Chapter 17
Management of Saltwater Intrusion in Coastal Aquifers: An Overview of Recent Advances

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Abstract The demand for freshwater is very high in the coastal regions due to the high population density in coastal areas. To meet this demand for freshwater, the coastal aquifers are often heavily pumped without any regulation, resulting in saltwater intrusion. Therefore, the biggest challenge in the management of coastal aquifer is to meet the demand for freshwater by pumping the coastal aquifer without causing saltwater intrusion. In this study, a brief overview of various methods for identification, prediction, and management of saltwater intrusion is presented. Detection of saltwater intrusion is largely hindered due to insufficient spatiotemporal monitoring because of budgetary constraints. Application, merits, and demerits of the newer cost-effective techniques as well as conventional techniques for identifying saltwater intrusion are discussed in this chapter. The application of various prediction models and their computational difficulties is also presented in this study. Finally, advanced techniques for identification and sustainable management practice in saltwater intrusion are discussed. Though significant progress has been made in the recent past in the management of coastal aquifers, they still show gaps in addressing real-life scenarios. An attempt has been made to highlight the suitability of a developed methodology and their respective limitations.

Keywords Saltwater intrusion · Coastal aquifers · Density-dependent model · Sharp interface model · Water resources management

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17.1 Introduction

From the dawn of civilization, coastal areas continue to be the hub of economic activities and global transport which has led to massive population growth in these coastal areas. It is estimated that about 70% of the world population live near coastal areas (Bear et al. 1999). The high population density in coastal regions results in greater demand for freshwater. Groundwater resources play an essential role in meeting this freshwater demand. These coastal aquifers (aquifers hydraulically connected to the sea) are pumped heavily to meet this demand for freshwater. The natural flow of groundwater is generally toward the sea, thus preventing the flow of saltwater toward the coastal aquifer. Due to heavy pumping, this natural gradient of flow gets disturbed, which allows the saltwater from the sea to flow toward the aquifer, thus causing saltwater intrusion. Once saltwater intrudes into a coastal freshwater aquifer, the reclamation of such aquifers is economically imprudent. Therefore, it is essential to develop a sustainable management policy for preventing saltwater intrusion in coastal aquifers.

The primary pollutant in the case of the coastal aquifer is saline water from the sea, which is already present near the coast. There is a natural balance wherein the lighter freshwater overlays the denser saline water forming a wedge between the two (Fig. 17.1). A mixing zone is present between saltwater and freshwater interface referred to as the transition zone or diffusion zone. In this, zone density of water gradually increases from the density of freshwater to density of saltwater (Shamir et al. 1984). The equilibrium between saltwater and freshwater is maintained due to the natural decreasing gradient of water table toward the sea (Fig. 17.1). This natural gradient is disturbed due to excessive pumping of the groundwater often bringing the water table below the mean sea level (MSL), causing a reversal of the gradient (Fig. 17.2). This allows movement of the denser saltwater from the sea toward the pumping wells. When saltwater reaches the pumping wells, saltwater along with freshwater gets pumped out. This phenomenon is called saltwater intrusion (Bear 2012). If the salt concentration in the pumped water reaches 1% of the total density of the water, then pumped water is deemed unsafe for human consumption without prior treatment (WHO 2004). Such a scenario is highly undesirable, and reversing

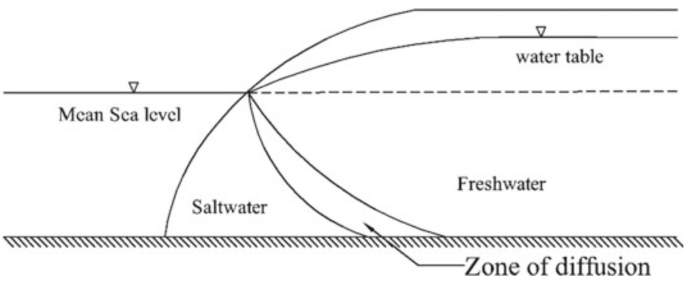

Fig. 17.1 Schematic vertical cross sections of unconfined coastal aquifer in natural condition

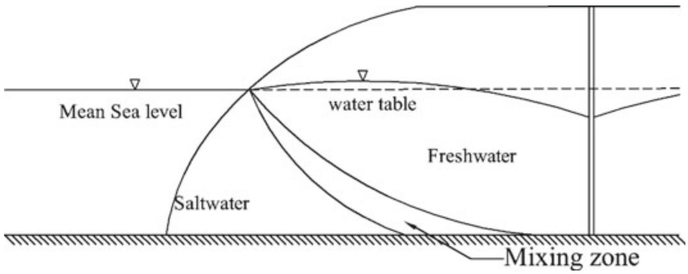

Fig. 17.2 Schematic vertical cross sections of unconfined coastal aquifer while pumping

the impact or reclamation of aquifers polluted by saltwater intrusion is economically unviable.

Thus, the primary challenge in the management of coastal aquifers is to pump large amounts of freshwater from the coastal aquifers to meet the freshwater demands of the growing population, without causing saltwater intrusion. Saltwater intrusion is first detected at an arbitrary water supply well or a group of wells installed for pumping freshwater from the coastal aquifers (Sreekanth and Datta 2015). By the time, saltwater intrusion is detected, and the aquifer is already polluted from the saltwater from the sea. Continuous spatiotemporal monitoring is required for on-time detection of saltwater intrusion. However, long-term spatiotemporal monitoring along with long coastlines is overlooked because of budgetary requirements. Other measures adopted to prevent saltwater intrusion is by hindering the movement of saltwater by the construction of physical barriers along with the coast and increasing water table with the help of artificial recharge of freshwater (Todd 1959).

