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Assessment of Groundwater Vulnerability to Seawater

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United Arab Emirates University
College of Graduate Studies

Assessment of Groundwater Vulnerability to Seawater Intrusion in the Coastal Aquifer of Wadi Ham, UAE

A Thesis submitted to the
College of Graduate Studies

by

Ayeda Matar Al-Hosani

In partial fulfillment of the requirements for the degree of
Master of Science in Water Resources

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

Page

CHAPTER 1
INTRODUCTION

Chapter 1. Introduction

1.1 Importance and Distribution of Water on Earth

The earth might be named as the "water planet" or the "blue planet." Water is essential for life, it is the most precious and irreplaceable resource on Earth. It is estimated that 99.4% (1.4×10^9 km^3) of the total available is surface water. Groundwater occurs only as 0.6% (9×10^6 km^3) of the total. However, of the vast amount of surface water, most of it is in the form of saltwater in oceans and inland seas (97 %). Fresh surface water accounts for only 2% of the total volume of water (Bear et al., 1999). Table 1.1 provides estimates for water availability on earth. Similar numbers have been obtained by Shiklomanov and Rodda (2003).

The fresh surface water resources that are accessible for human consumption are the water in lakes (0.3%) and streams (0.003%). These are dwarfed by the amount of groundwater (22%). As a water crisis is forecasted in the near future (Gleick, 1993), the welfare of the world's population is closely tied to a sustainable exploitation of groundwater. Lakes, rivers, reservoirs, and aquifers account for less than one-third of all fresh water, with the rest locked in glaciers and permanent snow covers (Raskin et al. 1995).

Earth has a limited supply of water and, in most cases, water is considered a renewable resource as it circulates through various parts of the environment. As the human population has expanded and the use water to meet various demands, including domestic, agricultural and industrial demands has increased, there are growing concerns about the availability of usable water. The multitude natural and human uses of fresh water are linked by the unitary character of the water cycle (Rogers and Lydon 1995). The use and misuse of water in one location can have far-flung effects, altering downstream resources, affecting the reliability of water flows, and degrading water quality and aquatic ecosystems. As the competition for limited resources increases with expanding water use, water quality often deteriorates and ecosystem maintenance is compromised.

Freshwater resources in arid and semi-arid lands have three main components: rainfall, surface water, and groundwater. Water that enters the ground and occupies the free space in soil and sediment as well as openings in bedrock including cracks and spaces between the grains is known as groundwater. For humans, groundwater forms an important part of Earth's water supply. Groundwater is usually freshwater, available nearly everywhere on the continents, and it is usually free of organic pollution, disease, and dangerous contaminants. However, groundwater is vulnerable to various source of pollution. It usually takes a long time for the pollutants and other substances disposed of, or spilled on or near the surface, to reach underground systems. Once groundwater is polluted or used up, recovery will be very slow. In some cases, it may not be possible to restore the depleted groundwater systems. Understanding groundwater and the hydrologic cycle may enable to conserve water for future use.

Table 1.1: Global water reservoirs; Mather (1984).

Category	Storage ($10^3 km^3$)	%of total	% of the liquid freshwater
Total global volume of water	1,384,000	100	-
Oceans and salt water lakes	1,350,000	97.5	-
Glaciers and ice caps	25,000	1.8	-
Freshwater:	9,000	0.65	100
a. Groundwater	8,847	0.64	98.3
b. Freshwater lakes	0,126	0.009	1.4
c. Soil moisture	0,0225	0.001	0.25
d. Man-made reservoirs	0,0027	-	0.03
e. Rivers	0,0018	-	0.02

1.2 Physical Conditions and Water Resources in the GCC Countries

The countries of the Arabian Peninsula, with specific reference to the Gulf Cooperation Council (GCC) countries, have similar physiographic, social, and economic characteristics, including arid climates, sparse natural vegetation, and fragile soil conditions. The natural water resources consist of limited quantities of surface water run-off resulting from flood events, groundwater in the alluvial aquifers, and extensive groundwater reserves in the deep sedimentary aquifers which are mainly non-renewable. The supplementary non-conventional sources include desalination of seawater and brackish water, and renovated wastewater. Water availability is governed by rainfall distribution in time and space, in relation to run-off generation, as well as topographic and geological features that influence water movement and storage.

The total area of the GCC countries is estimated as 2 557 470 km^2 (Al-Rashed and Sherif, 2000), Figure 1.1. The peninsula is largely desert with the exception of the coastal strips and mountain ranges. In basic climate terms, a desert can be defined as an area which receives little or no rainfall and experiences no season of the year in which rain regularly occurs (Nicholson, 1995). The climate in the GCC countries is characterized by long, hot, dry summers and short, cool winters for the interior regions, and hot, somewhat more humid, summers and mild winters for coastal regions. Hydrometeorological parameters exhibit great variation, seasonal temperatures may range from -5° to 46°C in the north, central, and eastern parts of the peninsula. The coastal areas and mountainous highlands have lower and less extreme temperatures, ranging from 5° to 35°C. Humidity is generally low in the interior, ranging from 10 to 30 percent. In the coastal areas it may range between 60 and 95 percent. The low percentage of cloudy days and the high solar radiation over the region result in high evaporation rates. The average annual rainfall in the GCC countries varies between 70 to140 mm. The total annual evaporation ranges from 2,500 mm in the coastal areas to more than 4,500 mm in desert of Saudi Arabia (Al-Rashed and Sherif, 2000).

Figure 1.1. Physical setting of the GCC Countries.

The main topographic features of the Arabian Peninsula are the western, southwestern, and south-eastern mountain ridges, as well as the central plateau. The mountain ridges divide numerous moderate-sized drainage basins that empty towards the Red Sea, Arabian Sea, and the Gulf of Oman, as well as larger basins that drain towards the central plateau and, in some cases, continue eastward towards the Gulf. Generally, the coastal drainage basins have steep reliefs and narrow coastal plains as compared with the mild slope and large catchments area of the inland region. Steep slopes and well-defined topographic features control the availability of surface run-off as well as the modes of groundwater recharge. The remainder of the peninsula is characterized by low relief and poor drainage. In the Arabian Gulf region, the Tertiary sediments are made up of limestones, dolomites and evaporites. Although the Oligocene sediments were removed by erosion due to a worldwide drop in sea level, some areas remained submerged and Oligocene sediments crop out in Abu Dhabi and Oman and in the subsurface in off shore United Arab Emirates (Al-Ruwaih and Talebi, 2007).

The other major features that influence the availability of groundwater resources are the peninsulas igneous and metamorphic basement rock known as the "Arabian Shield," and the sequences of sedimentary layers known as the "Arabian Shelf". The shield, which covers one-third of the peninsula, consists of an outcrop of hard rock that begins in the western part of Saudi Arabia and extends from the Gulf of Aqaba in the north to the Gulf of Aden in the south. The shield has limited groundwater storage in the alluvial deposits of wadi channels, and weathered joints and fracture zones.

The GCC countries, including United Arab Emirates, Bahrain, Kuwait, Saudi Arabia, Oman and Qatar, are suffering from water shortages. They are already under the water scarcity line as defined by the World Health Organization (WHO) (having renewable water resources <1000 m^3/y/capita). Rainfall scarcity and variability coupled with the harsh climatic conditions and high evaporation rates have characterized this part of the world as arid with a limited availability of renewable water. Surface water resources are scare to absent with the exception of mountain areas in southwestern part of Saudi Arabia, southern part of United Arab Emirates and northern and southern parts of Oman (Al-Rashed and Sherif, 2000). The total surface runoff generated from rainfall is estimated as 4.83 billion m^3/yr, of which 3.21 and 1.47 m^3/yr are generated in Saudi Arabia and Oman, respectively (Khouri and Deroubi, 1990). About 0.15 billion m^3 /yr of surface runoff are generated within the territory of the UAE (Al-Rashed and Sherif, 2000). The total amount of surface runoff in Kuwait, Bahrain and Qatar is less than 2 million m^3/yr (Abdulrazzak, 1995).

The increase of the gap between supply and demand of water in the GCC countries is attributed to the limited available surface water, high population growth and urbanization development, deficient institutional arrangements, poor management practices, water depletion and deterioration of quality, especially in shallow groundwater systems (Dawoud, 2005). To meet the increasing demands, water authorities have focused their efforts on the development and supply augmentation. Demands are being satisfied by the development of groundwater, installation of new desalination plants, expansion in wastewater treatment plants and reuse, in addition to construction of dams to collect, store, and utilize surface water runoff. Currently, groundwater resources are being over-exploited to meet the increasing agricultural demands. This has led to a continuous deterioration in the quantity and quality of groundwater. Many aquifers in GCC countries are being mined, either because it has not been possible to regulate the pumping or the aquifers are non-renewable (Dawoud, 2005). The total volume of groundwater extracted from deep aquifer in the area over the period between 1980 and 2000 is estimated around 300 billion m^3, of which 254.5 billion m^3 were pumped from Saudi Arabia alone to satisfy the needs for the expansion in agriculture sector (Al-Rashed and Sherif, 2000).

To meet domestic water requirement, the GCC countries have shifted to freshwater production through desalination plants. The GCC countries, by necessity, have become the world leader in desalination of seawater and brackish water and currently have more than 65% of total world's desalination capacity (GWI, 2000). Experience with desalination in many of the Gulf states, particularly Saudi Arabia and Kuwait began as early as 1938 (El Nashar, 2004). Desalination production in the GCC countries ranges between 78 and 88% of the designated plant capacity (Al-Rashed and Sherif, 2000). However, desalination remains as capital intensive and costly projects. In terms of wastewater recycling, available treated wastewaters are still not being reused to their potential; planning for full utilization of treated effluent is still in the early stages. Introduced in the early 1980s in most of the GCC countries, treated wastewater represents one of the most important alternatives that can be used to meet some of the present water requirements and to lessen the long-term supply versus demand imbalance faced by these countries (Al-Zubari, 1997). Large and small treatment plants were constructed in the GCC countries for wastewater treatment at the tertiary and secondary levels (Al-Saati, 1995; Al-Muzaini and Ismail, 1994; Al-Hajj, 1995; and Al-Zubari, 1997).

Domestic and industrial freshwater requirements for the GCC countries are mostly satisfied through desalination in additional to some limited amounts of groundwater from both shallow and deep aquifers. In UAE, the agricultural sector consumes about 85% of available water resources followed by domestic water use, 14% and 4% for commercial and industrial use (Dawoud, 2005). In all GCC countries, agricultural requirements are mainly met through abstraction of water from shallow alluvial aquifers located in the coastal strips and inland basins, and from deep aquifers covering most of the Arabian Peninsula. In Saudi Arabia, rapid expansion of agricultural activities has resulted in substantial increases in water demands, leading to extensive mining of the deep aquifers. Likewise, agricultural water demand has sharply increased in the countries of Bahrain, Qatar, Oman, and the United Arab Emirates, where groundwater reserves are being mined. Government incentives and subsidies have encouraged farmers to cultivate large areas, placing great strain on existing groundwater resources.

1.3 Physical Setting of UAE

The United Arab Emirates (UAE) lies in the southeastern part of the Arabian Peninsula between latitudes 22° 40' and 26° 00' north and longitudes 51° 00' and 56° 00' east. It is bounded from the north by the Arabian Gulf, on the east by the Sultanate of Oman and the Gulf of Oman and on the south and the west by the Kingdom of Saudi Arabia. The total area of the United Arab Emirates is about 83,600 km^2 (Sherif et al., 2005)

The United Arab Emirates is a federation of seven Emirates: Abu Dhabi, Dubai, Sharjah, Ras Al Khaimah, Fujairah, Umm Al Quwain and Ajman. Six of the seven Emirates lie on the coast of the Arabian Gulf, while the seventh, Fujairah Emirate, is situated on the eastern coast of the peninsula and has direct access to the Gulf of Oman.

By far the largest Emirate in UAE is Abu Dhabi. It occupies about 75% of the total area of the country. Abu Dhabi City is the capital of both the Emirate and the country. It also has the largest population numerically, but at the same time the lowest population density among the other Emirates. Dubai, has the highest population density, and is considered the business capital and the most important port in the country.

The climate is arid with very high summer temperatures. The unique location of UAE, with the characteristic land-sea distribution and tropic of cancer passing through it, subtropical anticyclone above it, provides this region a tropical desert climate with several typical climate features. Generally, the temperature in the UAE varies between 41 and 50 °C during the summer and around 13 °C in the winter (Howari et al, 2006). The geomorphic features have a major role in the movement of both surface water and groundwater. The geology determines the characteristics and patterns of the storage layers and structural zones of the hydrogeological systems. Such factors greatly affect surface water runoff from rainfall events, infiltration rates, storage capacity, and groundwater table fluctuation in the system. The infiltration rate and the vertical hydraulic conductivity of the upper layer of the unsaturated/saturated soil determine, to a great extent, the ability of the system to be recharged from rainfall events. In most arid regions, the amount of recharge would be in the order of 2% to 10% from the total volume of annual precipitation (Sherif et al., 2005).

Previous studies (Sherif et al 2005 and 2006) indicated that the percentage of recharge form rainfall events is believed to be around 2% of the total precipitation in the UAE. However, this amount could be significantly increased through the proper implementation of water harvesting schemes including the construction of retention and detention dams across the main wadis. Under favorable conditions, where infiltration rates are very high and evaporation rates are relatively small, the recharge rate could be significantly enhanced.

Topographically, the UAE consists of two domains: low plains that cover 95% of the country, and high mountain covering about 5% and constitutes a natural barrier that isolates the Gulf of Oman to the east from the rest of the country (Baghdady and Abu-Zeid, 2002). The larger sandy desert zone covers over 90% of the country's surface area extending from the Al Mayann region in the northwest across to the eastern part of the UAE, where it is truncated by the mountain zone. The mountain zones consist of N-S mountain ranges parallel to the east coast. It has a north–south extent of about 150 km and east-west extent of 50 km. These mountains, from northern part of the Oman Mountains and evolve to form networks of wadis of which Dibba, Al Bih and Ham are the most significant (Ministry of Communication, 1996). If the topography is flat, rainwater would continue to accumulate on the ground surface causing ponds in the depression areas. Otherwise, if the land surface is mountainous, the rainwater will be collected through a number of tributaries and would reach the course of the main wadi causing floods. Depending on the hydrological conditions, rainfall intensity and duration of the rainfall event, the total volume of flood water varies from one rainfall event to the other.

Large areas of inland sabkha occupy the low land region. The largest of these is known as Sabkhat Matti, located in the western part of the country. The lands of Sabkhat Matti extend from the coast for about 120 km and reach an elevation of about 40 m (a.s.l) at its southern tip. The area adjacent to the Arabian Gulf Coast comprises a number of salt domes. These features often form islands in the sea and isolated hills on land. The highest of which is Jabal Dhana and it rises to 99 m a.s.l. Where the low-lying zone merges gradually with the mountains zone, several isolated anticlinal hills and mountains (trending generally, in N-S direction) occur. The highest and most extensive of these is Jabal Hafit with a maximum altitude of over 1000 m a.s.l (Al Shamsei, 1993).

The desert zone ranges in altitude from sea level up to 300 meters. This region is characterized by sand dunes which rise gradually from the coastal plain, reaching elevation up to 250 meters above the sea level in Liwa–Al Batin basin (Ministry of Communication, 1996).

The main aquifers in the UAE include the limestone aquifer in the north and east. Fractured Ophiolite rocks in the east, Gravel aquifers flanking the eastern mountain ranges on the east and west and sand Dune aquifers in the south and west Figure 1.2. The largest reserve of fresh groundwater in UAE occurs in the gravel alluvial deposits extending along the western side of the Oman mountain chain from Ras Al Khaymah to Al Ain. The sand dune aquifer covers about 74% of the total area of UAE (Sherif et al., 2005). It receives most of its recharge from the western side of the mountain, whereas the Arabian Gulf and Gulf of Oman are the main discharge area. The Limestone Aquifers are seen in the northern region at Wadi Bih catchment, as well as Jabal Hafit catchment in Al-Ain region. Most of the natural recharge to the aquifer systems is received at the heads of alluvial fans by infiltration from wadi's flows originated in the mountain zone.

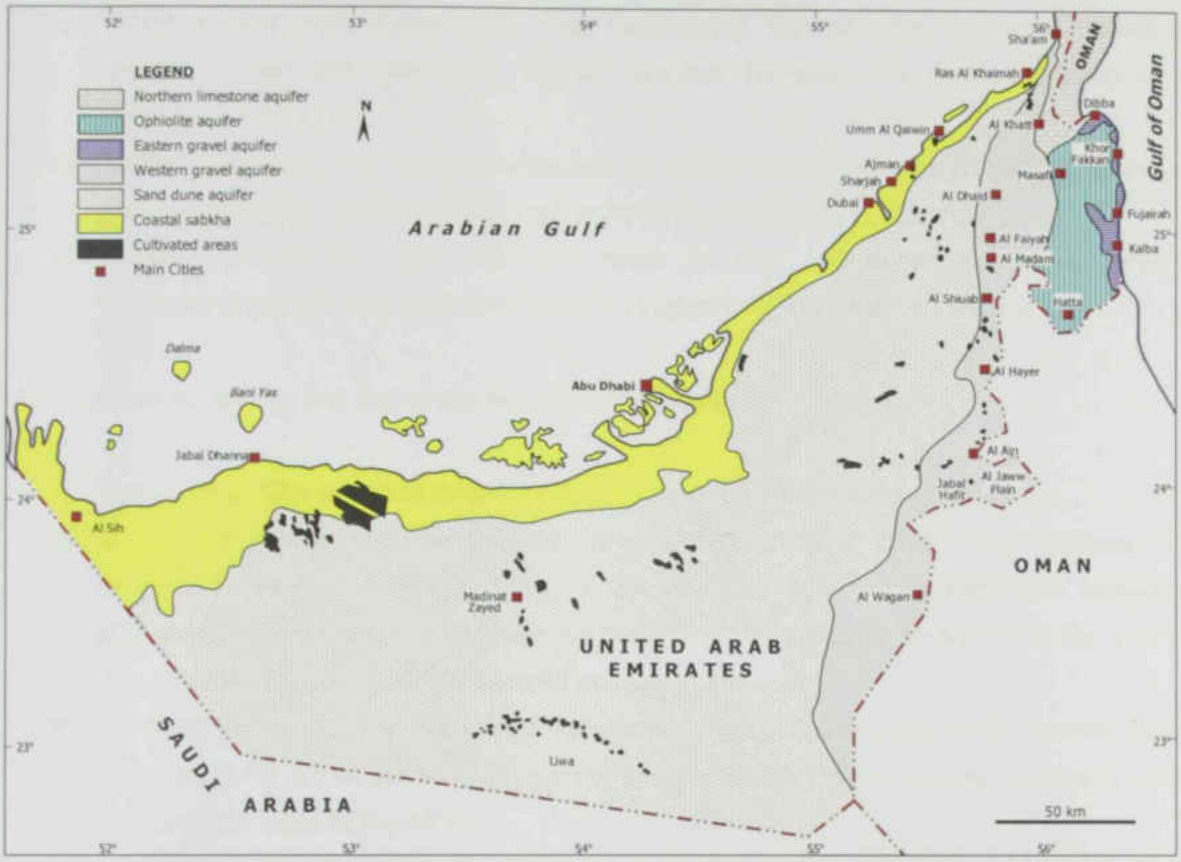

Figure 1.2. Main aquifers in the United Arab Emirates.

The groundwater units in U.A.E include a rock sequence ranging in age from the Permian to Quaternary periods. This sequence is ordered from oldest to youngest strata as follows (Al Shahi, 2002):

a) *Paleozoic:* It deals with the Permian rocks Khuff Formation.
b) *Mesozoic:* The age of rock sequence can be divided into Triassic (Lower, Middle and Upper), Jurassic (Lower, Middle and Upper) and Cretaceous (Lower, Middle and Upper). The Triassic includes the Triassic and Permian rocks and the Jurassic contains Hith anhydrite and Arab Formations. The Cretaceous age of rock sequence includes limestone marl and sandstone, Semail igneous complex, Hawasina complex, and Musandam limestone.
c) *Tertiary-Cenozoic:* This age sequence can be divided into Paleocene, Eocene, Oligocene and Miocene. Marl and limestone are the common rocks in this sequence.
d) *Quaternary-Cenozoic:* The common rocks in this sequence are Eolian sand, unconsolidated surficial deposits of gravel, gypsum deposits, and sabkha deposits.

The UAE can be divided into five structural provinces, (Rizk et al., 1997):

1) Rus Al Jibal: This area has thrusrt faults sloping in the east and south directions.
2) Diba Zone: It is a topographically low area and extends for 30 km from northeast to southwest, with an average width of 20 km. The Diba zone separates between Musandum calcareous sequence in the north and the ophiolite sequence in the south. The stratified rocks of tectonic boundaries exist in this zone.
3) Ophiolite Sequence: The Wadi Ham fault (north west-southeast) and Wadi Thawban fault (east-west) represent the northern part of this sequence. This is a clear change in rock type on both sides of the valley.
4) Hatta Zone: The folding and rock stratification in this zone are parallel to the longitudinal direction of the tectonic opening in the west-northwest direction.
5) Western region: The past late Maestrichtion calcareous deposits and associates rocks were subjected to folding along the western slopes along the northern Oman Mountains.

The upper cretaceous-lower tertiary boulder beds and calcareous rocks represent the boundary between the north Oman Mountains in the south and Schisa sands in the north. There are several aquifers in the UAE; each aquifer has its own characteristics and water potentiality. Aquifers can also be classified as given hereafter.

A- The Limestone aquifers

These aquifers are found in the north and east, and are composed of limestones and dolomites. The rocks of these aquifers are well stratified, hard, dense and non-porous at the surface in Wadi Al Bih. The Jabal Hafit area south of Al Ain city is an example of this aquifer (Al-Shahi, 2002).

B- Ophiolite aquifer

The Ophiolite sequence in the east is jointed and subjected to faulting. Groundwater in this area occurs only in joints and fractures, (Entec, 1995). The Ophiolite aquifer has been described to be of good quality due to jointing, faulting and weathering of the 'Semail beds' of the Northern Oman Mountains (Rizk et al., 1997). A survey by Electrowatt Engineering Services Ltd. (1981) showed that the Ophiolite Suite consists of medium-grained gabbros and fine to medium-grained diorites.

C- Gravel aquifers

A large quantity of fresh groundwater in UAE occurs in the alluvial deposits of the piedmont plains bounding the eastern mountains from the east and west. These aquifers can be distinguished into the eastern gravel aquifer, the northwestern gravel aquifer and the western gravel aquifer (Al-Shahi, 2002).

D- Sand Dune aquifer

The sand dunes cover about 74%of total area of UAE. The elevations of sand dunes change from sea level at the western coast to 250 m above ground level (sea level) at the Liwa-Al Batin basin in the south central part (Al-Shahi, 2002).

1.4 Water Resources in UAE

UAE is located in an arid area where the rainfall is very limited. The average annual rainfall varies between 20 to 140 mm/y. Measurement of rainfall in the UAE started in 1934; the maximum recorded average rainfall was 671.2 mm in 1995, measured at Fhor Fakkan (Ministry of Agriculture and Fisheries, 1995). The mean daily pan evaporation is estimated as 9.75 mm (Rizk et al., 1995).

Renewable water resources in the UAE are very limited. No surface water in the form of rivers or lakes is available. The rainfall is very scarce, random and infrequent. The UAE receives an average volume of 6.72 billion m^3 of annual rain water (Al-Rashed and Sherif, 2000). However, this annual volume of rainfall is mostly encountered in few events. Rainfall represents the main source for recharging groundwater systems. Phreatic aquifers are recharged directly through rainwater infiltration, while confined aquifers are recharged through their outcropping areas. Apart from the quantity of rainfall, its distribution in space and time plays a vital role in the planning and management of water resources. When rain falls with heavy intensities and short durations, surface water runoff is generated. The infiltration rate of the upper soil layer may not allow large quantities of the accumulated rainwater to percolate down through the soil and reach the aquifers.

The UAE has a low groundwater recharge rate and high evaporation rate (2000-3000 mm/y) with no reliable perennial surface water resources. To increase the groundwater recharge a number of dams have been built at various locations in the country. Many dams and embankments of various dimensions with a total storage capacity about 122 million cubic meters were build during the last three decades,(Environment & Agriculture Information Center, 2007). These dams are basically built for recharge purposes. They also provide protection against damage caused by flash floods.

The UAE is the second largest producer of desalinated water in the Arabian Gulf Region, with a production of about 5,465,784 million m^3/y (Mohamed et al., 2005). Because of a rapid increase in domestic and industrial water demand, several plants were installed, particularly in Abu Dhabi and Dubai. Currently, desalination plants produce about 98 % of the total drinking water supplies in the UAE (Sommariva and Syambabu, 2001).

Treated wastewater is used for irrigation of green areas along the highways, greenbelts, and city gardens. The annual production of treated wastewater in the UAE was 106 Mm^3/y of which 63 Mm^3/y were used to irrigate golf courses, parks, and green areas (Al-Rashed and Sherif, 2000).

The UAE has very limited potential for agricultural development since over 90% of the land is desert. In spite of the harsh weather conditions and soil and water constraints, a remarkable progress has been made in the agricultural sector, particularly during the last two decades. The main agricultural areas are located in the northeast (Ras Al Khaimah), in the east along the coast from Kalba to Dibba (Fujairah), in the southeast (Al Ain/Abu Dhabi) and in the central region (Dibba/ Sharjah). About 85% of the total water consumption for irrigation purpose in UAE is groundwater (Rizk et al., 1999).

1.5 Objectives of the Current Study

The objective of this thesis is to study and simulate the vulnerability of the groundwater resources to seawater intrusion in the coastal aquifer of Wadi Ham, Fujairah Emirate. This aquifer is of specific importance for agricultural development. Irrigation in the area of Wadi Ham is mostly based on groundwater pumping from the aquifer of Wadi Ham. The specific objectives of this study include:

1. Identify the geometric, geological and hydrogeological parameters that are relevant to the study and assessment of seawater intrusion in the area of Wadi Ham.

2. Select a numerical model and calibrate the model based on available groundwater levels during the last two decades in the study area.

3. Assess the vulnerability of the groundwater resources in the study area to seawater intrusion under different pumping scenarios.

4. Propose recommendations to decelerate the seawater intrusion process in Wadi Ham.

To achieve the above objectives the following tasks will be performed:-

1. Conduct a comprehensive review of all previous investigations and publications related to groundwater resources in the area of Wadi Ham with specific reference to the coastal zone.

2. Identify, store and present the geometric, geologic and hydrogeological information of the aquifer under consideration. All information and data will be stored as geographically referenced data such that they could be directly used by the selected model.

3. Review the available groundwater flow and solute transport models with specific reference to variable density models and select a suitable one based on the available data and model requirements.

4. Calibrate the selected model against the available data of groundwater levels over the last two decades.

5. Employ the numerical model to assess and study the vulnerability of groundwater resources to seawater intrusion in the study area.

6. Examine the effect of different pumping scenarios on the seawater intrusion in the study area.

7. Propose (or make) recommendations and guidelines to reduce the possible impacts of seawater intrusion in the Wadi Ham coastal aquifer.

1.6 Thesis Outline

This thesis encompasses six chapters. Chapter 1 discusses the importance and distribution of water resources at the global level and presents a summary about the water resources availability in the Gulf Cooperation Council (GCC) countries, including UAE. The objectives of the study are also included.

Chapter 2 discusses seawater intrusion and its mechanism under the dispersion-zone approach. Salinization sources are elaborated and the components of seawater are defined. Modeling approaches are presented and the causes of the seawater intrusion problem are outlined. The groundwater flow and solute transport governing equations are presented. SUTRA and other available numerical models are briefly discussed.

Chapter 3 reviews the geological and hydrogeological settings of the aquifer system in the area of Wadi Ham. The geometric and hydrogeological parameters are defined and the historical records of groundwater levels in a number of observation wells are discussed. Previous geophysical investigations to assess the seawater intrusion in the Wadi Ham aquifer are presented.

Chapter 4 discusses the calibration and validation of numerical models in general and elaborates the calibration and validation of SUTRA model in the area of Wadi Ham in particular. SUTRA-Argus One modeling environment is presented and its capabilities and limitations are outlined. The calibration and validation of SUTRA are conducted on a set of groundwater level data.

Chapter 5 discusses the simulation runs that have been conducted under steady- and unsteady-flow and solute transport conditions. The simulation is conducted for different scenarios in the horizontal (area) 2D view as well as in the vertical (cross sectional) 2D view. The effects of groundwater pumping from Khalba well field, hydraulic conductivity and dispersivity are elaborated.

Chapter 6 presents a summary, conclusions and recommendations of the study. A list of references that have been used in this study is also included.

CHAPTER 2
LITERATURE REVIEW

Chapter 2. Literature Review

2.1 Introduction

During the last century, rapid urbanization and population growth have resulted in many environmental problems. Among those, the water shortage and pollution are most serious. People around the world are beginning to realize the interactions between human beings and the environment. Human activities are affecting the natural water ecological cycle in many ways. Overexploitation of groundwater resources has decreased groundwater levels and caused seawater intrusion in coastal aquifers. Human interactions can however change the natural balance and in special cases this can lead to degradation of the environment, including lower quality of the drinking water and degradation of agricultural land and crops.

Degradation of groundwater quality is a very serious problem. In many countries, groundwater is the main supply of freshwater. Today more groundwater wells are abandoned as a consequence of degradation of the quality, and at present it is very hard to find new unpolluted groundwater reservoirs. Livestock and humans have fundamentally the same requirements with respect to water quality. The content of dissolved solids should in general not exceed 6 g/l, but animals can drink water up to 10 g/l of total dissolved solid if the main constituent is NaCl (Matthess, 1982). A plant's requirement to the water quality is similar to the requirement of livestock and humans. Irrigation water with a content of 0.5 g NaCl is always usable and becomes unusable above 4 g NaCl (Matthess, 1982). Saltwater intrusion occurs when freshwater is overpumped from a freshwater reservoir which is adjacent to a saltwater reservoir. This is the situation in many large cities situated next to the ocean. Large cities have high demand for freshwater and usually have limited freshwater supplies such as good aquifers, lakes or rivers. Many regions in the world are facing the challenge of water shortage and pollution. The United Nations Environment Programme (UNEP) identified water shortage and global warming are the two most critical problems for the coming few decades.

Intrusion of seawater into coastal aquifers is a widespread contamination phenomenon that increasingly causes groundwater salinization problems. Seawater intrusion is especially severe in semiarid regions where high pumping extraction rates are coupled with low freshwater recharge. Seawater intrusion, or encroachment, is defined as the migration of salt water into freshwater aquifers under the influence of groundwater development (Freeze and Cherry, 1979). One of the major concerns most commonly found in coastal aquifers is the induced flow of seawater into fresh water aquifers caused by groundwater development. In places where groundwater is being pumped from aquifers that are in hydraulic contact with the sea, the induced gradients may cause the migration of seawater from the sea toward production wells.

Seawater intrusion is a natural process, by which seawater displaces and mixes with the fresh groundwater in coastal aquifers due to the density difference existing between waters of different salinities. In heavily exploited coastal aquifers, where groundwater pumping consistently exceeds recharge, the water table falls and seawater intrusion becomes a major concern. Eventually, the limits for salinity in drinking water (established at 500 ppm of total dissolved solids, TDS, by the American Environmental Protection Agency) as well as for agricultural uses may be exceeded in the pumped groundwater, thus making it unsuitable for human uses. A 3% mixing of seawater with freshwater in coastal aquifer would render the freshwater resource unsuitable for human consumption (Sherif and Kacimove, 2006). Salinity in irrigation water can be detrimental to agriculture, reducing yields and damaging crops of low tolerances to salt. In some cases, conditions may necessitate a change to crops that are more salt tolerant. Salt water has also been shown by Jenkins and Moore (1984) to reduce soil erodibility. A common management approach when a pumping well becomes contaminated by saltwater is to relocate pumping further inland (Barlow, 2003). If no additional fresh water sources are available to satisfy the demands, the high groundwater extraction rate needs to be maintained or increased at the new location thus putting also groundwater further inland at the risk of saltwater intrusion.

2.2 Mechanisms of Seawater Intrusion

Seawater intrusion into freshwater supplies has become a cause of concern within the last century. The salinity distribution of the groundwater in coastal and deltaic areas is capricious as a result of past and ongoing natural processes including climate change, geologic processes and land subsidence, resulting in changes of the sea level relative to the land surface (Van Dam, 1993). This problem is intensified due to population growth, and the fact that about 70% of the world population occupies coastal plains (Bear et al., 1999).

Coastal aquifers are hydraulically connected to the adjacent marine water body. Consequently, they contain both fresh and saline (salty) groundwater. Fresh groundwater normally flows seaward within coastal aquifers, eventually intercepting saline groundwater. The lighter, freshwater (1 gram of salt per cubic centimeter - g/cm^3) tends to override and "float" on the denser, saline water (1.025 g/cm^3), but mixing also occurs. This mixing zone is known by several names, including the "freshwater-seawater interface," the "zone of transition," and the "zone of diffusion". The zone of diffusion is typically located near the marine shoreline. The exact location depends on several conditions, including the volume of freshwater discharge and the nature of the aquifer (confined or unconfined). In a typical coastal aquifer, the zone of diffusion dips down beneath the land surface. In the case of an island or peninsula, the zone of diffusion can extend beneath the entire land surface. As with most aquifers, coastal aquifers are recharged primarily by precipitation. Under natural conditions, aquifer recharge is in equilibrium with groundwater discharge. Consequently, the zone of diffusion maintains a position of relative stability, moving slightly landward or seaward in response to varying climatic and tidal conditions. When groundwater is pumped from coastal aquifers, freshwater that would normally discharge to the sea is intercepted, disrupting the natural equilibrium. This causes the zone of diffusion to migrate landward and/or locally upward. Groundwater drawn into pumping wells can become increasingly saline. Over time, the water can become unfit for consumption.

The initial model for seawater intrusion was developed independently by Ghyben in 1888 and by Herzberg in 1901. This simple model is known as the Ghyben–Herzberg relationship and is based on the hydrostatic balance between fresh and saline water in a U-shaped tube. They showed that the seawater occurs at a depth h below mean sea level represented by:

$$h = \frac{\rho f}{\rho s - \rho f} h_f \qquad (2.1)$$

where ρ_f and ρ_s are, respectively, the density of fresh and sea water, and h_f is the elevation of fresh water level above mean sea level. Substitution of ρ_f (1000 kg/m^3) and ρ_s (1025 kg/m^3) in Eq. (2.1) shows that $h = 40h_f$. In other words, the depth to the fresh-saline interface below mean sea level (h) is 40 times the elevation of the water table above sea level (h_f) (Freeze and Cherry, 1979). In general, if the water table in an aquifer is lowered by 1 foot, the freshwater-seawater transition zone will rise about 40 feet, and the total vertical thickness of the freshwater lens will be reduced by about 41 feet (Freeze and Cherry, 1979).

This simplistic model ignores convection, dispersion and diffusion phenomena responsible for the distribution of salinity in coastal aquifers. In coastal aquifers, freshwater usually overlies the seawater separated by a transition zone. Management of limited groundwater resources in such situations is a delicate task and requires special attention to minimize the movement of the seawater wedge into aquifers and upconing of seawater near pumping stations (Reilly and Goodman, 1987).

