dominantly laminar, thus the thickness of the saltwater/freshwater interface is small compared to the thickness of the aquifer. Therefore, in some analyses, an abrupt, sharp, well-defined, distinct interface in the aquifer cross section is assumed. Two fluids on either side of this sharp interface are assumed to be immiscible. If a coastal aquifer consists of two or more distinct aquifers, each aquifer will have its own interface. This interface is assumed to be static and, practically, there is no flow across the interface due to pressure balance on both its sides. Under these conditions, a steady-state seaward freshwater flow occurs discharging freshwater to the sea as a result of a constant hydraulic gradient. In unconfined aquifers, freshwater discharges to the sea across the sea floor. However, in confined aquifers, freshwater flows out to the sea by

![Figure 1: A schematic view of saltwater intrusion.](image-url)
upward leakage through an overlying layer or by flowing to the sea directly. A wedge-shaped denser body of saltwater will develop beneath the lighter freshwater in each aquifer forming an interface between freshwater and saltwater.

Anthropogenic effects such as excessive groundwater withdrawals and/or severe reduction of aquifer recharge, distort the saltwater/freshwater interface into irregular patterns and sizes depending on location, manner and degree of those effects. Although groundwater pumping is the main cause of saltwater intrusion along the coasts, lowering of water table by drainage canals also leads to saltwater intrusion [1]. Generally, the shape and extent of saltwater-intruded zones depend on several factors such as the magnitudes of freshwater flow rates from the aquifer to the sea, the total rate of groundwater withdrawal compared to total freshwater recharge to the aquifer, the distance of pumping wells and drainage canals from the coast, rainfall intensities and frequencies, land-cover type over natural recharge areas, rates of evaporation, physical characteristics of aquifer materials, presence of confining units that may prevent saltwater from moving toward or within the aquifer, and tidal effects.

In addition to lateral intrusion from the sea, saltwater intrusion can occur through several processes including upward intrusion from deeper, more saline zones of an aquifer system, and by downward intrusion from brackish surface waters. “Saltwater encroachment” and “saltwater intrusion” have been used in the literature to refer to lateral and vertical movements of saltwater, respectively [2]. Another specific term “saltwater upconing” was also used to describe the movement of saltwater from a deeper saltwater zone upward into the freshwater zone in response to pumping at a well or well field [2].

Saltwater intrusion is a contamination source of freshwater resources when concentrations of dissolved solids exceed drinking and/or irrigation water standards. The timing and degree of saltwater intrusion varies widely depending upon the hydrogeological setting of aquifers. The time required for saltwater to move through the aquifer and reach a pumping well can be quite long, therefore many years may pass before saltwater intrusion is detected at a particular location.

3 Hydraulic approaches to treatment of saltwater intrusion

The first saltwater-intrusion problem was reported by Braithwaite [3] that was caused by overpumping many production wells in London and Liverpool, England. He attributed the degradation of water quality of these wells to the lowering of the groundwater levels to below the sea level. It is now well known that saltwater intrusion occurs even if the water-table level is higher than that of the saltwater. The first analytical solution developed to study saltwater intrusion was formulated by Ghyben [4] and Herzberg [5] independently. Thus, it is called the Ghyben–Herzberg relationship. Their analytical solutions approximate the saltwater-intrusion behavior based on a number of simplifying assumptions, which, most of the time, are not applicable to real conditions. Although analytical solutions oversimplify the real problems, they have been used as tools for first-cut engineering analyses in preliminary hydrogeologic investigations. They may serve also as teaching tools as well as verification tools for numerical models. Cheng and Ouazar [6]
provided a number of analytical solutions to saltwater-intrusion problems that are of historical and practical importance. There are some key textbooks dealing with saltwater intrusion in the literature including Bear [7, 8], Todd [9], and Bear and Verruijt [10]. Advances in numerical solution techniques and computing capabilities enabled groundwater modelers to solve more general partial differential equations that realistically describe groundwater problems such as saltwater intrusion.

In a coastal aquifer there are three distinct zones; the freshwater zone, the mixing or diffusion zone, and the saltwater zone. The freshwater zone lies over the saltwater zone nearby the coastline. The salt concentration in the diffusion zone, where hydrodynamic dispersion exists, varies gradually from that of the saltwater to that of the freshwater. The thickness of the diffusion zone may change from 1 m to more than 100 m from system to system based on the hydrodynamic dispersion process [9]. The presence of seawater intrusion and tidal variations in coastal aquifers cause the solutes to move upward towards the coastline when it approaches the saltwater interface and then exits around the coastline [11]. Tidal variations of sea level cause groundwater fluctuation as well as oscillation of the saltwater/freshwater interface. Any stress applied within the freshwater region such as changes in discharge (i.e. pumping) or recharge causes movement of the freshwater–saltwater interface.

Saltwater–freshwater systems in coastal aquifers can be conceptualized by two basic hydraulic approaches depending on the relative thickness of the transition zone between freshwater and saltwater. The first approach is the freshwater-saltwater sharp-interface approach [12–14] and the second is the variable-density approach with solute transport (or hydrodynamic dispersion). The former approach ignores the mixing zone and considers the transition zone to be negligible compared to the aquifer thickness. In this approach, the saltwater-intrusion phenomenon is modeled as a two-region fluid flow separated by a sharp interface. However, hydrodynamic dispersion often causes a mixing zone across the interface between saltwater and freshwater. If the thickness of the mixing zone expands to a considerable extent than the sharp-interface approach may not be valid. In this case, the latter approach needs to be used to model the saltwater-intrusion problem using the density-dependent miscible flow and transport approaches. The latter approach assumes a mixing or transition zone and a solute-transport mechanism to model the hydrodynamic dispersion process [15–17]. Because of the improvements of computer technology and numerical solution techniques, this method has been used widely. It also represents the physical system more realistically compared to the sharp-interface approach. Several simulation models based on this approach were developed [18–20].

The appropriateness of either of these approaches and their method of analysis depends on the characteristics of the aquifer system under investigation and the problems of which solutions are being sought. The most important issue in modeling saltwater-intrusion problems is the proper conceptualization of the hydrogeologic system and definition of boundary conditions. Volker and Rushton [21] compared steady-state solutions using disperse- and sharp-interface approaches and showed that as the coefficient of dispersion decreases the solutions of those two approaches converge. Approaches and challenges for studying variable-density groundwater flow are summarized by Simmons et al. [22], Diersch and Kolditz [23], and Simmons [24].
3.1 Sharp-interface approach

In the sharp-interface conceptualization, the saltwater intrusion system can be formulated using two immiscible flow fields, namely freshwater and saltwater flow fields. The interface between these two flow fields acts as a common boundary (Fig. 2). Along this boundary, the two flow fields are coupled through the interfacial boundary condition of continuity of flux and pressure. In three dimensions, this boundary condition is highly nonlinear [7]. However, assuming horizontal flow and integrating the flow equations over the vertical dimension reduces the problem to a two-dimensional problem. The sharp-interface approach does not give information about the nature of the transition zone but simulates the regional flow dynamics of the groundwater system and the response of the interface to applied stresses.