Management of coastal aquifers requires a proper understanding of saltwater intrusion problem. The movement of the saltwater interface can be understood by developing a density–salinity relationship which is affected by small-scale factors, i.e., beach-scale dynamics (Carey et al. 2009). The density–salinity relationship in saltwater intrusion was first observed by a French teacher named Du Commun (1828), but Ghyben (1888) and Herzberg (1901) provided a mathematical equation relating density and salinity to determine the location of the saltwater–freshwater interface in a coastal aquifer. The relationship given by Ghyben–Herzberg assumed saltwater and freshwater as two immiscible liquids separated by a sharp interface. However, in actual field condition, a mixing zone is present between saltwater and freshwater where the density of water in the mixing zone increases from that of freshwater to saline water (Fig. 17.1). Models using a sharp interface concept are referred to as sharp interface models, whereas the models which consider the variation of density in the mixing zone are classified as density-dependent models (Dagan and Bear 1968).

Various analytical and numerical techniques have been developed for solving both sharp interface and density-dependent model (Barlow and Reichard 2010). In recent times, with increased computational power, complex numerical techniques of solution have gained popularity. These numerical simulation models are used in predicting the saltwater intrusion for different pumping scenarios and thus come up with a sustainable pumping strategy. As there is an infinite number of pumping scenarios possible, sequentially evaluating all possible pumping management scenarios is impractical or unrealistic. Thus, optimization algorithms are linked with simulation models to find an optimal pumping strategy. Optimization algorithm performs an organized search within possible scenarios, iteratively solving the simulation models to find optimal pumping scenario and corresponding location of saltwater interface (Das and Datta 1999). Every iteration of the density-dependent simulation models is computationally taxing (Kopsiaftis et al. 2019), and several iterations of the simulation model within the optimization model may render the linked simulation optimization method inefficient.

This inefficiency of the linked simulation model is overcome by the use of surrogate as simulation models within the optimization framework (Datta et al. 2014). However, surrogate models are computationally efficient but require large datasets for training and validation. In most practical scenarios of saltwater intrusion, such large datasets are seldom available. However, considerable work has been done in the management of coastal aquifers against saltwater intrusion; there are still gaps which need to be addressed. This study attempts to put in perspective the various steps involved in coastal aquifer management, i.e., identification of the saltwater intrusion, prediction of saltwater–freshwater interface, and management of the saltwater intrusion.

17.2 Identification of the Saltwater Intrusion

Chance of saltwater intrusion increases with heavy unregulated pumping of freshwater from coastal aquifers. The first necessary step in any coastal aquifer management is to assess the current state of the aquifer and check if the aquifer is already affected by saltwater intrusion. Identification of saltwater in groundwater aquifers constitutes of two main parts, i.e., determining salt content and finding the origin of the salt in the groundwater samples. Determining the origin of the salt in groundwater is vital as non-costal aquifers can also exhibit high concentrations of salt. Thus, not all salt content in a groundwater aquifer can be attributed to saltwater intrusion. Different geophysical information and geochemical indicators are used in conjunction with Geographical Information System (GIS) to determine saltwater intrusion. Based on the type of data used, the saltwater intrusion identification can be broadly classified as: (1) geochemical techniques (analysis of water quality data); (2) vertical electrical sounding techniques (analysis of resistivity data); and (3) combined approach (analysis of both water quality and resistivity data).

17.2.1 Geochemical Technique

Analysis of water quality data plays a vital role in identifying the origin and concentration of the salt content. Collected water samples from wells are analyzed for total dissolved solids (TDS), electrical conductivity (EC), pH, anions $\left(Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}\right)$, and cations $\left(Cl^{-}, HCO_3^{-}, SO_4^{2-}, NO_3^{-}, PO_4^{3-}\right)$. The concentration of cations and anions, TDS, pH, and EC in the sampled water is analyzed to determine the salinity and origin of the sampled well water. TDS is a good indication for determining the presence of salt in the water. Seawater generally contains about 3.5% salt (Bear 2012), i.e., about 35 g of salt dissolved in one liter of water. Due to this high salt concentration, observed TDS value for seawater ranges between 10,000 and 100,000 mg/l but generally TDS for freshwater is below 1,000 mg/l. TDS value between 1,000 and 10,000 mg/l is considered brackish water (Fetter 2000; Konikow and Reilly 1999; Rao et al. 2017). Therefore, TDS greater than 1,000 mg/l is considered as freshwater affected by salt. Ebraheem et al. (1997) first showed the applicability of TDS in the northern part of the Nile Delta of Egypt by calibrating the geophysical information with the help of TDS. The above-mentioned method of calibration of the geophysical information with the help of TDS was also performed by Khalil (2006), Sherif et al. (2006), Song et al. (2007), Cimino et al. (2008), and Rao et al. (2011) in Abu Zenima area, West Sinai, Egypt, Wadi Ham, UAE, Byunsan, Korea, Northern Sicily, and Godavari Delta Basin, India, respectively. Apart from the calibration of the geophysical information TDS is also used to develop a simulation model based on the sharp interface concept by Ranjbar and Ehteshami (2019). High level of TDS also can be observed in groundwater due to the high concentration of any pollutant (Huang et al. 2019; Jung 2001; Radfard et al. 2019; Shyamala et al. 2008). Therefore, it is challenging to use TDS as a sole identifier for saltwater intrusion, which is the reason TDS should be used only in conjunction with other geochemical methods to identify the presence of the saltwater in the coastal aquifer.