The transition zone between seawater and freshwater in a coastal aquifer start at the coast line and moves inland and downward (Figure 2.1). The width and length of the transition zone depend on many factors including oscillating sea level, dispersivity, freshwater recharge, hydraulic conductivity and other hydrogeological parameters. When the transition zone moves due to an oscillating sea level, change in freshwater flux, or change in recharge/pumping activities, a very intensive ion exchange and a following dissolution or precipitation of minerals can occur. In the seawater zone the dominating ions are sodium and chloride. In the freshwater zone the dominating ion are calcium and bicarbonate and the soil in the freshwater zone are typically dominated by calcium. When seawater intrudes into a freshwater zone the sodium in seawater ion-exchanges with calcium on the soil, the typical ions in the soil water become calcium and chloride. When the seawater re-draws, the calcium in the freshwater exchanges with sodium on the exchanger, the dominating ions in solution become sodium and bicarbonate. Research has shown that clay have different properties at different salinities, which in the case of seawater intrusion can result in clogging of pores (Goldenberg, 1985; Frenkel and Rhoades, 1978).

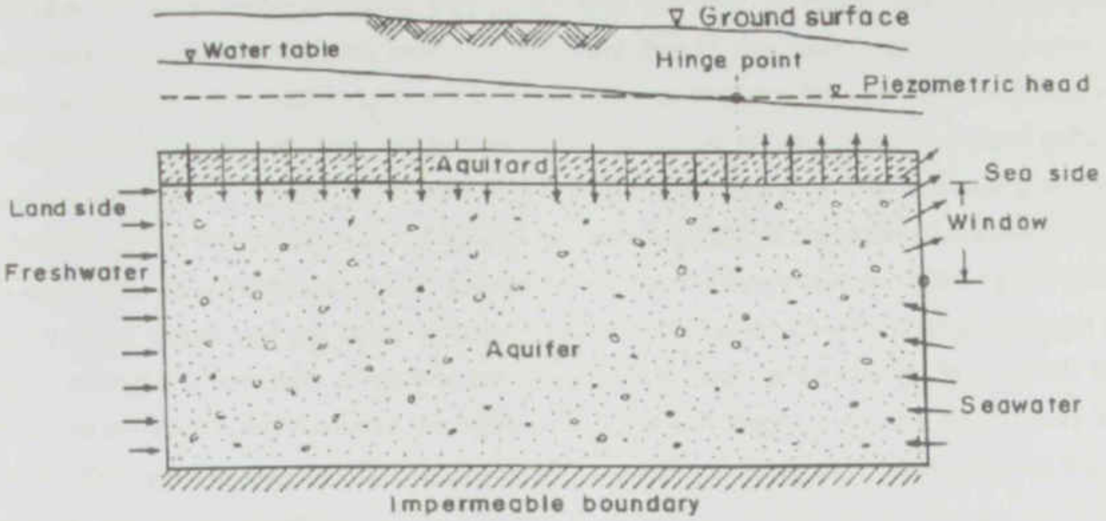

Figure 2.1. Mechanism of seawater intrusion (Sherif et al., 1999).

In a seawater intrusion case the seawater flows into the freshwater and mixes with freshwater. This result in seawater gets less dense, move upwards and flow parallel with the freshwater. The outflow of freshwater is of great importance, as it gives the ultimate potential for groundwater recharge in an aquifer. The larger outflow, the larger potential for freshwater recharge. The flow of groundwater to the sea is driven by the difference in pressure between the freshwater and the sea. As the freshwater recharge comes from the rain and the rain varies over the year, the potential for groundwater recharge also varies over the year. In many cases the sea level is assumed constant, but the sea level can also vary due to e.g. tides, waves and storms. The general setup of the water table and a description of the coastal boundary in groundwater modeling have been studied by Nielsen (1999) who considered tidal forces and wave setup and found an expression for the mean water table which ought to be used in regional groundwater modeling. Saline water makes up 97.25% of all water on earth. It is present in oceans, seas and estuaries and as groundwater in their subsoils and in land areas where seawater has occurred in the geologic history and not been replaced by freshwater so far. Climate change can also bring about changes in the rate of natural recharge of the fresh groundwater (Bear et al, 1999). Changes of climate have caused changes in sea levels throughout the geologic history in the present time the sea level arises; rising temperatures make the seawater expanding and the polar ice caps and glaciers melting.

2.3 Identification of Salinization Sources

The distinction of different salinization mechanism is crucial to the evaluation of the origin, pathways, rates and future salinization of the coastal aquifer. The interpretation of salinization process should be based upon geological and hydrochemical criteria. Several geochemical criteria can be suggested to identify the origin of salinity, especially detection of seawater intrusion as opposed to other salinity sources in coastal aquifer. Yechieli and Sivan (2008) documented that it essential to determine first the origin of salinity in order to be able to cope with this problem. They discussed the chemical and isotopic evidences for identifying seawater intrusion. Chemical and isotopic analyses are best tools for identifying the specific sources of salinity and their geochemical evolution (Jones et al., 1999). Like Cl/Br , Na/ Cl , Ca/Mg, Ca (HCO3+ SO4) ratios, O and H isotopes and boron isotopes. Morell et al (1996) argued that Br is the best indicator for tracing seawater.

2.3.1 Salinity

Because of the contrast in marine and typical continental anion matrices, the clearest indication of possible seawater intrusion is an increase in Cl^- concentration as a proxy for salinity, although other processes may lead to a similar phenomenon. In coastal aquifers, where continuous over-exploitation causes a reduction of the piezometric levels, intrusion of seawater results in a salinity breakthrough. Thus a time-series of chloride concentrations can record the early evolution of rapid salinization processes. Several prior studies evaluate the groundwater characteristics in the eastern part of the UAE (Jones and Marrei, 1982, Elschami, 1990, Rizk and El-Etr, 1997, Rizk et al., 1997 and Brunke and Schhelkes, 1999). Based on these studies, it has been suggested that evaporation and seawater intrusion are possible sources of salinity in the groundwater of the region.

2.3.2 Cl/Br ratio

The Cl/Br ratio can be used as a reliable tracer as both Cl and Br usually behave conservatively except in the presence of very high amounts of organic matter. Seawater (Cl/ Br weight ratio= 297) is distinguished from relics of evaporated seawater (hypersaline brines Cl/Br <297, Dead Sea = 40; Starinsky et al.(1983), evaporate-dissolution products (over 1000) and anthropogenic sources like sewage effluents (Cl/Br ratios up to 800;(Vengosh and Pankratov (1998)) or agriculture-return flows (low Cl/Br ratios). It should be noted that the Cl/Br signal can be modified by degradation of organic matter (Davis et al., 1998). Ben Hamouda et. al., 2008 documented that the Br/ Cl ration is often used for identifying a possible seawater intrusion because of its relativity constant value in the present seawater.

2.3.3 Na/Cl ratios

Na/Cl ratios of saltwater intrusion are usually lower than the marine values. Thus low Na/Cl ratio, combined with other geochemical parameters, can be an indicator of the arrival of the saltwater intrusion, even at relatively low chloride concentrations during early stages of salinization. The low Na/Cl ratio of seawater intrusion is distinguishable from the high (>1) Na/Cl ratios typical of anthropogenic sources like domestic wastewater (Bear et al. (1999)).

2.3.4 Ca/Mg, Ca/(HCO_3 + SO_4) ratios

One of the most conspicuous features of saltwater intrusion is commonly the enrichment of Ca over its concentration in seawater. High Ca / Mg and Ca / (HCO_3+ SO_4) ratios (>1) are further indicator of the arrival of seawater intrusion. It should be noted however, that saline water with high Ca can originated by a different mechanism, not necessary related to the base–exchange reaction and modification of modern seawater (Bear et al. (1999)).

2.3.5 O and H isotopes

Rectenwald and Bennett (2008) showed that stable isotopes (H, O) were used to complement inorganic data to define different water masses in the Floridian aquifer system. Linear correlations are expected from mixing of seawater with ^{18}O depleted groundwater in correlation of δD versus $\delta^{18}O$ or Cl versus $\delta^{18}O$ different source with high salinity would result in different slopes due to evaporation processes that would change the isotopic composition of the saline end-member (Bear et al. (1999)).

2.3.6 Chlorine-36

Chlorine-36 is a useful tool to trace different sources of salinity in groundwater systems (Carlson et al., 1990; Mazor, 1992; and Phillips, 2000). Chlorine-36 is produced naturally by several mechanisms: (1) atmospheric production through the spallation reaction of ^{40}Ar and neutron capture of ^{36}Ar (meteoric), (2) lithospheric production by the spallation reaction of K and Ca and neutron activation of ^{35}Cl (epigene), and (3) subsurface production via neutron activation of ^{35}Cl (hypogene) (Bentley et al., 1986a; Bentley et al., 1986b; Carlson et al., 1990; Commander et al., 1994; and Lyons et al., 1998). In situ production of chloride could be another source of ^{36}Cl in groundwater that is low in chloride (Yechiel et al., 1996). Also, ^{36}Cl has been produced anthropogenically in the 1950's during nuclear weapons testing (Lyons et al., 1998).

The distribution of ^{36}Cl in the subsurface is controlled by the above sources as well as by the evapotranspiration and dissolution of halite (Bird et al., 1989 and Mazor, 1997). In general, high ^{36}Cl/Cl ratios are associated with areas of high precipitation, whereas low ratios are observed in regions of low precipitation (Phillips, 2000). The meteoric ^{36}Cl tends to increase from the continental interiors toward coastal areas (Bentley et al., 1986a and Bentley et al., 1986b). Lehmann et al. (1995) suggested several other possible external sources of chloride to groundwater system which include seawater incursion, aquitard infiltration, and mixing with high salinity water from outside the aquifer.

2.3.7 Boron isotopes

The boron isotopic composition of groundwater can be a powerful tool for discrimination of salinization source, in particular distinguish seawater from anthropogenic fluid such as domestic wastewater. The δ^{11}B values of seawater intrusion range over 30% to the seawater value (δ^{11}B=39%), reflecting mixing of freshwater and seawater in coastal areas. Saline groundwater from coastal aquifer of Israel has high δ^{11}B values, up to 60%. The high δ^{11}B content of saltwater intrusion differ from the boron isotopic composition of sewage effluents (δ^{11}B=0-10%) and sewage-contamination groundwater (5-25%), and thus can be used to trace the origin of the salinity (Vengosh et al., 1994, 1998).

The geochemical features of brackish water within the transition zone of the seawater intrusion serve as an excellent tool to detect seawater intrusion. The most striking phenomena that characterize seawater intrusion is the difference between the chemical composition of the resulting brackish water and the simple mixture of seawater and groundwater. Several geochemical criteria are suggested to identify the origin of salinity, especially detection of seawater intrusion as opposed to other salinity sources in coastal aquifers. Table 2.1 gives the geochemical criteria for distinguishing the origin of saltwater (Sherif and Kacimov, 2006)

Table 2.1. Geochemical criteria for distinguishing saltwater origin (Sherif and Kacimov, 2006b)

Measurement	Criteria
Chloride, Cl	A time-series of Cl concentration can record the early evaluation of relatively rapid Salinization
Cl/Br ratios	Cl/Br=297 :seawater Cl/Br<297 Hypersaline Brine Cl/Br>1000 Evaporate Dissolution product Cl/Br up to 800 Anthropogenic Source (e.g. sewage)
Na/Cl ratios (Molar)	Na/Cl= 0.86 seawater Na/Cl<0.86 seawater intrusion Na/Cl>0.1 Anthropogenic Source (e.g. sewage)
Ca/Mg	Ca/Mg>0.1 86 seawater intrusion
Ca/(HCO_3 + SO_4)	Ca/(HCO_3 + SO_4)>1 Seawater intrusion

2.4 Component of Seawater Intrusion

2.4.1 Freshwater

Composition of freshwater is dependent on regional and local conditions. There are many factors that can influence the composition of freshwater such as distance to seawater, weather, saltwater intrusion, soilmatrix and others.

The more the seawater is dominating in the region the more salty is the water. Dazy et al. (1997) found out that atmospheric input of salt had a considerable effect on the freshwater composition in the Cyclades (Greece). They measured precipitation with total dissolved solids content (TDS) of 45-223 mg/l. Appelo and Postma (1993) found chloride iso-concentration in precipitation from 30mg/l at the shoreline to 2 mg/l, 150 km inland. Warm weather enhances the evaporation and the freshwater becomes more salty. This becomes more evident when groundwater is used for irrigation. The mineral in the soilmatrix is another important factor. When rocks degrade due to physical and chemical reaction, the degradation-products are minerals that are added to the groundwater. If pyrite is present in the soil, the freshwater could be dominated by sulfate and aqueous iron. In a carbonate rock, the freshwater would be dominated by calcium, magnesium and trace elements.

When seawater mixes with freshwater, the latter becomes more salty. When saltwater intrudes a freshwater aquifer, ion exchange and mineral dissolution and precipitation changes the composition of the freshwater Table 2.2 shows different ion-composition freshwater reported in various studies. Large differences between in the composition of the freshwater are observed and some ions vary up to a factor of 10. In a saltwater intrusion an ion-exchange between Ca^{2+} and Na^{+} is generally occurring, because Ca^{2+} has a large affinity to a exchanger and therefore occupies the main part of the exchanger in the freshwater zone of an aquifer even though there are more Na^{+} in freshwater.

Table 2.2. Composition of freshwater and soil properties from various studies (Gomis et al., 1996, 1997; Appelo et al., 1987, 1990).

Parameter	Gomis 96	Gomis 97	Appelo 87	Appelo 90 Alphen sed.	Appelo et al.90 Ketelmeter sed.	Appelo et al. 90 Delft sed.
PH -	-	-	-	6.85	6.3	6.83
Na^+(mmol/l)	6.3	2.17	0.1	5.6	3.3	21.4
K^+(mmol/l)	0.8	0.0873	0.1	0.7	0.4	0.693
Ca^{2+}(mmol/l)	2.02	3.13	*	2	1.1	2.95
Mg^{2+}(mmol/l)	1.93	0.617	0.5	1.9	1.2	1.93
Cl^-(mmol/l)	4.3	2.96	0.1	4.5	4.5	21.05
SO^{2-}_4(mmol/l)	0.09	1.72	0.4	0.1	0.07	0.142
HCO^{3-}(mmol/l)	10.8	3.28	**	10	4.7	14.37
CEC(meq/100g)	7	13	2	1.19	10.2	0.51

*in equilibrium with calcite
**in equilibrium with a partial pressure of CO_2 of 0.01 atm

When saltwater with high concentration of Na^+ and Cl^- intrudes a freshwater zone the dominating ions in solution becomes Ca^{2+} and Cl^- as Na^+ exchanges with Ca^{2+} on the exchanger. When an aquifer is freshning the domination ions becomes Na^+ and HCO^{3-}.

The intrusion of seawater and formation of a relatively static interface zone between overlying fresh and underlying saline water, may produce local low redox conditions, due to decomposition of dissolved organic matter, fine suspended organic particulate, or organic rich sediments (Schhoeller, 1956; Custodio and Llamas, 1976; Hem, 1985).

According to Custodio et al. (1987), this process will cause increased P_{co2}, changes in pH, and the reduction of dissolved sulfate to H_2S, resulting in low SO_4/ Cl ratios. Such changes shift the calcium carbonate equilibrium and most commonly cause dissolution. The resulting increase in the Ca-content is frequently masked by exchange of Ca^{2+} for Mg^{2+} or Na^+ on clays previously equilibrated with more seawater-like cation matrices.

As noted by Whitaker and Smart (1994) for the Bahamas, intense and episodic nature of rainfall, lack of soil cover, well developed karstic fissures and shallow depth of vadose zone, all contribute to significant inputs of organic matter to the freshwater lens. This generates potential for dissolution considerably greater than that predicted solely by simulations of inorganic mixing between basal freshwater lens waters and underlying saline groundwater. Whitaker and Smart (1994) documented that surface-derived organic matter penetrates the aquifer in the Bahamas to a considerable depth, supporting both aerobic and sulfate-reducing heterotropic bacteria. They noted that processes, rates and distribution of organically mediated carbonate dissolution are controlled by the balance between rates of input and consumption of oxygen and organic matter.

Freshwater in relation to groundwater in coastal areas can vary significantly. Between the rainwater and open seawater, the freshwater changes from rainwater composition to saltwater composition.

2.4.2 Saltwater

The composition of saltwater is dependent on global and regional factors. Compared to freshwater the composition of saltwater is very much alike around the world. The amount of salt in saltwater is measured in concentration [mmol/g] or[mg/l] or in mass-fraction [kg/kg] or [ppt] (part per thousand) also called the salinity. The salinity of seawater depends on the regional area. The pH of saltwater around the world is approximately 8.2 and the variation of the pH is very small. Table 2.3 shows the composition of saltwater from various studies, indicating that the main constituents in saltwater are Na^+ and Cl^-.

2.4.3 Density

The density of water depends on the temperature and salinity, which is shown on Figure 2.2. The density increases for increasing salinities and pure freshwater with zero salinity have maximum at approximately 4°C. Saltwater with a density of 1.0245 kg/l contains approximately 34.8 g of salts (Reilly and Goodman, 1985)

2.4.4 Fresh / Saline water interface.

The seawater problem occurs both on regional or large scale and on the local or small scale. The regional or large scale effects occur in large areas where the interface between fresh and saline groundwater moves slowly and smoothly in upward and /or inland direction. The large scale displacement is caused by groundwater table as in reclamation projects, new polder or for land improvement by drainage, by large excavations, such as borrow-pits for sand and gravel, and by excavations at the inner sides of sand dune.

Table 2.3. Composition of saltwater from various studies (Gomis et al.,1996, 1997; Appelo et al., 1987, 1990).

Parameter	Gomis 96	Gomis 97	Appelo et al. 87	Appelo et al. 90
Na^+(mmol/l)	182.6	522	145	180.9
K^+(mmol/l)	3.90	10.3	3.23	3.9
Ca^{2+}(mmol/l)	4.20	11.3	*	4.2
Mg^{2+}(mmol/l)	20.6	61.7	16.3	20.6
Cl^-(mmol/l)	212	606	169	212
SO^{2-}_4(mmol/l)	11.6	20.2	0.9	11.1
HCO^{3-}(mmol/l)	0.9	2.13	-	-

* in equilibrium with calcite

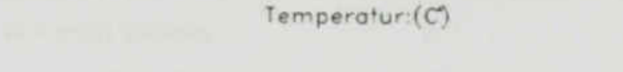

Figure 2.2. Density as a function of temperature and salinity (Harremoës et al.1990)

Seawater intrusion involves mixing between saline and freshwater components. Because of its significant salt content, a small fraction of seawater would dominate the chemical composition of the groundwater mixture. Contribution of the 1% of seawater would almost triple the salinity of typical groundwater (with an initial chloride content of 100 mg/l). Contribution of 5 % of seawater would result in water with salinity above 1000 mgCl/l. Consequently, chloride ion concentration is a very sensitive indicator, particularly if background salinity levels of the regional groundwater are low. In as much as seawater has a high salt content relative to local fresh groundwater, the ionic ratios of seawater dominate the chemical composition of saline groundwater, assuming conservative behavior of the ion species.

When mixing of two waters with different ion composition occurs, dissolution and precipitation can occur depending on prevailing conditions; e.g., when water in equilibrium with calcite and a water in equilibrium with gypsum are mixed, calcite may precipitate. This effect is called the common ion effect. Mixing of two groundwaters with different CO_2 pressures, both at equilibrium with calcite leads to subsaturation with respect to calcite. This effect is called mischungskorrosion in the German literature (Appelo and Postma, 1993).

Eeman et al. (2008) focused on the mixing zone between thin, shallow freshwater lenses and underlying, upward seeping saline groundwater, under homogeneous isotropic conditions. The stable isotopes of O and H can also be used to describe the mixing process between saline and fresh water. Fresh groundwater is generally depleted in both ^{18}O and ^{2}H relative to seawater. Mixing of fresh and seawater should result in a straight line connecting the two end members. Such relationships have been utilized by Manzano et al. (1990) and Izbicki (1996) to distinguish different water sources in coastal mixing zones, and to signal possible variance from truly conservative behavior. The interface produced between fresh and saline waters is the product of the physical and chemical properties of these two water bodies, as well as external processes. The interaction between these waters is produced by the flow seaward of fresh waters derived from recharge. This flow towards the sea is the product of pressure gradients within the coastal mediums, which prevent the seawaters from infiltrating the coastal setting within natural conditions. At this interface between the two water bodies, fresh water pressures exceed that of the denser saline water, which produces the flow from land to sea. This flow is determined by the levels and gradients of the water table and piezometric levels, which in turn are determined by boundary conditions, such as surface water levels, rates of exchange and abstraction (Bear et al, 1999). Due to the contrasts in volume and densities of the two water bodies, the fresh water overlies the saline waters producing the saline wedge or intrusion phenomenon (Domenico and Schwartz, 1990).

A salinity transition zone is formed with salinity ranging from that of seawater at the land-sea interface to that of freshwater at some distance into the aquifer. Within the transition zone, at least some of the intruding seawater re-circulates back into the sea, following flow patterns that are determined by the freshwater flow, density differences, thermal convection, tidal oscillations and wave set-up. The shape and position of the transition zone depend on many factors, but at steady-state the transition zone is stationary indicating a dynamic equilibrium between the natural fresh groundwater flow towards the sea and the re-circulating seawater inflow. Any disturbance of this water flow balance in the aquifer will change the position and shape of the transition zone.

2.4.5 Transition zone

The position of the salt wedge is usually indicated by the two lines, the 35000 ppm. interface indicating boundary of seawater influence, and the 500 ppm interface making fresh water. Between these lines occur the so-called transition zones. The transition zone normally is a result of hydrodynamic dispersion (Bear and Dagan, 1964). Cooper (1959) developed a hypothesis stating that "where a zone of diffusion exists between the salt water and the fresh water, the saltwater is not static but flows in a cycle from the floor of the sea, to the transition zone, and back to the sea. This mixing phenomenon is related to the tidal fluctuations." He described the transition zone as the occurrence of circulation of saltwater.

The most important factors that affect the transition zone are summarized next.

Heterogeneity: A heterogeneous environment allows for a variable penetration of the seawater wedge. The effect is very negligible in shallow, low-permeability formations. The aquifer transmissivity, T, does not properly describe the conditions, since the transmissivity is not linearly correlated with the thickness.

In case of stratified aquifers, if the upper layer is of low hydraulic conductivity, it favors the formation of a fresh water body; and if the upper layer is of high hydraulic conductivity, it favors the saltwater. In thick low-permeability aquifers, an important upward fresh water flow exists near the coast. If they are covered by a layer of highly permeable material, an extreme situation of the stratified aquifers appears and a salt or brackish water body may develop in the upper layer, especially in dry climates, floating on fresh water.

Anisotropy: Both the flow pattern and the interface position are influenced by anisotropy. In general terms, a low hydraulic conductivity to vertical flow would tend to increase the depth of the interface; while a high hydraulic conductivity to vertical flow would tend to reduce its depth.

Sea bottom: Sea bottom conditions can alter the fresh/saltwater relationships in coastal aquifers. A cover of low hydraulic conductivity cap acts as an obstacle to fresh discharges into the sea, thus tending to decrease the width of the transition zone and vice versa. Entrapped old sea water in deep lenticular or discontinuous permeable formations, not open to the sea, saltwater penetrated in earlier times (connate or infiltrated), cannot be expelled, and only disappears by upward diffusion towards the fresh water body. This can last for centuries (Meinardi, 1976). In low permeability recent formations, such as clay and silt deltaic lenses, salty connate water remains for a long time between fresh water bodies.

Density and viscosity: Kempers and Haas (1994) stated that" if the fluids have equal viscosity and density and the heterogeneity of the porous medium is statically homogenous, the length of the dispersion zone between the fluids known to increase as (BX)1/2, where B is the dispersivity and X is the average displacement distance. Dispersivity is a soil characteristic parameter with a dimension of length and is regarded as a fundamental transport property of the soil matrix. For the case where the fluids differ in density, viscosity or both, there is effect on the dynamic of the fluid flow on the magnitude of the dispersivity B and validity of the X1/2 dependence of the dispersion zones length. Measurements demonstrate that the dispersivity does indeed depend on the displacement velocity. The dispersion zone does grow as X1/2 in presence of density contrast and viscosity contrast. Experiment, simulation and the model show that the dispersivity is strongly dependent on the displacement velocity in the conditionally stable flow regime. They also show that a nearly non-dispersive development of the shock front between the fluids occurs when gravity segregation dominates the dispersive effect of the porous medium. Even a very small difference in density, such as that between water and brine, can suppress the dispersivity significantly.

Groundwater temperature: The temperature of groundwater is mainly a function of its mobility in an aquifer. Water temperature in the ground depends on many factors but can be related to recharge temperature, low velocity, and depth of circulation and it is over imposed on geothermal heat flow.

Custodio (1986) state that "there is generally a clear temperature change between groundwater in movement and groundwater which is almost stagnate." In very thick aquifer, the geothermal heating of water, especially when the geothermal gradient is higher than normal, may lead to thermal induced convective water movements that increase the fresh/saltwater mixing or the exchange with ocean (Kohout, 1985). He also stated that "At great depth the temperature stays constant with time but increases with depth. This is mainly because heat is being generated, and therefore the temperature rise with depth varies from place to place."

Usually, regional groundwater flow and contaminant transport studies in the vicinity of the coastal zone assume that the coastal boundary water level is equivalent to the mean sea level and that tidal- and wave induced variations have a negligible effect. As the position of a beach water table is an important factor in cross-shore sediment transport and beach stability on a sandy beach, a significant amount of work on beach groundwater has been done by researchers who have tended to concentrate on the nearshore water-table position and its transient variations. Grant (1948) noted that a high groundwater table accelerates offshore sediment transport and beach erosion, and conversely, a low water table may result in pronounced aggradation of the foreshore.

2.5 Seawater Intrusion Modeling

Many researchers have attempted to model the physics of groundwater flow processes in beaches. Dominick et al. (1971) used an implicit finite-difference numerical solution of the Boussinesq equation to simulate beach water-table response to tidal forcing. Their model was for a beach with a vertical face and, hence, this produces substantial differences from the real case of a sloping beach when predicting water tables and discharges. Fang et al. (1972) used a two-dimensional finite-element model to solve the beach water-table response to tidal fluctuations. They considered a homogeneous beach with a vertical face. Li et al. (1997) presented a boundary element model for simulating tidal induced fluctuations of the beach water table. The model solves the two-dimensional fully saturated flow equation subject to free and moving boundary conditions, including the seepage dynamics at the beach face. Baird and Horn (1996) reviewed previous works on groundwater behavior in sandy beaches and discussed how models of beach groundwater process can be improved by using a numerical approach. The aforementioned studies were concerned particularly with the relationship between tides and beach water-tables emphasizing the tidal-induced fluctuations of the water table near the shore and the consequences for processes affecting beach stability. None of them give an accurate picture of groundwater hydraulics and seepage velocity patterns near the sea boundary.

Philip (1973) showed for the first time that a sinusoidal tidal motion on a vertical beach would cause an inland water table over-height relative to mean sea level. Before him a number of investigators (e.g. Jacob, 1950) used a linearization of the Boussinesq equation to study the influence of tides on groundwater level in coastal regions and the result of this linear analysis is a constant groundwater level, at points far enough inland, in equilibrium with the mean sea level. Philip (1973) showed that for an aquifer with a horizontal impermeable base, a vertical interface between sea and land, and zero net discharge through the system, the linear analysis may be considerably in error and the groundwater levels are significantly above the mean sea level. Smiles and Stokes (1976) confirmed Philip's prediction using a Hele–Shaw experiment that models groundwater flow. The physical explanation for the phenomenon is that with high water levels the effective transmissivity of the aquifer is greater and so water flows in from the sea more readily than it flows out at low tide (Knight, 1981). Parlange et al. (1984) used second-order theory to describe the propagation of steady periodic motion in a porous medium, driven by the oscillating level of a reservoir in contact with it. Nielsen (1990) found an analytical solution to the one-dimensional Boussinesq equation for the sloping beach case using a perturbation technique. He showed that real beaches that are sloping produce an additional overheight as a result of the slope. He explained that the asymmetry of the tidal infiltration/draining process for a sloping beach results in a further rise of groundwater level. Nielsen (1990) acknowledged that if decoupling of water table and sea level occurs, the analytical solution will probably fail.

Analytical approaches to model coastal water table fluctuations usually are based on the one-dimensional Boussinesq equation or its approximations. In addition to assumptions such as uniform thickness of aquifer, uniform hydraulic conductivity and a single inland boundary condition at which water-table oscillations reduce to zero, all analytical models are based on the assumption that the exit point of the water table on the beach face is coupled with the tidal sea level.

In cases where there is a constant water level at the landward end of the aquifer, the overheight resulting from tidal pumping may have a significant effect on groundwater discharge to the sea and consequently on contaminant transport. This effect will be magnified where the aquifer is shallow and therefore the effects of tidal fluctuations are more significant. This problem has not been fully addressed, because of the limitations of the analytical solutions (Philip, 1973; Nielsen, 1990) and the assumption that the landward boundary is far inland. Also none of the previous numerical simulations (Dominick et al., 1971; Li et al., 1997) have discussed this problem. In addition, most of the previous studies, including the numerical ones, have focused on the behavior of the water table in coastal beaches. Turner et al. (1996), who showed that neglecting the effects of tidal fluctuations will lead to errors in predictions of groundwater discharge to the ocean. However, in their numerical simulation the complexities of the beach face such as mild beach slope, seepage face and the unsaturated zone were neglected.

Ataie et al. (1999) used Glover's solution to quantify the sharp interface in relation to tides, and found that tidal activity causes the saltwater interface to move inland and become more dispersed. The magnitude and the tidal activity were measured by the ratio between amplitude of tides and aquifer depth. Ataie et al. (1999) also studied the influence of a sloping beach and found that saltwater intrusion increases with sloping beaches. Saltwater intrusion is more noticeable near top of aquifer at a sloping beach.

The width of the transition zone increases with increasing dispersivities both longitudinal and transversal (Ataie et al., 1999; Korsbech, 1991). Korsbech (1991) also concluded that the penetration length decreased for increasing dispersivities. Ataie et al. (1999) found that constant freshwater flux on the landward side increases saltwater intrusion due to reduced groundwater gradient caused by over pumping.

Mathematical modeling is an efficient and inexpensive method of predicting intrusion (Bear 1979). Two basic approaches in modeling are known - a variable density model (Voss, 1984) and a sharp interface approximation. The former deals with the advective dispersion equation for solute transport. The latter, if applied in a homogeneous rock, solves the Laplace equation with a priori unknown free boundaries (interface and phreatic surface). If vertically averaged according to the Dupuit-Forchheimer (DF) concept, the free boundary problem is reduced to nonlinear partial and ordinary differential equations in a domain with two fronts, a tip and a toe (Bear 1979).

2.6 Cause of Seawater Intrusion

In coastal aquifers freshwater is hydraulically connected to seawater. Under most natural conditions the hydraulic gradient ensures the net water flow is towards the sea, which protects the freshwater. However, the gradient is usually relatively small and any excessive net withdrawal can alter the hydrostatic balance. In this situation seawater can enter the aquifer and replace the freshwater. This phenomenon, known as seawater or salt water intrusion, can have adverse and long-term impacts on coastal groundwater systems and limit their use as a supply of good quality water for human and agricultural uses. The coastal areas of the world are characterized by high populations with about 50% of the world's population living within 60 km of the shoreline (Essink, 2001). What makes saltwater intrusion different and more complex than other solute transport problems is that the variation of concentration causes water density to vary in space and time. Density differences cause freshwater to float over seawater. This effect was first addressed by Ghyben (1888) and Herzberg (1901), who empirically found that the depth to saltwater correlates with freshwater head.

Field investigations have indicated that groundwater as an important source of water and solute input to coastal waters (Lewis, 1987; Moore, 1996; Kim et al., 2003). According to Church (1996), these scientific findings challenge our understanding of coastal and oceanic chemical mass balance and ecosystem functioning. On the one hand, the seaward flow of fresh groundwater to coastal waters may carry land-generated pollutants, which constitute a serious threat to coastal ecosystems, in addition to limit the available fresh groundwater resources. On the other hand, the quality and availability of these fresh groundwater resources in coastal areas are also threatened by seawater intrusion from the seaside (Bear et al., 1999).

The contamination to the fresh water resources of coastal aquifers around the world has driven research, to gain a comprehension of the main contamination processes that occur. Several studies have been undertaken to identify natural and human induced contamination processes. These consist of internal and external processes of natural and anthopogenic origin, which promote pollution within coastal aquifers. Of these, three main processes have been identified as being specific importance in respect to the pollution of these resources:

1. Chemical weathering of the natural geological deposits.
2. Leaching of agricultural chemicals, accidental spillage/leaks and industrial processes such as mining.
3. Intrusion of saline waters.

Of the three main polluting processes highlighted above, the intrusion of saline waters has become one of the most characteristic types of water quality degradation occurring within coastal aquifers (Fetter, 1994). This process has been recognized as the major constraint in the abstraction and utilization of the freshwater resources contained within coastal aquifers.

Surface water has been the traditional primary water source for agricultural use in tropical environments. The main reason for intensive use of surface water is its easy access an associated low cost. However, increasing pressure on surface water resources generated by economic and population growth has lead to diversification of the water supply sources. Thus, during the second half of the 20th century, groundwater withdrawals have increased and currently groundwater accounts for about one-third of the world's freshwater consumption (Essink, 2001). This increase in groundwater extraction rates, often higher than natural recharge thresholds, has resulted in substantial decline in aquifer levels in many areas (Hiroshiro et al., 2006; Sethi et al., 2006; Zhang et al., 2004; Sadeg and Karahanoglu, 2001; Zhou et al., 2000).

A second effect associated with concentration differences is the mixing between the two fluids. Mixing is caused by diffusion/dispersion processes and results in some of the salt being driven seawards by freshwater. The result is a vertical convection cell formed by seawater that flows landwards at depth and disperses into the freshwater flowing zone, where salt is flushed out by the discharging freshwater flow. The equilibrium assumption of Ghyben-Herzberg is not valid because seawater flux causes an energy loss. Therefore, the depth of the seawater wedge is underestimated. Seawater movement in coastal aquifers is caused by the combination of density driven flow and hydrodynamic dispersion. This effect was first discussed by Cooper (1964). When taking into account density effects, the groundwater flow and solute transport are coupled by the presence of the density in the gravity (buoyancy) term in the momentum balance equation of fluid.

Three-dimensionality may be a critical factor that has often been ignored when analyzing seawater intrusion processes. Irregular patterns of salinity can be caused by many factors such as variable thickness formations, heterogeneity or variations in the depth of the aquifer boundaries. In homogeneous aquifers where the horizontal extension is large when compared with the thickness, aquifer topography may become critical. The effective gravity is controlled by the slope and shape of the boundaries. When the lateral slope is large, vertical flow can diminish with respect to the lateral flow. The effect of variations in hydraulic conductivity on seawater intrusion is small compared to variations in volumes of groundwater pumping and recharge. Heterogeneity in hydraulic conductivity has only a short-term effect on the inward migration of the seawater wedge and associated concentration profiles.