The Ghyben–Herzberg [4, 5] relation between saltwater and freshwater states that, assuming hydrostatic condition in a homogeneous unconfined aquifer, the weight of a unit column of freshwater extending from the water table to the sharp interface is balanced by a unit column of saltwater extending from sea level to the same depth as the sharp interface [25]. As seen in Fig. 2, the Ghyben–Herzberg relationship can be written as:

$$ z = \frac{\rho_f - \rho_s}{\rho_s - \rho_l} h $$

where $\rho_f$ and $\rho_s$ are fresh and saltwater densities (ML$^{-3}$), respectively, $z$ is the depth of sharp interface from sea level for a given point (L), $h$ is the distance...
Effects of Land-Use Changes and Groundwater Pumping

between water table and sea level \((L)\). At about 20 °C, for \(\rho_f = 1.0 \text{ g cm}^{-3}\) and \(\rho_s = 1.025 \text{ g cm}^{-3}\), Eq. (1) becomes \(z_s = 40 h\). This means that if the water table in an unconfined coastal aquifer is lowered 1 m, the freshwater–saltwater interface will rise 40 m. The main drawback of the Ghyben–Herzberg relationship is that it assumes a hydrostatic condition for the saltwater phase, which is not valid in many real cases so that this relationship generally underestimates the depth to the interface [25]. This equation is reasonable away from the shoreline for mainly horizontal flow.

Hubbert [26] provided a more realistic relation describing steady-state groundwater flow in both fresh and saltwater zones. He formulated the following equation:

\[
z = \frac{\rho_f}{\rho_s - \rho_t} \cdot h_t - \frac{\rho_s}{\rho_s - \rho_t} \cdot h_s, \tag{2}
\]

where \(z\) is the elevation of a point on the interface at which head is measured \((L)\), \(h_t\) and \(h_s\) are the freshwater and saltwater heads \((L)\), respectively. Hubbert’s relation must hold true at the interface to ensure the continuity of pressure field at the interface. For the stationary saltwater condition without any saltwater head gradient, if \(h_s\) is taken as zero by choosing the sea level as the datum, Eq. (2) becomes identical to Eq. (1), which is the Ghyben–Herzberg relationship.

When the saltwater is not static, then the freshwater and saltwater flow systems on both sides of the freshwater–saltwater interface need to be reformulated and solved simultaneously. In that case, the sharp interface is considered as a common boundary between those flow systems and it must satisfy Hubbert’s [26] relation, Eq. (2). The saltwater and freshwater flow systems can be defined with the following equations [27]:

\[
S_t \frac{\partial h_t}{\partial t} + \nabla \cdot \bar{q}_t - Q_t = 0, \tag{3a}
\]

\[
S_s \frac{\partial h_s}{\partial t} + \nabla \cdot \bar{q}_s - Q_s = 0, \tag{3b}
\]

where \(h\) is the head \((L)\), \(t\) is time \((T)\), \(\bar{q}\) is the specific discharge \((LT^{-1})\) determined by Darcy’s law for constant density fluid as \(\bar{q} = -K \cdot \nabla h\), \(K\) is the hydraulic conductivity \((LT^{-1})\), \(S\) is the specific storage \((LT^{-1})\), \(Q\) is the source sink term \((T^{-1})\), \(\nabla = \left(\frac{\partial}{\partial x} \hat{i} + \frac{\partial}{\partial y} \hat{j} + \frac{\partial}{\partial z} \hat{k}\right)\), and subscripts \(f\) and \(s\) refer to freshwater and saltwater, respectively. Equations (3a) and (3b), are parabolic partial differential equations, they must be solved simultaneously for the freshwater head \((h_t)\) and saltwater head \((h_s)\). Once the freshwater and saltwater head distributions are known, the interface elevation at any \(x-y\) location in the aquifer can be obtained using Eq. (2). In regions away from the interface, only one type of fluid exists and the flow field can be described by a single equation without the interface [28]. The sharp-interface approach was used by some researchers, i.e. Wilson and Da Costa [29], and Essaid [13, 14].
In regional applications, this approach has advantages because of its relative simplicity, ease of development and use, and less intense data requirements. In addition to overlooking the dispersion zone, its main disadvantage for practical purposes is that it cannot evaluate chloride concentration of individual production wells. Overall, this approach is useful in developing regional water-management plans.

### 3.2 Variable-density and dispersion approach

In reality, and as described earlier, the region between the freshwater and saltwater zones is not a sharp interface but instead it changes gradually over a finite distance, and is known as the zone of diffusion, zone of dispersion, or the zone of mixing (Fig. 3). Unlike the sharp-interface approach, this approach assumes saltwater and freshwater as miscible fluids. Water in this approach transports a solute (salt) that influences its density and viscosity; therefore, partial differential equations governing the groundwater flow system are used to simulate variable-density flow and solute transport with a relationship defining the density as a function of solute concentration. Thus, this approach requires coupling density-dependent groundwater flow and solute-transport equations. In this approach, the flow field is modeled using a single fluid with variable density, which is a function of concentration and pressure applied on the fluid.

The mass-balance equation for a variable-density fluid can be derived by combining the continuity equation, in which the storage term is written as a function of
Effects of Land-Use Changes and Groundwater Pumping

change in pressure and concentration, with a pressure-based density-dependent form of Darcy’s law as follows [30]:

\[
\left(\rho_{sp}\right) \frac{\partial P}{\partial t} + \left( n \frac{\partial \rho}{\partial t} \right) \frac{\partial C}{\partial t} = \nabla \left[ \left( \frac{\rho k}{\mu} \right) (\nabla P + \rho g \nabla z) + Q \right] \tag{4}
\]

where \( \rho \) is fluid density (ML\(^{-3}\)), \( S_{op} \) is specific pressure storage (LT\(^2\)M\(^{-1}\)), \( n \) is porosity (dimensionless), \( C \) is the mass-based solute concentration (MM\(^{-1}\)), \( t \) is time (T), \( k \) is the intrinsic permeability of the solid matrix (L\(^2\)), \( \mu \) is the fluid viscosity (ML\(^{-1}\)T\(^{-1}\)), \( P \) is the pressure applied on the fluid (ML\(^{-1}\)T\(^{-2}\)), \( g \) is the gravitational acceleration vector (LT\(^{-2}\)), \( z \) is the upward coordinate direction (L), and \( Q \) is fluid mass source (ML\(^{-3}\)T\(^{-1}\)).

In the same way, the mass balance for a solute stored in solution can be expressed as

\[
\frac{\partial (n\rho C)}{\partial t} = \nabla \left[ n \rho D \nabla C \right] - \nabla \cdot (n \rho v C) + QC^* \tag{5}
\]

where \( D \) is the dispersion tensor that also includes molecular diffusivity of solutes (L\(^2\)T\(^{-1}\)), \( v \) is the fluid velocity (which is the specific discharge, \( q \), divided by porosity, \( n \)) (LT\(^{-1}\)), and \( C^* \) is the solute concentration of the fluid sources (MM\(^{-1}\)).