A plot of cations' and anions' concentrations on a piper diagram helps to identify the origin and concentration of salt in groundwater samples. Piper diagram helps to classify the waters sample based on the concentration of $CaHCO_3$, $NaCl$, $CaNaHCO_3$, $CaMgCl$, $CaCl$ and $NaHCO_3$ in the water samples (Hoyle 1989; Kelly 2005). As the concentration of the NaCl in saltwater from the sea is very high, therefore a plot of constituents of the water samples on the piper diagram can easily identify the salt concentration and origin of the salt. The above method was applied by Arslan et al. (2012) on Bafra Plain, Turkey, in conjunction with other geochemical indicators for identifying saltwater intrusion. Same approach was used by Wen et al. (2012) in Eastern Laizhou Bay of China, Sola et al. (2013) in Andarax Delta of Spain, Kumar et al. (2014) in the southern portion of Chennai City in India, Askri (2015) in Al Musanaah in Sultanate of Oman, Hamzah et al. (2017) in Terengganu of Malaysia, Sarker et al. (2018) of southwest Bangladesh, and Chidambaram et al. (2018) in Cuddalore District of Tamil Nadu in India. Though piper diagram is a very good identifier of saltwater intrusion, it does not consider the secondary ions in the sample (Pulido-Leboeuf 2004). Also, the use of concentration as a percentage often overlooks the actual level of salinity (Singhal and Gupta 2010). Above-mentioned difficulties were rectified by Tomaszkiewicz et al. (2014) using piper diagram to develop a new method named GQI_{SWI}. The advantages of the GQI_{SWI} will be discussed in the later part of this section.

Another handy indicator for the identification of origin and location of salt is cluster analysis of cations and anions in water samples. In cluster analysis, cations and anions are grouped to find a correlation between them and plotted on a dendrogram. To detect salt in the water, sample cluster analysis was performed by Arslan (2013), Triki et al. (2014), and Askri (2015).

Chlorine (Cl^-) concentration is also an important indicator of saltwater intrusion as saltwater from the sea contains a very high concentration of NaCl. Therefore, the plot of cations and anions especially Br^+ and Na^+ against chlorine concentration helps to categorize the water sample for saltwater intrusion (Sola et al. 2013). Cl^- concentration is also used to calculate the seawater fraction, which is a ratio between chlorine concentration difference in sample water–freshwater and seawater–freshwater. Seawater fraction is extensively used to identify saltwater intrusion by Arslan et al. (2012), Askri (2015), Mondal et al. (2011a), Mondal et al. (2011b), Sola et al. (2013), Tomaszkiewicz et al. (2014), Wen et al. (2012). However, seawater fraction does not identify the main cation exchange that happens due to saltwater intrusion (Appelo and Postma 2004). Generally, saltwater contains a high level of Na^+; therefore, sodium adsorption ratio (SAR) coupled with seawater fraction is used to identify the origin of salt in groundwater samples. High SAR value (>26) is used to identify the presence of a high level of Na^+ for the water sample (Arslan 2013; Kumar et al. 2014; Mondal et al. 2011a; Rao et al. 2017; Triki et al. 2014).

The dominant feature in seawater is the high level of concentration Na^+ and Cl^- which can be easily identified by a multivariate technique called principal component analysis (PCA) (Anderson 1958; Sharma 1995; Singh et al. 2005). PCA helps to identify the dominant components in water samples by establishing a correlation between components of the water samples (Arslan 2013; Askri 2015; Kumar et al. 2014; Mondal et al. 2011b; Triki et al. 2014). Apart from PCA analysis, groundwater quality index (GQI) is also done on various dominant ions present in the water sample. Generally, GQI is calculated from the ratios of various cations and anions in the water sample. Among various GQIs, $GQI_{Piper(mix)}$ is used to classify between freshwater and saltwater and $GQI_{Piper(dom)}$ used to identify high Ca, Cl and $NaHCO_3$ concentrations in water. The concentration of the Na^+, K^+, and HCO_3^- is used to calculate $GQI_{Piper(dom)}$, and the concentration of Ca^{2+}, Mg^{2+}, and HCO_3^- is used to calculate $GQI_{Piper(mix)}$ (Hussien 2015). As discussed, the dominant constituents of saltwater from the sea are of Cl and Na which is not properly described in $GQI_{Piper(mix)}$. Therefore, Tomaszkiewicz et al. (2014) developed a new index named GQI_{SWI}, which is based on $GQI_{Piper(mix)}$ and $GQI_{Piper(dom)}$. GQI_{SWI} reduces difficulties in piper diagram and seawater faction.

The main objective of the geochemical method is to find the relation between two dominant fractions in the water samples. Ionic ratios of cations and anions such as Ca/Mg, Cl/HCO_3, Na/Cl are used for identifying saltwater in the water samples.