The extent of intrusion depends on a number of factors such as aquifer geometry and properties (hydraulic conductivity, anisotropy, porosity and dispersivity), abstraction rates, depth, recharge rate, and distance of pumping wells from the coastline (Ghassemi et al., 1993). Complex models are required to quantify these factors.

During the last three decades, numerous studies have been published dealing with various aspects of solute movement in aquifers. Modeling of seawater intrusion into groundwater systems has also received much attention and several mathematical and numerical models have been developed. These models predict the interface or transition zone between fresh groundwater of meteoric origin and seawater in the subsurface of coastal areas. Reilly and Goodman (1985) provided a historical perspective of quantitative analyses of seawater– freshwater in groundwater systems. Bear (1979) provided mathematical description of the problems related to seawater intrusion in coastal aquifers.

The development of these models was largely motivated by groundwater issues; that is, assessment of fresh groundwater reserves, and prediction of seawater intrusion—the landward or upward movement of the interface in response to groundwater exploitation practices (e.g. Volker and Rushton, 1982; Custodio et al., 1987; Ghassemi et al., 1990, 1993; Bear et al., 1999; Zhou et al., 2000; Sadeg and Karahanoglu, 2001; Gotovac et al., 2001; Paniconi et al., 2001).

Coupled simulation-optimization models of aquifer pumpage have been reported by Gorelick et al. (1984); Ahlfeld and Heidari (1994); Gordon et al. (2000); Mayer et al. (2002); Cheng et al. (2000); Mantoglou (2003); Mantoglou et al. (2004). Determination of optimum pumping rates from coastal unconfined aquifers have been based mostly on linear and nonlinear optimization techniques using the concepts of a sharp interface and the Ghyben–Herzberg approximations.

Groundwater pumping near the coast must be controlled to limit inward migration of the seawater wedge. The utilization of the freshwater resources of coastal aquifers by abstraction has been constrained by the intrusion of saline waters. These intrusions events directly pollute the resources by the interactions of the abstraction programs with the natural internal and external processes of the aquifer.

The saline or brackish groundwater which is present below fresh groundwater in coastal and deltaic areas can also be abstracted. Such abstractions cause the volume of fresh groundwater to decrease. Complete control of the interface is possible by simultaneous abstraction of fresh and saline groundwater, in mutually adjusted proportions. The effect of pumping saline groundwater is described by the extreme, theoretical, situation in confined groundwater. The saline groundwater is pumped, theoretically, in the tip of the saltwater wedge at such a rate, in this extreme theoretical situation, that the piezometric level of the fresh groundwater above the saltwater wedge is horizontal. In that case the fresh groundwater above the saltwater wedge is stagnant. There is no loss of fresh groundwater by outflow into the sea and the flow of fresh groundwater can be abstracted totally.

The abstraction programs undertaken in the utilization of the freshwater resources entail the sinking wells into the aquifer to allow direct abstraction via surface pumps. Due to these methods, alterations in the subsurface pressure gradients or "head" occur. These changes in head have a direct affect on the position and movement of the interface between the fresh and saline waters. These alterations in head and their relationship to the position and movement of the interface were characterized by the Ghyben-Herzberg principle (Domenico and Schwartz, 1990). This principle states that any loss in head produces a rise in the interface between fresh and saline waters. This rise in the interface is referred to as "upconing" and the scale of which it occurs is in direct proportion to the head loss (Price, 1998). In many cases of saline pollution within coastal aquifers, the scale at which abstraction has been undertaken has not been monitored (CDWR, 1958). This lack in monitoring has lead to over abstraction producing losses in head that generate an upconing of the interface that reaches into the abstraction zone. With the occurrence of this upconing the resources within the aquifer become polluted with the chemical changes induced by the intrusion of the saline waters.

This intrusion of the interface does not only render the resource within the aquifer useless, but it also alters the chemical composition of the groundwater (Bear et al, 1999). These compositional changes in groundwater cause secondary detrimental effects, which threaten industrially sensitive structures within coastal settings. This threat is produced from the change of the water chemistry from a fresh water environment towards a saline or marine environment.

This change in groundwater chemistry could cause acceleration in predicted corrosion times and a reduction in the overall life span of the structures (Sandberg et al, 1998). This process would therefore produce a possible hazard in the long term to sites containing such structures.

One such site is the British Nuclear Fuels low–level radioactive waste disposal facility, situated at Drigg, West Cumbria. Within this site, aquifer type media and processes have been identified (Sears, 1998). These characteristics along with the understanding of the 'active' processes of the saline intrusion would be classified as being at risk from chemical changes in groundwater. These changes in the groundwater chemistry could therefore produce acceleration in the predicted corrosion times of the facility's structure.

With the understanding of the effects that are caused through abstraction of fresh water resources from coastal aquifers and the awareness of secondary detrimental effects, monitoring of the saline intrusion phenomena is deemed essential. By monitoring the saline intrusion position over periods of activity such as tidal cycles, abstraction and recharge within the subsurface profile, mitigation programs can be put in place. These would allow abstraction of the resource to be reduced or stopped to thus allowing pressures between the two water bodies to equilibrate. The return to equilibrium of the pressures gradients would permit a consequent reduction of the upconing effect, thus preventing pollution of the resource. To undertake a monitoring exercise to enable the identification and assessment of the saline intrusion, clear objectives have to be identified and attained.

One characteristic identified within the coastal aquifers, which promotes these events, is the interface between the freshwaters of the aquifer and those of the encroaching seawaters. At this interface, a natural 'wedge' or saline intrusion occurs due to contrasts in the two water bodies' volume and density (Fetter, 1994). These are in turn influenced by, the "nature of geological formations present, hydraulic gradients, rate of withdrawal and recharge of groundwater" (Choudhury et al, 2001). The resulting pressure gradients produced by these influences is an active and balanced phenomenon that regulates the storage and flow of freshwater within the aquifers.

The natural external processes of the tidal fluctuations in seawater levels have been seen to influence these pressure gradients within the aquifer. Though the work of Ataie-Ashtisni, et al (2001) it has been shown that a quasi-steady-state rise and fall in the mean water-table position is produced in direct proportion the tidal fluctuations. These proportional changes seen in the mean water table levels are produced due to the movement of the saline intrusion into and out of the aquifer over tidal periods. During these periods, studies have shown that regions of coastal aquifers utilized as a resource for freshwater, have been seen to increase in salinity to the point of pollution (Choudhury et al 2001).

The freshwater resources affected by an increase in salinity, have been utilized as a readily available supply of freshwater. These resources are abstracted by the implementation of pumping projects. The alteration caused by abstraction of the freshwater resource produces a depression around the wellhead in the potentiometric surface known as the 'cone of depression' (Price, 1998). With the reduction of the overlying pressure produced by the freshwater, a consequent 'upconing' of the interface between the fresh and saline water towards the pumping region takes place in direct proportion to a drop in pressure overhead (Domenico and Schwartz, 1990).

The production of this upconing of the fresh and saline water interface has lead to saline pollution of resources within coastal aquifers. These affects caused by abstraction occur when monitoring of the abstraction in relation to pressure gradients and tidal fluctuations are neglected. This neglected in monitoring may result in an alteration in the gradients and the interface position, to the point that it produces an upconing event that infiltrates the resources being utilized (CDWR, 1958). The consequences of these infiltrations have been recorded in the form of saline pollution events within the freshwater resources, which have resulted in the exclusion of the aquifers as a resource for freshwater.

These adverse effects caused by the interactions of human and naturally induced processes can be prevented. This has been accomplished by the introduction of monitoring programs to identify and monitor the position of the interface between the fresh and saline waters using several different methods.

2.6.1 Saltwater upconing

Saltwater upconing describes the phenomenon where saltwater is transported vertically upward under a well in response to pumping in a fresh water aquifer underlain by saltwater. Reilly and Goodman (1987) analyzed saltwater upconing beneath a pumping well. The upconing is in response to the pressure reduction due to drawdown of the water table around the well if the bottom of the well is close to the saline water level or the well discharge is relatively high. Where the regional fresh/saltwater system is in equilibrium, a pumping well screened in the fresh water zone can cause a disturbance of this equilibrium.

Most investigators of upcoming have assumed a sharp interface between the two fluids is situation could be existing between immiscible fluids. For miscible fluids such as fresh/ saltwater, a mixing or transition zone having a finite thickness occurs.

According to Schomorak and Mercado (1969), the upcoming is in the form of an abrupt interface. Bear and Dagan (1964) made some assumptions to develop an expression that describes the upcoming of the interface as a function of time and distance from the pumping well.

These assumptions are:

1. The porous medium is homogenous and non deformable;
2. The two fluids are incompressible and separated by an abrupt interface;
3. The flow obeys Darcy's law;
4. Velocity potential satisfies Laplace's equation.

The Dagan and Bear (1968) expression for the rise of the cone below the center of the well is

$$z\,(\,r = 0\,,\,t \rightarrow \infty\,) = \frac{Q}{2\pi d K (\frac{\Delta\gamma}{\gamma})} \qquad (2.3)$$

Schomarak and Mercado (1969) deduced that the linear relation between z and Q is

$$\frac{z}{d} > (\frac{1}{3} \rightarrow \frac{1}{2}) \qquad (2.4)$$

Limited to a certain critical rise z_{cr} , values. Where, z is upcoming rise (L), Q is well discharge (L^3T^1), γ is water density (M/L^3), and d is aquifer saturated thickness

The rise accelerated and a certain critical rise z_{cr} reaches the bottom of the pumping well with a sudden jump. Bear and Dagan (1964a) stated that the maximum value of (z/d) does not exceed ¼ in order to ensure the safe and salt-free operation of coastal wells. They gave the maximum permissible pumping rate which will ensure salt-free water by:

$$Q_{max}{}^{s}\, 2\, \pi\, d z_{max} (\frac{\Delta\gamma}{\gamma}) k_z \qquad (2.5)$$

Muscat and Wyckoff (1935) studied the problem of upcoming. Their analysis of brine coning beneath oil wells is hardly applicable to water wells. Bennet et al., (1969) developed type curves for determination permissible steady pumpage for partially penetrating wells. Rubin and Pinder

(1977) made an analysis of the upcoming phenomenon with accounting of the miscibility of the two fluids. They described upcoming as migration of a sharp interface penetrated by small disturbances due to dispersion. Diersch et al. (1984) simulated upcoming using the advection-dispersion approach. They described upcoming as migration of a sharp interface penetrated by small disturbances due to dispersion.

Louis (1992) studied the upconing in an aquifer overlain by a leaky confined bed. He used Ghyben-Hezerberg relation and studied the critical pumping rate under different factors. He assumed that the critical pumping rate occurs when the interface rise is equal to 0.3 d from the distance between the well bottom and the interface. This value (0.3 d) is changed due to the changes in anisotropy (the rate of critical rise increases with decrease of anisotropy). He also concluded that the critical pumping rate decreases with the increase of well penetration degree. Reilly and Goodman (1987) stated "The dispersion phenomenon is not the reason for the upcoming phenomenon; and that the saltwater intrusion may occur without any water extraction from the aquifer, due to the effect of the dispersion phenomenon."

Diersch et al. (1984) used a finite element model to analyze the saltwater upcoming mechanism as a result of pumping and to determine the salinity of the pumped water. They took into consideration hydrodynamic dispersion and density dependencies. They found a good match between their results and those obtained by Bear and Dagan under subcritical conditions. Diersch et al. (1984) stated that their methodology is capable of describing critical or supercritical conditions. Stakebeek (1988) stated that "Upconing below abstraction wells is a very local phenomenon which can only be accurately measured by electrode cables located in the abstraction well. Measure when the upconed brackish zone is moving horizontally or vertically, when the abstraction has been finished, the use of at least two observation wells applied with electrode cables is recommended. When upcoming occurs, it takes the saltwater zone a long time (one year or more) to return to its original position". Custodio (1986) concluded that "in order to prevent saltwater upcoming in a pumping well, it is recommended to construct a second well at the same place. A careful control of fresh and saltwater discharge can maintain the fresh/saltwater interface at a convenient position between the two screens. In order to eliminate the second well, the single well must be used with two screens and two pumps, with a packer between them in order to isolate the upper and lower parts." Rushton and Redshaw (1979) studied the effect of the layered aquifers on the upconing. They investigated the movement of saline water towards a pumping well by studying the flow patterns of a single liquid with the variable hydraulic conductivity and boundary conditions of the aquifer. They applied numerical relations of a finite difference technique with a vertical mesh spacing of 10 m and logarithmic radial mesh spacing with six mesh intervals for each tenfold increase in radius (Rushton and Redshaw, 1979). The authors concluded that "in this case, the upward rise of the interface is markedly reduced. If constant, a similar layer occurring above the well between the top of the well screen and the water table the time taken for saline water to reach the well is reduced."

2.7 Cases of Seawater Intrusion

One of the earliest saltwater intrusion studies was the Biscayne aquifer in Florida, US which started in 1960 with field studies by Kohout (Kohout, 1965) and later Henry simulated the Biscayne aquifer intrusion numerically (Reilly and Goodman, 1985). Henry also posed a hypothetical aquifer intrusion problem which has become a benchmark for numerical models. The problems mainly arise in areas with a high population and a subsequently high demand for water. In China there are examples of area with a high population density. Xue et al. (1995) present a case where a freeky aquifer in China has been contaminated by seawater. The transition zone was measured to be approximately 1.5 km. In their study they developed a 3-dimensional model, which were able to model the observed concentration. Three production wells with a production of 10000 m^3/day were installed which reduced the freshwater recharge. Spain has some examples where seawater has been drawn into a freshwater aquifer. The basin of the river Verde in southern Spain has been overexploited due to the area profitable agriculture and tourism (Padilla et al., 1997). The seasonal fluctuation of rainfall and water consumption in this area is very unfortunately as the rainfall is very scarce in the summer where the water consumption is high. An affected well, 1 km inland, showed a very large variation of chloride concentration from 20 mg/l up to 5 g/l. Padilla et al. (1997) also observed an unusual speed at which the salinization and desalinization took place. In their work they used a two-dimensional horizontal model using a sharp interface approximation of the fresh-saltwater interface and made some overall mass balances over a regional aquifer.

The Djibouti aquifer in Somali is an example of over-exploitation (Housein and Jalludin, 1996). The climate in Somali is arid which enhances the processes leading to saltwater intrusion. The Djibouti aquifer has since 1960 been exploited which has lead to saltwater intrusion and chloride concentration from 15 mg/l in 1960 to 38 mg/l in 1995. Housein and Jalludin (1996) made a chemical analysis of the water quality from various wells in the area and found an increased concentration of calcium, which is attributed to dissolution of plagioclase or calcite.

Holland is example of a country where saltwater intrusion occurs because of land reclamations and lowering of the groundwater table (Stuyfzand, 1995; Appelo et al., 1987). When land is reclaimed from the sea, the land still contains many salts that have to be flushed away with the freshwater; this process may take a very long time.

Saltwater intrusion changes the groundwater chemistry via mixing, ion exchange, redox reaction and mineral dissolution/precipitation. Barker et al. (1998) examined the impact of saltwater intrusion had on a sandstone aquifer in Liverpool, and found that the governing processes near the intrusion was mixing between salt and freshwater, SO_4 reduction and calcite reduction. Further inland ions from previous saltwater intrusion, probably from the 70'ies where the freshwater abstraction peaked, were detected on the exchanger.

In 1997, the U.S. Geological Survey (USGS), in cooperation with the San Juan County Conservation District, studied the possibilities of seawater intrusion on the Island and found that 46 percent of 185 freshwater samples had chloride concentrations indicating seawater intrusion. Lopez Island lies among the San Juan Islands, an archipelago in the coastal waters of Washington State, just offshore of Seattle and of Vancouver, British Columbia. Its scenic views and relatively little precipitation have made it one of Washington's premier places to live. The Island's main freshwater source is the groundwater. Local surface water cannot be developed to meet the increasing needs of freshwater because the Island lacks of lakes and continuously flowing streams. However, there are concern that pumping more groundwater will affect its availability and quality. Because many wells are located near the shore and the recharge rates to the aquifers are low, there is a great potential for seawater intrusion

In Australia, coastal Queensland is fortunate to have extensive groundwater resources. Many rivers have well developed alluvial tracts and deltas with extensive sand and gravel aquifers. The river delta systems usually contain rich soil and were an obvious target for agricultural development, particularly plantations of sugarcane in the late 19th century. Groundwater use for irrigation commenced shortly after settlement, but it was not until the expansion of the sugar industry in the mid-20th century that we saw a rapid increase in irrigation from groundwater and the emergence of serious problems of seawater intrusion in many coastal areas of Queensland (Volker and Rushton, 1982; Hillier, 1993; Arunakumaren et al., 2000; Murphy and Sorensen, 2001; Zhang et al., 2004). Werner et al., 2008 documented that there is evidence of extensive seawater intrusion problem in Australia, most noticeable in Queensland and South Australia, but also in region of Western Australia, Victoria and Tasmania. Their studies had been used to underpin water resources management plans, which aim to control groundwater resources in these areas.

A number of geophysical studies provided evidence that seawater penetrates further inland in the deepest parts of coastal aquifers. Flores-M´arquez et al. (1998) compared the three dimensional shape of the basement of the Costa de Hermosillo aquifer (Mexico) with geochemical and geophysical data. The crystalline basement presents a structure of alternating horsts and grabens and the integration of all available data indicates that preferential pathways for seawater intrusion correspond to the lineation of basement depressions (grabens). Yet, only two-dimensional density dependent flow cross sections of the aquifer were modeled. Thus, the three dimensionality of the flux due to the irregularity of aquifer bottom was not considered.

Using the Direct Current method, Benkabbour et al. (2004) determined the depth of the bottom of the coastal aquifer of the Mamora Plain, Morocco and the lateral and vertical distribution of salinity. Seawater penetrates further inland in the proximity of the Sebou River, where the substratum is deeper. This further penetration was attributed to a greater aquifer thickness since seawater penetration inferred from Ghyben-Herzberg approximation is proportional to the square of the aquifer thickness. However, according to this assumption the total freshwater flow in each vertical section is constant. Buoyancy effects due to density differences were not taken into account. The hydrogeological literature contains no qualitative analysis of the effect of aquifer morphology on seawater intrusion, although the need of such an analysis has been highlighted in a number of heat transport studies. In these studies, variable density was taken into account and the effect of aquifer slope in the heat plume movement and velocity was addressed. Bachu (1995); Bachu and Karsten (2002) studied density driven flow in sloping aquifers, applying the results to two sedimentary basins: Alberta (Canada) and Los Llanos (Colombia). Malkovsky et al. (2002) highlighted the importance of natural convection in a heat-generating liquid waste plume in a sloping aquifer, which could cause acceleration as well as a slowing down of the plume depending on the parameters of the system. The role of the aquifer slope has also been addressed in brine movement in continental basins. Lahm et al. (1998) studied the role of salinity, derived variable, density flow in the displacement of brine from a shallow, regionally extensive aquifer and argued that density dependent flow causes a decrease in groundwater velocities and a reorientation of local flow directions of the aquifer within the mixing zone.

Assouline and Shavit (2004) studied the effects of management policies, including artificial recharge, on salinization in a sloping aquifer in Israel. Although seawater intrusion processes were not taken into account, the importance of the thickness variations in the sloping aquifer in the salinization process was addressed. There has many much research on to saltwater intrusion in regions such as: in the Mediterranean coast of Israel by Shamir et al. (1984), in the Waialae aquifer of southern Oahu, Hawaii by Essaid (1986), Emch and Yeh (1998), in southern Oahu, Hawaii by Souza and Voss (1987), in Hallandale, Florida by Andersen et al. (1988), in the Yun Lin Basin, Taiwan by Willis and Finney (1988), in the Soquel-Aptos basin, Santa Cruz County, California by Essaid (1990a, 1990b), in the Jakarta Basin by Finney et al. (1992), in the Dutch coast by Essink (1998), etc. Amongst these, the aquifer systems are characterized by either single layer (unconfined) or multiple layers with varying hydraulic properties.

Luc et al. (2008) discussed the evolution of the seawater distribution around the Zwin estuary mouth which is modeled for a period of about five centuries. The evolution is simulated by the 3D density depended groundwater flow model MOCDENS3D (Lebbe & Oude Essink, 1999). They finalized that the historical evolution of seawater distribution around the Zwin estuary mouth results in a large number of different inverse density problem. Larabi et al. (2008) showed that the coupled flow and transport code (SEAWAT) was applied to the study seawater intrusion in the Rmel coastal aquifer in both qualitative and quantitative aspects. The result showed that seawater intrusion started in 1992 in the northern sector due to intensive pumping from the wells and reduced the recharge. The model is also applied to test the aquifer response to three planning scenarios for a period of 20 years. Yager and Misut (2008) presented the variable density/viscosity simulations were conducted to investigate processes controlling the migration of brine and saline water in the aftermath of a salt mine collapse, which threatens to contaminate an overlying glacial- drift aquifer. Model result indicates that movement of brine and saline water is controlled by displacement of brine from the mine and mixing of water from bedrock fracture zones.

According to Yang (2008) a three-dimensional, finite- element model capable of predictive simulation of the effects of reservoirs on the groundwater system for the Ping Tung Plain in southwestern Taiwan was developed. The finite-element numerical model FEMWATER was selected as an appropriate model for simulating feature at the Plain. The result of numerical modeling indicate that the advancing chloride front will move north in more permeable sand and gravel near the mouth of the Kaoping River. The chloride concentration will increase substantially over the next 50 years.

2.8 Groundwater Modeling

2.8.1 Introduction

Models are helpful in simplifying and schematizing real systems using a set of assumptions. Models can then be applied to understand the flow system and the relationships with other systems and to get answers to different exploitation or preparation action. Modeling of saturated flow in porous media is generally straightforward with few conceptual or numerical problems. The applicability of flow models involving two or more liquids in porous media are even more complicated in terms of the process and parameters involved. Nevertheless, such models have been applied successfully.

Mass transport is controlled by a variety of physical, chemical, and biological processes. Quantitative descriptions of the processes concerned with mass transport (advection, diffusion) are today well understood. Various types of models exist, including physical, analytical and numerical models. In modeling two systems are considered: prototype (the real system) and the analog system.

Physical model, except the electrical ones are mostly laboratory device and experimental apparatus, though they have been used for the solution of practical problem. Among the different possible models and analogs (Custodio and Llamas, 1976), only a few are used to study problems directly related to coastal aquifers. Hele-Shaw analog, especially the vertical ones, are the best suited. Bear and Dagan (1964) used a Hele-Shaw model to compare the results of an approximate solution for the movement of the interface in confined aquifer. The movement was caused by a sudden change in the rate of seaward flow of fresh water. Bear (1979) used a Hele-Shaw model to compare the results of approximate solutions for the shape of the interface in which a thin semiprevious layer is present, and for the extent of the freshwater region above it under steady-state conditions.

Through the process of model calibration and verification the values of the different hydrogeologic conditions are varied to reduce the disparity between the model simulations and field data, and to improve the accuracy of the model. The model can also be used to simulate possible future changes to hydraulic head or groundwater-flow rates as a result of future changes in stresses on the aquifer system. These are referred to as "predictive simulations." Monitoring of hydraulic heads, hydraulic gradients, and groundwater-flow rates (where appropriate) will be required to support predictive simulations using groundwater-flow models.

Fate-and-transport models simulate the migration and chemical alteration of contaminants as they move with groundwater through the subsurface. Fate-and-transport models require the development of a calibrated groundwater-flow model or, at a minimum, an accurate determination of the velocity and direction of groundwater-flow based on field data. The model simulates the movement of contaminants by advection and diffusion, spread and dilution of contaminants by dispersion, removal, or release, of contaminants by sorption, or desorption, of contaminants onto, or from, subsurface sediment or rock, addition or removal of contaminants by contaminant sources or sinks, and chemical alteration of the contaminant by chemical reactions which may be controlled by biological processes or physical-chemical reactions.

As with groundwater-flow models, fate-and-transport models should be calibrated and verified by adjusting values of the different hydrogeologic or geochemical conditions to reduce the disparity between the model simulations and field data. This process may result in a re-evaluation of the model used for simulating groundwater-flow if the adjustment of values of geochemical data does not result in an acceptable comparison with contaminant migration direction or rate. Predictive simulations may be made with a fate-and-transport model to predict the expected concentrations of contaminants in groundwater as a result of implementation of a remedial or corrective action. Monitoring of groundwater chemistry will be required to support predictive simulations using fate and transport models.

2.8.2 Numerical models

The equations that describe the groundwater-flow and fate-and-transport processes may be solved using different types of models. Some models may be exact solutions to equations that describe very simple flow or transport conditions (analytical model), some models may use exact solutions of equations that described sources and sinks and other parameters that are solved together using the superposition principle (analytic element model), and others may be approximations of equations that describe very complex conditions (numerical models). Each model may also simulate one or more of the processes that govern groundwater-flow or contaminant migration rather than all of the flow and transport processes.

Numerical models use approximations to solve differential equations describing groundwater flow or solute transport. They are capable of solving more complex equations that describe groundwater flow and solute transport. These equations generally describe multi-dimensional groundwater flow, solute transport and chemical reactions. Numerical models use approximations to solve the differential equations describing groundwater flow or solute transport.

The complete description of transport in porous media in mathematical models is made up of partial differential equation or a system of several partial differential equations together with initial and boundary conditions. To solve those equations, one can use analytical or numerical methods. Because of the irregular shape of the boundary, the spatial variability of the coefficients appearing in the equation and in the boundary condition, the nonuniformity of the initial condition and the nonanalytic form of the various source and sink terms, analytical solutions are virtually impossible, except for relatively simple problems. Solution of most problems can be obtained only by numerical methods. The oldest numerical method is the method of finite differences (Southwell, 1940; Forsythe and Wasow, 1960; Fox, 1962; Kantorovich and Krylov1964). In this method, the partial derivatives appearing in the basic differential equation are replaced by an algebraic equivalent, with a quotient of two finite differences of the dependent and an independent variable replacing the differential quotient.

The second very powerful numerical method is the finite element method. An elementary way of presenting this method is used in structural mechanics, where the element are the actual parts of a structure like the beams and columns in the framework of a building, or grid of beams in the floor of a bridge. The deformation of each element is then expressed in terms of the forces acting upon it at the two ends. This enables us to express the displacement of each nodal point in terms of those of the neighboring nodes and the deformation of the connecting elements. The final system of equation is obtained from conditions of equilibrium at each side.

The system of the linear equations obtained in the finite element method has the same structure as in the finite difference method. Actually, the two methods are very similar and from certain problem it has been shown that they can be considered as two representation of the same model. The finite element method is somewhat more flexible than the standard form of the finite difference method (Bear and Verruijt, 1987). Four major methods that can be used to solve the solute transport equation are: 1) the finite different method; 2) the finite element method; 3) the random walk method (Uffink, 1990); and 4) the method of characteristics (Konikow and Bredehoeft, 1978). In the last method, the particle tracking technique is also employed to solve the advective transport and either the finite difference or finite element approach is used to solve the dispersive equation.

The approximations require that the model domain and time be discretized. In this discretization process, the model domain is represented by a network of grid cells or elements, and the duration of the simulation is represented by a series of time steps. Numerical solutions are much more versatile and with the widespread availability of computers, are now easier to use than some of the more complex analytical solutions (Anderson and Woessner, 1992). The accuracy of numerical models depends upon the accuracy of the model input data, the size of the space and time discretization (the greater the size of the discretization steps, the greater the possible error), and the numerical method used to solve the model equations. During a sensitivity analysis, calibrated values for hydraulic conductivity, storage parameters, recharge, and boundary conditions are systematically changed within a previously established plausible range. Figure 2.3 outlines the steps involved in the solution of a groundwater model.

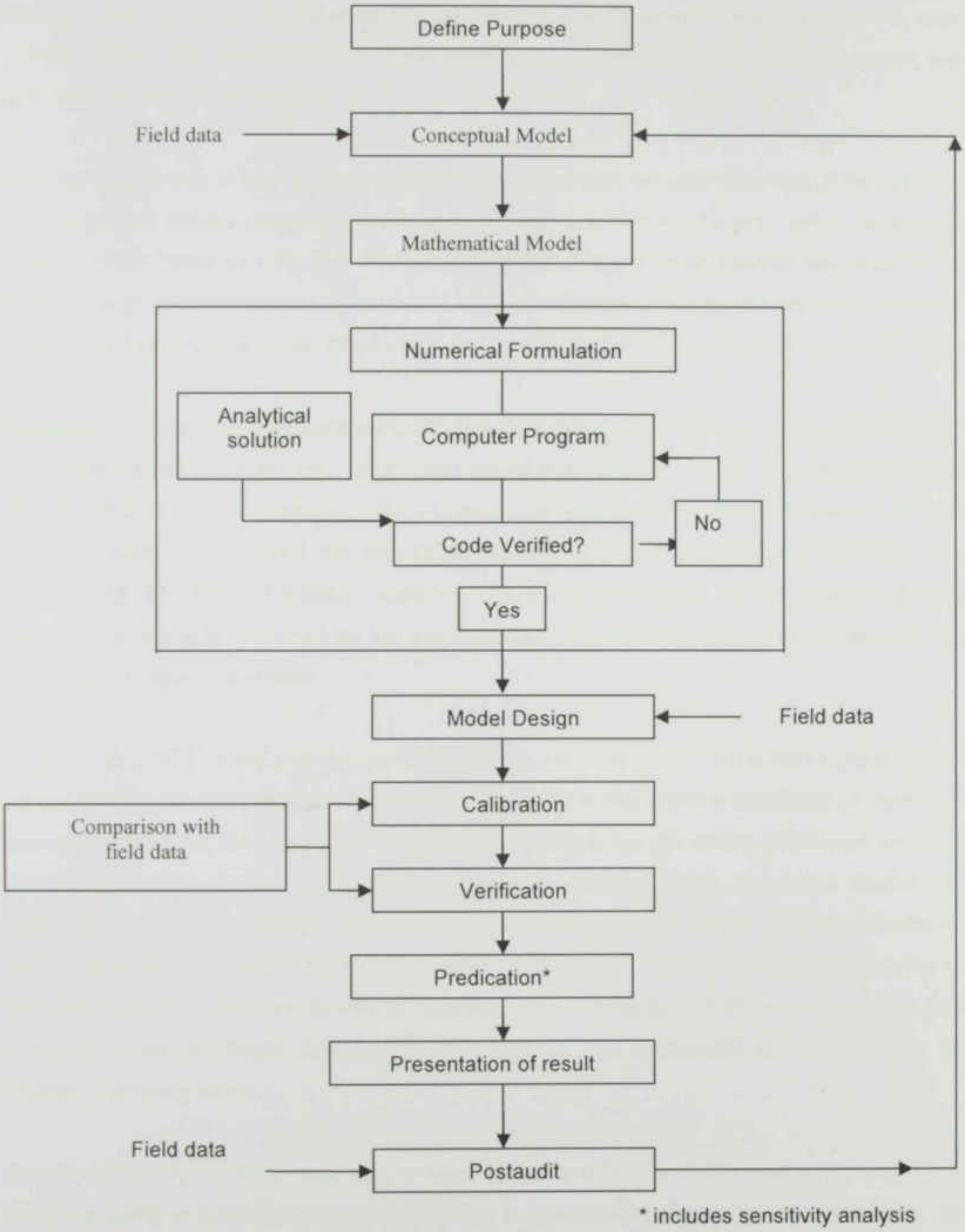

Figure 2.3. Steps in groundwater model (Anderson and Woessner, 2002)

Numerical models are used to simulate complex two- or three-dimensional groundwater-flow and solute-transport problems, steady-state or transient groundwater flow or solute transport, assess regional- or local-scale flow or transport, estimate fluxes at simple or complex hydrogeologic boundaries, and simulate problems which cannot be adequately described using analytical models.

Sharp-interface models are based on the Ghyben-Herzberg principle that assumes a sharp interface between fresh and saline groundwater, which is able to represent the actual situation. The one-fluid models are based on freshwater dynamics only. These were used by Glover (1959), Henry (1959), Shamir and Dagan (1971), Volker and Rushton (1982) and Ayers and Vacher (1983). It assumed that the water table and the sharp interface maintain continuous equilibrium and that the salt water is static.

Alternatively, the two-fluid method may be used, in which coupled freshwater and salt water flow equations are solved simultaneously (e.g. Wilson and Sa da Costa, 1982; Contractor, 1983; Essaid, 1986; Willis and Finney, 1988). Most coupled two-fluid sharp-nterface models are limited to a quasi-three-dimensional single layer or a two-dimensional vertical section; however, Essaid (1990a, b) developed a quasi-three-dimensional model that allows for multiple aquifer layers.

Saltwater dynamics are important during the transient period; hence, a two-fluid model may be more appropriate for examining short-term responses (Essaid, 1986). Pinder and Cooper (1979) developed a numerical solution for the movement of the transition zone between fresh/saltwater in porous media. They solved the two-dimensional problem by considering both the equation of motion and the solute transport equation. They assumed that: a) the release of water from storage has a negligible effect on the movement of the interface; and b) the dispersion coefficient is constant in space and time.

Segol et al. (1976) analyzed the general problem described by Pinder and Cooper (1970). They solved pressures and velocities simultaneously in order to generate continuity of velocity between elements, and then they used the velocity field to solve the advection-dispersion equation to find the concentration. Bennett et al. (1969) made an electric-analog simulation model for a fresh water aquifer of variable depth with uniform varying penetration depth. They developed a solution with graphs for the design of skimming wells. Rivera and Ledox (1988) described the motion of saltwater and freshwater as a case of two-phase flow. Simulations were made both in steady and unsteady state conditions to establish the position and movement of the interface toe under different pumping rates.

Rivera and Ledox (1988) originally examined the position and the movement of the saltwater interface using a simplified model combining a numerical solution with an analytical one. They found that the numerical solution could give acceptable results for the position of the interface in a vertical cross section in steady state and that it was in agreement with an analytical solution. Stakelbeek (1988) used two different groundwater programs to simulate the movement of brackish zone under a pumping well. One of these programs neglected the density differences on flow. He concluded that when upconing is simulated, the influence of the distance from the brackish zone (transition zone) to well filter on upconing has to be calculated and the vertical flow below the wells has to be considered. The degree to which the fresh/saltwater interface is dispersed could be important to a study.

Henry (1964) developed the first solution for the steady salt distribution in an idealized aquifer, taking into account dispersion, based on the assumption of a constant dispersive mechanism. Rubin and Pinder (1977) used a dispersion tensor which was linearly dependent on velocity to analyze upconing due to well abstraction. They showed that the dispersive mechanism actually comprises (a constant) molecular diffusion term and a velocity-dependent mechanical dispersion term. Tompson and Gray (1986) developed a more general representation of dispersion in the macroscopic transport equation.

Reev (1988) used a numerical model to investigate, both in the presence and absence of well abstraction, the effect of representing the hydrodynamic dispersion tensor as constant, linearly-dependent on velocity in a cross-section perpendicular to the coast, and concluded that the effect of the increasing the dispersion coefficient is similar in both the abstraction and non abstraction cases. He stated that in the non abstraction case the greater inland penetration of saltwater occurs at the highest dispersion coefficient, while in the abstraction case it occurs at the lowest dispersion coefficient. When the abstraction is simulated, the results indicated that the low dispersion coefficient becomes significantly different as the magnitude of the dispersion coefficient increases.