The variable-density approach requires the simultaneous solution of the flow equation (Eq. (4)) and the transport equation (Eq. (5)) to obtain the pressure distribution and concentration distribution in the aquifer ranging from pure freshwater to pure saltwater. The equations are nonlinear in nature and require an iterative solution technique. Since there are three unknowns (\( P \), \( C \), \( \rho \)) and only two equations (Eqs. (4) and (5)), another equation is required that can relate fluid density to solute concentration. Such an equation of state relating density to concentration was given by Voss and Provost [31]:

\[
\rho = \rho(C) = \rho_o + \frac{\partial \rho}{\partial C} (C - C_o), \tag{6}
\]

where \( \rho_o \) [ML\(^3\)] is reference fluid density at reference solute concentration \( C_o \) [M/L]. Usually, \( C_o = 0 \), and the reference density is taken as that of pure water. The factor \( \partial \rho/\partial C \) is a constant value of density change with concentration. For example, for mixtures of freshwater and seawater at 20 °C, when \( C \) is the mass fraction of total dissolved solids, \( C_o = 0 \), and \( \rho_o = 998.2 \) [kg/m\(^3\)], then the factor, \( \partial \rho/\partial C \), is approximately 700 [kg/m\(^3\)] [31].

In large regional simulations, a steady-state solute distribution can be assumed, in which case the concentration defining the density of fluid is known and will not change over the period of the analysis. Based on this assumption, there is no need to solve the transport equation, and the solution of the density-dependent flow equation (Eq. (4)) is sufficient to describe the flow system [27]. This approach has been used by several researchers, i.e. Weisss [32], and Kontis and Mandle [33].

The advantage of the variable-density method is that the response of the system to stresses influencing the thicknesses of the transition zone can be better analyzed.
and understood. Density-dependent flow and transport conceptualization represents the actual physical system more accurately than the sharp-interface approach. Also, the concentration distribution of solutes is simulated, and estimates of solute concentrations in individual wells are obtained. However, more input parameters are required, and the method is computationally more complex.

3.2.1 Equivalent freshwater heads approach

Equivalent freshwater head refers to a height of freshwater column that exerts the same pressure applied by saltwater at a given point. With the equivalent freshwater head approach, density effects can be converted into equivalent freshwater heads by which the density-dependent flow model turns into a constant-density flow model. Weiss [32] was one of the first to reformulate the groundwater flow equation in equivalent freshwater head form. Langevin and Guo [34] presented a methodology to couple a constant-density groundwater flow code with a solute transport code to simulate variable-density groundwater flow and solute transport in three dimensions using the equivalent freshwater head concept. They coupled a popular groundwater flow model, MODular three-dimensional finite-difference groundwater FLOW model (MODFLOW) [35, 36], and a popular transport model, Modular 3-D Transport model with Multi-Species structure (MT3DMS) [37], on a platform called the SEAWAT computer program [38].

Many existing models used for density-dependent groundwater simulation formulate the groundwater flow equations in terms of pressure. Langevin et al. [38] reformulated the flow equation in terms of freshwater heads to be able to use MODFLOW’s flow equation routines. The equivalent freshwater head, \( h_f \) [L], can be defined as:

\[
h_f = \frac{P}{\rho_f g} + z,
\]

where \( z \) is the elevation of the point at which head is measured (L) and \( P \) is the fluid pressure at the point of measurement (ML\(^{-1}\)T\(^{-2}\)). The equivalent freshwater head formulation leads to a system of variable-density flow equations that can be solved relatively easily using the existing constant density groundwater flow equations in MODFLOW. The final form of the flow equation in terms of equivalent freshwater head is [39]:

\[
\rho S_{st} \frac{\partial h_f}{\partial t} + n \frac{\partial \rho}{\partial t} + \frac{\partial C}{\partial t} = + \nabla \left( \rho K_f \left( \nabla h_f + \frac{\rho_f - \rho_t}{\rho_t} \nabla z \right) \right) + Q
\]

where \( S_{st} \) is the freshwater specific storage [L\(^{-1}\)] defined as the volume of water released from storage per unit volume per unit decline of freshwater head, and \( C \) is the concentration of solute mass per unit volume of fluid [ML\(^{-3}\)]. For a constant-density system, Eq. (8) reduces to the flow equation solved by MODFLOW.

Motz [40] numerically investigated hydraulic heads in the freshwater part of saltwater/freshwater interface using MODFLOW and SEAWAT. He demonstrated that it is possible to represent the effects of the saltwater/freshwater interface in
MODFLOW simulations by specifying the boundary conditions as equivalent freshwater heads at the coastal boundary over the full thickness of the aquifer. This approach does not give any insight into the location or thickness of the interface and saltwater concentration at the interface, but it can be used as a guide during the construction of new regional groundwater flow models to ensure correct calculation of heads in the freshwater part of the aquifer.

The general form of the solute transport equation used in SEAWAT, which is solved in MT3DMS, is identical to Eq. (5) without the density term:

\[
\frac{\partial (nC)}{\partial t} = + \nabla \cdot [nD \nabla C] - \nabla \cdot (qC) + QC^*.
\]

For conditions with large spatial density gradients, the \( \nabla C \) term in Eq. (9) should be formulated as \( \rho \nabla (C/\rho) \) [41]. For most practical applications with moderate density variations, Zheng and Bennett [41] suggest that Eq. (9) represents a suitable approximation of the solute mass balance.

Equations (8) and (9) are coupled to model saltwater intrusion in variable-density groundwater systems. Fluid density is a function of solute concentration, transport is dependent on the flow field, and the storage term in the transient-flow equation incorporates changes in concentration. Concentrations resulting from the solution of Eq. (9) are used by an equation of state to calculate fluid density using the following linearized relationship between fluid density and solute concentrations similar to Eq. (6):

\[
\rho = \rho_t + \frac{\partial \rho}{\partial C} C.
\]

This equation does not include the dependence of fluid density on temperature or pressure, and thus it should not be used for other than isothermal systems with an incompressible fluid. For other conditions Diersch and Kolditz [23] provided a summary of more rigorous forms of the equations of state. The partial differential equations presented in Eqs. (8) to (10) are simultaneously solved by numerical methods to obtain head and solute concentration distribution in the system.

4 Numerical models and case studies

Numerical modeling helps in analyzing coastal aquifer systems and provides quantitative insights as to their best management. It is one of the essential tools that have been used to understand groundwater flow and saltwater movement in coastal aquifers. It generally lies at the heart of any planning or research process of coastal aquifers. It provides a means to analyze complex systems of groundwater flow and saltwater movement in coastal aquifers. Such analysis is often impractical or impossible to do by analytical models or field studies alone. Field studies are costly and analytical models have limited practical use for aquifers with heterogeneous complex geometry; therefore use of numerical models is generally a
Coastal Watershed Management

necessity before additional more detailed field studies. Moreover, models provide significant insights into the potential mechanism of intrusion and clear guidance on the need for additional data, and the type and location of data needed [42]. The exclusion of seawater intrusion in numerical modeling results in an underestimation of solute mass rate exiting around the shoreline and unrealistic migration paths under the seabed [11]. Models are one of the best and most effective tools to design hydrogeologic investigations, locate and select well depths, and answer different “what if” questions. Models are also good tools for understanding the sensitivity of coastal aquifer systems to changes in their hydrologic components and anthropogenic effects such as land-use changes and groundwater exploitations. Without numerical models, testing different management scenarios and system analysis would be almost impossible with field experiments due to their prohibitive cost.