The ionic ratio of Ca/Mg, Cl/HCO_3, Na/Cl was used by Hamzah et al. (2017), Cl/Br and Cl/HCO_3 ratio by Sarker et al. (2018), and Cl/HCO_3 ratio by Chidambaram et al. (2018) to identify saltwater intrusion. Stable isotopes of oxygen (O^{18}) and hydrogen (H^2) are also suitable identifiers of origin of saltwater. The isotopic ratios of water sample are compared with international standards for identifying origin and the presence of saltwater in the groundwater (Kanagaraj et al. 2018; Sarker et al. 2018; Sola et al. 2013). Some other notable methods for identifying the origin of saltwater are Gibbs diagram (Rao et al. 2017) and Chadda's hydrographical process plot (Chidambaram et al. 2018). Geochemical methods are applied successfully to understand the presence and origin of saltwater intrusion. Being point source data, geochemical methods are useful in identifying only the horizontal distribution of the saltwater intrusion. It is difficult to quantify the vertical distribution of salt in the water sample using geochemical methods. It is also economically expensive and time-consuming.

17.2.2 Vertical Electrical Sounding

The shortcomings mentioned above are easily tackled by vertical electrical sounding (VES). In the case of VES, a low voltage direct current is passed through the ground by an array of electrodes and resulting change in potential due to the resistivity of the medium is measured via a micro-ohmmeters. Analyzing change in resistivity with different arrays of the electrodes provides information regarding properties of earth crust like density, magnetism, elasticity, and electrical resistivity (Todd 1959). Saltwater is a good conductor of electrical current; therefore, aquifers bearing saltwater pose a very low resistivity (0.2 Ω m) which clearly distinguish saltwater bearing medium from the surrounding area. Swartz (1937) first used VES to identify saltwater intrusion in Hawaiian Islands. Subsequent work by Hallenbach (1953), Flathe (1955), Dam and Meulenkamp (1967), Ginsberg and Levanton (1976), El-Waheidi et al. (1992) showed the applicability of 1-D resistivity method to identify the presence of saltwater in the aquifer.

In the case of 2-D or 3-D models, large array of electrodes are used and computer-driven algorithms automatically measure the change in resistivity by passing direct current to different electrodes. By analyzing different resistivity of different orientations of electrodes, a 2-D or 3-D image is produced. This system is also known as electrical resistivity tomography (ERT) (Werner et al. 2013). ERT was first used in coastal areas by Acworth and Dasey (2003) in a tidal creek of New South Wales, Australia, to identify mixing zone of saltwater with infiltrated rainwater. Day-Lewis et al. (2006) used ERT to identify submarine groundwater discharge. Comte and Banton (2007) used ERT images to validate the variable-density flow model SUTRA. Satriani et al. (2011) used ERT to identify the location of saltwater in coastal areas of Ionian Plain of Southern Italy. Belaval et al. (2003) and Manheim et al. (2004) used ERT method to identify the presence of freshwater under aquifers in offshore waters of South Carolina, Massachusetts, and Delmarva Peninsula (USA).

ERT systems are useful to find the distribution of the saltwater along with the coastal aquifers. The dynamics of saltwater can be observed by time-lapse electrical resistivity tomography (TERT) method. TERT is a modified version of ERT where an array of electrodes is permanently placed in the study area and resistivity information in the specified interval is measured. De Franco et al. (2009) and Poulsen et al. (2010) used TERT system in southern Venice Lagoon and coastal dune in Denmark to identify the behavior of saltwater with rainfall, tidal inundation, and channel stages. An automated TERT system was developed by Ogilvy et al. (2009) to understand saltwater dynamics on lower catchment of river Andarax, Almeria, Spain.

In resistivity method, electrodes are taped on the ground surface. Therefore, a physical ground connection via the electrodes is required to be set up to record the subsurface information, which is time-consuming. In electromagnetic (EM) method, electrodes are laid over the ground surface and do not require any physical ground connections to operate. This results in speedy operation, which increases the popularity of EM methods in recent times. EM methods are also able to detect a better local and near-surface inhomogeneity compared to resistivity method. Besides, EM methods outperform resistivity methods in desert and permafrost conditions (Goldman et al. 1991). EM methods are mainly of two types, i.e., time domain and frequency domain. Time-domain EM methods have been widely used for several decades; but in recent times, frequency-domain EM is gaining popularity (Stewart 1982). Goldman et al. (1991) first used time-domain EM (TDEM) method for detecting saltwater interface in Mediterranean coastal strip of Israel. TDEM was also used by Melloul and Goldenberg (1997) in coastal areas of Israel and by Al-Sayed and El-Qady (2007) in southwestern part of Sinai Egypt. As EM device does not require any physical ground connection, airborne EM measuring system has been developed. Airborne EM measuring system was used for saltwater intrusion by Paine (2003) in Red River Basin in Texas, Fitterman and Deszcz-Pan (1998) in Everglades National Park, Florida, and Viezzoli et al. (2010) in Venice coastal lagoon. The airborne EM system is cost-effective and can be deployed over a large area very quickly.

17.2.3 Combined Method

VES methods are used extensively to determine the location of the saltwater interface. As the observation of salinity is performed through resistivity information, therefore, any other low resistivity geological strata like ore mines, sedimentary rocks may give the same resistance signal as saltwater bearing strata. Therefore, validation of the geophysical data with some physical data is of paramount importance, which is the reason behind the application of VES method in conjunction with a geochemical analysis. Ebraheem et al. (1997) first tried to determine exact resistivity with the help of TDS data in the Nile Delta Region. TDS was also used by Khalil (2006), Sherif et al. (2006), Rao et al. (2011) to calibrate the electrical resistivity data.