Uffink (1990) described the development of a transition zone in two-dimensional flows, starting from the known sharp interface. For calculating the velocity distribution, he used a similarity between the boundary layer problem in hydrodynamics and the flow of the groundwater near an interface. From this velocity distribution, the dispersion was determined by means of the random walk method.

A narrow transition zone is simulated by means of a finite-element model of two-dimensional density flows by Voss and Souza (1986). They stated that a narrow transition zone amplifies any inconsistencies, inaccuracies or instabilities inherent in a given simulation model. They likely sources of simulation error are threefold:

- Vertical discertization is typically too large for the desired level of transversal dispersion.
- Inconsistent approximations of terms involved in the fluid velocity calculation can lead to the large artificial velocity and dispersion components in a simulation.

- The process of flow driven by density differences in the fluid may not be accurately represented by the simulation.

Voss and Souza (1986) added that the following modeling approach rectifies these difficulties.

- Vertical discertization must be in the order of the transversal dispersivity value when flow is predominantly horizontal. Transport simulation studies should always begin with a steady-state simulation for the case of zero transversal dispersion to check for the sharpest transition zone possible with a given mesh and flow field.
- A numerical method that gives a consistent velocity approximation must be employed. The standard Galerkin finite-element method gives an inconsistent velocity approximation that can generate overwhelming artificial velocities in a simulation.

Bruggeman (1990) described a general method for calculating a transition zone in three-dimensional flow, making use of the so-called pressure generation. He assumed the following:

1. The aquifer is homogenous and isotropic
2. The viscosity of the fluid is constant
3. The density of the fluid is related to the concentration of the solute
4. The soil skeleton and the fluid are incompressible

2.9 Governing Equations of Solute Transport in Groundwater

When problems involve miscible fluids, it is necessary to solve the solute transport equation. To solve a solute transport problem one has to solve the groundwater flow and a solute transport equation. The governing equations of the dispersion zone and the flow pattern in coastal aquifers subjected to saltwater intrusion under the unsteady state conditions are (Bear and Veruijt, 1987):

1. The general Darcy equation for ground water flow,

$$q = -\frac{k}{\mu}(\nabla p + \rho g \nabla z) \qquad (2.6)$$

where q the specific discharge vector (LT^{-1}) , k is the permeability tensor (L^{2}) , μ is the dynamic viscosity ($ML^{-1}\ T^{-1}$), p is pressure ($ML^{-1}\ T^{2}$), ρ is the fluid density (ML^{-3}), g is the gravitational acceleration (LT^{2}) and z is a space coordinate (L) . Substitution of

$$\psi = \frac{p}{\rho_f g} + z, \quad k = \frac{k\rho_f g}{\mu}\ and \quad \rho_r = \frac{\rho}{\rho_f} - 1$$

into the general Darcy equation, then equation (2.6) can be written as,

$$q = -K(\nabla \psi + \rho_r \nabla z) \qquad (2.7)$$

where K is the hydraulic conductivity tensor (LT^{-1}), ψ is the equivalent hydraulic head (L), and ρ_r is the relative density (dimensionless).

2. The basic fluid continuity equation or the mass balance equation for the fluid which can be written as:

$$\frac{\partial n\rho}{\partial t} = -\nabla \rho q + R\rho * + p\rho \tag{2.8}$$

Where, n is the effective porosity(dimensionless), R and P are the recharge and pumping rates per unit volume of aquifer medium, respectively (T^{-1}), and ρ^* is the density of the recharged water (ML^{-3}).

3. The hydrodynamic dispersion equation or the mass balance equation for the salt ions can be written as,

$$\frac{\partial nc}{\partial t} = -\nabla(qc - nD\nabla C) + RC * - PC \tag{2.9}$$

where C is the solute concentration in mg/l. In the above equation, the effect of adsorption on dispersion process and solute transport is neglected. For two- dimensional vertical cross-calculated as follows (Bear, 1979):

$$D_{XX} = \alpha L \frac{V_X^2}{|V|} \alpha T \frac{V_Z^2}{|V|} + D^*$$

$$D_{ZZ} = \alpha T \frac{V_X^2}{|V|} \alpha L \frac{V_Z^2}{|V|} + D^* \tag{2.10}$$

$$D_{XZ} = D_{ZX} = (\alpha L - \alpha T) \frac{V_X V_Z}{|V|}$$

4. A constitutive equation relating fluid density to solute concentration, which is expressed as:

$$\rho = \rho_f + a(c - c_f) \tag{2.11}$$

where c_f is the freshwater (reference) concentration (ML^{-3}), and a is known constant (dimensionless) which can be calculated as

$$a = \frac{\rho_s - \rho_f}{C_S - C_f} \tag{2.12}$$

where ρ_s is the seawater density (ML^{-3}), and c_s is the seawater concentration (ML^{-3}). A linear relationship for the density and concentration is assumed in equation (2.11). Baxter and Wallace (1916) developed an empirical relation which relates the salt concentration to fluid density as:

$$\rho = \rho_f + (1 - E)\rho_S \tag{2.13}$$

where E is a constant (dimensionless)and has a value of 0.3 for concentrations as high as seawater. Examination of the main equations (2.7, 2.8, 2.9 and 2.11) reveals that there are four unknowns (ψ, V, C and ρ) in four equations. However, these equations can be combined into two nonlinear partial differential equations in only two variables, namely, the hydraulic head ψ and the concentration C.

Simulation of flow involving water with high TDS or higher or lower temperatures requires that the effects of density be included in the model. This is the case of density-dependent flow of miscible fluids that may be necessary to solve three models - flow, solute transport, and heat transport. Models that simulate density-dependent flow require an initial pressure and density distribution. At the beginning of a time step, these initial values are used to generate the first approximation of the flow field. The resulting head values are input to the transport models, which redistribute solute and/or temperature. A new density distribution is calculated from the transport results, ending the first iteration of the first time step. The second iteration begins with the substitution of the newly calculated densities into the flow model. Iteration is continued until closure is attained. This process is repeated for all time steps (Anderson and Woesner, 1992).

2.10 Seawater Intrusion Modeling

Two general approaches have been used to analyze saltwater intrusion in coastal aquifers: the disperse interface and sharp interface approaches. The disperse interface approach explicitly represents a transition zone that is a mixing zone (brackish water) of the freshwater and salt water within an aquifer due to the effects of hydrodynamic dispersion. In the transition zone there is a gradual change in density. The freshwater and saltwater are considered to be two immiscible fluids of different constant densities.

The analysis, simulation and management of coastal aquifers are usually based on two assumptions:

- Aquifer parameters, especially hydraulic conductivity, remain constant throughout the time-span under consideration.
- The saltwater/freshwater interface is a movable boundary, modified by the effects of diffusion and hydrodynamic dispersion (Volker et al., 1982).

Since the beginning of the twentieth century, exclusive studies about saltwater intrusion have been performed and different mathematical models have been used to investigate this phenomenon quantitatively. The saltwater intrusion phenomenon in groundwater systems has been conceptualized by two general approaches: the sharp interface approach and the dispersed interface approach. In the former it is assumed that the saltwater and freshwater are immiscible fluids separated by a sharp interface. In the latter a transition zone of mixed salt and freshwater is considered to be present at the interface. In this approach, the diffusion and hydrodynamic dispersion effects, density dependent fluid flow and solute transport are incorporated. A historical perspective of salt-water intrusion is presented by Reilly and Goodman (1985). The models of Volker (1980), who employed the finite element method for the saltwater intrusion problems in coastal confined and unconfined aquifers, Volker and Rushton (1982) and Taigbenu et al. (1984) who applied the boundary integral method, and the models of Mercer et al. (1980), Polo and Ramis (1983), Ledoux et al. (1990), who used the finite difference method are based on the first approach. Also the recent model of Masciopinto (2006) is based on the sharp interface approach. Numerical models based on the dispersed interface approach have been used extensively to investigate different aspects of seawater intrusion by including the density difference between seawater and fresh groundwater (Segol et al., 1975; Volker and Rushton, 1982; Frind, 1982; Voss and Souza, 1987; Konikow and Arevalo, 1993). Ataie-Ashtianiet al. (1999a) studied the effect of tidal oscillations on seawater intrusion in coastal aquifers based on the dispersed interface approach. It was noted that the effect can be significant on near-shore groundwater hydrodynamics and saltwater intrusion, especially for a low relief beach. Custodio (1987) mentioned that in many real situations, such as slow freshwater flow, stresses caused by tidal oscillations and recharge events, and enhanced dispersivity by macroscopic heterogeneities, the sharp interface approach is a crude one. The sharp interface approach is computationally less demanding in comparison to dispersed interface approach. Ataie-Ashtiani et al. (1999a, b) presented a numerical model for simulation of groundwater flow in coastal aquifers that could handle tidal fluctuations and the seepage-face condition at the seaward boundary. In their model the seawater intrusion in coastal aquifer was simulated using dispersed interface approach. However, the model can be used either for simulation of contaminant transport or the seawater intrusion. Besides, solving density-dependent flow for dispersed interface approach is computationally demanding and therefore it imposes severe limitations on the scale of an aquifer considered for simulation. Also, Ataie-Ashtiani et al. (2001, 2002) studied the influence of tidal fluctuation effects on groundwater dynamics and contaminant transport in unconfined coastal aquifers. In their studies the seawater intrusion interface into the coastal aquifers was not considered.

2.10.1 SUTRA model

One of the most commonly used models for simulation of density-dependent groundwater flow is a two-dimensional, finite-element model by Voss (1984). The computer code named SUTRA (Saturated–Unsaturated Transport) is a product of the US Geological Survey and has become the widely accepted variable-density groundwater flow model throughout the world (Essink, 2003). SUTRA (Voss, 1984), in conjunction with the Argus-One Graphic User Interface, is generally chosen as the basis for numerical modeling because of its ability to solve density-dependent groundwater flow and variably saturated flow, and also because it is readily available in source code form. This model implements a hybridisation of finite element and integrated finite difference methods employed in the framework of a method of weighted residuals. In the model, standard finite element approximations are employed only for terms in the balance equations that describe fluxes of fluid mass, solute mass and energy. All other non-flux terms are approximated with a finite element mesh version of the integrated finite difference methods. The hybrid method is the simplest and most economical approach, which preserves the mathematical elegance and geometric flexibility of finite element simulation, while taking advantage of finite difference efficiency. The finite element method allows the simulation of irregular internal discretisation. This is made possible through use of quadrilateral elements with four corner nodes (Voss, 1984; Ataie-Ashtiani et al., 1999). The SUTRA model has been successfully applied to solve seawater intrusion problems (e.g. Voss and Souza, 1987; Souza and Voss, 1987; Bush, 1988; Ghassemi et al., 1990; Kacimove et al., 2008).

Terry (2008) examined a density-dependent solute transport model to evaluate potential response to varying pumping stressing the upper Floridan aquifer in southwest Florida. Two-dimension, axisymmetric SUTRA models and a Monte Carlo statistic approach are used to evaluate upconing potential. The model was used to examine an upconing response to a pumping stress in a brackish water aquifer.

Gingerich (2008) used a three-dimensional solute transport (3-D SUTRA) computer code to simulate the freshwater and the underlying brackish-water transition zone and incorporates hydrologic feature such as valley-fill barriers and the sediments that form a caprock and a barrier between the lavas of the West Maui and Haleakala Volcanoes. The code is capable of simulating variable-density groundwater flow solute transport in heterogeneous, anisotropic aquifers. He concluded that the groundwater flow model is useful as a tool to forecast the effects of future groundwater withdrawal and changes in recharge distributions. Sherif and Kacimov (2008) applied SUTRA to verify new methodology for controlling the seawater intrusion and enhancing the quality of the groundwater in the coastal aquifer. They concluded that seawater intrusion problems could be controlled through proper pumping of fresh/saline/brackish groundwater from the coastal zone.

Kumar et al. (2007) described the use of SUTRA to define the current and potential extent of seawater intrusion in the Burdekin Delta under various pumping and recharge conditions. A 2D vertical cross-section model, which accounts for groundwater pumping and recharge, was developed for the area. The Burdekin Delta aquifer consists mainly of sand and clay lenses with granitic bedrock. The model domain used vertical cross-sections along the direction of groundwater flow. The initial conditions used in the model are based on land use prior to agricultural development when the seawater wedge was in its assumed natural state. They demonstrated the effects of variations in pumping and net recharge rates on the dynamics of seawater intrusion. Simulations were carried out for a range of recharge, pumping rates and hydraulic conductivity values. Modeling results showed that seawater intrusion is far more sensitive to pumping rates and recharge than to aquifer properties such as hydraulic conductivity. Analysis also shows that the effect of tidal fluctuations on groundwater levels is limited to areas very close to the coast. Tidal influences on saltwater intrusion therefore can be neglected when compared with the effects due to groundwater pumping. They recommend that a 3D model which accounts more realistically for the actual pumping regime and key spatial and temporal features of the lower Burdekin be used as a follow up to this study to refine their understanding of seawater intrusion in order to help optimize location and management of groundwater withdrawals,, including the distance from the coast in which no groundwater should be extracted.

2.10.2 Examples of other seawater intrusion models

Many other codes are available, including, FEFLOW (Diersch and Kolditz, 1998), ROCKFLOW (Kolditz et al.,1998), HST3D (Kipp, 1986), TVDT3D (Ackerer et al., 1999), METROPOL (Sauter et al., 1993), MVAEM (Strack, 1995), SWICHA (Huyakorn et al., 1987), SWIFT (Ward, 1991), CODESA (Gambolati et al., 1999) and d3f (Fein and Schneider, 1999). Improvements in computer speed have facilitated the construction of adequately refined grids to reduce problems of numerical dispersion, which accounts for the emergence of 3D benchmark problems (Johannsen et al., 2002; Oswald and Kinzelbach, 2004) for density dependent codes. Variable density 3D models of real cases are also becoming increasingly frequent (Essink, 2001; Xue et al., 1995; Sciabica et al., 1994; Gambolati et al., 1999; Paniconi et al., 2001; Gingerich and Voss, 2002; Milnes and Renard, 2004).

CHAPTER 3

GEOLOGICAL, HYDROGEOLOGICAL AND GEOPHYSICAL INVESTIGATION IN WADI HAM

Chapter 3. Geological, Hydrogeological and Geophysical Investigations in Wadi Ham

3.1 Introduction

Groundwater resources constitute about 81% of the total water supply in the UAE. The agriculture development in UAE is mainly dependent on the availability of groundwater. Many productive farms are located in the coastal areas of the Emirates of Ras Al Khimah and Fujairah. Groundwater levels have declined significantly during the last decade due to the lack of rainfall. Current annual rainfalls are significantly lower than the average recorded rainfall over the last 50 years. On the other hand, groundwater salinity has also increased during the last decade due to seawater intrusion problem. As a result, many farms have been abandoned.

Wadi Ham is located in the Emirate of Fujairah. The valley floor is a flat-gravelly plain with triangular shape broadening to the sea and draining the surrounding mountains. It rises from sea level at Fujairah to approximately 100 m above sea level; to the northwest. Few hills are scattered in different parts of the wadi. These hills subdivide the wadi into communicative zones. Along the coast, the inward land becomes a river terrace or alluvial plain. It is locally dissected by stream channels filled with cobble and gravel. The number and the depth of channels decrease towards the coast. The wadi plain is used for extensive agricultural activities. Some new industries have commenced in the vicinity of the coastal zone. Figure 3.1 provides a remote sensing image for the catchement area of Wadi Ham.

3.2. Geological Setting

Geology is defined by the physical and chemical properties and distribution of local rocks, as well as prevailing tectonic conditions (Toth, 1970). Geology influences the flow paths of groundwater because water will flow more readily through materials of higher permeability for a given hydraulic gradient. Gravel and sand mixtures have much higher permeability than silt and clay mixtures. The Emirate of Fujairah may be regard as the Land of wadis and dams in UAE, where more than thirty major Wadis are formed among many small wadis. The geologic map of Fujairah Emirates and the tectonic profile of the Oman Mountains are presented in Figures 3.2 and 3.3. The limestone of Northern Oman Mountains are sediments deposited on the autochthonous shelf (unit a). To the south these are cut off by the Dibba fault zone, which shows large olistholite blocks mixed with deep-sea sediments of the Hawasina Complex (unit c). To the south, a metamorphic sheet appears (unit d), overlain by the ocean floor rocks of the Semail Ophiolites (unit f). This consists of a several kilometers thick slice of igneous basic and ultrabasic rocks, presumed to have originated as part of an upper mantle sequence beneath an ocean floor. Six major lithologic units are distinguished, of which three are exposed in Fujairah: Gabbros, transition Peridotites-Gabbros, and Peridotites. The ophiolite sequence is jointed and fissured as it has been subjected to faulting. In the wadi floor it is overlain by recent to Pleistocence Wadi Gravels ranging in thickness from 22 to 57 meters (Sherif et al., 2005).

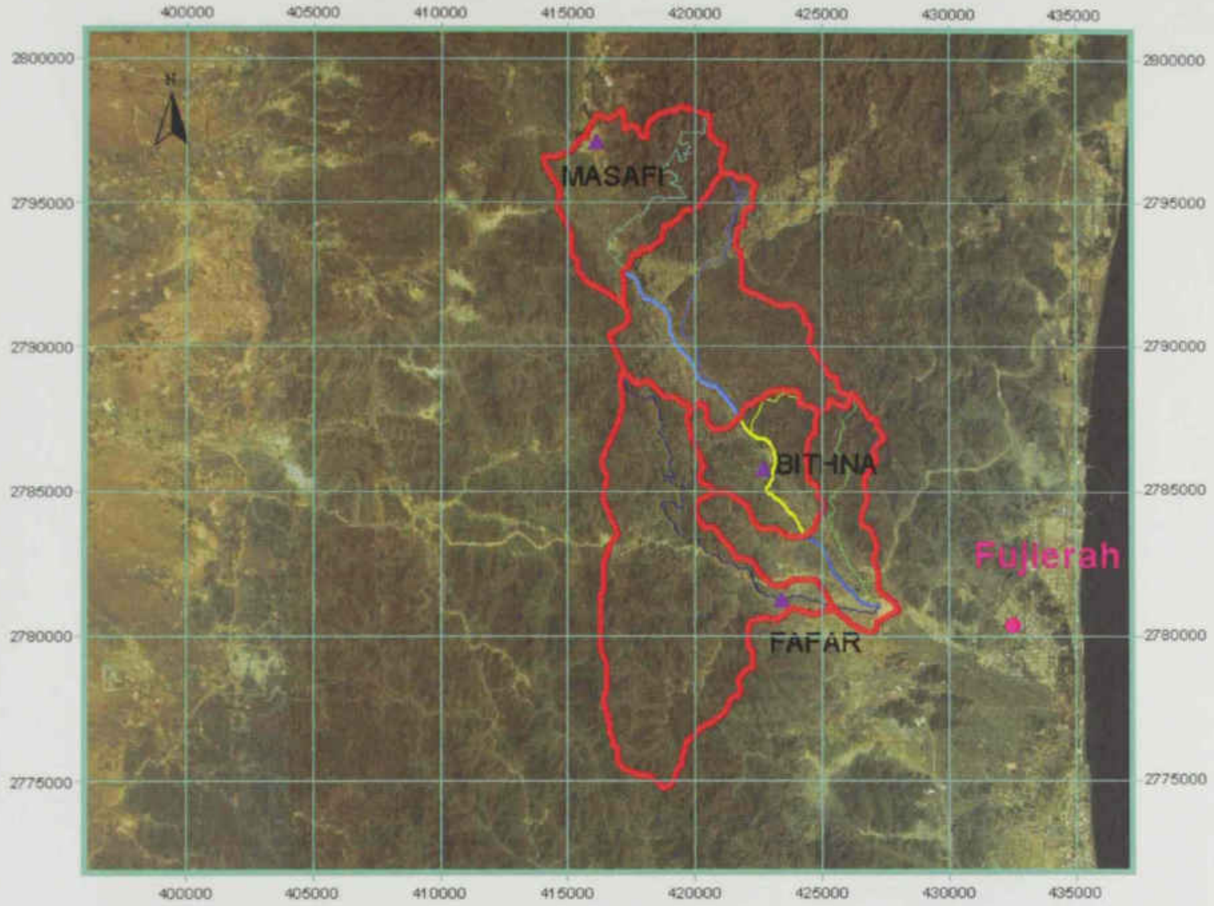

Figure 3.1. A remote sensing image for the catchment area of Wadi Ham.

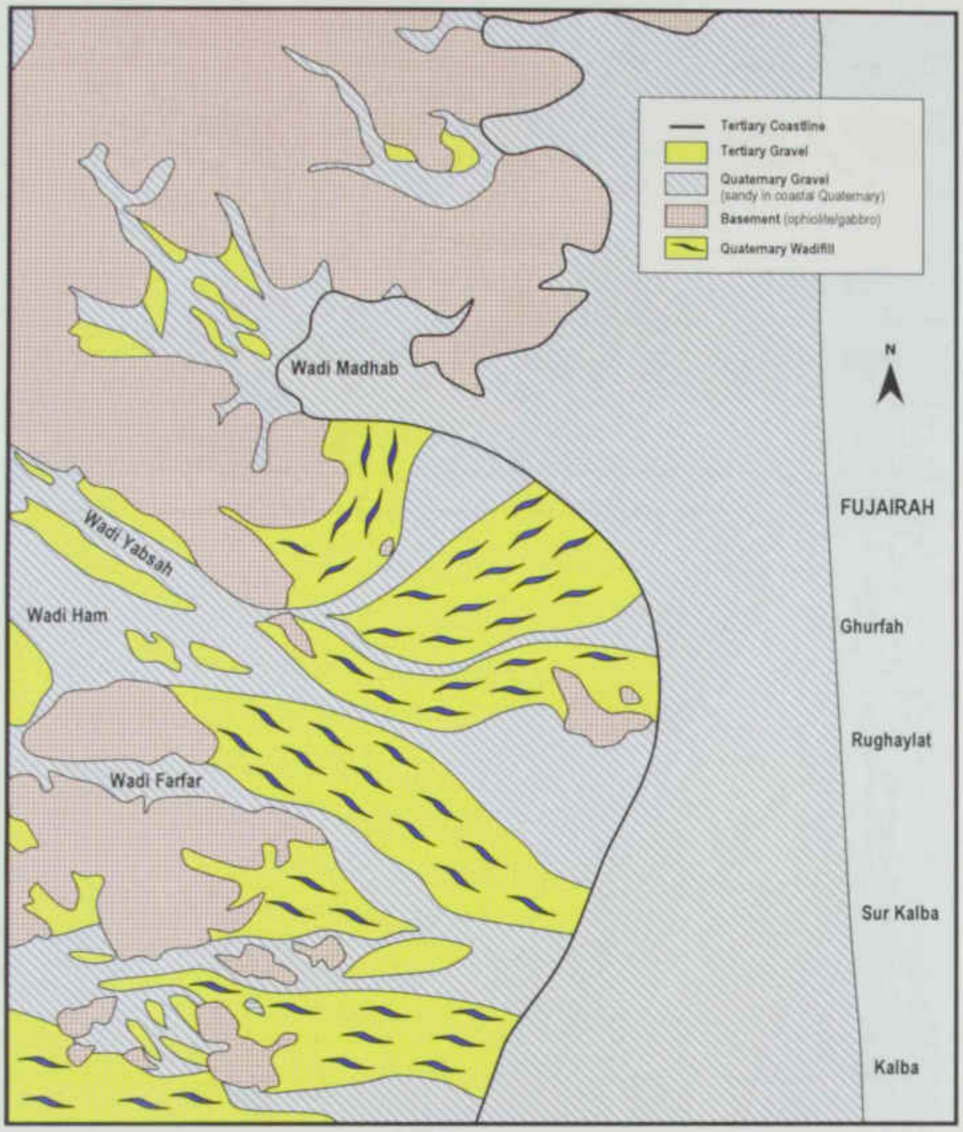

Figure 3.2. A sketch for the geological map of Fujairah (revised after Geoconsult and Bin Ham, 1985).

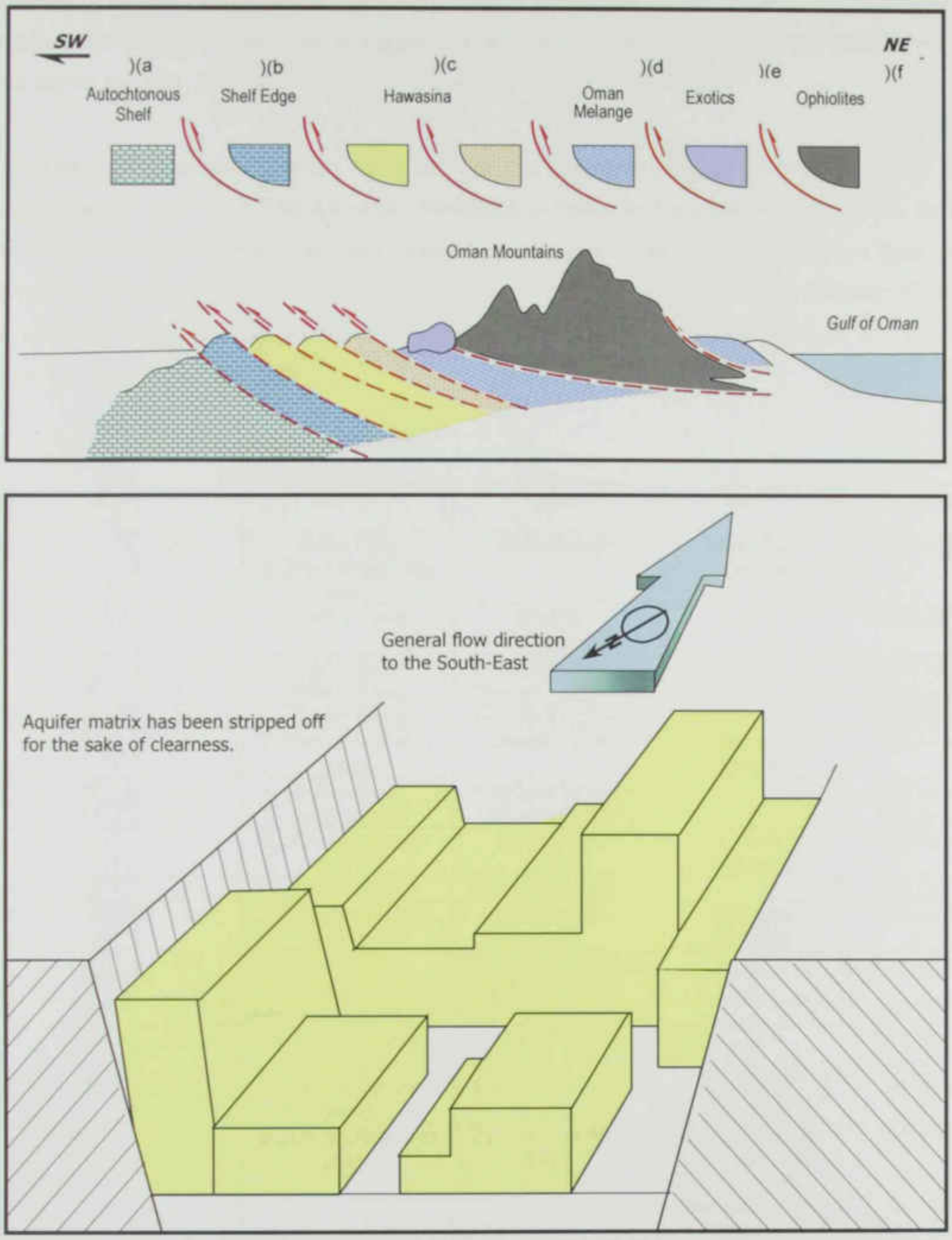

Figure 3.3. Schematic tectonic profile of the Oman Mountain (upper) and blockfolding in Wadi Ham (lower), (revised after Geoconsulat and Bin Ham, 1985).

The degree of consolidation varies from recent uncemented sandy gravel to the older well cemented and consolidated gravels. Clastics size ranges from silt grade to boulder sized material with a very high sand content. The gravels are typically composed of basic igneous clasts with other clasts of very well cemented sandstone and conglomerates (Entec 1996). The lithological information of available borehole in the Wadi Ham is presented in the Table 3.1. The locations of available boreholes are presented in Figure 3.4. It should be noted that HRP1, HRP2 and HRP3 are the same as OB1, BH1 and BH2, respectively.

The thickness of wadi gravel varies from 18 m at the upstream side of the dam to about 100 m near the coast (Figure 3.5). The minimum thickness is found in the area of well number BHF-19, at the upstream of Wadi Ham dam and close to the mountain series. The maximum thickness is observed in the area of well number BHF-14 which is very close to the coast of Oman Gulf. The cross-sectional depth of wadi gravels and sand along the wadi course varies from 45 m to 64 m as shown in Figures 3.6 and 3.7.

Table 3.1. Lithological information of Wadi Ham

Sl No.	BHF No.	M.S.L	Layer-I	Layer-II	Layer-III	Remarks
1	1	58.0	0-5m Clay 5-22m Gravel with sand	22-64m Sand	64m plus Ophiolite	Left bank
2	3	12.0	0-63m Sand	63m plus ?Ophiolite	-	Left bank
3	4	11.0	0-57m Sand	57m plus Ophiolite	-	Right bank
4	5	8.0	0-24m Sand	24m plus Ophiolite	-	Right bank
5	7	26.0	0-24m Sand	24m plus Ophiolite	-	Right bank
6	9	45.0	0-25m Gravel with sand	25-49m Sand	49m plus Ophiolite	Left bank
7	10	20.0	0-48m Sand	48m plus Ophiolite	-	Left bank
8	11	24.0	0-42m Sand	42m plus Ophiolite	-	Left bank
9	12	25.0	0-38m Gravel with sand	38-73m Sand	73 plus Ophiolite	Right bank
10	14	8.0	0-99m Sand	99m plus ophiolite	-	Left bank
11	15	68.0	0-28m Gravel with sand	28-53m Sand	53m plus Ophiolite	Left bank
12	16	550	0-47m Sand	47m plus Ophiolite	-	Right bank
13	17	8.0	0-63m Sand	63m plus Sand	-	
14	18	36.0	0-58m Sand	58m plus Ophiolite	-	Left bank
15	19	86.0	0-15m Boulders with gravel	15m plus Ophiolite	-	Left bank
16	20	52.0	0-5m Boulder with gravel	5-45m Gravel, fine to coarse	45m plus Ophiolite	Left bank
17	HRP1 (OB1)	-	0-21m Boulder with gravel	21-40m weathered Gabbro	40-45m Boulder with gravel, 46–58m Ophiolite, and 59–137 m Gabbro	Left bank
18	HRP-2 (BH1)	-	0-15m Boulder	15-33m Ophiolite	33m plus Gabbro	Right bank
19	HRP-3 (BH2)	-	0-18m Gravel with boulder	18-21m Boulder	21m plus Gabbro	Right bank

Note: HRP1, HRP2 and HRP3 are the same as OB1, BH1 and BH2, respectively.

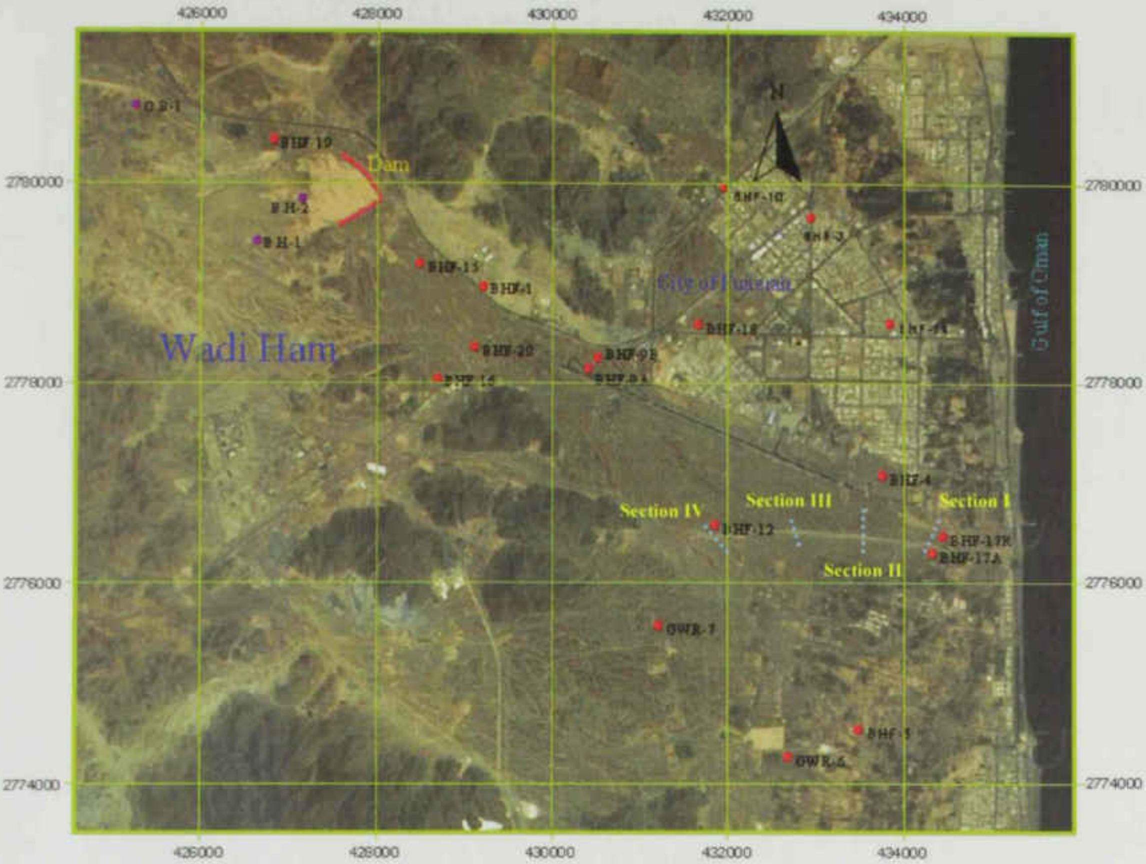

Figure 3.4. Location of observation wells in Wadi Ham.

Figure 3.5. Variation of gravel thickness in selected boreholes.