Developments of numerical models for saltwater-intrusion problems showed a parallel progress with groundwater-solute transport and groundwater–surface water interaction models over the years. Simmons [24] summarized the current research challenges and future possibilities in variable-density groundwater flow modeling. He illustrated the widespread importance, diversity and interest in applications of variable-density flow phenomena in groundwater hydrology. These applications include seawater intrusion, freshwater/saline-water interfaces and saltwater upconing in coastal aquifers, subterranean groundwater discharge, dense contaminant plume migration, and density-driven transport in the vadose zone. Sorek and Finder [43] provided a survey of 15 computer codes that simulate saltwater-intrusion problems. Several authors have written groundwater modeling textbooks, including Anderson and Woessner [44] and Konikow and Reilly [45].

A critical step in any numerical hydrologic investigation is to select the appropriate model, which depends on setting clear modeling objectives. According to Maimone et al. [42], practical coastal aquifer modeling studies may have one or more of the following objectives: (1) determining the causes of existing saltwater intrusion and the mechanism that caused it, i.e. lateral intrusion, upconing, or downward leakage; (2) estimating the location of the interface; (3) evaluating the stability of interface in response to pumping; (4) determining the potential for intrusion based on current pumping or projected pumping; (5) estimating expected time of impact for specific well locations based on various pumping scenarios; and (6) testing various approaches to stopping or reversing intrusion, or assessing strategies for sustainable use of coastal aquifers as viable water supplies even with ongoing intrusion.

The sharp-interface approach has been used successfully in many regional modeling studies to analyze the long-term stability of coastal wells. This modeling approach can provide insight into horizontal movement of saltwater under the influence of both sea-level rise and coastal pumping. It can help estimate the optimum rate of long-term viable pumping without amplifying saltwater intrusion further inland. Models based on this approach have been used as primary planning tools in Florida, New York [42], and Hawaii [28].

In analyzing upconing of saltwater (Fig. 4), it is important to calculate the maximum sustainable pumping rate that avoids saltwater upconing, or to calculate the
expected levels of salt in the wells. In this case, an analytical model can be a solution to the saltwater upconing problems [46]. However, numerical models including sharp-interface models, density-dependent groundwater flow models, and coupled flow and transport models are more useful in simulating this situation in more complex hydrogeologic environment. In upconing problems, the density-dependent groundwater flow models, or coupled flow and transport models may have to be used if the salt concentration gradients underneath the freshwater zone need to be quantified and mapped.

Saltwater-intrusion models often do not have sufficient data to provide appropriate calibration and verification; therefore most of the models have been tested against standard analytical solutions to verify the correctness of the numerical approximations. This type of verification procedure is called benchmarking. Verifying a density-dependent flow model by testing it against data from either a laboratory-scale experimental or a field-scale case study is a difficult task because availability of these types of data sets is limited [47]. Simpson and Clement [48] proposed and used the term benchmarking as a way that the numerical algorithm can reproduce the prior history of a well-defined problem. It also refers to model testing against standard problems and/or well-controlled field and laboratory studies that have been sufficiently tested and are widely accepted by model developers. The most popular benchmarking problems are the Henry’s saltwater-intrusion problem [15], for which an analytical solution exists, and the Elder’s salt-convection problem [49], for which laboratory and numerical data are available.

Three commonly used models and their example application in a few case studies are briefly discussed below. These models are: 1) USGS SHARP model, a quasithree-dimensional, finite-difference model to simulate freshwater and saltwater
flow separated by a sharp-interface model [13, 14], 2) the USGS finite-element variable-density flow and transport simulation code, SUTRA, [30, 31], and 3) SEAWAT model [38], which is a platform coupling the groundwater flow model, MODFLOW [35, 36], and the transport model MT3DMS [37].

**SHARP** is a quasithree-dimensional finite-difference model developed by Essaid [13, 14] to simulate freshwater and saltwater flow separated by a sharp interface in layered coastal aquifer systems. The model is quasithree-dimensional because each aquifer is represented by a layer in which flow is assumed to be horizontal. This model is a good example of the sharp-interface approach in modeling saltwater intrusion. The main assumption in this model is that the width of the freshwater–saltwater transition zone is small relative to the thickness of the aquifer so that it can be assumed that freshwater and saltwater are separated by a sharp interface and they do not mix. This modeling approach, in conjunction with vertical integration of the aquifer flow equations, can be used in regional-scale studies of coastal areas. This approach does not give information concerning the nature of the transition zone but does reproduce the regional flow dynamics of the system and the response of the interface to applied stresses.

SHARP was used to simulate the coastal aquifer of the Soquel-Aptos basin, Santa Cruz County, California. The topography of the Soquel-Aptos basin, Santa Cruz County, California, ranges from very steep valley slopes and angular land forms to nearly flat marine-terraced, sea cliffs and narrow beaches along Monterey Bay. The region is a populated urban area with increasing demand for freshwater. The principle aquifer is a layered aquifer with variable thickness. Saltwater has not intruded onshore and the position of the interface offshore was not known. An analysis was performed by Essaid [14] to estimate the amount of freshwater flow through the system, the position of the saltwater interface offshore, the quantity of discharge to the sea that must be maintained to keep the interface at or near the shore, and the rate at which the interface will move due to onshore development.

Initial conditions were obtained by simulating predevelopment conditions by letting the system reach the steady-state condition. Transient conditions were simulated for the period of 1930 to 1985. Historical well-pumping values were inputted to the model according to the time they were tapped to the aquifer and went into production. Prior to the predevelopment, the recharge to the system was 0.50 m$^3$ s$^{-1}$, of which 0.47 m$^3$ s$^{-1}$ discharged onshore to creeks and to the overlying Aromas Sand. Only 0.03 m$^3$ s$^{-1}$ of the water flowed offshore to the ocean as fresh groundwater discharge. This small proportion of recharge was sufficient to maintain the freshwater/saltwater interface position offshore. With the development and increase in groundwater pumping, the 1981 onshore and offshore discharge rates decreased to 0.43 m$^3$ s$^{-1}$ and 0.01 m$^3$ s$^{-1}$, respectively. The 1930 to 1985 simulations predicted almost no movement of the interface despite significant changes in the groundwater flow system. Essaid [14] concluded that interface response was quite slow and took place over a long time. The slow response of the saltwater zone was a result of the considerably low horizontal and vertical hydraulic conductivities that impede the flow of saltwater into the interface zone.
SUTRA simulates fluid movement and transport of either energy or dissolved substances in a subsurface environment. The original versions of SUTRA [30, 31] are capable of simulating variable-density flow and transport of either heat or one dissolved species through variably to fully saturated porous media. Hughes and Sanford [50] modified SUTRA to simulate multispecies transport in which species may or may not affect fluid density and viscosity. SUTRA solves the two- or three-dimensional form of density-dependent saturated or unsaturated groundwater flow, and solute- and energy-transport equations using two-dimensional finite-element and three-dimensional finite-difference methods. Solute-transport simulation with SUTRA may be used for cross-sectional modeling of saltwater intrusion in aquifers at near-well or regional scales, with either dispersed or relatively sharp transition zones between the freshwater and saltwater. SUTRA provides, as the primary output, fluid pressures and either solute concentrations or temperatures as functions of time and space. SUTRA requires specification of pressures rather than hydraulic heads in saltwater-intrusion simulations.