Ratios of cations and anions provide comprehensive information about the occurrence and distribution of the salt in groundwater. Song et al. (2007) showed the applicability of ionic ratios to calibrate resistivity survey data in Byunsan, Korea, using ratio of chloride to bicarbonate and calcium to sodium ions. Bromide ion index, ratio of chloride ion to bicarbonate-plus-carbonate ions and ratio of strontium to chloride ion, was used to calibrate resistivity data by Shahsavari et al. (2015). Cimino et al. (2008) determined saltwater through PCA of water samples and used this information to calibrate VES data. As salt in saline water mainly comprises of sodium chloride, chlorine concentration also plays a vital role in determining the presence of saltwater. Samsudin et al. (2008) used chlorine concentration to calibrate VES data to determine the location of the saltwater interface in Kelantan, Malaysia. Stable isotopes of hydrogen and oxygen were used to calibrate VES data and identify the extent of the saltwater intrusion in Chennai, Tamil Nadu, India, by Kanagaraj et al. (2018).

Well log data can provide useful information about the vertical and horizontal distributions of the salinity. Well log data was first used by Edet and Okereke (2001) in Southern Nigeria to calibrate VES data. Hermans et al. (2012) applied this technique in Westhoek, Belgium, Shahsavari et al. (2015) in Kharg Island in Iran, and García-Menéndez et al. (2018) in Spanish Mediterranean coast.

17.3 Prediction Models

The next step of developing a sustainable management model for the coastal aquifer is to use spatiotemporal salt concentration/resistivity data and to assess the extent of saltwater intrusion in the coastal aquifer. Transport of dissolved salts in groundwater is governed by various physical processes such as advection, hydrodynamic dispersion, and density gradient. Numerical models are used to simulate these processes in coastal aquifer. These numerical models are solved forward in time with appropriate initial condition and boundary condition to estimate the extent of saltwater intrusion and to predict the fate and transport of the salt under various pumping stresses.

Most of the salt in groundwater aquifer is transported by advection, which essentially is the carrying of dissolved salt along with the flow of groundwater. The flow of groundwater is caused due to the naturally existing seaward pressure gradient, causing the freshwater to flow from the land toward the sea. The zone where the fresh groundwater meets the saline groundwater, the pressure is constant. This is also referred to as the freshwater–saltwater interface. The movement of this interface is calculated based on the change in pressure throughout the aquifer. Most numerical models can be broadly classified as sharp interface model or density-dependent model. In sharp interface, saltwater and freshwater are considered as two immiscible fluids. Therefore, a sharp interface is present between saltwater and freshwater (Fig. 17.3); but in the actual scenario, a mixing zone is present (Fig. 17.4) which is better described by density-dependent models. Sharp interface models are less accurate compared to density-dependent models as they do not consider mixing between saltwater and freshwater due to hydrodynamic dispersion. Density-dependent model combines flow with transport due to advection and hydrodynamic dispersion making

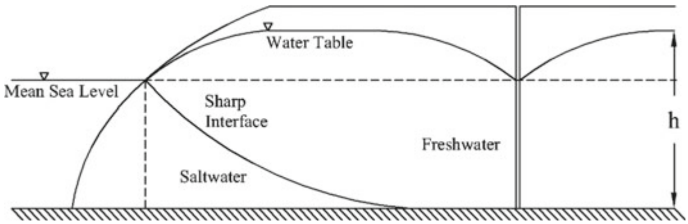

Fig. 17.3 Illustration of sharp interface of an unconfined aquifer

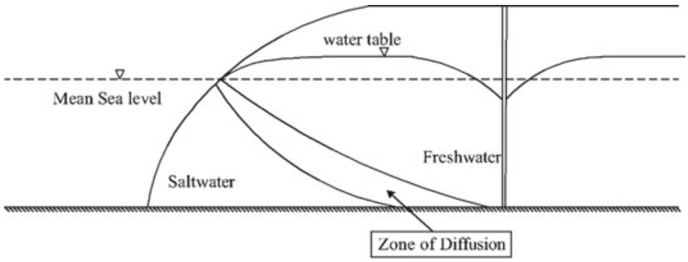

Fig. 17.4 Illustration of a density-dependent model for unconfined aquifer

it more realistic as well as nonlinear and computationally expensive to solve. The solution technique of these simulation models can be broadly classified as graphical methods, analytical methods, and numerical methods.

17.3.1 Graphical Method

Henry (1959) first developed a graphical solution based on sharp interface for single homogeneous aquifer and was first to conceptualize a single velocity potential (φ) for a two-dimensional flow system. This was further extended by Bear and Dagan (1964). Bear and Dagan (1964) introduced the concept of upcoming in saltwater intrusion process. These methods may be applied to very small homogeneous aquifers. However, nonlinearity caused due to change in boundary conditions cannot be handled by these approaches.