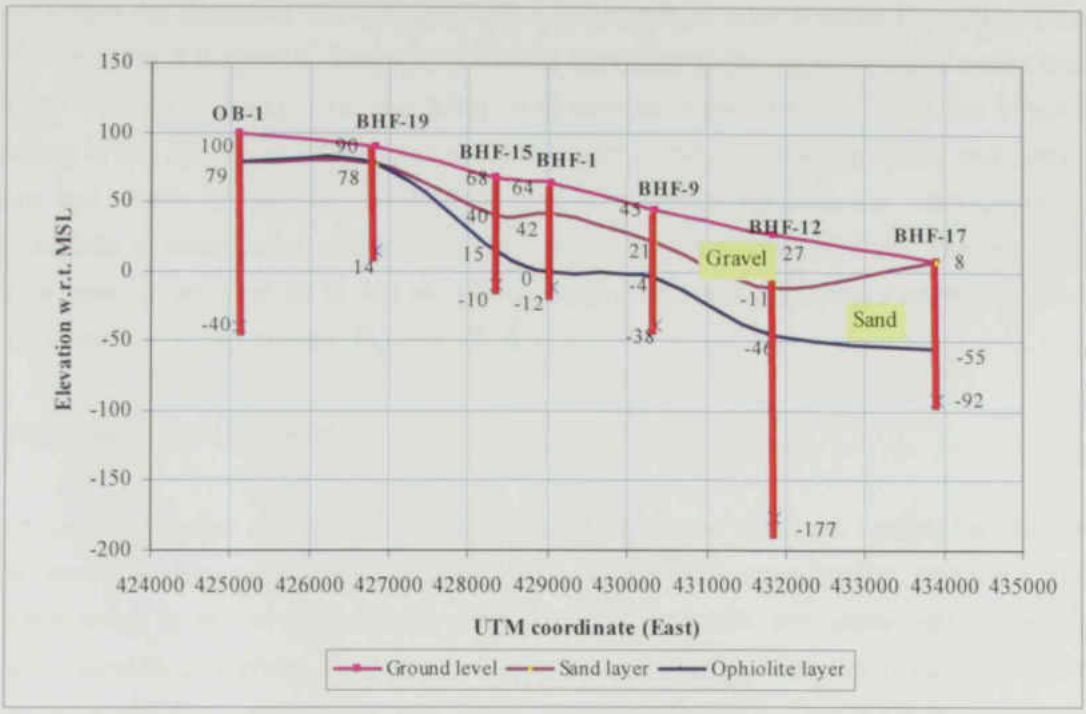

Figure 3.6. Alluvium layer along the course of Wadi Ham.

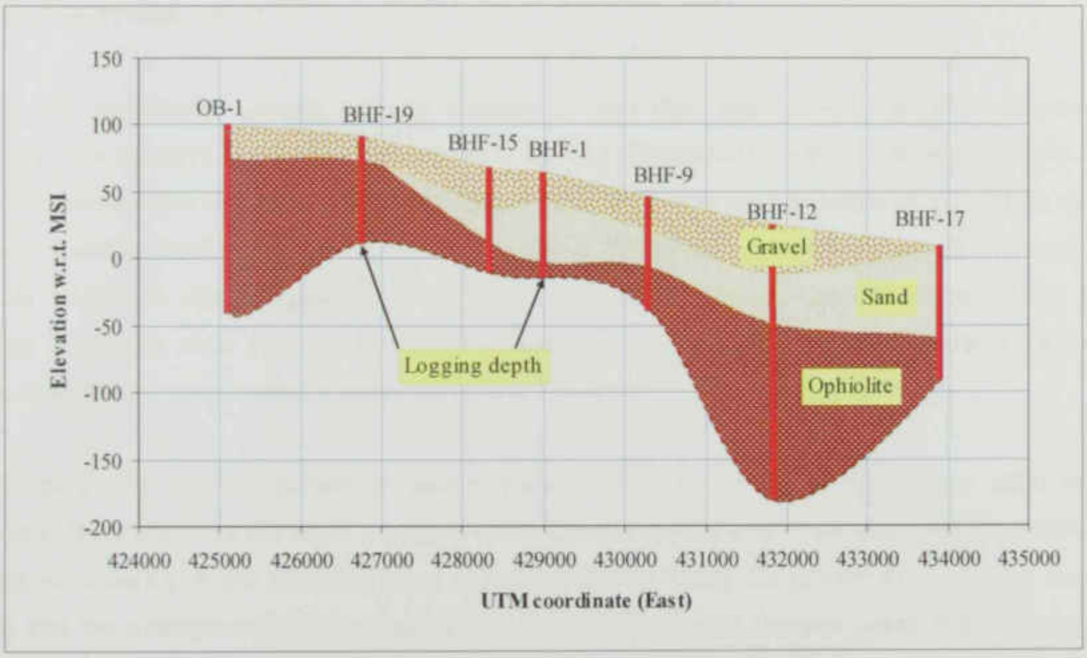

Figure 3.7. Geological cross-section along the course of Wadi Ham.

The cross-sections (longitudinal profiles) in the area near to the Oman Gulf show that the gravel depth is varying from 24 to 99 m (Figure 3.8 and 3.9). Its thickness decreases with increasing the distance from the shoreline. For example, within a distance of 3 km it varies from 24 to 73 m as shown in Figures 3.8 and 3.9. This is attributed to the regional dipping of Ophiolite series towards the wadi channel (Figures 3.10 and 3.11). It should be noted, however, that the information presented in this section is restricted to the total depth of the available boreholes and wells. The gabbro and diorite of the Samail Ophiolite are encountered beneath the wadi gravels. The gabbro/diorite is likely to be confined in some places by the cemented units. The depth to the ophiolite layer varies from 15 to 100 m. The ophiolite basement is dipping towards the coast as well as towards the wadi course (Figures 3.8 -3.11).

3.3. Hydrogeological Parameters

Based on interpretation of the above data, two aquifers can be identified, namely the Quaternary aquifer which is composed of wadi gravels and constitutes the main aquifer, and the Fractured Ophiolite which is of low groundwater potentiality. The gravels are highly permeable and of variable hydraulic properties. They tend to be unconsolidated at the ground surface, becoming better cemented and consolidated with depth. Electrowatt (1981) subdivided them into recent gravels, being slightly silty sand gravel with some cobbles; young gravels, which are silty sandy gravels with many cobbles and boulders and finally old gravels, which are weathered and cemented. Values of the hydraulic conductivity of the unconsolidated gravels tend to be very high, typically being 6 to 17 m/day and in the range 0.086-0.86 m/day for the cemented lower layers m/day (Electrowatt, 1981).

In the unconsolidated gravels primary porosity is very high when compared to the cemented gravels. The storativity typically ranges from 0.1 to 0.3 (Electrowatt 1980). At a distance of 3.5 km directly downstream of the dam the saturated aquifer thickness ranges between 10 and 40 m with a transmissivity ranging from less than 100 to about 200 m^2/day. In sections where the saturated aquifer thickness varies between 50 to 100 m, the transmissivity may reach more than 1000 m^2/day. Fourteen short duration (8 to 300 minutes) pumping tests performed by IWACO (1986) were analyzed by using both Cooper-Jacob and Theis methods.

A pumping test was conducted on well number HAM-OB1 (HRP1) in December 2003 for 75 minutes. Drawdown curves were analyzed using Cooper-Jacob and Theis methods. The data and results of analysis of the pumping tests are presented in Table 3.2 (Sherif et al., 2005). Results show that the transmissivity varies from 4.3 to 11700 m^2/d by the Cooper-Jacob method and from 3.83 to 9120 m^2/d by the Theis method. The hydraulic conductivities range from 4.73 to 203 m/d using the Theis method and from 2.26 to 259 m/d using the Cooper-Jacob method. Transmissivity values estimated by the IWACO (1986) ranged from 8.3 to 6959 m^2/d.

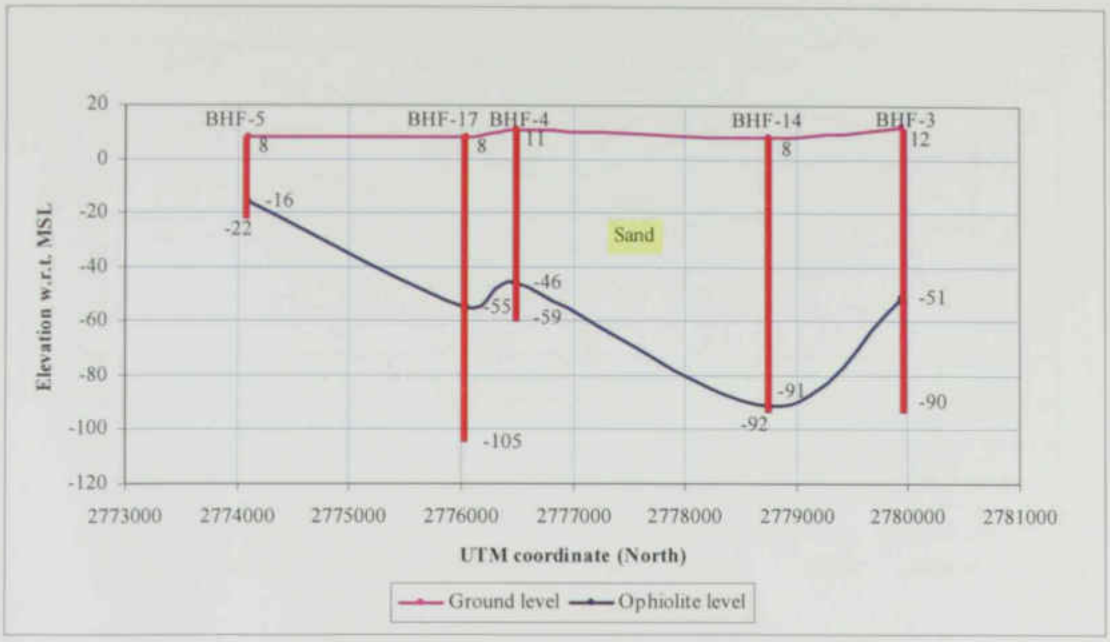

Figure 3.8. Depth of alluvium layer in Wadi Ham (close to the Oman Gulf coast).

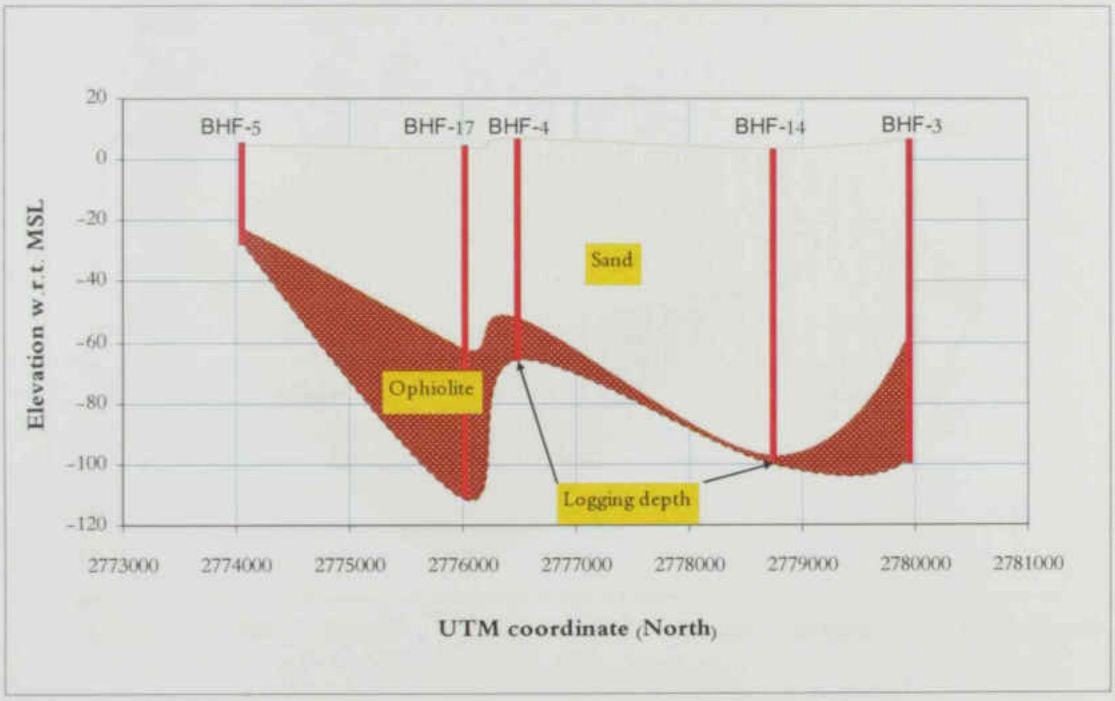

Figure 3.9. Geological cross-section across Wadi Ham (close to the Oman Gulf coast).

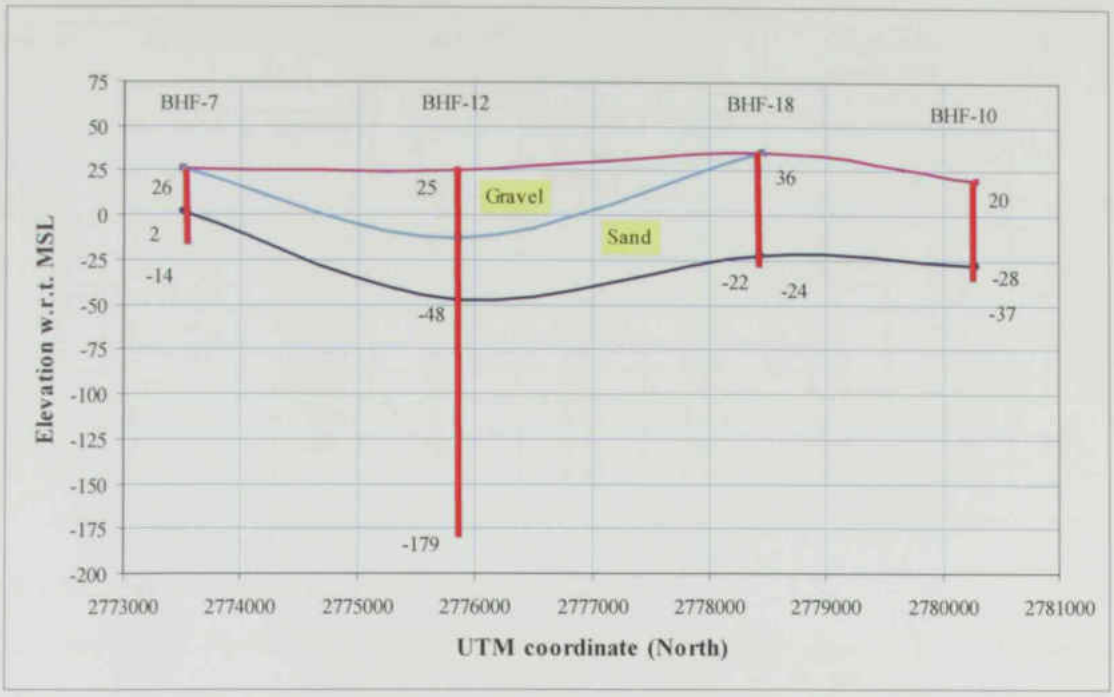

Figure 3.10. Depth of alluvium layer across the Wadi (about 3 km from the Oman Gulf coast).

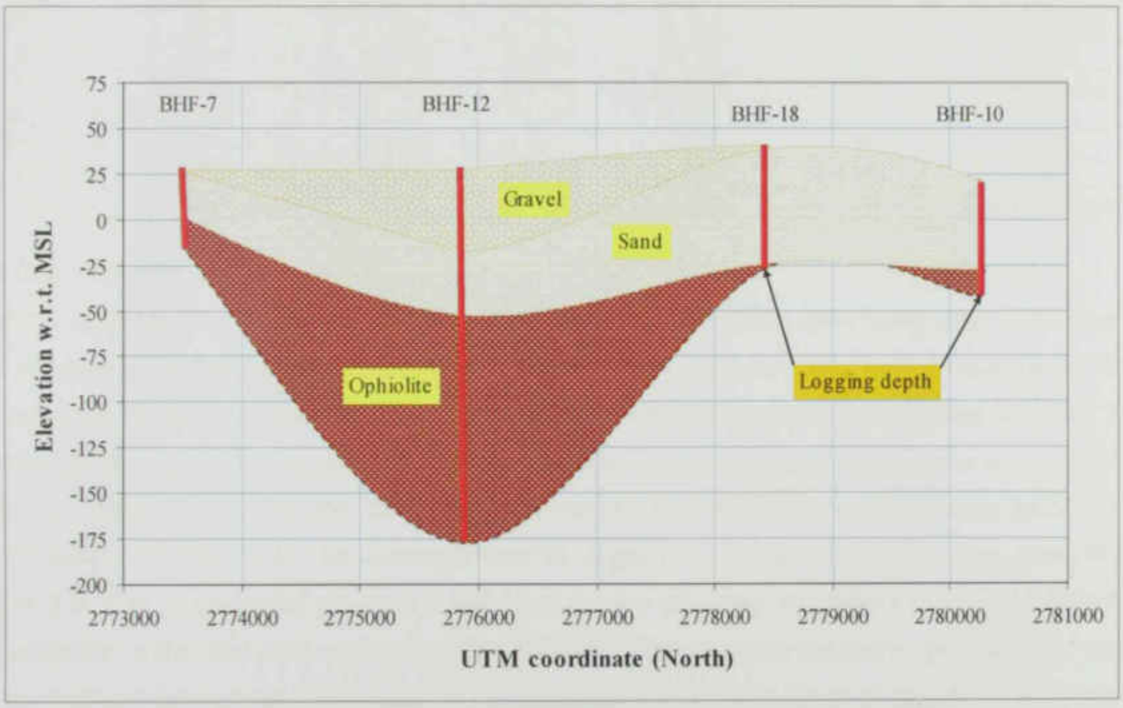

Figure 3.11. Geological cross-section across Wadi Ham (about 3km from the Oman Gulf coast).

Table 3.2. Estimated aquifer parameters

Borehole	UTM coordinate		T EWACO m^2/d	Transmissivity (m^2/d)		Hydraulic conductivity (m/day)		Storativity
	Northing	Easting		Cooper-Jacob method	Theis method	Theis method	Cooper-Jacob method	
BHF-1	2779163	429211	30	101	148	7.76	11.4	0.00161
BHF-4A	2776995	433773	8630	11700	9120	203	259	$2.01\text{x}10^{-15}$
BHF-12	2776432	431865	1340	947	789	15.5	18.6	0.00239
BHF-3A	2779800	432900	3450	6246	744	14	118	$6.21\text{x}10^{-8}$
BHF-5	2773400	432950	4347	8940	1260	105	745	$1.33\text{x}10^{-20}$
BHF-10	2780250	431800	1230	480	258	6.79	12.6	0.00197
BHF-11	2781000	430450	386	101	151	4.73	3.16	0.00666
BHF-13	2774900	427800	8.5	4.13	3.83	25.6	27.5	0.0101
BHF-14	2778750	433900	2882	4750	3630	39.4	51.6	$3.03\text{x}10^{-7}$
HAM-OB1 HRP1	2780166	427151	-	0.49	0.51	0.0058	0.0056	0.00128

Table 3.3. Minimum and maximum values of the water table

Obs. well	Period	Max. water table		Min. water table		Remark
		level	Month/yr	level	Month/yr	
BHF-1	1987-2003	53.066	5-1996	8.876	7-1994	Active
BHF-4	1988-2003	5.805	8-1996	2.585	7-2002	Active
BHF-4A	1990-2003	5.777	8-1996	2.347	7-2002	Active
BHF-9A	1987-2003	35.798	5-1996	0.728	6-1994	Active
BHF-9B	1990-2003	35.59	5-1996	3.02	7-2002	Active
BHF-12	1987-2003	11.329	7-1996	-1.191	8-2002	Active
BHF-15	1988-2002	64.106	4-1996	13.516	10-2001	Abandoned
BHF-16	1988-2003	54.317	5-1996	33.627	9-2002	Active
BHF-18	1988-2000	12.364	8-1996	-0.256	11-1989	abandoned
BHF-19	1995-2003	86.618	3-1996	52.27	10-2002	Active
BHF-20	1995-2002	53.969	5-1996	21.559	9-2002	abandoned
BHF-17R	1988-2003	3.153	7-1996	0.823	8-2000	Active
BHF-17A	1989-2003	3.897	6-1997	1.477	9-1999	Active
GWR-6	1977-2002	5.015	9-1996	-3.955	12-1984	Active
GWR-5	1977-2002	3.602	5-1996	-0.168	12-1980	Dry/abandoned

3.4 Groundwater Levels

Monthly groundwater levels for 16 observation wells in Wadi Ham area have been collected by the Ministry of Agriculture and Fisheries,. There is a significant variation in the groundwater level in response to recharge events. The maximum groundwater levels were observed in 1996. The maximum measured water table fluctuation reached 51 m and was found in observation well BHF-15, which is close to the dam site. The minimum water table variation was recorded at observation well BHF-17A. The complete records of groundwater level fluctuation are given in the Table 3.3. The groundwater gradient in the plain area is very mild as compared to the gradient of groundwater within the wadi valley close to the dam area. At some locations, groundwater levels have declined below mean sea level. An example of the general trend of groundwater levels in observation wells located in Wadi Ham is presented in Figure 3.12. It is generally noticed from this figure that groundwater levels are declining with time in all the cases. The monthly rainfall and the groundwater levels are plotted on the same Figure to identify the relationship between them. A clear relationship is observed between rainfall events and groundwater levels.

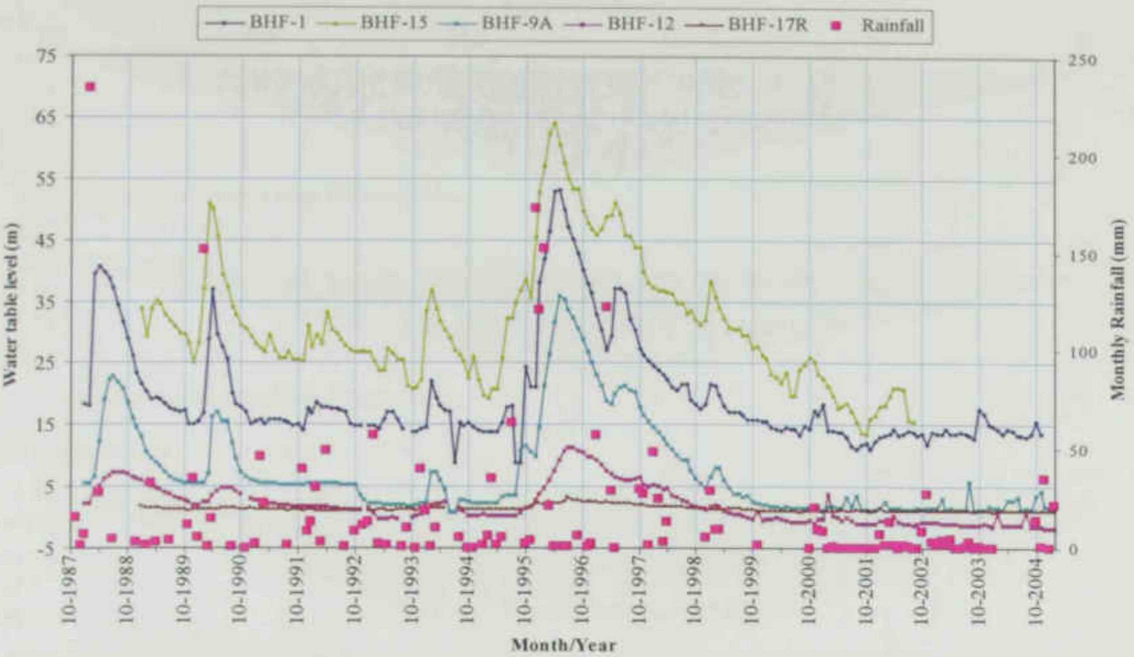

Figure 3.12. Variation of monthly water table and rainfall events, Wadi Ham.

3.5 Geophysical Studies

Sherif et al. (2006) conducted comprehensive geophysical investigations in the area of Wadi Ham to identify the salinity distribution and delineate the effects of seawater intrusion on the groundwater quality. Four 2D Dc-resistivity profiles were run (Profiles 1-4; Fig. 3.4) to assess the groundwater quality and seawater intrusion in the coastal aquifer of Wadi Ham. Profile 1 was aligned parallel to the shore line and approximately at a distance of 700 m from the sea boundary. For profiles 1, 2 and 3 (Figure 3.4), forty three electrodes spaced 5 m apart were used. In profile 4 thirty five electrodes spaced 10 m apart were used. The 2D apparent resistivity data were inverted to create a model of the resistivity of the subsurface using Res2dinv software. Res2dinv uses an iterative smoothness-constrained least-squares method (deGroot-Hedlin and Constable, 1990; Sasaki, 1992).

To test interpretation, resistivity models were created based on the inversion results. The resistivity models were used to generate synthetic apparent resistivity data. The synthetic apparent resistivity data were inverted using Res2dinv and the resulting inversions were compared with the original inverted resistivity section. The resistivity models were adjusted and simplified to qualitatively match the field-data inversions. Generating resistivity models helped constrain interpretation of the field-data inversions to identify locations and orientations of resistivity anomalies. The 2D Dc-resistivity field-data inversions, resistivity models and synthetic-data inversions for profiles 1 (near the shore line and observation well BHF-17, Figure 3.4) are presented in Figure 3.13. The depth of penetration is approximately 50 m. Therefore, it did not reach the top surface of the Ophiolites bedrock which is more than 50 m in the area of profile 1.

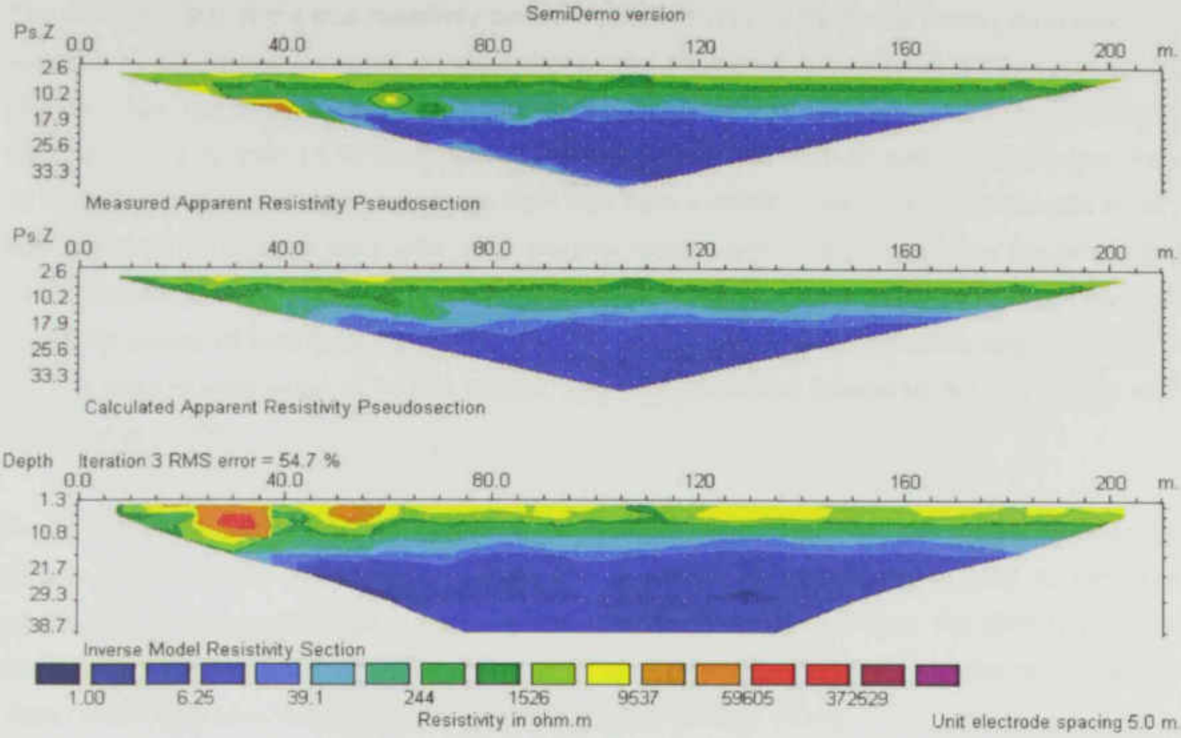

Figure 3.13. Results of 2D Dc-resistivity data and modeling for Profile 1 in Wadi Ham

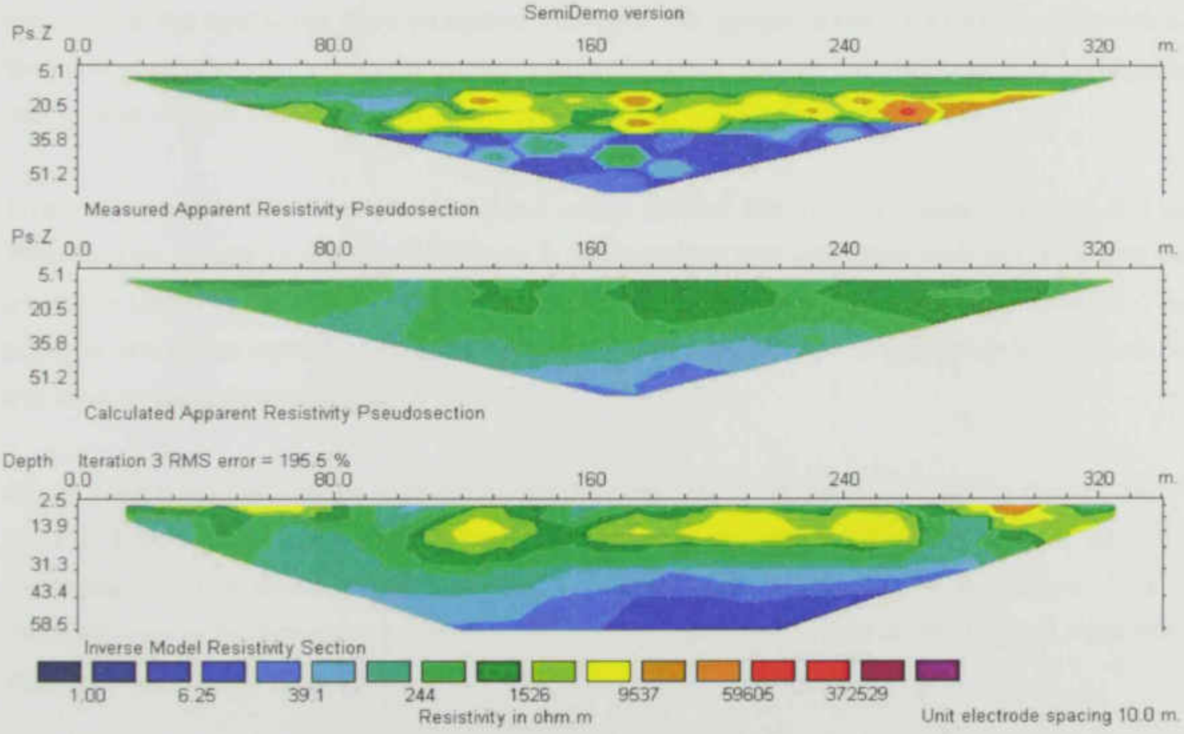

Figure 3.14. Results of 2D dc-resistivity data and modeling for Profile 2 in Wadi Ham.

The cross-section of the true resistivity beneath profile 1 (near agricultural farms) represents the variation in the lithology, degree of saturation of the alluvium gravel layer and the water quality (Figure 3.13). The depth to water table (the layer coloured in light blue in the depth-true resistivity section is ranging from 11 to 15 meters below ground surface. Its true resistivity is ranging from 30 to 70 Ωm and its thickness is ranging from less than a meter in the center of the profile to only few meters at the sides of the profile. The brackish water zone is very thick below this profile and has a resistivity range of 1-30 Ωm. Only in the central area the saline water zone which has true resistivity values of less than 1 Ωm can be seen (dark blue). The 1 Ωm refers approximately to electric conductance value of 10,000 µS/cm, approximately Total Dissolved Solids of 6,400 mg/l (Sherif et al., 2006).

Profile 2 (Figure 3.14) is located 800 m to the west of profile 1 and parallel to it. The inversion results of this profile data indicates that the thickness of fresh zone started to increase, particularly in the northern part near Fujairah Airport. At the same time the thickness of the brackish water zone decreases. The saline water zone can only be seen in the southern side near Kalba town (dark blue colour and has a resistivity range of less 1 Ωm).

Profile 3 (Figure 3.15) is located 800 m to the west of profile 2 and is parallel to the others. The interpretation results of this profile indicate that along the total length and to its maximum depth of penetration which is about 40 meters only the fresh and brackish water zones can be seen. The thickness of the freshwater zone increased. However, the upward coning of brackish water due to the cone of depression caused by intensive pumping (well field) in the middle part of the area is remarkable (Figure 3.15).

Profile 4 is parallel to all the other sections and is located 800 m to the west of Profile 3. The interpretation results of this profile (Figure 3.16) indicates that along the total length and to its maximum depth of penetration which is about 40 m only the fresh water zone can be seen. The brackish and saline water zones were not observed in this profile. The quality of groundwater in this area is relatively better.

Figure 3.17 presents a fence diagram of the true resistivity, including the salinity distribution in profiles 1 through 4 respectively. The groundwater contamination due to seawater intrusion increases mainly eastward toward the shore line and with a less degree southward. This is probably caused by excessive pumping in Kalba area or due to its bay which is about 3 km from the shore line (Sherif et al., 2006).

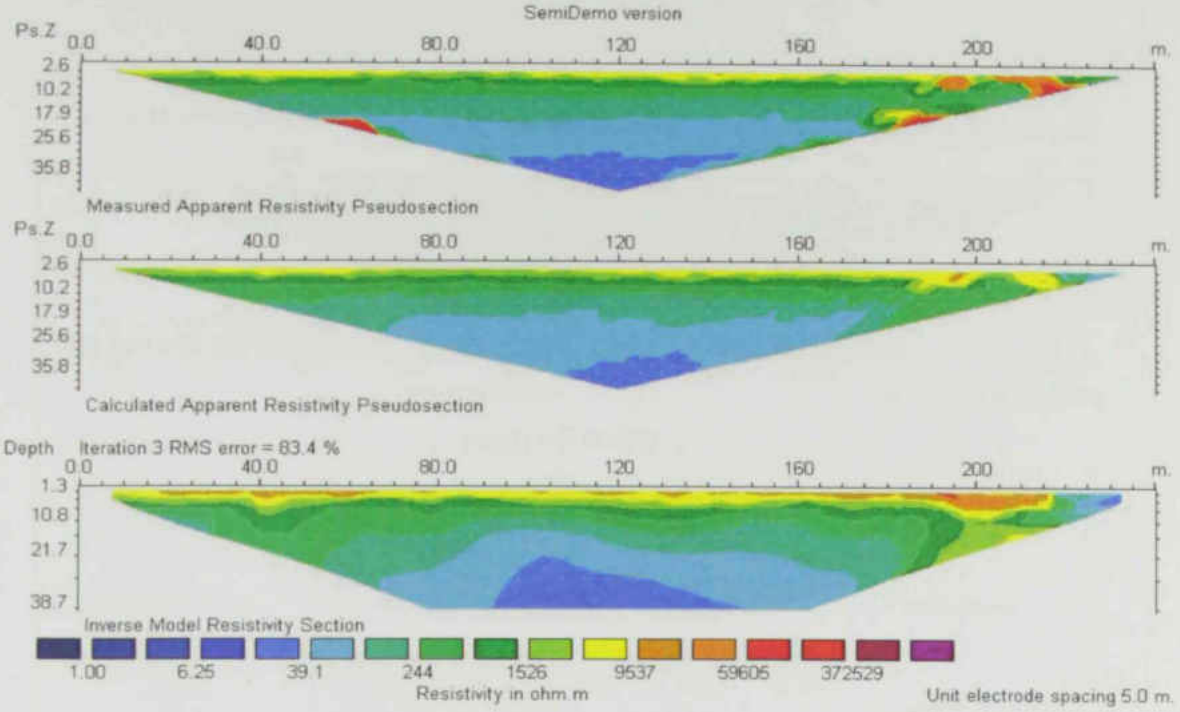

Figure 3.15. Results of 2D Dc-resistivity data and modeling for Profile 3 in Wadi Ham.