Gingerich and Voss [51] applied SUTRA to model groundwater flow and solute transport in the Pearl Harbor aquifer, southern Oahu, Hawaii. They showed that the readjustment of the freshwater–saltwater transition zone would take a long time following changes in pumping, irrigation, or recharge in the aquifer system. They claimed that the Ghyben–Herzberg estimate of the freshwater/saltwater interface depth is not a good predictor of the depth of potable water. Their simulations showed that the transition zone moved upward and landward compared to the predevelopment period of year 1880.

Hunt et al. [52] and Voss [53] used SUTRA in cross section to numerically evaluate Oahu’s southern coastal aquifer hydraulics and saltwater intrusion by analyzing and tracking the movement of the freshwater–saltwater transition zone. The purposes of this analysis were to give Oahu water managers reliable scientific data to help them decide on water allocation, managing Oahu’s aquifers, and to quantify the amount of groundwater that would be safely produced from each aquifer. Oahu has high rainfall, highly permeable aquifers, and a coastal semiconfining layer overlaying many of the aquifers. The major aquifers on Oahu are composed of hundreds of thin lava flows that were extruded onto the land surface forming dike-type vertical aquifers. The layers, generally several meters thick, form a matrix of thin overlapping tubular units commonly tens to hundreds of meters wide and up to 30 km long that dip about 5 to 10 degrees from the mountainous recharge areas to the ocean. The coastal confining unit keeps heads high at the coast and creates a very thick freshwater lens. The combination of high recharge, high permeability and impeded discharge provides a rich freshwater supply for both drinking and irrigation. Other than anthropogenic contamination, saltwater exists at the bottom of each of Oahu’s coastal aquifers and represents a potential threat by means of saltwater upconing.

This Southern Oahu’s aquifer system was analyzed as a 2D cross section that represents the basal aquifer and caprock containing the basal freshwater lens, saltwater–freshwater transition zone, and deeper saltwater. Recharge to the aquifer
Coastal Watershed Management

enters through the water table, and through the upstream boundary of the aquifer representing inflow from dike compartments and the central plateau. Withdrawals occur near the coast. Recharge and withdrawal rates were changed with time according to historical data. The seaward boundary was held as zero pressure at the hydrostatic saltwater pressure. The rest of the boundaries were assumed to be impermeable. The transition zone was analyzed to predict the water quality of freshwater for eight different scenarios for the period of 1980–2080. These scenarios have different withdrawal and/or recharge rates [54]. The limiting concentration of potable water was defined as 2% saltwater concentration.

Some scenarios predicted sustainable potable water supplies, while others resulted in significant saltwater intrusion. Up to about 75% of the assumed recharge can be withdrawn before any significant saltwater intrusion affecting the water quality of production wells. According to this analysis, the key parameter controlling the saltwater intrusion was the net system discharge, not the percentage of recharge pumped. It was also found that the maximum safe withdrawals vary depending on the actual long-term recharge rate.

SEA W AT was developed by combining MT3DMS and a modified version of MODFLOW into a single program that solves the coupled variable-density groundwater flow and solute-transport equations [39]. MODFLOW was modified to solve the variable-density flow equation by reformulating the matrix equations in terms of fluid mass rather than fluid volume and by including the appropriate density terms. Fluid density is assumed to be solely a function of the concentration of dissolved constituents; the effects of temperature on fluid density are not considered. Temporally and spatially varying salt concentrations are simulated by SEAW AT using routines from the MT3DMS program. SEAW AT couples the groundwater flow equation with the solute-transport equation. The basic assumptions in SEAW AT development are the followings; Darcy’s law is valid; the diffusive approach to dispersive transport based on Fick’s law can be applied; isothermal conditions prevail; the porous medium is fully saturated; and a single, fully miscible liquid phase of very low compressibility is assumed.

Dausman and Langevin [55] constructed a variable-density groundwater flow model of Broward County, Florida using SEAW AT. In that study, SEAW AT was used to evaluate the relationship between water-level fluctuations and saltwater intrusion. The model was representative of many locations in Broward County that contain a well field, a control structure, a canal, the Intracoastal Waterway, and the Atlantic Ocean. The model was used to simulate short-term and long-term movements of the saltwater interface resulting from changes in rainfall, well-field withdrawals, sea-level rise, and upstream canal stages.

Long-term simulations, i.e. periods greater than 10 years, revealed that the upstream canal stage controls the movement and location of the saltwater interface. If the upstream canal stage is decreased by 30 cm, the saltwater interface takes 50 years to move inland and stabilize. If the upstream canal stage is then increased by 30 cm, the saltwater interface takes 90 years to move seaward and stabilize. If sea-level rises about 48 cm over the next 100 years as predicted, then inland movement of the saltwater interface may cause well-field contamination.
These results show that the upstream canal stage substantially affects the long-term position of the saltwater interface in this surficial aquifer system. The saltwater interface moves faster inland than seaward as a result of changes in the upstream canal stage. The saltwater-intrusion problem in the Florida Biscayne aquifer does not seem to be severe if the well-field withdrawal is increased for short-term drought problems based on the assumption that well-field withdrawals will decrease once the drought is over. Sea-level rise may be a potential problem to the water supply in Broward County because of the movement of the saltwater interface inland toward well fields.

The number of SEAWAT applications is increasing rapidly because SEAWAT is taking advantage of the widely used and accepted groundwater models: MODFLOW and MT3DMS models. Among these applications are: a study of movement of the saltwater interface in Broward County, Florida [55]; a second study that investigated the freshwater–saltwater interaction and the effects of pumping and sea-level change in the Lower Cape Cod Aquifer System, Massachusetts [56]; a third study that assessed the natural and the anthropogenic impacts on freshwater-lens morphology on Dog Island and St. George Island, Florida [57]; and lastly a study of salt transport in the Okavango Delta [58].

5 Land-use changes and groundwater pumping

Land-use changes on coastal aquifers are direct results of population increase and urban and agricultural developments. Increasing demand for water requires more groundwater exploitation from coastal aquifers. Population growth on coastal areas creates a chain reaction by triggering land-use changes and exploitation of more groundwater resources and deteriorating the coastal ecology and groundwater resources. As population increases, lands used for urban and agriculture increase and consequently demand for freshwater supplies increases. These stresses change the hydrologic and hydrogeologic characteristics of the coastal aquifers. They would have pronounced adverse effects on freshwater and saltwater quality, if proper precautions and preventative measures are not taken.