17.3.2 Analytical Solution

Dagan and Bear (1968) first developed exact equations for solving saltwater intrusion based on sharp interface model. The developed equations were nonlinear, so Dagan and Bear (1968) used perturbation technique to solve the problem. Haatush

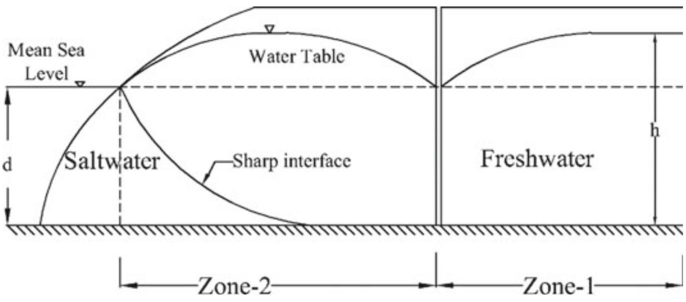

Fig. 17.5 Illustration of an unconfined coastal aquifer according to Strack (1976)

(1968) extended this method for cases where sharp interface between saltwater and freshwater does not reach the bottom of the aquifer. The developed partial differential equation is nonlinear, which only gives an approximate solution of the model. Schmork and Mercado (1969) gave experimental proof supporting this method. However, while incorporating boundary conditions, the above method became nonlinear leading to inaccuracies for large areas.

Instead of deriving the interface equation, Strack (1976) first developed a single potential concept, which reduces nonlinearity. The main advantage of using velocity potential (φ) lies in the fact that the solution can be written as one expression. Besides, Strack (1976) potential can handle three-dimensional problems. Instead of using the whole aquifer as a single system, Strack (1976) divided aquifer into two parts (Fig. 17.5) based on the presence of saltwater and freshwater. In zone 1, the only freshwater is present, and in zone 2, both denser saltwater and freshwater are present. The single potential is continuous throughout the aquifer. Strack (1976) method was extended for layered aquifer by Mualem and Bear (1974). Subsequently, Collins and Gelhar (1971) developed analytical solutions for the layered aquifer.

The inherent problem using sharp interface approach is that, it overestimates the location of the interface. This problem is overcome by the use of density-dependent model-based analytical solutions. Paster et al. (2006) developed a steady-state solution for density-dependent model using boundary layer approach. This was further extended to handle 3-D problem for field applications by Paster and Dagan (2008). The applicability of Strack (1976) approach was enhanced by Pool and Carrera (2011) with the introduction of density in Strack (1976) potential.

17.3.3 Numerical Solution

17.3.3.1 Numerical Solution for Sharp Interface Models

Mercer et al. (1980a) used finite difference technique to solve partial differential equation based on mass and momentum transfer for a two-dimensional single layer aquifer. Further, the model was extended for the three-dimensional problem and integrated with USGS's modular hydrogeological flow model (MODFLOW) by Mercer et al. (1980b). In this case, finite difference scheme was used to solve the partial differential equation. The main drawback of this method was that it could only locate the toe of the interface instead of locating the entire interface. In addition, it lacked stability in convergence. These limitations were overcome by Polo and Ramis (1983) who used a backward difference scheme to solve the problem. However, the developed methods were time-intensive in terms of computation and data preparation. These limitations were overcome by used of boundary integral equation method by Taigbenu et al. (1984) in a Strack potential approach. Mantoglou et al. (2004) solved this method with the help of the finite difference method.

Strack potential-based approach could not handle multilayered aquifer system. Essaid (1990) overcame this problem by solving the partial differential equation given by Mualem and Bear (1974). Authors used central in space and backward in time scheme to solve the partial differential equation. Huyakorn et al. (1996) used the Newton–Raphson method, and Guvanasen et al. (2000) used finite element method to solve the equation.

17.3.3.2 Numerical Solution for Density-Dependent Models

Sharp interface approach generally overestimates the extent of saltwater intrusion as it does not consider hydrodynamic dispersion causing mixing of saltwater and freshwater. This is overcome by the use of flow and transport equations coupled with density coupling term, which increases the complexity in simulation. Pinder and Cooper (1970) first solved Henry's problem for the transient condition using the method of characteristics. Volker and Rushton (1982) used finite difference scheme for estimating saltwater intrusion for a two-dimensional system. Guo and Langevin (2002) developed a three-dimensional finite difference model called SEAWAT to solve the coupled equation. This model combines MODFLOW and MT3DMS into a single program that solves the coupled flow and solute transport equations. This is one of the most widely used and is an efficient density-dependent model.

Lee and Cheng (1974) solved a steady-state two-dimensional problem using finite element-based triangular technique. The above method was solved using Galerkin finite element by Segol (1975) dominated by advective transport. Segol and Pinder (1976) used the above method in an actual field area. Huyakorn and Taylor (1976) presented a model using reference hydraulic head and concentration as dependent variables. Kawatani (1980) presented density-dependent model saltwater intrusion process for an aquifer–aquitard system, which was solved using Galerkin finite element technique by Frind (1982). Voss (1984) developed finite element-based Saturated–Unsaturated TRAnsport (SUTRA) code for the simulation of saltwater intrusion process in two-dimensional coastal aquifers. Galeati et al. (1992) used implicit Eulerian–Lagrangian method to solve the density-dependent model. Lin et al. (1997)

developed three-dimensional finite element-based model, FEMWATER that combines density term with transport term. Sherif et al. (1988) developed finite element model for a leaky confined aquifer.