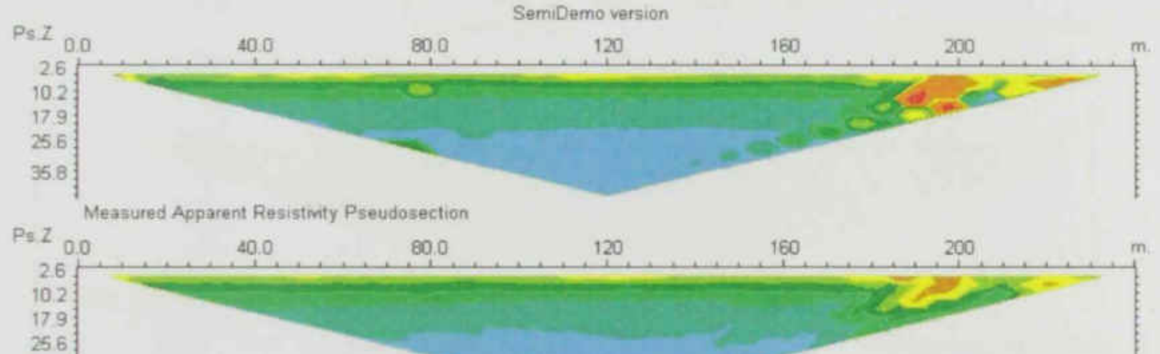

Figure 3.16. Results of 2D Dc-resistivity data and modeling for Profile 4 in Wadi Ham.

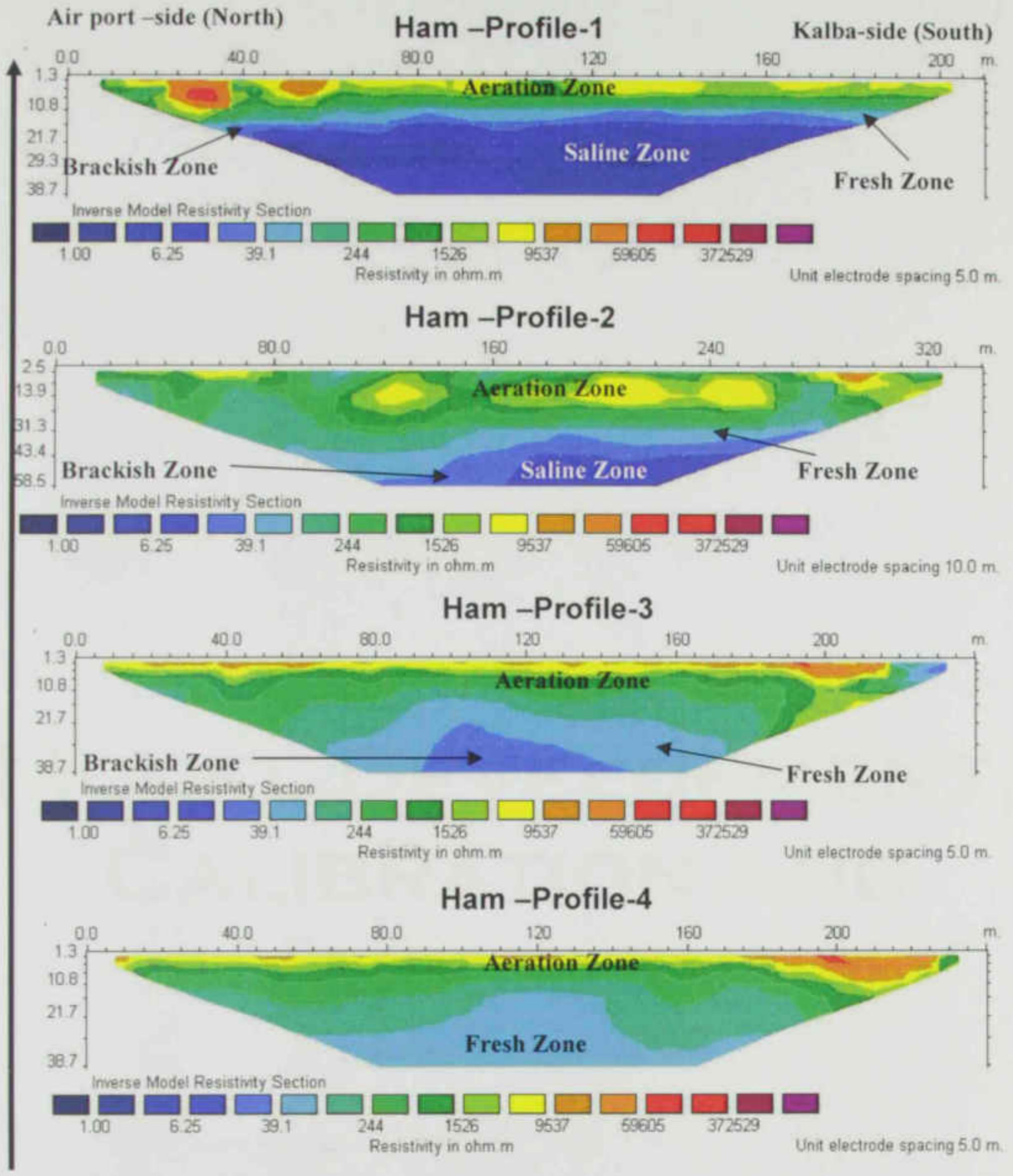

Figure 3.17. Combined diagrams of four true resistivity sections (increasing seawater intrusion eastward).

CHAPTER 4

MODEL DEVELOPMENT, CALIBRATION AND VALIDATION

Chapter 4. Model Development, Calibration and Validation

Groundwater flow and solute transport in porous media can be simulated both analytically and numerically. Analytical methods are based on the solution of closed form equations, while numerical methods are based on the approximate or iterative solutions. Each method has its own advantages and disadvantages as compared to the other. Analytical models are generally simpler in formulation and application and do not require detailed input parameters (data). They provide accurate results when applied to small-scale problems, e.g., upconning phenomenon below pumping wells. Analytical models often require simplified assumptions and idealized domains.

Numerical models are relatively adaptable and flexible and could be applied to heterogeneous systems and irregular domains. Many well-developed and verified numerical codes are available and can be employed to almost every case. These models include, among others, MODFLOW, MOC, MOCDENSE, and SUTRA. The main disadvantage of the numerical methods is the need for detailed field data in space and time that might not be available. Analytical solutions are often employed to verify numerical models for idealized domains, flow and boundary conditions.

4.1. Calibration and Validation of Numerical Models

Model calibration consists of changing the values of model input parameters, within a reasonable range, in an attempt to match a given aquifer hydraulic state or solute behavior within some acceptable criteria. This requires that field conditions at a facility be properly characterized. Lack of proper characterization may result in a model that is "calibrated" to a set of conditions which is not representative of actual field conditions. The calibration process typically involves calibrating to both steady-state and transient conditions. With steady-state simulations, there are no observed changes in hydraulic head or contaminant concentration with time for the field conditions being modeled. Transient simulations involve the change in hydraulic head or contaminant concentration with time. These simulations are needed to narrow the range of variability in model input data, since there are numerous choices of model input data values which may result in similar steady-state simulations. Models may be calibrated without simulating steady-state flow conditions, but not without some difficulty.

Calibration includes comparisons between model-simulated conditions and field conditions for the hydraulic head data, hydraulic-head gradient (magnitude and direction), and water mass balance and for fate and transport models the solute concentrations, contaminant migration rates, contaminant migration directions, and degradation rates. Typically, the difference between simulated and actual field conditions (residual) should be less than 10 percent of the variability in the field data across the model domain. Errors should be randomly distributed, such that model results are not biased high or low within particular regions or over the entire model domain. A "calibrated" model having a residual error less than 10 percent should not be considered accurate and without error. In our case there are not enough data for calibration.

A second step in the calibration process is the "history-matching" process. This process has been referred to by others as "model verification". A calibrated model uses selected values of hydrogeologic parameters, sources and sinks, and boundary conditions to match field conditions for selected calibration time periods (either steady-state or transient). This choice of "calibrated" model parameters is referred to as a "realization." However, the choice of the parameter values and boundary conditions used in the calibrated model is not unique. There may be an infinite number of statistically-similar realizations that give very different predictive model results. History matching uses the calibrated model to reproduce a set of historic field conditions, other than those used in the initial model-calibration process, in an attempt to reduce the number of realizations and variability in simulation results.

The most common history-matching scenario consists of reproducing an observed change in the hydraulic head or solute concentrations over a different time period, typically one that follows the calibration time period. The best scenarios for model verification are ones that use the calibrated model to simulate the aquifer under stressed conditions. The process of model verification may result in the need for further refinement of the model. After the model has successfully reproduced measured changes in field conditions for both the calibration and history-matching time periods, it is ready for predictive simulations.

4.2 Sensitivity Analysis

A sensitivity analysis is the process of varying model input parameters over a reasonable range (range of uncertainty in values of model parameters) and observing the relative change in model response. Typically, the observed changes in hydraulic head, groundwater flow rate, or contaminant transport (migration rate and concentrations) are noted. The purpose of the sensitivity analysis is to demonstrate the sensitivity of the model simulations to uncertainty in values of model input data. The sensitivity of one model parameter relative to other parameters is also demonstrated. Some common parameter estimation programs incorporate a quantitative analysis of parameter sensitivity as part of the parameter estimation output. According to Anderson and Woesser (1992), the purpose of the sensitivity analysis is to quantify the uncertainty in the calibrated models caused by uncertainty in the estimation of aquifer parameters.

A sensitivity analysis may be performed at any point in the model development process. Perhaps the greatest utility of a sensitivity analysis is in determining the direction of future data-collection activities. Parameters for which the model is relatively sensitive could require additional characterization; model-insensitive parameters would not require further field characterization. It is also useful to conduct a sensitivity analysis during predictive simulations to demonstrate the impact of varying pertinent model parameters on the simulation outcome.

4.3 The SUTRA-Argus Environment

In this study, the USGS model SUTRA (Voss, 1984) is employed to simulate the groundwater conditions. The model is used with Argus-One GUI to represent the domain and develop the input data. Argus-One was also used to view the output of the different simulation runs. The study domain was discretized into a number of smaller quadrilateral elements. SUTRA simulates fluid movement and the transport of either energy or dissolved substances in subsurface environment. It employs a two-dimensional hybrid finite element and integrated finite difference method to approximate the governing equations that describe the two interdependent processes:

1- Fluid density-dependent saturated or unsaturated groundwater flow, and either,

2- a- Transport of a solute in the groundwater, in which the solute may be subject to equilibrium adsorption on the porous matrix, and both first-order and zero order production or decay, or

b- Transport of thermal energy in the groundwater and solid matrix of the aquifer (Voss and Provost, 2002).

4.3.1 Capabilities and limitations

SUTRA may be employed for areal and cross-sectional modeling of saturated zone flow and solute transport. It can be employed to model natural or man-induced chemical species transport, including processes of solute sorption, production and decay.

SUTRA uses quadrilateral elements in cartesian or radial cylindrical coordinate systems. The mesh may be coarsened employing (pinch nodes) in areas where transport is unimportant or considerably small. Hydraulic conductivities may be anisotropic and may vary both in direction and magnitude throughout the system as may other aquifer and fluid properties. The boundary conditions and other stresses such as sources and sinks can be time-dependent in the sense that they can vary from one time step to the other. One should always notice that SUTRA requires spatial and temporal discretization for the simulation of rapid variation either in the piezometric heads or in the concentrations. A special reference is made here to the region near the shore boundary where the cyclic flow exists and the concentration gradient is relatively high (Voss and Provost, 2002).

Although SUTRA is mainly developed to simulate two-space dimensions, the thickness of the two-dimensional region may vary from one point to the other. In other words, the third dimension is introduced, while all hydraulic and transport parameters are not allowed to vary in that direction. Fluid density may be constant or vary as a function of solute concentration or fluid temperature. SUTRA tracks the transport of either solute mass or energy in the flowing groundwater through a unified equation which represents the transport of either solute or energy (Voss, 1984).

Solute transport is simulated through numerical solution of a solute mass balance equation where solute concentration may affect fluid density. The single solute species may be transported conservatively, or it may undergo equilibrium sorption (through linear, Freundlich or Langmuir isotherms). The solute may be produced or decay through first- or zero-order processes. Dispersion processes modeled by SUTRA include diffusion and two types of fluid velocity dependent dispersion. The first type is the dispersion process for isotropic media in which direction independent values for longitudinal and transversal dispersivities are assumed. The second type is the dispersion process for anisotropic media. This process assumes that longitudinal dispersivity varies with the angle between the flow direction and principal axis of aquifer permeability when anisotropic conditions exist.

SUTRA is structured in a modular, top-down programming style that allows for code readability and eases any desirable modifications. Fluid pressures and solute concentrations or temperatures at each node in the studied domain after each time step are obtained. The velocities are evaluated at the centroid of each element (Voss, 1984).

4.3.2 Organization of SUTRA

SUTRA (V06902D) is written in ANSI-STANDARD FORTRAN-77 and may be compiled and executed under most operating systems and on most computers. Many SUTRA applications require considerable array storage and computational effort. These applications must be carried out on large, fast scaler machines such as mainframes, minicomputers, work stations and 386-or-better microcomputers with math co-processors and at least a few Mbytes of memory, or on vector/array processing machines.

SUTRA package contains 25 files (including one that contains a copy of a text file (SUTRA.DOC). The set of files includes:

(1) SUTRA main routine (MAIN.FOR),
(2) 24 SUTRA subroutines contained in three files: a) USUBS.FOR, with two user- programmable routines, and b) SUBS1.FOR and SUBS2.FOR, with all other subroutines,
(3) two mesh data generation routines (MGENREC.FOR and MGENRAD.FOR),
(4) nine input data sets consisting of three data sets required to run each of three examples from the SUTRA documentation,
(5) three output data sets with results from these three examples,
(6) one routine for calculation of hydrostatic pressure data at specified pressure boundaries (PBCGEN.FOR),
(7) a file for compiling and loading SUTRA problems under 640 K bytes using DOS/Microsoft-Fortran-4.0 or 5.0 (MSFOR.BAT), a file for running SUTRA which has been compiled under Microsoft Fortran (MSUTRA.BAT),
(8) a file for compiling and loading SUTRA problems up to available extended memory size on a 486 microcomputer using Lahey F77L-EM/32 Fortran 3.0 (L3FOR.BAT), a file for running SUTRA which has been compiled under Lahey Fortran (LSUTRA.BAT).

(9) a file executable under DOS on PC systems with an 8087/287/387 co-processor that was created using the SUTRA routines listed in (1) and (2), and the Microsoft-Fortran-5.0 system with the utility files listed in (7), above (SUTRA.EXE, requires 531 Kbytes).

4.3.3. Modeling via SUTRA

For problems in regional scale the real situation is geometrically simplified to be easier to solve. First the simplified domain must be discretized in space and time.

Discretization: Adequate discretization is vital for two reasons (Voss 1984):

1. The ability of a model to represent the variations in system parameters and to simulate complex processes depends on the fineness of discretization.
2. The accuracy and stability of the numerical methods used to represent system processes, in particular, transport, depends on the spatial and temporal discretization. A better discretization is always obtained by making existing discretization finer, but the finer the discretizations are, the more computationally expensive the simulations become. The only way to explicitly check for inadequate discretization of a system is to simulate with a discretization that is assumed to be adequate and then with a significantly finer discretization and compare results. If there are no significant differences in the results, then the coarser simulation indeed has been adequately discretized.

Guidelines: For adequate discretization, the following guidelines should be considered:

1. Nodes are required where boundary conditions and sources are specified. As accurate simulation of processes near these specified points to be required, then a finer mesh is needed in these areas.
2. A finer mesh is required where parameters vary faster in space. Thus, finer mesh is required at high concentration gradient (near sea side) in saltwater intrusion problems. A rule-of-thumb is that at least five elements should divide the front in order to guarantee that the simulated front width arises from simulated physical processes rather than from spreading due to inadequate discretization.
3. The spatial stability of the numerical approximation of the unified transport equation depends on the value of a mesh Peclet number, Pe_m, given by:

$$Pe_n = \left(\frac{\Delta L_L}{\alpha_L} \right) \tag{4.1}$$

where ΔL_L is the local distance between element sides along a streamline of flow. α_l is the longitudinal dispersivity of the porous medium. Stability is guaranteed in all cases when $Pe_n \leq 2$, which gives a criterion for choosing a maximum allowable element dimension, ΔL_L, along the local flow direction. This criterion significantly affects discretization. Spatial stability is usually obtained with SUTRA when

$$Pe_n \leq 4 \tag{4.2}$$

A discretization rule-of-thumb for simulation with SUTRA which guarantees spatial stability in most cases is:

$$\Delta L_L \leq 4\alpha_L \tag{4.3}$$

Taken in combination with the considerations of guideline (2) requiring at least five elements across a front, the previous rule implies that a minimum front width which may be simulated when the mesh is designed according to ΔL_L. $4\ \alpha_L$ is $20\ \alpha_L$.

4. Discretization for transverse dispersion also may be related to dispersivity. Although an exact guideline is not given, the object of transverse discretization is to make the local element perpendicular to a streamline small relative to the total transverse dispersivity:

$$\Delta L_L \prec \alpha_T + \frac{1}{|V|}\left[\varepsilon S_w \sigma_w + (1-\varepsilon)\sigma_s\right] \tag{4.4}$$

   where ΔL_L is the local element dimension transverse to the flow direction. α_T is the transverse dispersivity in 2 D [L], *V* is the magnitude of vector, σ_w is diffusivity is fluid phase in undefined transverse equation, ε is the porosity [dimensionless] and σ_s is diffusivity is solid phase in undefined transverse equation and S_w is the water saturation (volume of water per volume of voids) . In the case where the transverse mixing rather than diffusion dominates the transverse dispersion an adequate but stringent rule-of-thumb may be, $\Delta L_T < 10\ \alpha_T$.

5. Radial meshes with a well require very fine discretization near the center axis to accommodate the sharply curving pressure distribution. The radial element dimensions may increase outward and become constant at, for example, a size of $4\ \alpha_L$.
6. Discretization in time is done by choosing the size of time steps. The adequacy of temporal discretization may be tested only by comparing results of simulations carried out with different time step sizes. For saturated flow simulation, temporal discretization begins with fine time steps which may become significantly larger as the system response slows. For transport simulation, changes in concentration or temperature at a point in a space are often due to the movement of fronts with the fluid flow. Therefore, adequate discretization of these parameters in time is always related to both fluid velocity and spatial gradients in the parameters. The higher the longitudinal spatial gradient and fluid velocity, the smaller the time step required for adequate temporal discretization. A general guideline is that relatively sharp fronts require time discretization which allows them to move only a fraction of an element per time step. Broad fronts with low gradient in concentration or temperature have adequate temporal discretization when time steps are chosen to move the front one or more elements per step.

After preparing mesh and choosing adequate discretization, nodes and elements in the mesh must be numbered. As SUTRA uses the method of banded matrix for solving equations, careful numbering of the nodes is necessary for minimizing the bandwidth which is critical to computational efficiency.

4.3.4. Argus-One

Argus-One is an independent Geographical Information System (GIS) for numerical modeling. Using a conceptual model approach, combined with export capabilities, Argus-One can be considered as an application development environment for development and deployment of graphical user interfaces for numerical models. Argus-One provides a user environment where geospatial (map-type) information (or coverage) may be synthesized in preparation for use as input to numerical models. Like other GIS systems, the various types of geospatial information are stored and viewed in coverages or layers which can be viewed and interact with directly from the screen. Export scripting of Argus-One enables to export the synthesized information to input files for numerical modeling at the exact format the model requires. Combining the export scripting and the conceptual model approach, Argus-One offer a model independent environment which enables to use it as a pre-processor for the model. At the same time, it also enables to interchangeably use geospatial coverages with Argus-one. Argus-One is composed of the two main modules. The first is the GIS module encompassing information layers (nodal information, boundary conditions, domain out lines, and other), data layers (data on grid and interpolation of data), and maps layers (import text, DXF, GIS shape files and images). The second is the mesh and grid module encompassing finite element mesh layers and finite difference grid layers. Complex finite element and finite difference meshes can be created.

4.4. Wadi Ham Presentation and Discretization

The study domain of Wadi Ham aquifer comprises an area of 117.81 km^2 with a length of 11.9 km east to west (Dam to coast of Oman Gulf) and a length of 9.9 km north to south (Fujairah to Kalbha) as shown in Figure 4.1. The study area and the aquifer boundaries were delineated by digitizing the remote sensing image of Wadi Ham. The model domain includes the Gulf of Oman and the ophiolite sequence rock outcrops. The ophiolite outcrops are separated as inactive or noflow area. The area of separated outcrop is about 6.56 km^2. At the coast, many cells are located in the sea which is considered to be constant-head cells with a head of 0.0 m (sea level). Ponding area was delineated and marked on the study domain. The total area of ponding zone at the flood level is about 0.40 km^2. Two well fields (Saraah and Kalbha) are identified. The Saraah well field is located about 800 m downstream (south-west) of the dam and the other in the Kalbha area is about 2.5 km west of the sea coast. Inflow boundaries were also considered as shown in the Figure 4.1, at the entry of Wadi Ham upstream of dam, at Wadi Al Hayl and at Wadi Hald in which underflow from high lying branches of wadis is encountered. To compensate the inflow to the aquifer from upstream area, specified flow boundaries are simulated by using injection/recharge wells with positive pumping rates. The Wadi Ham dam is located 8.5 km from the coast of Fujairah and the catchment area of the dam is approximately 195 km^2. The main dam is situated at an elevation 75 m above mean sea level with a length of 600 m. The elevation of the dam crest is situated at 88.5 m above mean sea level. The height of the dam is 13.5 m.

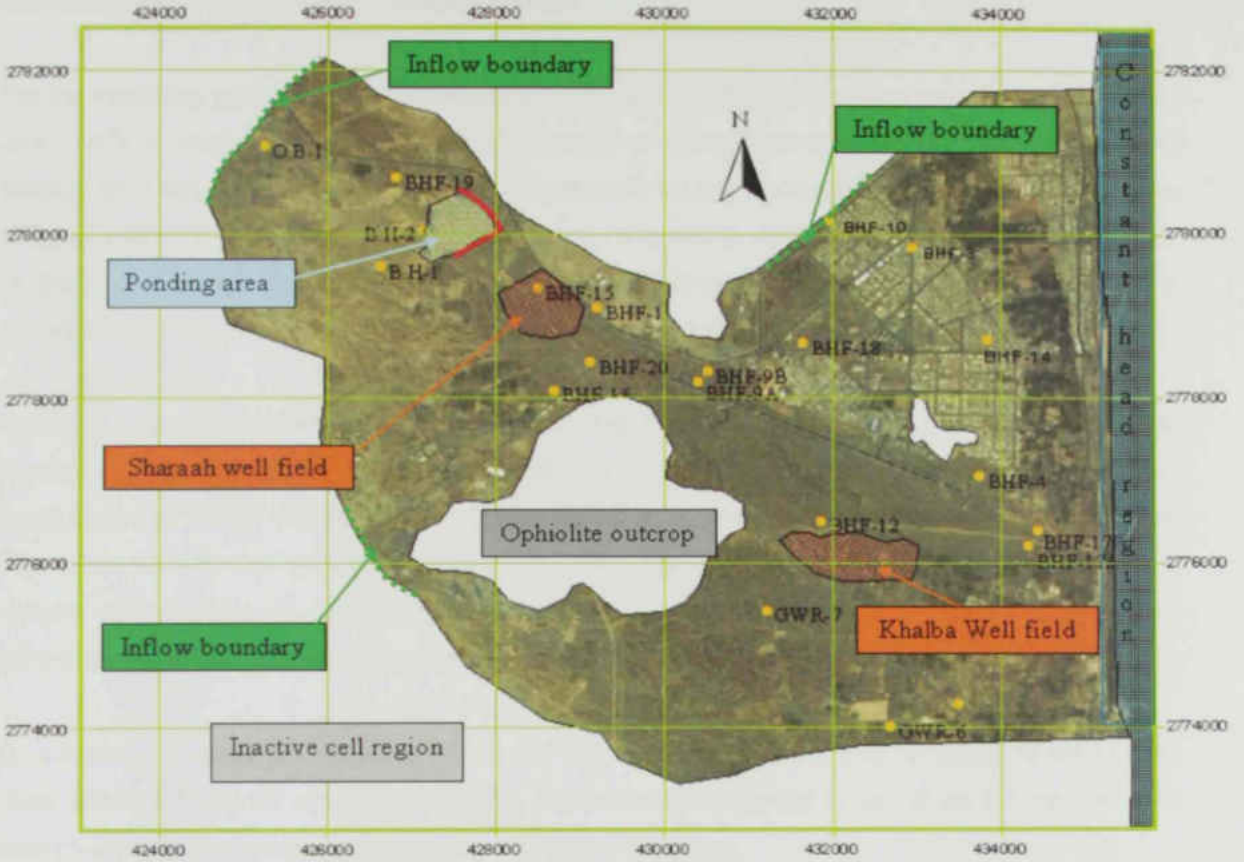

Figure 4.1. Study domain and boundary conditions.

Data regarding dam storage, rainfall, observation wells, aquifer parameters, and well field pumping were assembled. The recharge to the aquifer was assigned through two zones corresponding to the recharge from rainfall and from the dam storage for the modeling period. In the vicinity of the study area, six rain gauges namely Fujairah, Farfar, Kalbha, Masifi, Bithna and Hamraniyah are available. However, in this case only Farfar daily rainfall was considered as it better represents the study area when compared to other gauging stations. The rainfall (m/day) was assigned on daily basis in the study domain.

For the modeling purpose, the storage depth has been distributed in space over the ponding area (0.40 km^2) of dam and in time to the total period of storage as m/day. The average period of storage for each storage event in the dam is considered approximately 60 days. The model area of lower plains of Wadi Ham composed of recent Pleistocene wadi gravels. This layer is underlain by the consolidated rocks of the Semail formation (Ophiolitic sequence). The thickness of wadi gravel varies from 15 to 100 m in the upstream side of the dam to the coast.

For the horizontal simulation, fine finite element mesh with a total number of 27277 quadrilateral elements of which 26026 elements were interior and 1251 were located on the boundaries. The mesh included 27897 nodes of which 26655 nodes were interior and 1242 were boundary nodes. The bandwidth of this grid system was 245. This fine discretization was made to ensure the stability of numerical solution and satisfy equations 4.1-4.4. A stress period is defined as a time period during which all time-dependent processes such as pumping and recharge are constant.

Groundwater is exploited intensively from the sand and gravel aquifer for irrigation in the coastal plain between Fujairah and Khawr Kalbha. Several well fields are in operation for the domestic water supply by the Ministry of Electricity and Water including:

a. Fujairah well field with a total pumping of 3.2 million m^3/year until 1988. Very limited groundwater extraction was encountered after 1988.
b. Shaara well field, 2 km downstream of Wadi Ham with a pumping of 1 million m^3/year since 1988. The pumping duration was about 10 hr per day. However, out of the 9 wells 5 wells were dried up in the year 2003. Discharges of the wells were drastically reduced from 1988 to 2003.
c. New well field with about 60 wells is operated since 1995 near Kalbha. The total draft is about 6 million m^3/year. A number of wells were in operation before 1995.

To initiate the simulation, SUTRA requires initial assumptions for the groundwater level and solute concentration throughout the study domain. Proper initial assumptions for the starting heads and concentrations of the simulation can reduce the required simulation time significantly. Initial head values were also used to calculate drawdown values. A stable piezometric surface head and concentration values may be obtained by the steady-state simulation.

4.5. Calibration of SUTRA

Based on the availability and continuity of data in the study area, the calibration period was selected for 5 yrs from January 1989 to December 1993 (1826 days). A time interval (stress period) is defined as a time period during which all time-dependent processes such as pumping and recharge are constant and could not be changed by the user. The length of the stress period in this simulation exercise was taken as one real month.

SUTRA allows to change the time step by a user-defined multiplier to reduce the computational time at a later time of simulation when the system approached steady-state conditions. However, smaller time steps at the beginning of the simulation would ensure numerical stability. The time step multiplier is a factor that can be used to increment the time step size within each stress period. A time step multiplier of 1.2 was considered in the calibration period.

The daily rainfall of Farfar was considered for the calibration period. The rainfall in m/day was assigned to the appropriate areas. For the calibration purpose, the ponding area (0.40 km^2) was discretized in to a smaller quad elements to enhance the accuracy of simulation. The average period of storage for the each event in the dam is considered approximately 60 days. Evapotranspiration of 0.014 m/day and extinction depth of 2 m were considered for the model domain. The area occupied by the Gulf of Oman in the model domain was considered as a constant-head boundary in the model with head at sea level (0.0 m) for 1826 days. The most common type of transient flow calibration starts the simulation from the calibrated steady-state solution to derive stable initial head conditions. However, there is no steady-state situation in the Wadi Ham study area. Water levels react almost instantly with great magnitude to rainfall events. Therefore, steady-state calibration was not performed in this simulation exercise.

Based on the available geological information, the top layer is gravel and sand and the lower layer is ophiolite. The bottom ophiolite layer is impermeable in nature. Therefore one layer model of wadi gravel and sand is considered. Figures 4.2 and 4.3 show the contour map surface elevation and aquifer bottom in meters with reference to the mean sea level. Groundwater levels during the month of December 1988 in all observation wells were considered to initial groundwater levels in the study area (Ministry of Agriculture and Fisheries, 1989). The concurrent data of only 8 wells were available out of 16 wells that have been considered for the calibration of model from January 1989 to December 1993.

The model calibration was achieved by changing three parameters, namely, hydraulic conductivity,, specific yield and pumping rates. In order to simulate the sudden response for the recharge in observation boreholes located in the vicinity of the dam, a rather small specific yield was imposed. Abstraction and inflow across the boundaries were also simulated by a number of simulation runs till the desirable calculated heads in each observation wells were achieved.

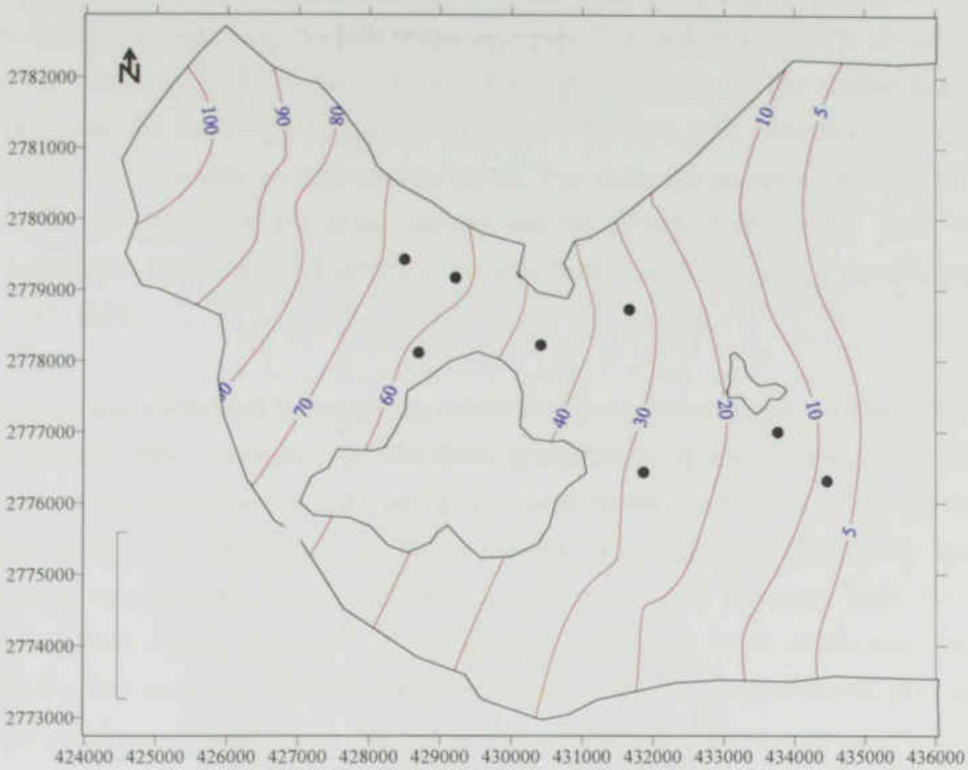

Figure 4.2. Ground-surface elevation contour map, meters with reference to seawater intrusion.

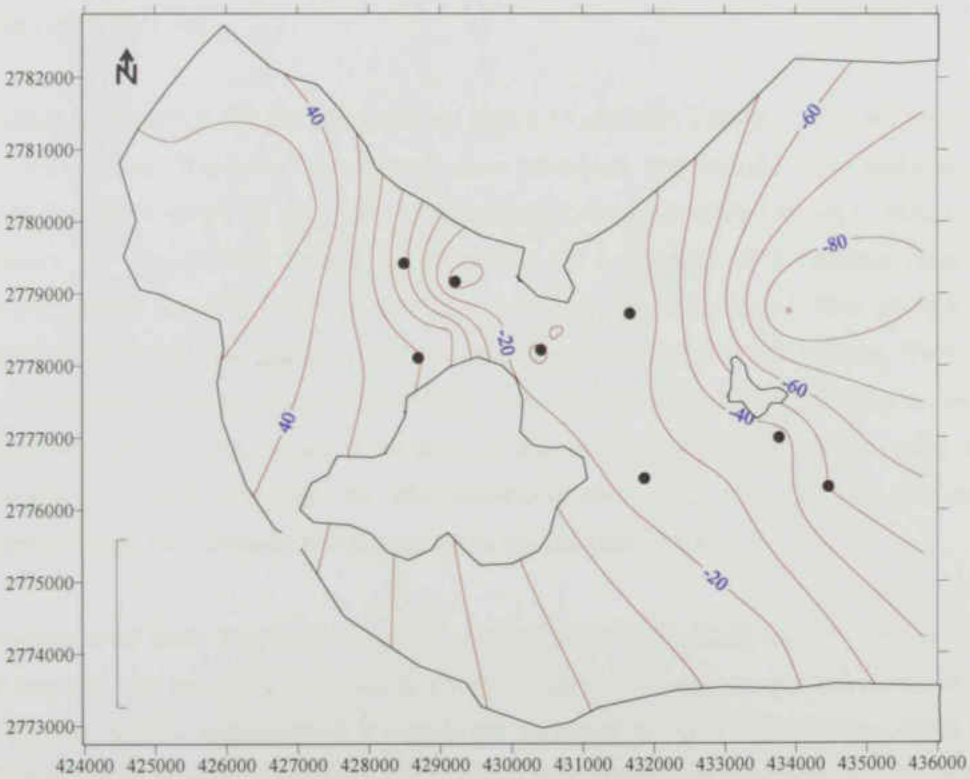

Figure 4.3. Contour map of aquifer bottom, meters with reference to seawater intrusion.

The recharge due to rainfall was also adjusted to ensure that the calculated heads at observation points are reasonably matching the field measurements. The recharge factor is about 20 percent of the rainfall. Although relatively high, the sand and gravel nature of the aquifer system in the study area allows for such high recharge. On the other hand, field observations indicated the direct effect of rainfall events on groundwater levels. The recharge factor in the ponding area was considered as 40 percent of the actual rainfall volume (Sherif et al., 2005). The calibration process indicated a hydraulic conductivity in the range of 150-10 m/d and a specific yield in the range of 0.01– 0.04.