The effects of land-use changes are primarily seen on land-cover type, i.e. a forest area can be turned into an urban or agricultural area, or a barren land can be turned into a golf course. Obviously, changes in land use influence water infiltration and recharge to groundwater, and probably may change the microclimate of a region. As described before, according to the Ghyben–Herzberg relationship, there is a strong linear relationship between elevation of the freshwater table and the depth of the saltwater/freshwater interface such that 1-m drop in an unconfined coastal aquifer is compensated by about a 40-m rise of the interface. The key parameter is the height of the freshwater table above sea level or the quantity of fresh groundwater flow towards the sea. For example, if the land use of an area changes from forest to urban, then recharge rates in the area reduce due to increase in the proportions of impervious areas. Consequently, the water table drops, which causes the saltwater interface to move inland. On the other hand, if the change in land use increases the soil permeability, such as under golf courses, this would
result in an increase of the recharge rates to groundwater, increasing groundwater flow to the sea and/or raising the water-table depth. As a consequence, saltwater intrusion will decrease.

Agricultural and urbanization activities on coastal zones might have a variety of effects on their aquifers depending on the hydrogeology of these areas, type of agricultural practices, and the degree of urbanization. If agricultural land is drained by lowering the water table, then saltwater intrudes inland by responding to low water-table levels. In coastal areas, urban and agricultural land uses might have parallel environmental conflicts. For example, in coastal urban areas, the underlying aquifer system must serve as a source of water supply, in which water levels drop due to pumping, while the aquifer water levels must be maintained at certain elevations to prevent saltwater intrusion. Changes in natural groundwater flow patterns and the associated reduction in groundwater flow toward coastal bays alter salinity and affect the local marine ecology. It can be concluded that land-use changes and groundwater pumping directly affect saltwater intrusion in the coastal aquifer systems as well as in coastal bays. Therefore, a careful water- and land-use management plan should be applied on coastal zones.

The Everglades in south Florida is a very good example for land-use changes and corresponding saltwater-interface response to those changes. A highly controlled water-management system has evolved during the 1900s largely to provide drained land for a rapidly expanding population in the Everglades. Draining of Everglades’s wetland areas during the last 75 years has provided the opportunity for westward expansion of agricultural, mining, and urban activities, which changed the land-use and water-use characteristics of the region substantially [59]. Urban and agricultural growth and land-use change can greatly impact the ecological health and stability of coastal areas. A review of 100 years of land-use and population changes in the Everglades illustrated impacts of those changes on water resources in a coastal area where urban areas are growing rapidly and replacing agricultural areas [59]. Some declines in water levels can be directly attributed to municipal groundwater withdrawals; however, water-level declines over wider areas were a direct result of canal drainage, i.e. dropping the water-table elevations. Landward movement of the saltwater interface has been an issue of local and regional concern since the 1940s. In decreasing importance, canal overdrainage, overpumping from wells located near the coast, and upconing of seawater are the primary sources of saltwater in the surficial aquifer system. Significant changes in land and water uses at the coastal zones contributed to the deteriorating conditions of the marine ecosystem in south Florida. Renken [59] showed that saltwater intrusion in the surficial aquifer system in south Florida was a direct consequence of water-management practices, concurrent agricultural and urban development, and natural drought conditions. These findings would be true for any other areas with similar conditions.

The seaward groundwater flow affects coastal ecosystems; it sustains the flow and aquatic habitats of coastal streams during periods when surface runoff is low. Groundwater discharge also helps to maintain water levels and water budgets of freshwater lakes, ponds, and wetlands near the coast. Dissolved chemicals in
groundwater coming from agricultural lands affect the salinity and geochemical budgets of coastal ecosystems and play a role in the biological-species composition and productivity of these systems. Although coastal groundwater systems have been contaminated by many types of chemical constituents, the current concern has focused on the discharge of excess nutrients, particularly nitrogen, to coastal ecosystems. Nutrient contamination of coastal groundwater occurs as a consequence of land-use and water-use changes. Wastewater disposal from septic systems and pollutant and sediment runoff from agricultural and urban areas are the main sources of the nutrient contamination. Nutrient overenrichment can lead to excessive production of algal biomass, loss of important habitats such as seagrass beds and coral reefs, changes in marine biodiversity and distribution of species, and depletion of dissolved oxygen and associated die-offs of marine life [60].

Groundwater-pumping effects on saltwater intrusion should be analyzed in two different scales; the regional scale and individual well scale. In order for an aquifer to supply freshwater to wells, the regional system must be capable of providing the required quantity of water. However, even though the regional system may be in equilibrium, saltwater upconing beneath pumping wells can make these wells produce high salinity water. Although, the regional system may be capable of sustaining the rate of production, the drawdown around an individual pumping well may cause saltwater upconing. Reilly and Goodman [2] showed that in a well analysis in Truro, Cape Cod, MA, although the regional system was in equilibrium and capable of sustaining 4200 m$^3$ d$^{-1}$ at a particular well field based on regional estimates, the actual well was not capable of this production because extensive drawdown occurred around the well causing saltwater upconing. Estimates from local analysis showed that the maximum permissible withdrawal from a single well should not exceed 1800 m$^3$ d$^{-1}$; thus, the withdrawal rate for a well should be limited, and additional wells should be installed if more capacity is required. To reduce the risk of upconing while producing water from freshwater lenses, horizontal shafts (sometimes called a Maui shaft) can be used. The horizontal shafts can produce large volumes of freshwater by skimming water from near the top of the freshwater lens [61].

The total amount of groundwater extraction, locations of wells, and individual well pumping rates should be taken into account all together in analyses of saltwater intrusion. This kind of analysis requires simultaneous use of optimization and numerical groundwater models that are capable of simulating density-dependent flow and solute transport. Optimization and management models should be able to optimize pumping rates, well locations, and number of wells. However, the non-linearity in the variable-density groundwater flow brings difficulties in optimization models due to computational time and burden. Therefore, the complex optimization models with the constraint of saltwater-intrusion problems have yet to become very practical. Nonetheless, saltwater intrusion into wells can also be dealt with in simpler and indirect approaches, i.e. by constraining drawdown at a number of control points, or by minimizing the overall intruded saltwater volume in the whole aquifer. A typical analysis of temporal and spatial variations of coastal saltwater intrusion generally follow the following steps; 1) identification of the
principal factors that control the extent of saltwater intrusion, 2) evaluation of long-term trends in groundwater-withdrawal rates, groundwater-level change, rainfall, and increases in chloride concentration; and 3) determination of causal relations between the position of the saltwater interface, water-management practices, and the expansion of agricultural and urban areas.

6 Tidal effects and sea-level rise on saltwater intrusion in coastal aquifers

Tidal activity in the oceans may force the saltwater to intrude further inland and creates thicker interface compared to the cases without tidal effect. Tidal fluctuations also change the configuration of the interface because of the changes in the flow pattern and the velocity of groundwater near the shoreline. If the depth of an aquifer is much larger than tidal amplitudes, the tidal fluctuation does not have much effect on how far the seawater intrudes into the aquifer but a significant change is observed in the configuration of salt-concentration contours [62]. This change is more pronounced at the top of the aquifer than at the bottom, and is caused by the infiltration of saltwater into the top of the aquifer during the high tides.