17.4 Management Model

Sustainable use of water resources requires design and implementation of well-planned management strategies. One of the challenges in management of costal aquifer is to pump a large amount of freshwater without causing saltwater intrusion, which is conflicting in nature. When a coastal aquifer is pumped heavily, water table goes below MSL and saltwater from the sea comes near pumping wells. Therefore, the prime management challenge is to design some barriers or raise water table above MSL such that saltwater does not reach the pumping wells. The barrier can be either a physical barrier or pumping barrier. In addition, the water table can be increased above mean sea level through artificial recharge of freshwater. As a management alternative, pumping of freshwater can be limited such that the saltwater–freshwater wedge never reaches the pumping locations. Most of the above methods for the management of saltwater intrusion can be used in conjunction. Creating physical barriers or artificial recharge along shorelines is economically imprudent, and often the selection of management strategy depends on the availability of funds. Thus, pumping barriers are the most commonly implemented technique used for the management of saltwater intrusion.

Prediction models are used to predict the location of saltwater for different sets of input conditions. The simplest way to find a suitable management alternative is by sequentially evaluating all the possible scenarios, which is unrealistic and impossible. To overcome this problem, numerical simulation models are linked with optimization algorithms. Optimization algorithm performs an organized search for new and improved pumping schemes or other management alternatives. During this search process, the simulation model is run many times to quantify the impact of the pumping on the movement of the saltwater front. Depending upon the type of incorporated simulation model in optimization algorithm, it is classified as: (1) embedding approach, (2) linked simulation optimization approach, (3) surrogate modeling approach, and (4) combined approach using density-dependent model and sharp interface model.

Initially, classical linear and nonlinear optimization techniques were used to solve most of the optimization problems. Aguado and Remson (1974), Alley et al. (1976), Willis and Finney (1988) used a classical optimization algorithm to sequentially evaluate all possible pumping combinations. The main disadvantage of classical methods is that they may converge to a sub-optimal or near-optimal solution. Heuristic optimization techniques such as genetic algorithm (GA), simulated annealing (SA) have gained popularity in recent times due to their capability to find the optimal global solution.

17.4.1 Embedding Techniques

In the embedding technique, the simulation model is incorporated within an optimization model where the flow and transport equations are embedded in the optimization code as binding constraints. An initial attempt at embedding techniques had applied by Aguado and Remson (1974), where a linear response matrix was used to approximate the response of saltwater against pumping. Applicability of embedding technique is shown by Alley et al. (1976), Willis and Finney (1988), Culver and Shoemaker (1993), Das and Datta (1999), Abd-Elhamid (2010), Aguado and Remson (1974), and Alley et al. (1976) using linear programming (LP) in both steady-state and transient conditions. Willis and Finney (1988) first used nonlinear programming techniques and applied the method on an actual field condition. Culver and Shoemaker (1993) used differential dynamic programming with quasi-Newton approximations (QNDDP) and finite element method (FEMWATER) to determine optimal time-varying pumping. Das and Datta (1999) first used nonlinear programming to solve a multi-objective management model. Abd-Elhamid (2010) used the GA to determine the optimum way to recharge freshwater in coastal areas. Embedding techniques are useful for a small-scale problem, but for a large-scale problem, the convergence of the optimization model is the biggest challenge.

The response matrix is based on the principle of superposition and linearity. A linear response matrix is developed by multiple runs of external numerical simulation model prior to the start of optimization. Applicability of this approach on groundwater management was observed by Deininger (1970), Maddock III (1972), Maddock (1974), Rosenwald and Green (1974), Heidari (1982), Willis (1984), Hallaji and Yazicigil (1996), and Kourakos and Mantoglou (2015). Hallaji and Yazicigil (1996) used this approach to the coastal aquifer management problem. Hallaji and Yazicigil (1996) considered seven different groundwater management models to determine the optimal pumping policy for a coastal aquifer in southern Turkey threatened by saltwater intrusion.

17.4.2 Linked Simulation Optimization Approach

In the linked simulation optimization technique, the simulation model is linked with the optimization algorithm externally. The simulation model acts as binding constraint such that any candidate solution from the optimization has to satisfy these constraints. Cheng et al. (2000) show the applicability of the method by linking heuristic optimization, i.e., GA with the analytical model by Strack (1976). Analytical model by Strack (1976) was used by Mantoglou (2003) and Park and Aral (2004) in a linked simulation optimization approach. Mantoglou (2003) used sequential quadratic programming as the optimization algorithm, and Park and Aral (2004) used GA.

Numerical simulation model of sharp interface approach is linked with an optimization algorithm by Willis and Finney (1988), Finney et al. (1992), Emch and Yeh (1998), Mantoglou et al. (2004), Reichard and Johnson (2005), Karterakis et al. (2007). Mantoglou et al. (2004), and Karterakis et al. (2007) optimized pumping and well locations, whereas others used this model to optimize pumping only. Linear programming was used by Reichard and Johnson (2005) and Karterakis et al. (2007). Various heuristic optimization techniques such as quadratic programming (Willis and Finney 1988), projected Lagrangian algorithm MINOS (Emch and Yeh 1998; Finney et al. 1992), sequential quadratic programming (Mantoglou et al. 2004), and evolutionary algorithms (EA) (Karterakis et al. 2007) were used in a linked simulation optimization approach.