The calibration was performed for matching measured groundwater levels in observation wells of complete records with corresponding simulated groundwater levels in space and time. Time series graphs of simulated versus observed groundwater levels are shown in Figures 4.4-4.6 for 3 observation wells. As illustrated by these figures, the model simulates the trends and groundwater levels resulting from groundwater abstractions and recharge from the reservoir storage and rainfall events. However, the limited discrepancies in some of the peak values may be attributed to the accuracy of observed groundwater levels as measurements are taken once every month and not necessary on the same day of every month. Based on the comparison presented in Figures 4.4-4.6, the model is considered to be calibrated for the period January 1989 to December 1993. It should be noted that no records for salt concentration in groundwater levels were available and hence the calibration was conducted for groundwater levels only.

4.6 Validation of SUTRA

The validation was carried out for a duration of about 11 yrs from January 1994 to March 2005 for a total of 4108 days. The time interval was also taken as one month. The average period of storage for the each event in the dam is considered approximately 60 days. Abstraction and inflow across the boundaries were also simulated by a number of simulation runs until the desirable calculated head in each observation well has been achieved. The pumping rate at Saraah well field during the initial period of validation was about 3150 m^3/day. This rate was gradually decreased down to reach 1700 m^3/day. This represents the closing down of few wells at Saraah well field either due to drying up of wells or the deterioration of the water quality. However, during the maximum pumping rate, the total volume of abstraction from the well field was 1.150 MCM which is about the same as the pumping during the year 1988.

In the Fujairah well field, the draft rate was scaled down from 2250 to 750 m^3/day during the validation period. The pumping from several wells in this field was terminated during this period. The well field at Kalbha experienced a significant increase in pumping rate from 4000 to 20000 m^3/day. The number of wells increased significantly after 1995. The present pumping rate is about 20000 m^3/day, which is more or less the same as the pumping rate that was provided by the Sharjah Electricity and Water Authority.

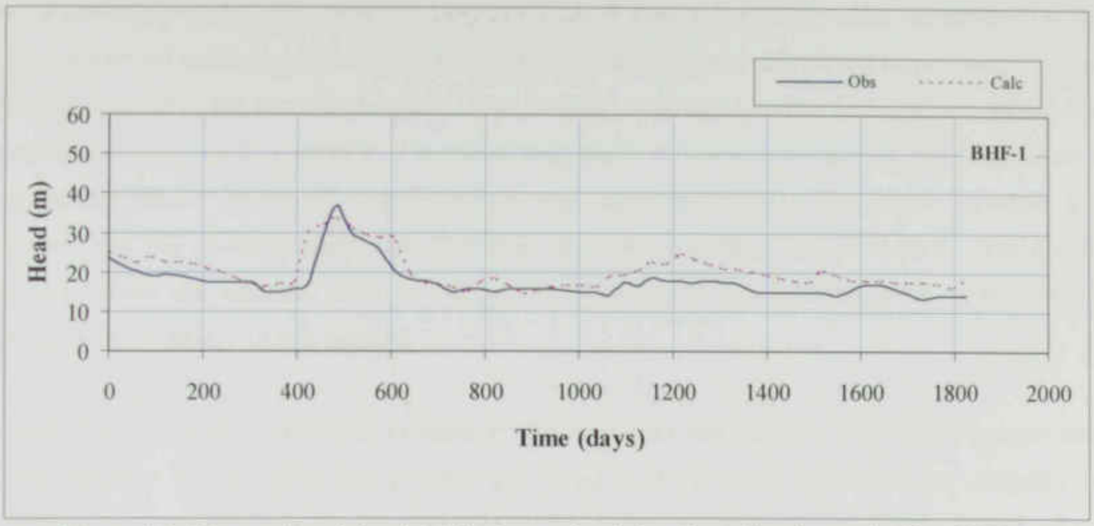

Figure 4.4. Observed and simulated hydrographs during the calibration period for BHF-1.

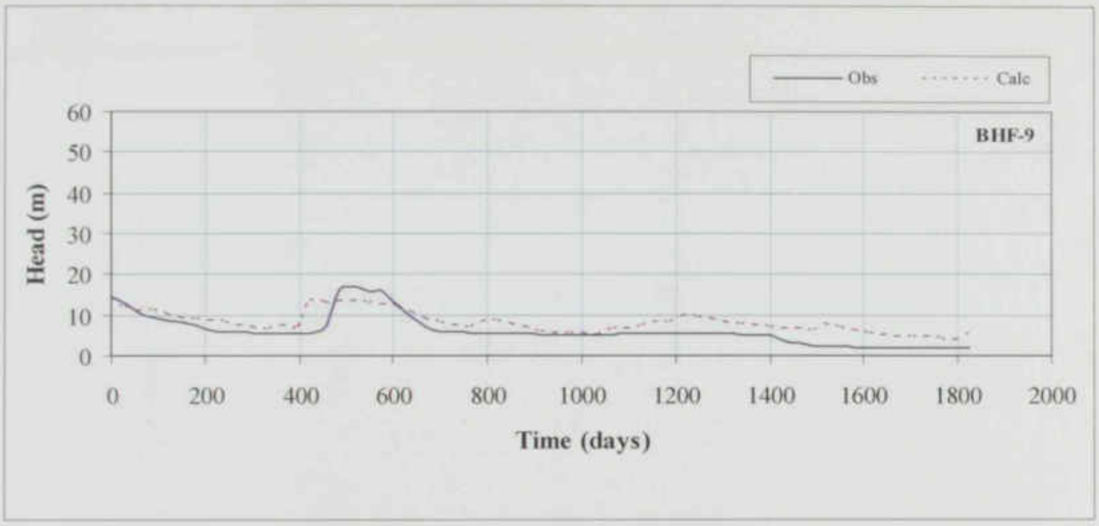

Figure 4.5. Observed and simulated hydrographs during the calibration period for BHF-9.

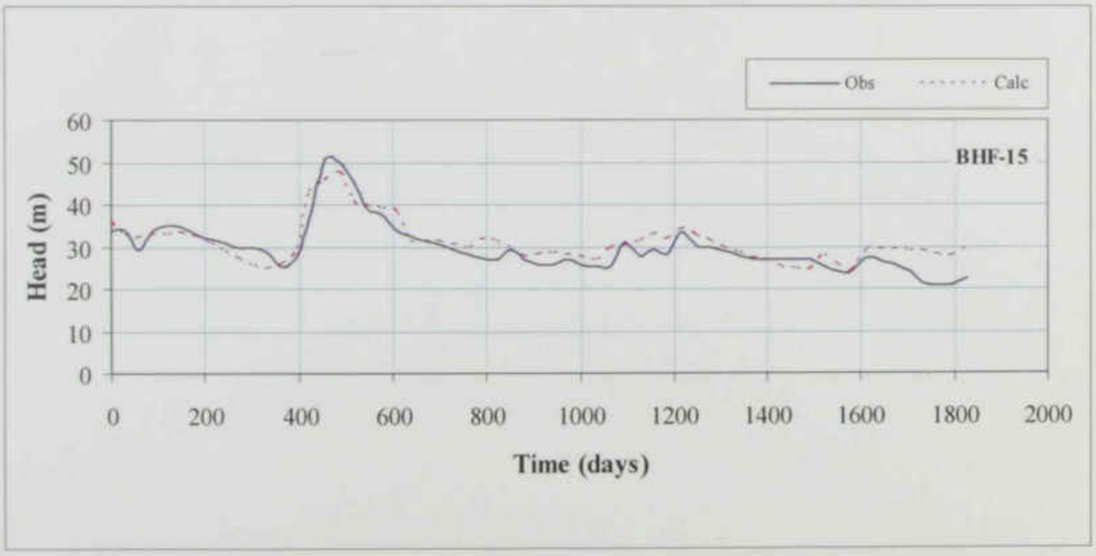

Figure 4.6. Observed and simulated hydrographs during the calibration period for BHF-15.

The observed field hydrograph and simulated hydrographs are presented in Figures 4.7-4.9 for three observation wells. Simulated hydrographs of all the observation wells exhibited the same trend of observed hydrographs. However, in some cases low and calculated peak head values did not match well with the observed heads. This may be attributed to the fact that the exact day of recording the groundwater level in the monitoring wells is not known and is not fixed from one month to the other. Observation wells which are close to the dam are more sensitive to the recharge from the ponding area than the wells situated far away from the dam. This could be attributed to the low specific yield of the aquifer, storage in the ponding area of the dam and extent and confinement of the aquifer.

The analysis of the time series of storage in the dam and rainfall over the study domain shows that the variation in buildup of the groundwater table is much more related to the contribution of the rainfall and the storage in the dam rather than the inflow from the upper reach which is relatively small. It is also noted that peak values of the simulated groundwater levels are slightly shifted from the peak values of the observed levels. This might be attributed to the effect of the unsaturated zone which is not considered in this modeling exercise.

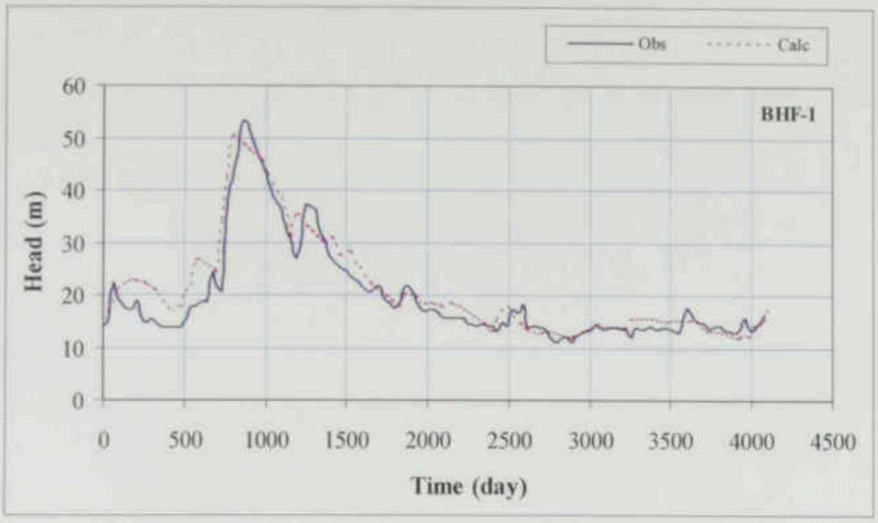

Figure 4.7. Observed and simulated hydrographs for the validation period (BHF-1)

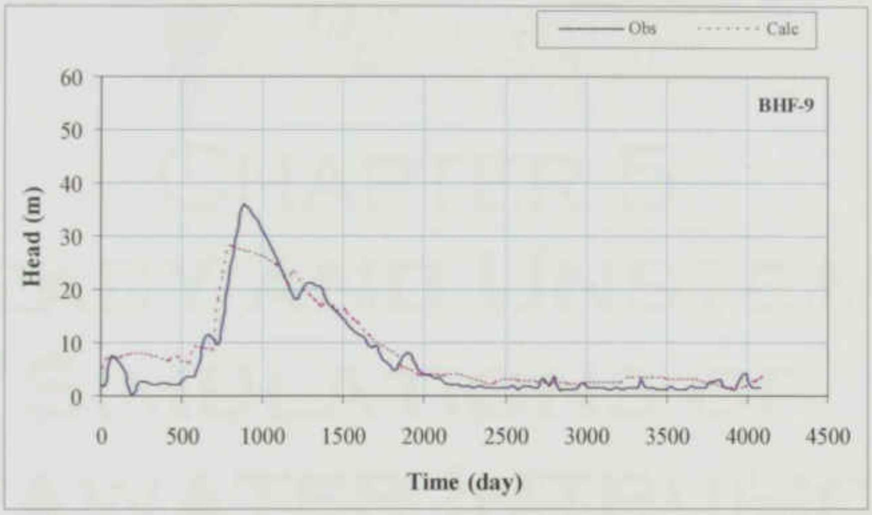

Figure 4.8. Observed and simulated hydrographs for the validation period (BHF-9)

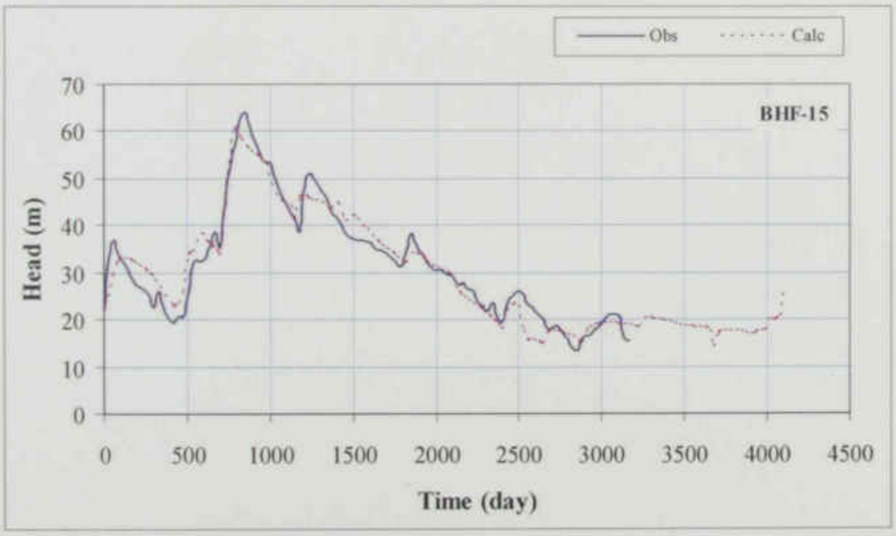

Figure 4.9. Observed and simulated hydrographs for the validation period (BHF-15)

CHAPTER 5

STEADY AND UNSTEADY SIMULATIONS OF SEAWATER INTRUSION

Chapter 5. Steady and Unsteady Simulations of Seawater Intrusion

5.1 Introduction

This chapter is devoted to the presentation and discussion of different simulation runs of steady and unsteady (transient) conditions of groundwater flow and solute transport in the aquifer of Wadi Ham. Simulation runs are conducted in two dimensional horizontal (areal) and two-dimensional vertical (cross sectional) flow fields. Although the new version of SUTRA can handle three dimensional flow fields, it requires comprehensive data and field measurements that may not be available in most cases. The lack of three-dimensional data, including among others, anisotropic hydraulic conductivities and dispersivities hinders the implementation of three-dimensional solute transport models except for a limited number of field cases.

It should also be noted that fully steady-state conditions of solute transport in groundwater systems cannot be achieved under most field (natural) conditions. Solute transport steady state conditions may require several decades (or even hundreds of years) to be achieved under constant boundary conditions and fixed excitations. In other words, seawater level and concentration, pumping and recharge activities including rainfall and irrigation practices, and inflow and outflow through the different boundaries of the domain under consideration should remain constant for several decades during which the steady-state conditions can be achieved. Because most of the parameters are not constant with time and cannot be fixed for several decades, the flow and solute transport will always remain under transient conditions.

Steady-state simulations, however, provide the extreme situation of any contamination or seawater intrusion problem. Therefore, such simulations would provide an allusion on whether the area under investigation might be exposed to groundwater quality deterioration on the long term. Steady-state conditions can be simulated in SUTRA either by selecting the steady state option or by allowing the transient simulation to continue until achieving the steady state under which the changes, between two successive iterations, in groundwater levels and concentrations is very small and can thus be neglected.

Hydrogeological systems are mostly heterogeneous and isotropic. On the other hand, estimation of most hydrogeological parameters including, porosity, specific yield, hydraulic conductivity, dispersivity and others is based on some measurements that are conducted at specific points. The measured values of the different parameters are then interpolated "or generalized" to cover the entire study domain. This approximation of parameters involves uncertainties and hence field experiences and prior knowledge about the performance and response of the groundwater system under different excitations are required to ensure meaningful results. Calibration and validation of numerical models ensure that models are capable of representing the system under investigation under different flow conditions. Independent sets of data should therefore be used.

Because of the limited availability of data, the results presented in this chapter should be considered in a qualitative manner rather than quantitative.

5.2 Areal Simulations

Areal (horizontal) simulations were conducted to assess the behavior of the Wadi Ham aquifer under different pumping scenarios. The effects of changing the hydraulic conductivities and dispersivities were also investigated.

5.2.1 Steady State Simulations

Three different sets of runs were conducted for areal simulation under steady-state conditions. In all runs the hydraulic head at the land side, upstream of the Wadi Ham Dam was set as 45 m (amsl). The hydraulic head at the Gulf of Oman was set equal to the seawater level. The hydraulic conductivity in the longitudinal direction, K_{xx}, was set equal to 86.4 m/d. In the lateral (transversal) direction the hydraulic conductivity, K_{yy}, was set equal to 8.64 m/d. The ration between the longitudinal and lateral hydraulic conductivities was set as 10:1. The longitudinal dispersivity, α_L, was set equal to 200 m while the transverse dispersivity, α_T, was set equal to 10 m. These values were set based on initial trials of SUTRA simulations to obtain an adequate concentration distribution in the aquifer. The ratio between the longitudinal and transverse dispersivities was set as 20:1. Values of the hydraulic conductivities were based on the obtained results from the calibration process, while values of the longitudinal and transverse dispersivities were based on data found in the literature. The seawater concentration was set as 35 kg/m^3. The following groups of runs were conducted:

Group 1: Effect of Pumping

Hydraulic parameters were kept constant. Pumping rate from Khalba well field was changed as:
Run 1: Total pumping was set as 450 m^3/d. (Basic Run)
Run 2: Total pumping was set as 900 m^3/d.
Run 3: Total pumping was set as 1800 m^3/d.
Run 4: Total pumping was set as 4000 m^3/d.

The results of the above four simulation runs are presented in Figures 5.1-5.5. Figure 5.1 presents the resulting equipotential lines under the steady-state conditions of Run 1, while the Figures 5.2-5.5 demonstrate the resulting equiconcentration lines under various pumping scenarios. An increase in the pumping from the Khalba well field has a major impact on the seawater intrusion in the Wadi Ham aquifer. Under the reduced pumping rate of 450 m^3/d, seawater intrusion was limited to the southern eastern part of the study domain as indicated in Figure 5.2. As the pumping increased, the seawater occupied more area. Under scenario 4, the seawater occupied about 50% of the study domain. The results indicated that seawater intrusion is very much dependent on groundwater pumping from the Khalba well field.

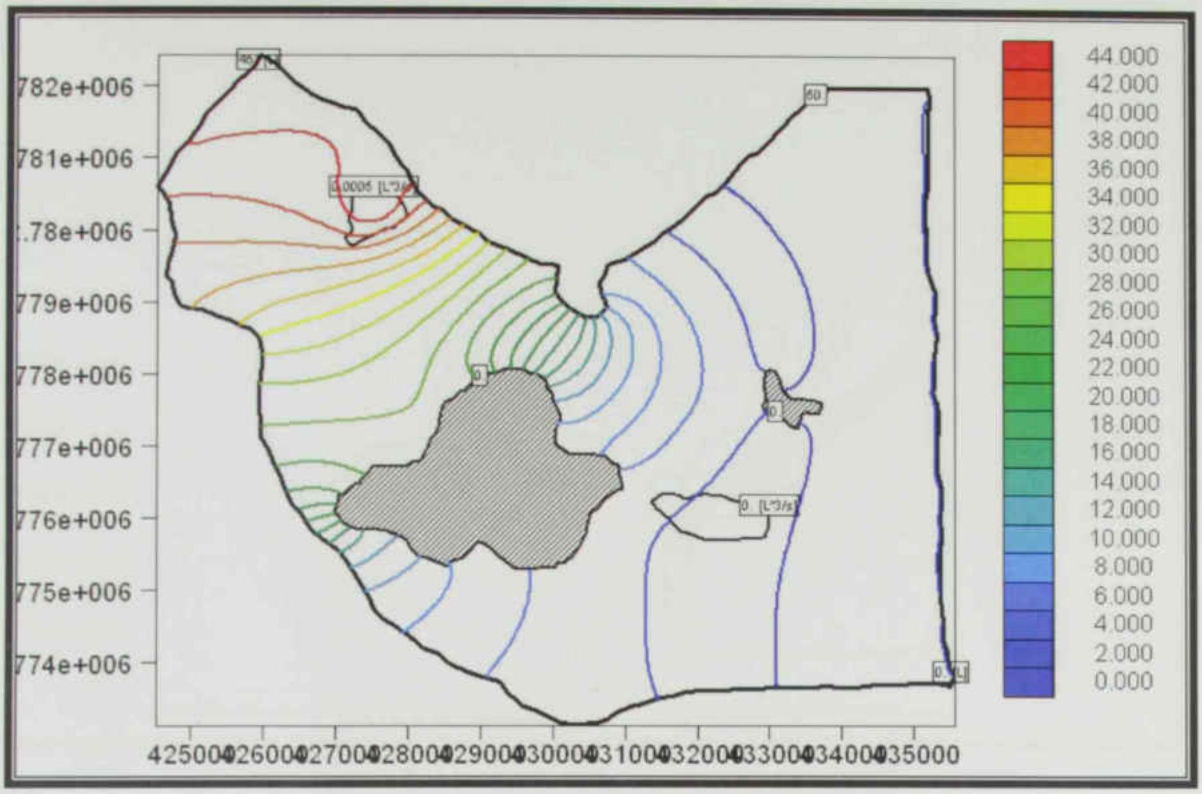

Figure 5.1. Equipotential under steady-state conditions (Run 1, Group 1).

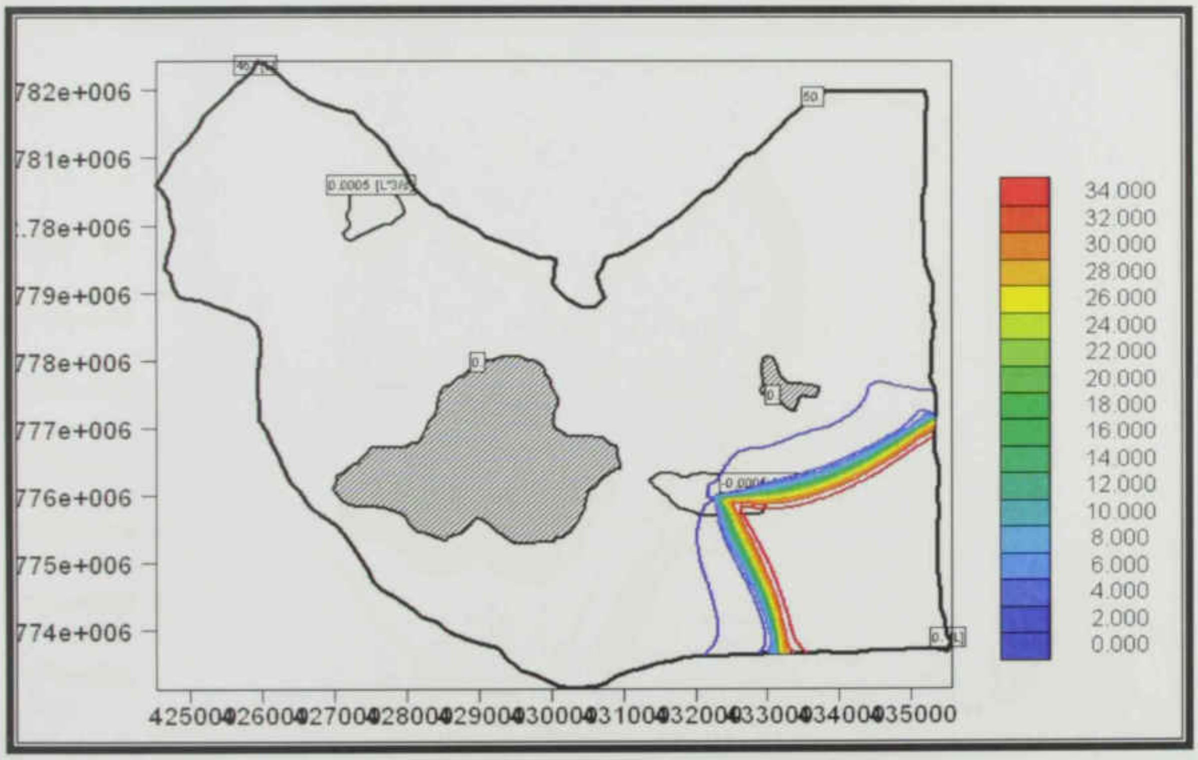

Figure 5.2. Equiconcentration lines under steady-state conditions (Run 1, Group 1).

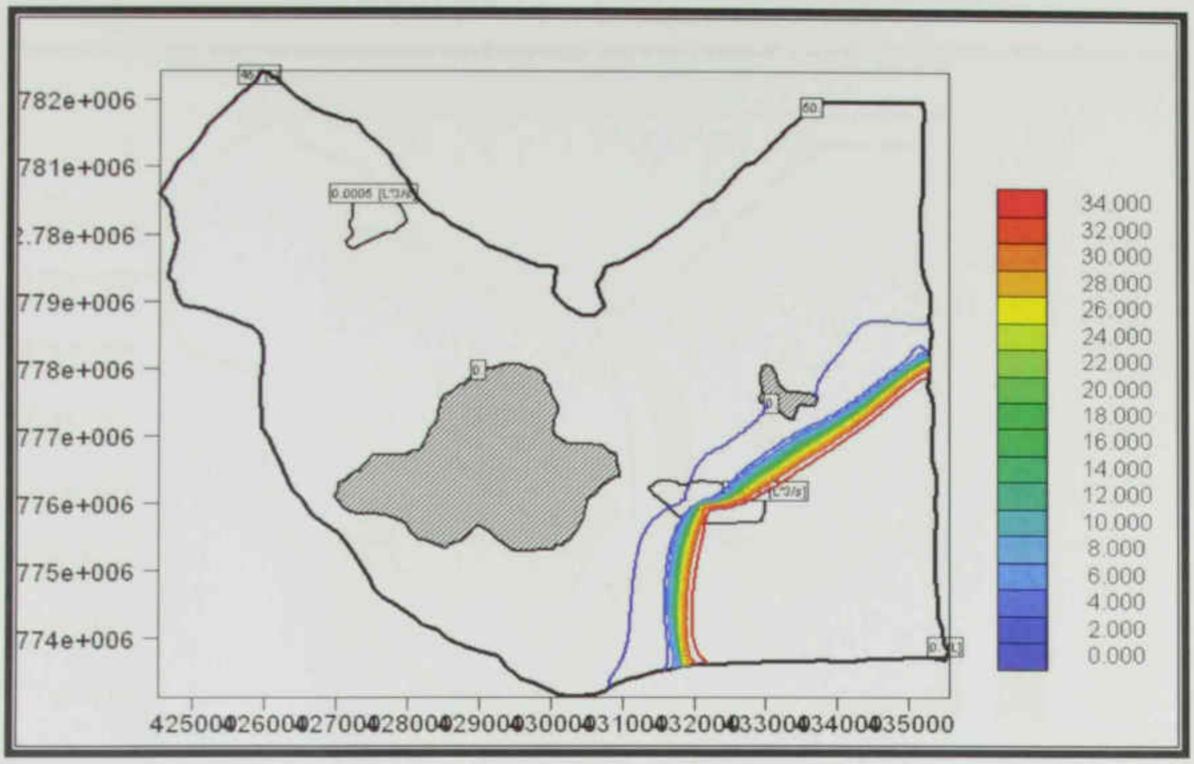

Figure 5.3. Equiconcentration under steady-state conditions (Run 2, Group 1).

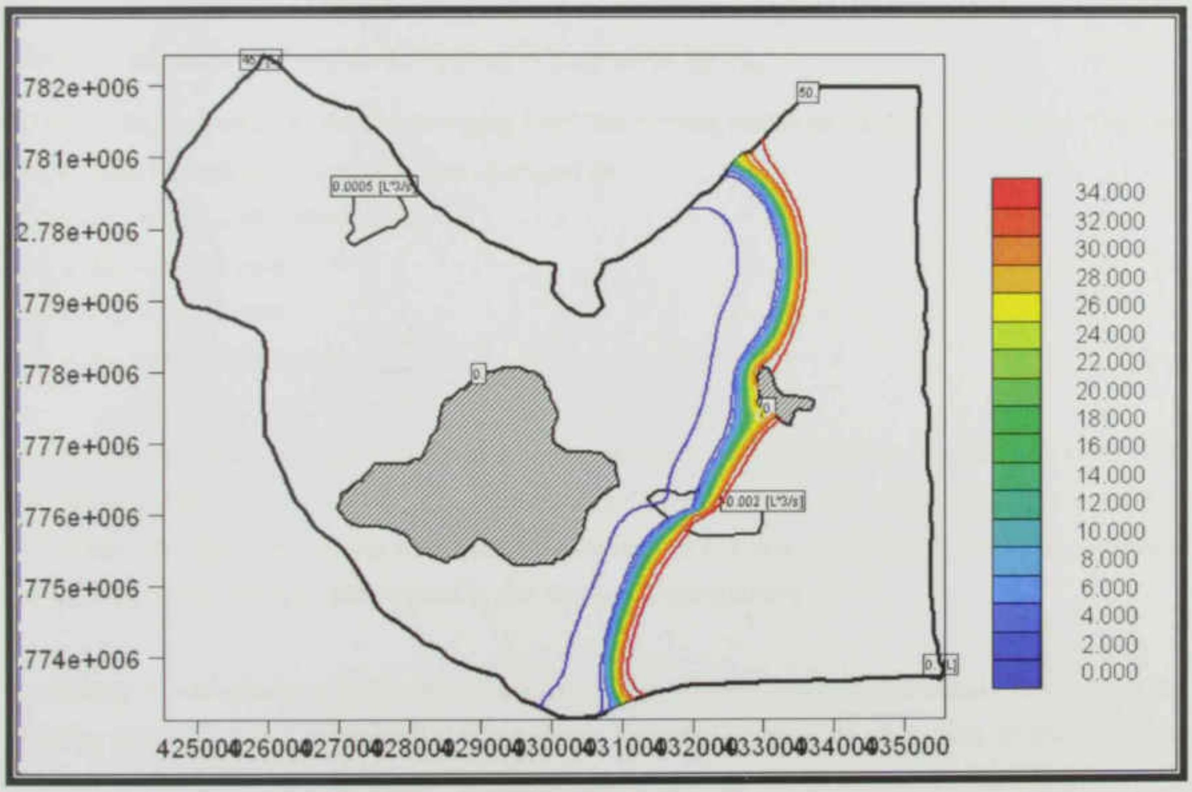

Figure 5.4. Equiconcentration lines under steady-state conditions (Run 3, Group 1).

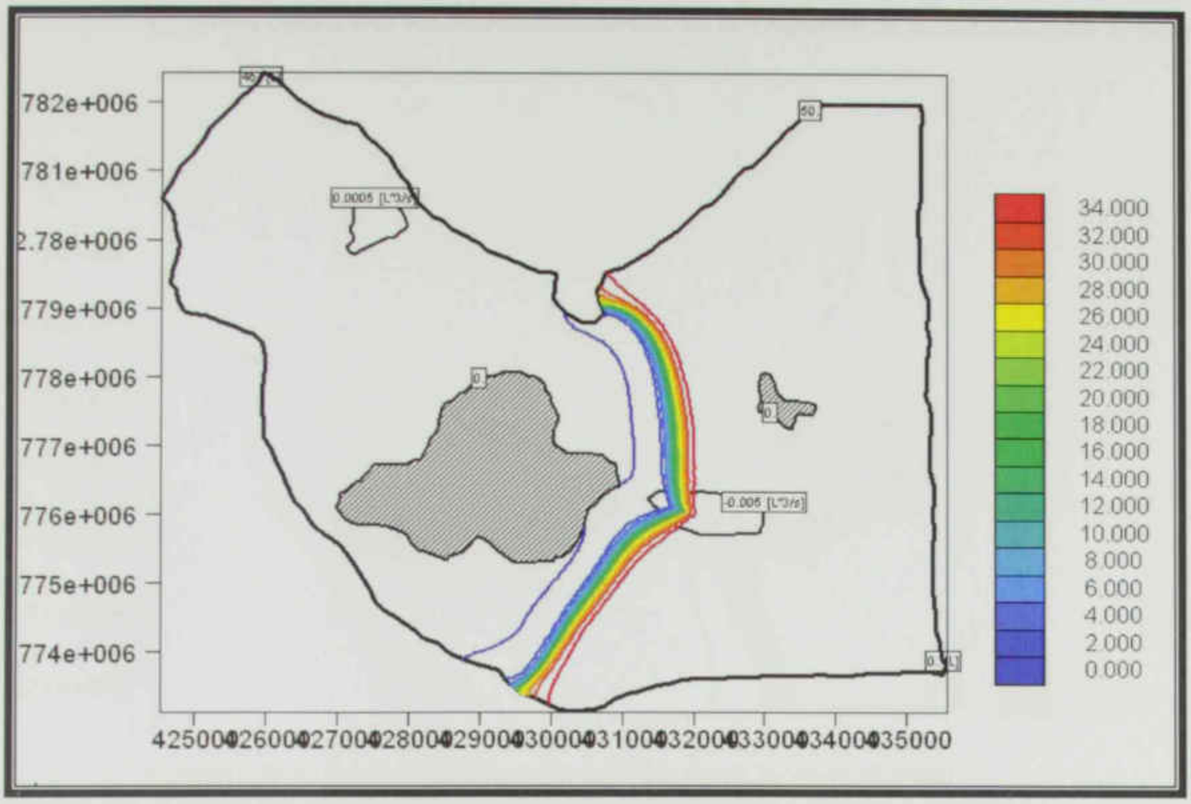

Figure 5.5. Equiconcentration lines under steady-state conditions (Run 4, Group 1).

Group 2: Effect of Longitudinal Hydraulic Conductivity, K_{xx}

All hydraulic parameters and the pumping from the Khalba well field were kept constant. Only the longitudinal hydraulic conductivity was changed as:

Run 1: K_{xx} = 86.4 m/d. (Basic Run)
Run 2: K_{xx} = 100.0 m/d.
Run 3: K_{xx} = 120.0 m/d.
Run 4: K_{xx} = K_{yy} = 8.64 m/d.

The results of the above four scenarios are presented in Figures 5.6-5.9. It should be noted that the lateral hydraulic conductivity, K_{yy}, was kept unchanged (8.64 m/d). Therefore, the degree of anisotropy changed among the four tested scenarios. In the last scenario (Run 4), the system is considered to be isotropic with regard to the hydraulic conductivity.

Increasing the longitudinal hydraulic conductivity, K_{xx}, allowed more freshwater to travel from the ponding area of the dam toward the seaside. Therefore, the less area was affected by the seawater intrusion. However, when K_{xx} was reduced and set equal to K_{yy}, the seawater intrusion occupied more area as less freshwater was able to move toward the seaside (Figure 5.9).