Tidal effects on coastal aquifers have been subject to numerous recent studies [63–68]. Inouchi et al. [68] presented an approximate analytical solution and a numerical model for analyzing seawater intrusion in a confined aquifer including the effects of tides. Nielsen [64] reported the first analytical investigation on the slope effects. Essink [69] studied the impact of sea-level rise on saltwater intrusion in the Netherland. From the year 1990 to 2100, an average of 49 cm of global mean sea-level rise is estimated, which will accelerate the salinization process in the aquifers and shift the mixing zone between freshwater and saltwater further inland. This situation will seriously impact every coastal aquifer by easing the upconing process where groundwater is heavily exploited [69]. Ataie-Ashtiania et al. [62] investigated the effects of tidal fluctuations on seawater intrusion and groundwater dynamics in an unconfined aquifer. They reported that the effects of tidal fluctuations are more significant for a sloping beach than for a vertical face with more pronounced saltwater intrusion. In the case of the sloping beach, unlike the vertical face beaches, the saltwater can easily move inland over the beach at the high-tide stage and then infiltrate vertically through the top of the aquifer. They also stated that neglecting the effects of tidal fluctuations underestimates the saltwater-intrusion impact on groundwater quality near the shore, because the large-amplitude tidal fluctuations force the seawater to intrude further inland and also create a thicker interface in shallow coastal aquifers. Kim et al. [70] conducted field studies in a multilayered coastal aquifer in the eastern part of Jeju Island, Korea, to observe the tidal effects on seawater intrusion. They reported that tidal effects on the groundwater level reached up to 3 km inland from the coastline. They found a zone where freshwater and saltwater moved alternately in opposite directions, as influenced by tidal fluctuations.

Although the tidal influence on groundwater dynamics has been studied extensively, the effects of tides on the fate of chemicals in the aquifer have not been
investigated adequately [71]. These effects should be quantified in order to determine the pathways of land-originated nutrients and contaminants entering seawater, and provide useful information for improving strategies for sustainable coastal resources management and development. Tide-induced groundwater fluctuations and oceanic oscillation affect the chemical transport and transformation in the aquifer near the shore. Li et al. [71] reported that a subsurface estuary may play an important role in determining the subsurface chemical fluxes to coastal waters. They used MODFLOW and PHT3D [72] to model contaminant transport and biodegradation in coastal aquifers affected by tidal oscillations. They used toluene as a representative biodegradable contaminant and oxygen as the electron acceptor. Their simulation results demonstrated that tidal fluctuations lead to the formation of an oxygen-rich zone in the near-shore aquifer area. Aerobic bacterial activity sustained by high oxygen concentration in this active zone degrades the contaminants. These effects may have significant implications for the beach environment [71]. El-Kadi [73] developed a model for hydrocarbon biodegradation in tidal aquifers. He modified SUTRA [30, 31] to simulate multispecies fate and transport and combined with a bacterial growth submodel. He applied the model to a hypothetical tidal aquifer. He reported that tides cause additional mixing for nutrients and oxygen-enhancing degradation in the unsaturated zone of the coastal aquifer. He also developed a quantitative approach to assessing the bioremediation in tidal aquifers.

7 Control and management of saltwater intrusion

The first step in control and management of saltwater intrusion is to collect sufficient data to adequately understand the coastal aquifer system and its associated problems. Existing data on water levels and salinity in coastal wells should be reviewed. A database consisting of the present situation of production wells and their current and historical pumping rates, recharge estimates, aquifer hydrogeologic parameters, and estimated position of the interface should be obtained. Once the available data and information have been collected and reviewed, a conceptual model of the mechanism of intrusion is hypothesized [42].

The second step is to use an appropriate numerical model to gain a deeper understanding of intrusion and to test the hypothetical conceptual model. Modeling lies at the heart of control and management processes. Measures considered for preventing intrusion must be tested using the models. It is recommended that a saltwater-intrusion model be developed before any field implementation of those measures is carried out considering their high cost [42]. The developed and calibrated intrusion model can provide a clear picture of the problem and potential threats. Once the problem of saltwater intrusion is clearly identified, potential means of mitigating intrusion can be investigated. The desired state after restoration, in terms of sustainable rates of withdrawals and the groundwater tables and piezometric levels should be determined by water managers. Each option considered to be a solution to the problem must be tested using the numerical model to determine its effectiveness and applicability.
Maimone et al. [42] gave examples of some of the potential measures against saltwater intrusion as follows:

1. Enhancing aquifer recharge to increase freshwater heads to resist saltwater intrusion. Aquifer recharge can be enhanced by spreading any available surface water, or treated waste water, or by capturing surface runoff in recharge basins and allowing them to infiltrate.

2. Lowering the demand for water to reduce pumping from the aquifer. This can be done by convincing or educating the public not to waste water, or increasing water prices, or supplying treated waste waters for irrigation and other nonpotable water uses. In some cases, legal action to forbid water uses for certain times and for certain activities (i.e. car washing) may be helpful to reduce water demand.

3. Creating hydraulic injection barriers to prevent intrusion into unaffected portion of the aquifer by injecting treated waste waters to form a narrow zone in which the freshwater gradient is toward the sea.

4. Tapping alternative aquifers that are located either below or above the impacted aquifer. This relieves the pumping stress on the impacted aquifer.

5. Relocating the wells to areas of higher freshwater heads or areas less susceptible to intrusion. Relocation can also be used to spread out the pumping cone of depression and reducing the potential localized intrusion of upconing.

6. Modifying pumping rates or pumping schedules to allow the well heads to recover.

7. Restricting the pumping rates or the placement of new wells.

8. Replacing deep wells with horizontal wells for skimming freshwater, i.e. Maui-type horizontal shafts.

9. Constructing physical barriers such as slurry walls or sheet piles that can be applied in shallow intrusion situations in small-scale projects.

10. Extracting saltwater using scavenger wells while freshwater pumping continues to stabilize or lower the upconing and increase the storage capacity of freshwater zone.

11. Conjunctive use of surface water and groundwater to offset the excessive reliance on groundwater.

12. The natural recharge can be increased by proper land use (natural vegetation and choice of crops), land-tillage practices, the installation of check dams, retention and detention basins for flood control, and weirs in surface waters, so as to raise the water levels therein and to divert water to adjacent spreading grounds. The main principle of these measures is to hold up the water as long and as much as possible in order to give it more time for infiltration, rather than to let it run off directly. Most of these measures are also favorable in erosion and flood-control terms, but the quality of the water infiltrating in urban areas may be doubtful.

The most important aspect in coastal-aquifer management is the selection of alternative solutions, all of which have some kind of complex tradeoff. The ideal selection of the best alternative requires running saltwater-intrusion and decision-support models. All aspects of decision-making criteria, including economic, environmental, social, technical, and political considerations, should be involved in the
Effects of Land-Use Changes and Groundwater Pumping

selection process of the alternative solutions. Groundwater development and restoration of deteriorated fresh groundwater resources in coastal aquifer systems requires integrated water management of surface water and groundwater, both in terms of water quantity and water quality. Integrated use of numerical models with a geographic information system (GIS) for the input data and presentation of the model output will be helpful in the decision-making procedure. If there is no comprehensive model and/or computer capacity is available, or if there is not enough knowledge or experience with such models, simpler models, such as those ignoring dispersion and assuming sharp interfaces and steady-state conditions, can be used. Those simpler models can be surprisingly helpful for preliminary analysis.