17.4.3 Surrogate Models

Density-dependent models can accurately simulate the physical behavior of saltwater intrusion, but due to the nonlinear nature of the density-dependent models, the CPU runtime for a density-dependent simulation model is very large. Due to the computational burden, it is unrealistic to apply density-dependent model linked with an optimization algorithm. To overcome this, problem density-dependent models were replaced entirely with a pattern recognizing algorithm like artificial neural network (ANN). Rao et al. (2003) first showed possibility of ANN as a surrogate in saltwater intrusion problem. ANN as a surrogate is used by Rao et al. (2004), Bhattacharjya and Datta (2005), Kourakos and Mantoglou (2006), Nikolos et al. (2008), and Ataie-Ashtiani et al. (2013). To develop the surrogate, ANN models are trained with pumping as input and saltwater concentration in observation as wells as output. Density-dependent models are used for generating training datasets. Heuristic optimization such as SA was used by Rao et al. (2003), Rao et al. (2004) as optimizer. GA was used by Bhattacharjya and Datta (2005), and Ataie-Ashtiani et al. (2013); sequential quadratic programming was used by Kourakos and Mantoglou (2006), and differential evolution (DE) algorithm was used by Nikolos et al. (2008).

A large number of datasets are required to fit a surrogate model. Generating data using density-depended model is very tiresome (Rao et al. 2004), as the CPU runtime for a density-dependent simulation model is very large. To reduce CPU runtime problem, Dhar and Datta (2009) used an undertrained ANN linked with non-sorting genetic algorithm-II (NSGA-II) as a screening model (metamodel), before linking NSGA-II with a density-dependent model. Another approach to overcome this problem had been addressed by Kourakos and Mantoglou (2009), where the authors used modular neural network (MNN) such that each module are trained for a different decision variable. Kourakos and Mantoglou (2009) used evolutionary annealing-simplex (EAS).

Genetic programming (GP) model was used as a surrogate by Sreekanth and Datta (2010; 2011). As predictions using a single surrogate model may sometimes show eccentric results, thus an ensemble of surrogates based on different realization, i.e., each surrogate is efficient for a segment was developed by Sreekanth and Datta (2010). Sreekanth and Datta (2010) used the nonparametric bootstrap method to generate different realizations. To calculate an optimal number of surrogate models, root-mean-square error (RMSE) of each surrogate was computed, and the coefficient of variation of these RMSE was considered as the measure of uncertainty in the ensemble of the models. Authors compared this result with ANN-based surrogates (Sreekanth and Datta 2011) and reported that this innovative method reduces the computational burden in training the ANN. Authors also applied this method on an actual area to showed the applicability of this method (Sreekanth and Datta 2014).

In the same procedure to reduce the training difficulties, Roy and Datta (2017) used fuzzy logic-based adaptive neuro-fuzzy inference system (ANFIS). The main advantage of fuzzy logic is that instead of using multiple surrogate models for multiple output problems one surrogate is efficient for all regions of solution. Other notable works on surrogate models have been performed by Bhattacharjya and Datta (2009), Papadopoulou et al. (2010), Ataie-Ashtiani et al. (2013), Kourakos and Mantoglou (2013), Hussain et al. (2015), Christelis and Mantoglou (2016b), Christelis et al. (2018), and Lal and Datta (2019).

17.4.4 Combined Approach Using Density-Dependent Model and Sharp Interface Model

Due to its linear nature, sharp interface models are less complicated compared to density-dependent models (Cheng et al. 2000). However, the simplification of the sharp interface model leads to an overestimation of the extent of the saltwater–freshwater wedge (Dausman et al. 2010; Dokou and Karatzas 2012). Results of the numerical model by the sharp interface in the estimation of the saltwater–freshwater wedge can be more realistic if dispersion can be included within the sharp interface model. Sharp interface model is a function of head and density ratio. The head is affected by pumping; therefore, modification of density ratio in sharp interface model based on dispersion such that outcomes of the sharp interface model will be similar as outcome of the density-dependent model (Pool and Carrera 2011). This idea was further extended by Christelis and Mantoglou (2016a), where density ratio is modified with the help of density-dependent model within an optimization framework. Christelis and Mantoglou (2016a) designed density ratio such that the location of sharp interface and iso-salinity contour (calculated by density-dependent model) will match.

17.5 Conclusions

Sustainable exploitation of coastal aquifer requires pumping out a large amount of freshwater keeping the saline water at bay. Managing saltwater intrusion is a complex task due to the conflicting nature of these objectives. A successful management strategy comprises of early detection of saltwater intrusion, predicting the extent and future course of intrusion. Though a majority of the identification technique focuses on the concentration of the salt ions and their origin, coupled approach using geophysical and geochemical approach proves to be better. New detection techniques based on EM method are easy and fast to apply. Airborne EM systems need to be explored more as they can produce reliable results over vast areas in a short span of time.

Predicting the freshwater–saltwater boundary accurately is a major challenge in any saltwater management scenario. Both sharp interface models and density-dependent model predict the boundary but have their own limitations. Sharp interface models are quick to simulate but they overestimate the intrusion. Though density-dependent models are more accurate, it consumes an enormous amount of CPU time in the simulation. Thus, direct use of density-dependent models in a linked simulation optimization is not preferred as a management technique. Surrogate models can be a viable alternative to the use of density-dependent model but require extensive data. MNN and fuzzy logic-based ANN may overcome some of the issues with the density-dependent models but still require training.

To overcome these difficulties, the combined use of density-dependent model and sharp interface model is applied. Capability of fast simulation speeds from sharp interface models, and accuracy of prediction in the density-dependent model is used simultaneously in this methodology. Still considerable work needs to be done for further application of multi-objective problems. Computational burden for seawater intrusion management models is still a challenge, especially when linked to a heuristic optimization algorithm in saltwater intrusion management problems. Use of classical optimization often leads to sub-optimal results. Therefore, further work needs to be done to overcome these limitations.

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