Saltwater barrier projects in Los Angeles County, California were among the good examples showing the complex challenges to be faced in managing a coastal aquifer system. The Central and West Coast groundwater basins are two coastal aquifer systems located adjacent to the Pacific Ocean in California. Severe groundwater exploitation from these basins from the early 1900s to the late 1950s caused saltwater intrusion taking coastal wells out of use, and threatening the sustainability of water supply to the area. Groundwater-management agencies took major steps to halt the intrusion and control the overdraft including construction of freshwater injection wells along the coast, limiting the annual amount of groundwater pumping by legislation, purchasing water from alternative sources for making up the annual and accumulated overdrafts and for use in artificial replenishment and injection wells [74].

The first injection well was tested in the early 1950s by the Los Angeles County Flood Control District using an abandoned water well in Manhattan Beach. The test was successful in creating a freshwater mound and reversing the gradient back to seaward [74]. With the success of the first test, a larger test was carried out including 9 recharge wells, spaced 152.4 m apart, and 54 observation wells from February 1953 to June 1954. This test successfully created a pressure ridge along the injection line, reversing the previous landward gradient to a seaward gradient, which stopped saltwater intrusion [75]. Three major saltwater barrier projects were successfully implemented in California. The West Coast project started in 1953 and had 153 injection and 276 observation wells along a 14.5-km stretch as of 2003. The Alamitos Gap (Los Angeles County) project started in 1964 and had 44 injection and 4 extraction wells, and 239 observation wells along a 3.5-km stretch as of 2003. Dominguez Gap (Los Angeles County) project started in 1969 and had 94 injection and 232 observation wells along a 6.9-km stretch as of 2003 [76]. In all of these projects, potable waters purchased from the Colorado River and north California were used to ensure the water quality of groundwater reserves.

Because of the high maintenance and operation costs of the injection wells, the search for new alternatives is underway in the saltwater-barrier projects in Los Angeles County. The cost of injection water was steadily increasing since 1960, from 1.7 cent m\(^{-3}\) to a maximum of 42.8 cent m\(^{-3}\) in 2001. In the 2001 water year, a total of 37.48 million m\(^{3}\) water was injected into barriers at a cost of nearly $15 million. Johnson and Whitaker [76] reported that nine alternative saltwater barriers were identified by URS Greiner and Woodward-Clyde [77], including
slurry walls, deep-soil mixing, grout curtains, jet grouting, in-situ vitrification, channel lining, rubber dams, nitrogen-gas injection, and biological barrier walls. After a thorough economic and technical analysis using numerical models and optimization models, the nitrogen-gas injection and deep-soil-mixing alternatives were recommended over traditional injection wells. Pilot testing of nitrogen-gas injection alternative is currently underway and preliminary designs for deep-soil mixing were completed [76].

This project showed that injection wells have been successfully used to both control saltwater intrusion and to replenish the overexploited aquifers in California since the 1950s. However, the rising cost of the injection water requires other alternatives to injection wells to be sought. Saltwater-intrusion models in conjunction with optimization models were utilized to maximize the groundwater production while minimizing injection requirements with the constriction of drinking-water quality standards. Based on these results, new alternatives were found to stop saltwater intrusion. Rechard and Johnson [78] applied simulation-optimization methods to the West Coast Basin of coastal Los Angeles to obtain useful quantitative guidance for controlling seawater intrusion. Their goal was to determine the most cost-effective way to raise water levels along the coast, either by increasing injection or reducing pumpage through substitution of delivery of surface water, so as to better control seawater intrusion. For the base-case optimization analysis, assuming constant groundwater demand, substitute delivery was determined to be most cost effective. These studies showed the complexity of the coastal-aquifer management due to saltwater intrusion, and they also showed the necessity of saltwater-intrusion and optimization models in decision making and management processes in coastal aquifers.

8 Summary and conclusion

The current understanding of the saltwater-intrusion process, interaction between saltwater–freshwater environments, and their hydraulic and hydrogeologic characteristics were presented in this chapter. In the saltwater-intrusion process, heavier saltwater moves inland until it is balanced by the freshwater along the saltwater/freshwater interface. Pressure balance occurs at both sides of this interface. Excessive groundwater pumping, changes in land uses, reducing the natural recharge rates, and other human activities reducing the freshwater heads in coastal aquifers change this balance in favor of saltwater. In many aquifers, the occurrence and movement of saltwater have been changed by groundwater pumping. Human activities not only cause saltwater intrusion but also cause adverse impacts on coastal ecosystems. Overexploitation of groundwater resources reduces the seaward flow of freshwater to the coastal ecosystem, increasing the salt concentrations and reducing dissolved oxygen and some nutrients carried by groundwater flow in estuaries. Agricultural practices may also influence the salinity and geochemical budgets of coastal ecosystems that may greatly affect the flora, fauna, and aquatic habitats in coastal ecosystem. Nutrient overenrichment can lead to excessive production of algal biomass, loss of important habitats such as seagrass beds and coral reefs, changes in marine biodiversity and distribution of species, and depletion of dissolved oxygen and associated die-offs of marine life.
Several numerical models were developed to simulate the saltwater-intrusion process, including SHARP, SUTRA, and SEAWAT. Numerical models simulate the saltwater/freshwater interface as either sharp or diffusive. The former approach assumes no mixing between saltwater and freshwater. The models that use this approach are called sharp-interface models. They couple saltwater and freshwater flow equations along the interface. In the regional scale, these models give successful results in terms of general movement of saltwater interface with less computational complexity. However, these models can not be used if saltwater concentrations need to be predicted, e.g. to assess water-quality problems in pumping wells. To simulate the transition zone (or mixing zone) between the saltwater and freshwater, one needs to use the latter type of models. These models simulate single fluid flow with variable density changing with salt concentration using a state equation describing the relationship between fluid density and its salt concentration. This approach is more realistic but it requires more computational efforts compared to the sharp-interface approach. There are three types of saltwater intrusion: lateral and upward saltwater intrusion (regional), downward leakage from brackish surface water (local), and saltwater upconing beneath a pumping well (local).

Water needs for increasing populations along coastal zones and economic developments produce a number of complex and unique challenges to hydrologists, water-resources managers, and public decision makers. Today, most of the coastal aquifers are generally under the threat of saltwater intrusion due to intensive groundwater pumping. A consensus on an effective management option by many stakeholders and other concerned parties must be developed to cope with the potential threat of saltwater intrusion. This effective management strategy must be supported by suitable numerical models and field investigations. The remedial or preventive measures against saltwater intrusion can be one or a combination of the followings: i) increasing recharge naturally or artificially, ii) decreasing the groundwater extraction by demand management or alternative water supplies, iii) creating barriers (hydraulic or physical) to stop or reverse inland saltwater movement, iv) relocation of pumping wells or changing the schedule of pumping, and v) implementing conjunctive use of surface and groundwater resources.

The time scale of observing the benefits of management decisions may be relatively long, but if these critical coastal resources are adversely impacted by over-exploitation then consequent saltwater intrusion will be very costly and last long term. Remediation measures for these impacted coastal freshwater resources will be very difficult and expensive to implement. In many cases, saltwater-impacted coastal aquifers cannot be restored to a viable freshwater condition and have to be abandoned.

References


Coastal Watershed Management