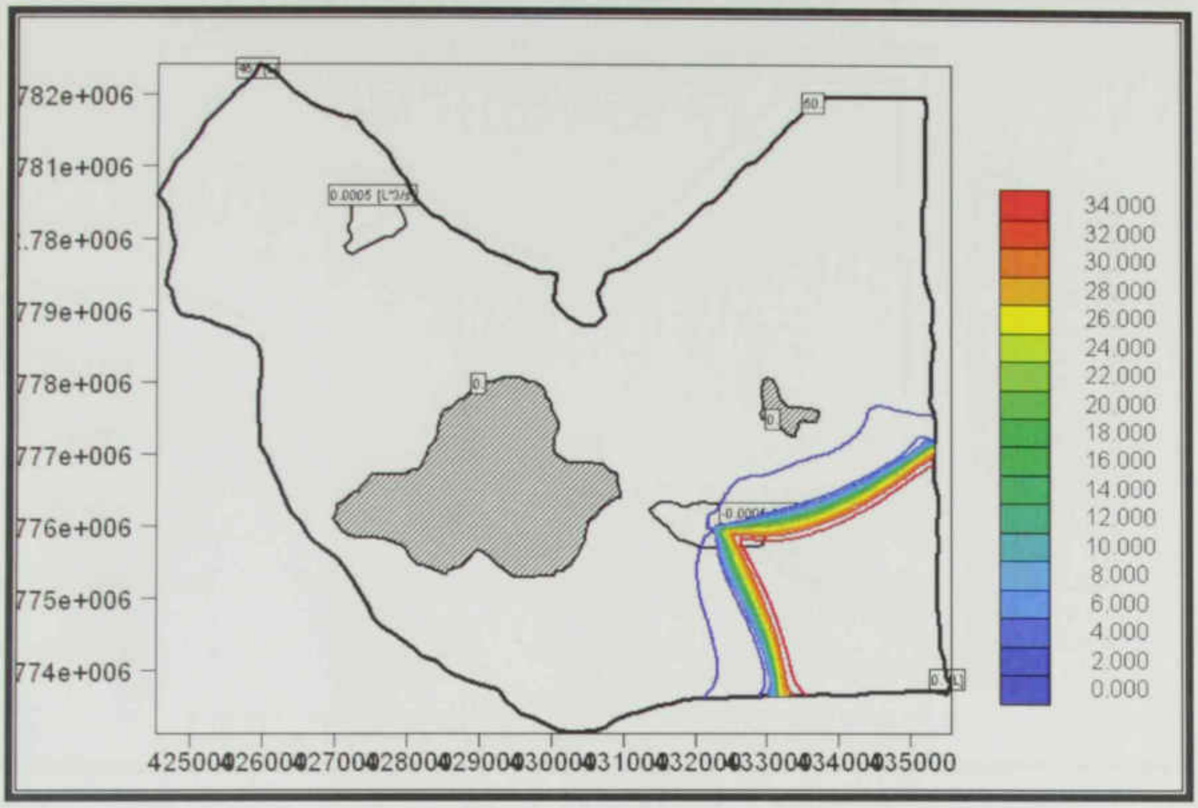

Figure 5.6. Equiconcentration lines under steady state conditions (Run 1, Group 2).

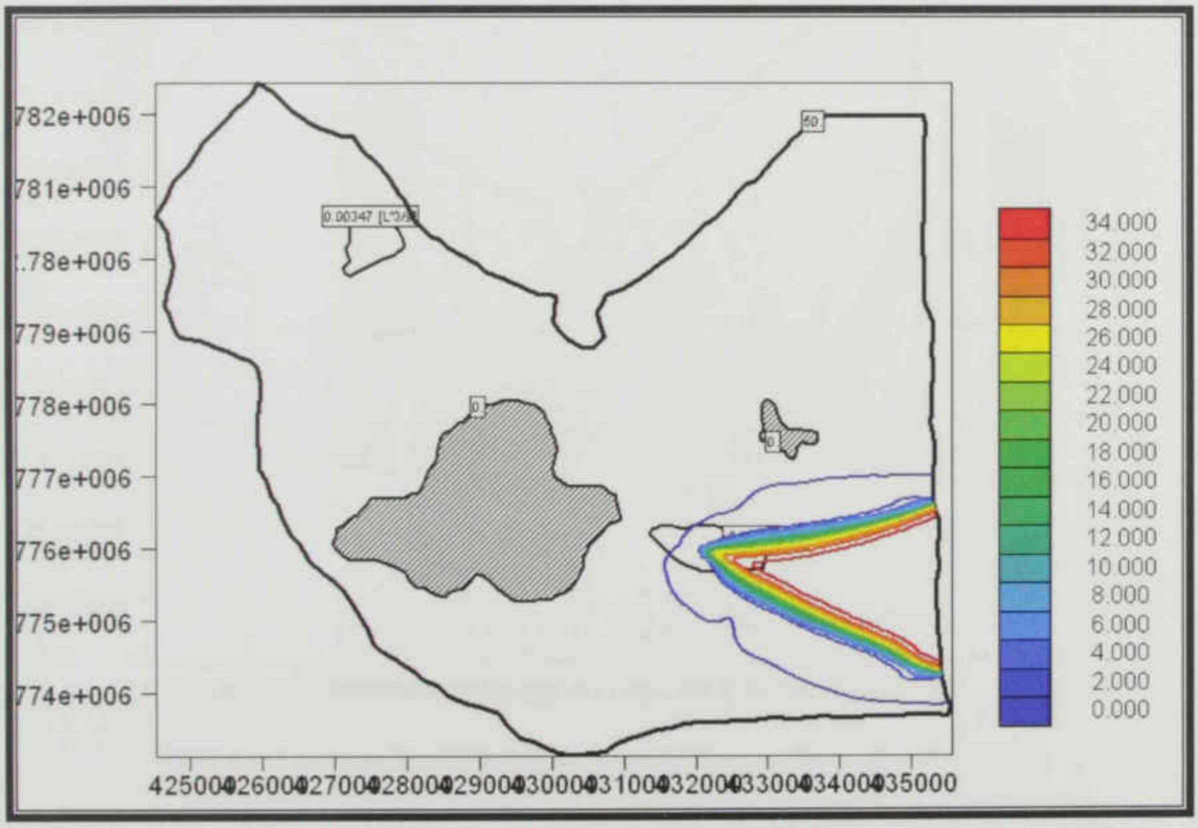

Figure 5.7. Equiconcentration lines under steady state conditions (Run 2, Group 2).

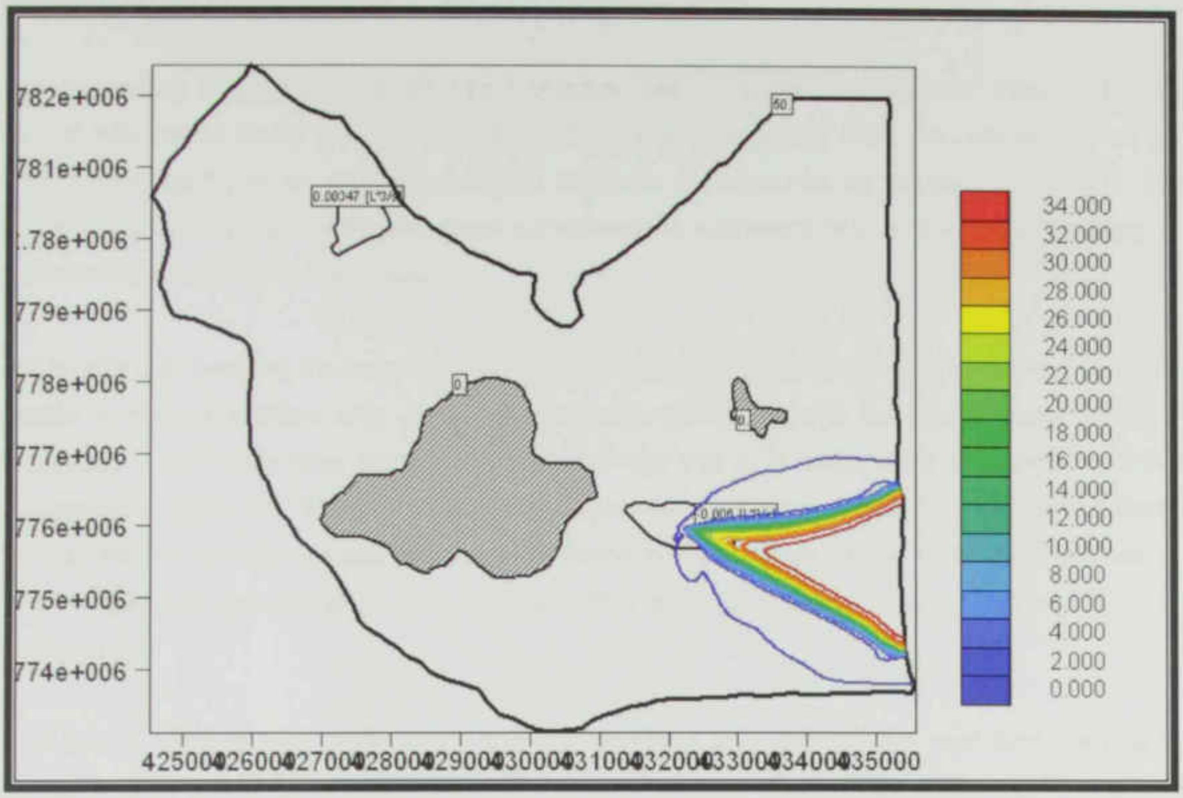

Figure 5.8. Equiconcentration lines under steady state conditions (Run 3, Group 2).

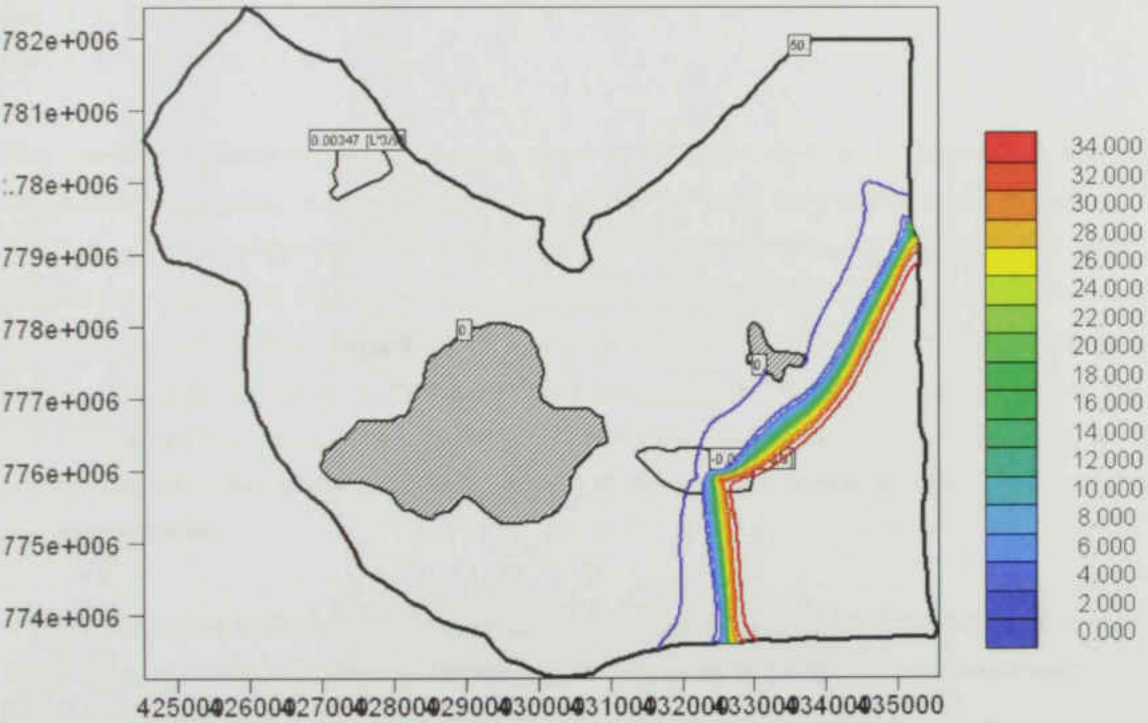

Figure 5.9. Equiconcentration lines under steady state conditions (Run 4, Group 2).

Group 3: Effect of Dispersivity

The dispersivity in the area of Wadi Ham has never been measured. Dispersivity accounts for the lack of information about the pore velocity fluctuation when passing from the microscopic to the macroscopic configuration of the solid-liquid interface. It measures dispersive properties of the system. Dispersivity has traditionally been considered as a characteristic single-valued property of the entire medium (Sherif et al. 1988).

For an isotropic medium, the number of non zero components of the dispersivity tensor is 21. All are related to two parameters only, the longitudinal dispersivity, α_L, and the lateral dispersivity, α_T. Laboratory experiments have shown (Sherif et al. 1990) that α_L is of the order of magnitude of the average sand grain size. Transverse or lateral dispersivity α_T is estimated as 10 to 20 times smaller than α_L. Values between 0.1 and 500 m can be found in the literature. In this study, the basic values of the longitudinal dispersivity, α_L, and the lateral dispersivity, α_T, were set equal to 200 m and 10 m, respectively.

In Group 3, all hydraulic parameters and the pumping from the Khalba well field were kept constant as for the first two groups. Only the longitudinal and lateral dispersivities were changed between among the different runs.

Run 1: $\alpha_l = 200$ m, $\alpha_t = 10$ m.
Run 2: $\alpha_l = 200$ m, $\alpha_t = 1$ m.
Run 3: $\alpha_l = 100$ m, $\alpha_t = 1$ m.
Run 4: $\alpha_l = 50$ m, $\alpha_t = 1$ m.

The resulting equiconcentration lines are given in Figures 5.10-5.13. In scenario 2, where the longitudinal dispersivity, α_l was kept constant as 200 m, while the transverse dispersivity, α_t was reduced to 1 m, the lateral spread of the seawater intrusion reduced significantly. The impact of intrusion was limited to the area in the proximity of the Khalba well field (Figure 5.11). In scenario 3, where the longitudinal dispersivity α_l was reduced to 100 m, the affected area remained as before. However, the width of the dispersion zone (distance between equiconcentration lines 0 and 34) decreased significantly (Figure 5.12). Reducing the longitudinal dispersivity, α_l, to 50 m while maintaining the lateral dispersivity, α_t, at 1 m did not have a tenable impact on the seawater intrusion process.

It should be noted that all the above simulations (presented in Figures 5.1-5.13) were conducted under the steady-state conditions. Therefore, the time scale of the seawater intrusion process has not be considered.

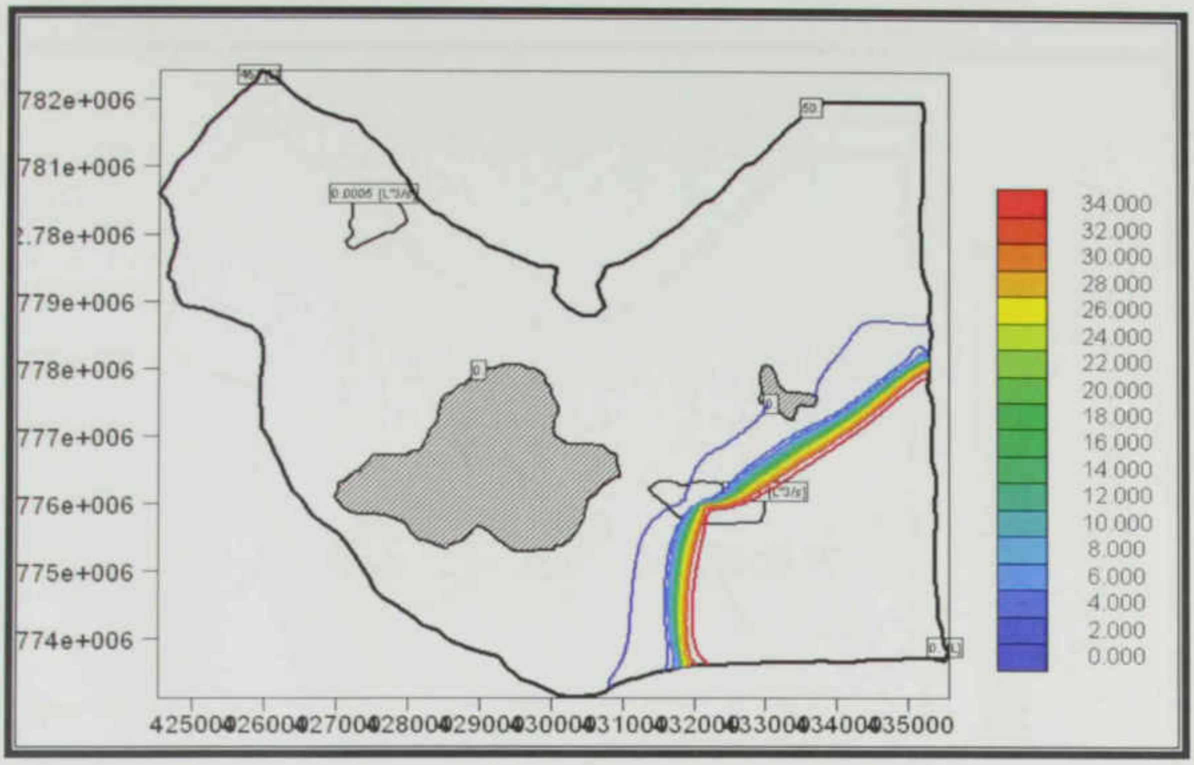

Figure 5.10. Equiconcentration lines under steady-state conditions (Run 1, Group 3).

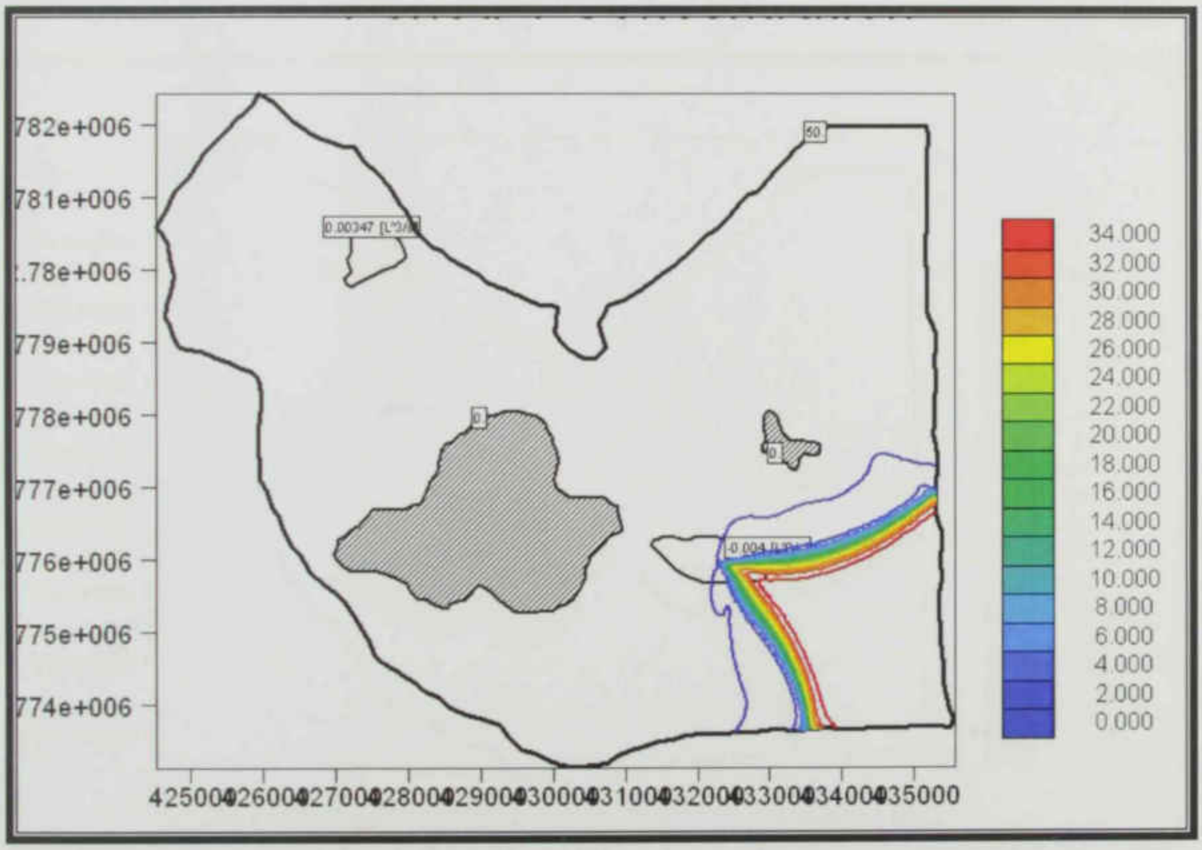

Figure 5.11. Equiconcentration lines under steady-state conditions (Run 2, Group 3).

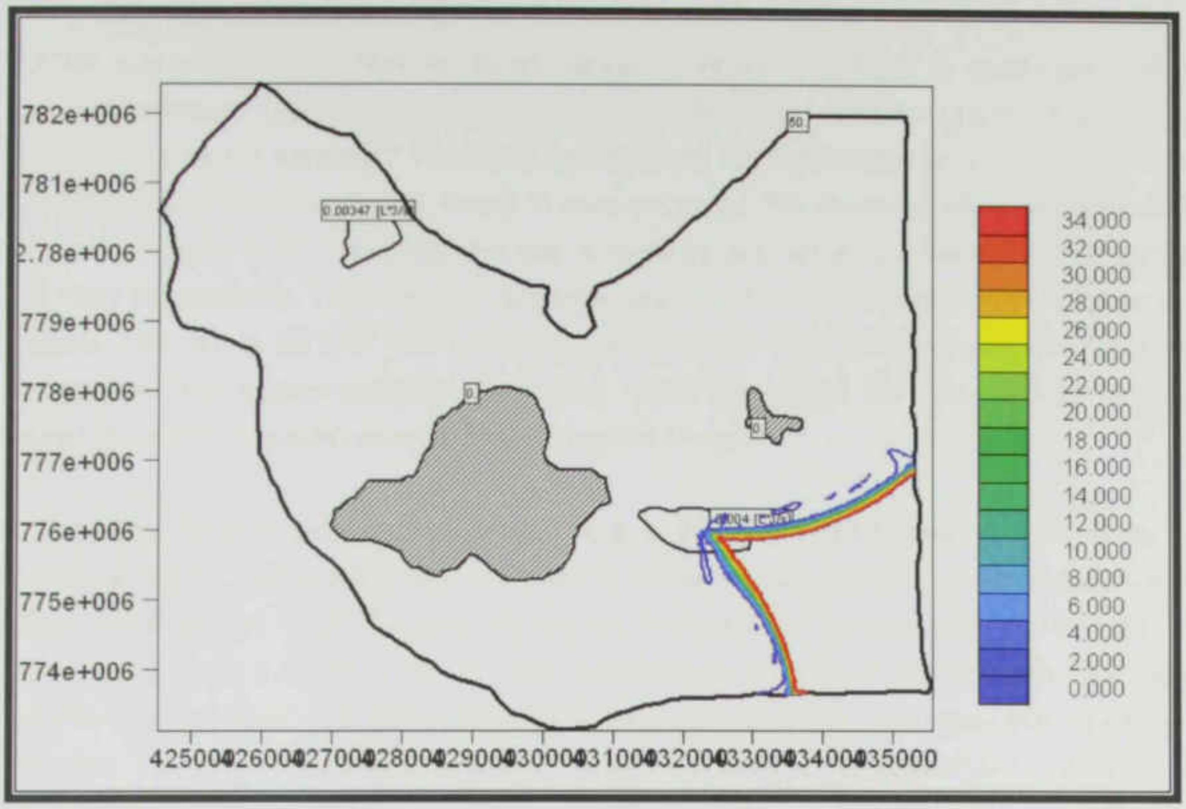

Figure 5.12. Equiconcentration lines under steady-state conditions (Run 3, Group 3).

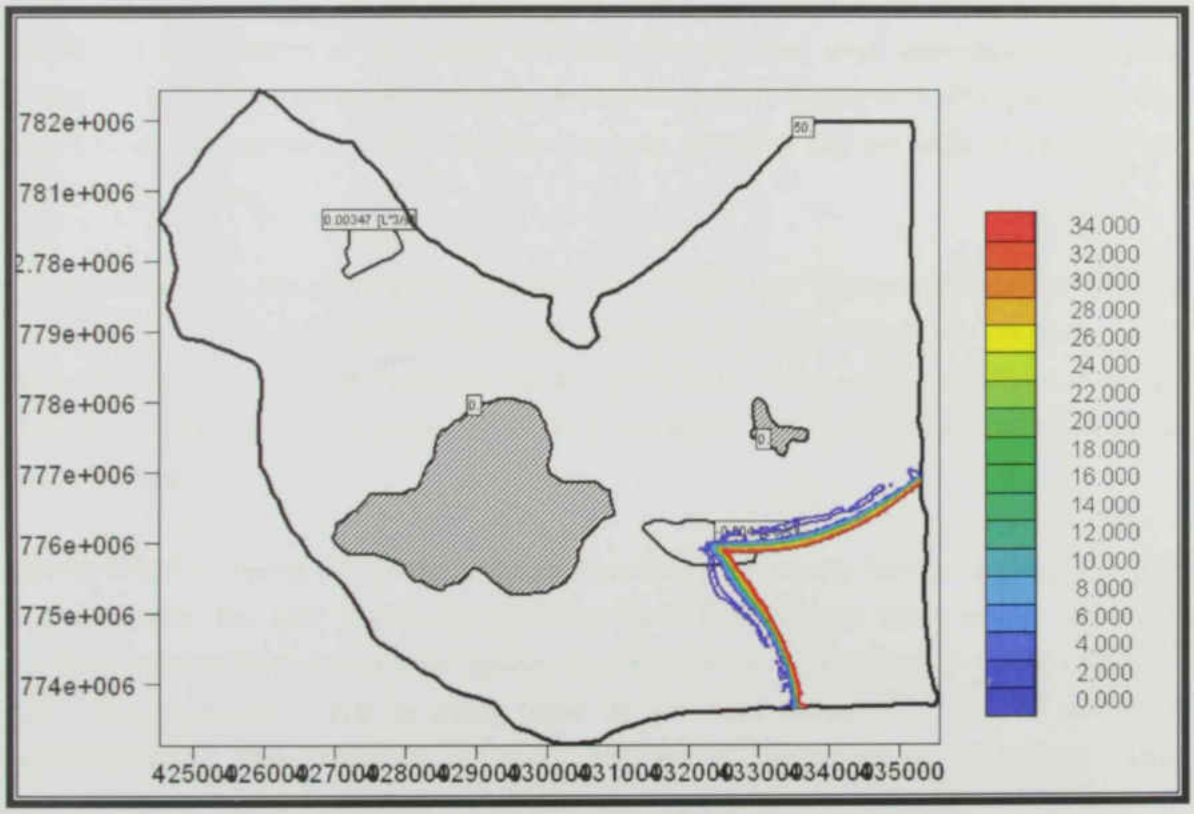

Figure 5.13. Equiconcentration lines under steady-state conditions (Run 4, Group 3).

5.2.2 Unsteady-State Simulations

SUTRA was employed to demonstrate the seawater intrusion process, in areal view, under transient conditions. The simulation was conducted with the option of "transient groundwater flow" and "transient solute transport." The same hydraulic and hydrogeological parameters that were used in the Basic Scenario (Run 1, Group 1) were employed. The maximum allowable simulation time was set equal to 10 years. The time step (in seconds) was set as one day with a multiplier of 1.2 every ten iterations. The maximum allowable time step (in seconds) was set as one month. Outputs "NPRINT in SUTRA" were produced every month. This allows to generate maps of equipotential and equiconcentration lines every month throughout the simulation period "10 years." As before, seawater concentration was set as 35 kg/m^3.

The resulting equiconcentration lines, after 1, 3, 6, and 12 years of simulation are presented in Figures 5.14-5.17, respectively. After one year of simulation, the maximum concentration in the study domain was 32 kg/m^3 (Figure 5.14) and this relatively high concentration was limited to a narrow zone along the Gulf of Oman. The concentration decreased rapidly inland. The high concentration (10 kg/m^3 and above) was limited to a zone of about 1 km away from the shore boundary. Otherwise, the concentration was relatively low (8 kg /m^3 and below) between the Gulf of Oman and the Khalba well field.

After three years of simulation (Figure 5.15), high equiconcentration lines started to migrate inland. Equiconcentration line 34 kg/m^3 migrated inland to a distance of about 1 km from the shoreline in the direction of the Khalba well field. The red lines (high equiconcentration lines) occupied a larger area as compared to the case presented in Figure 5.14. The green and blue lines (low equiconcentration lines) occupied less area indicating that the width of the dispersion zone has decreased.

The same seawater intrusion pattern continued after 6 years and 10 years of simulation (Figures 5.16 and 5.17) High equiconcentration lines continued to move inland and the width of the dispersion zone continued to decrease but at a lower rate. The seawater continued to occupy more area indicating more deterioration of the groundwater quality. The total affected area has also increased.

Figures 5.14-5.17 reveal that seawater intrusion occurs more rapidly during the early simulation time and then the rate of intrusion declines over the time. In other words, almost all equiconcentration lines moved faster during the first 3 years of simulation. It is also noted that lower equiconcentration tend to move faster as the early stage of simulation, while high equiconcentration lines continue to shift for a longer period. The system did not achieve steady-state conditions after 10 years. Such conditions might be fully achieved after several decades.

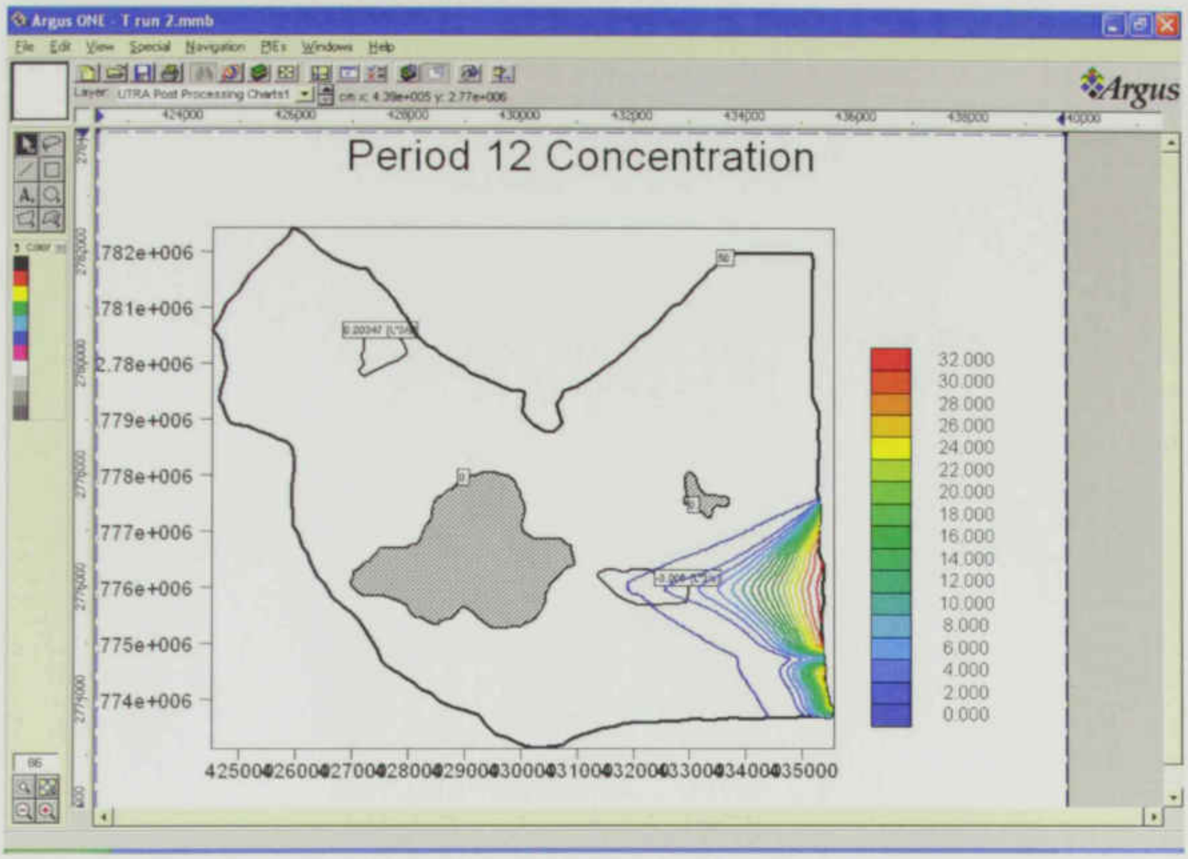

Figure 5.14. Equiconcentration lines after 1 year of simulation.

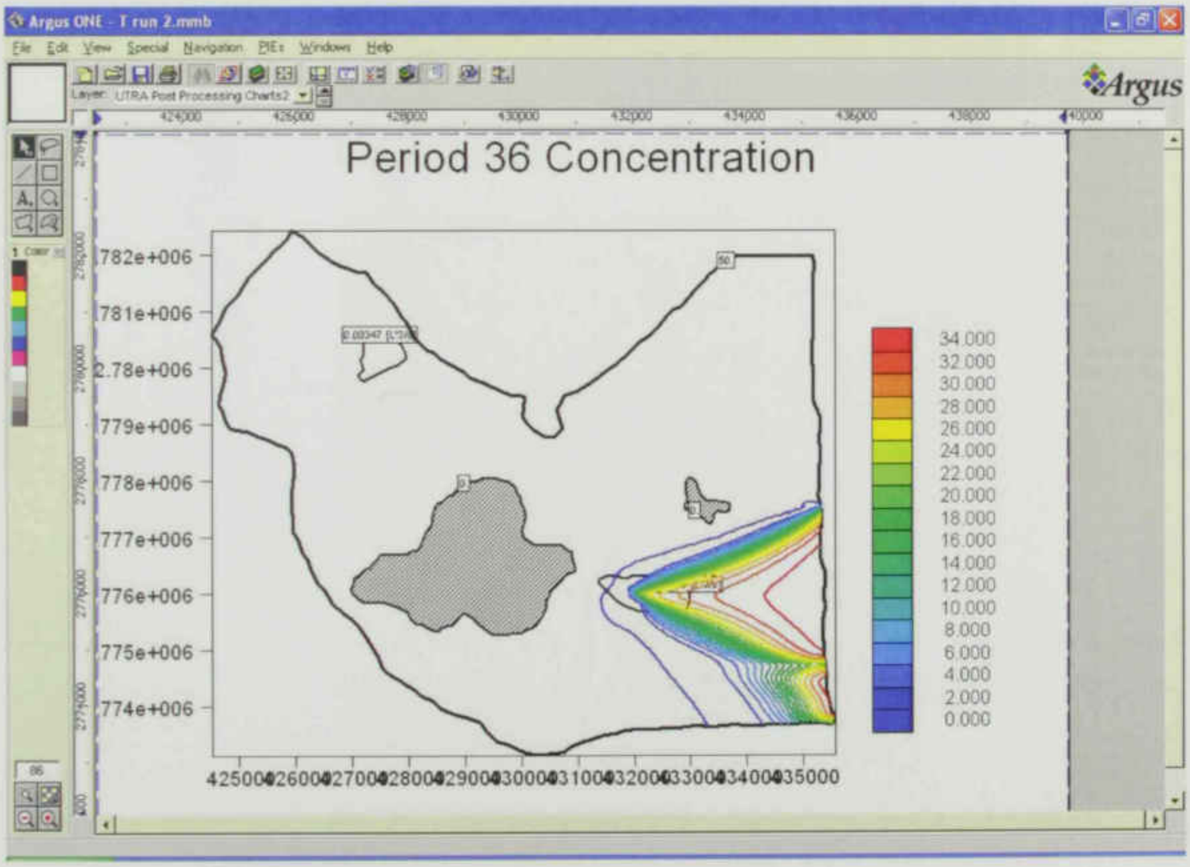

Figure 5.15. Equiconcentration lines after 3 years of simulation.

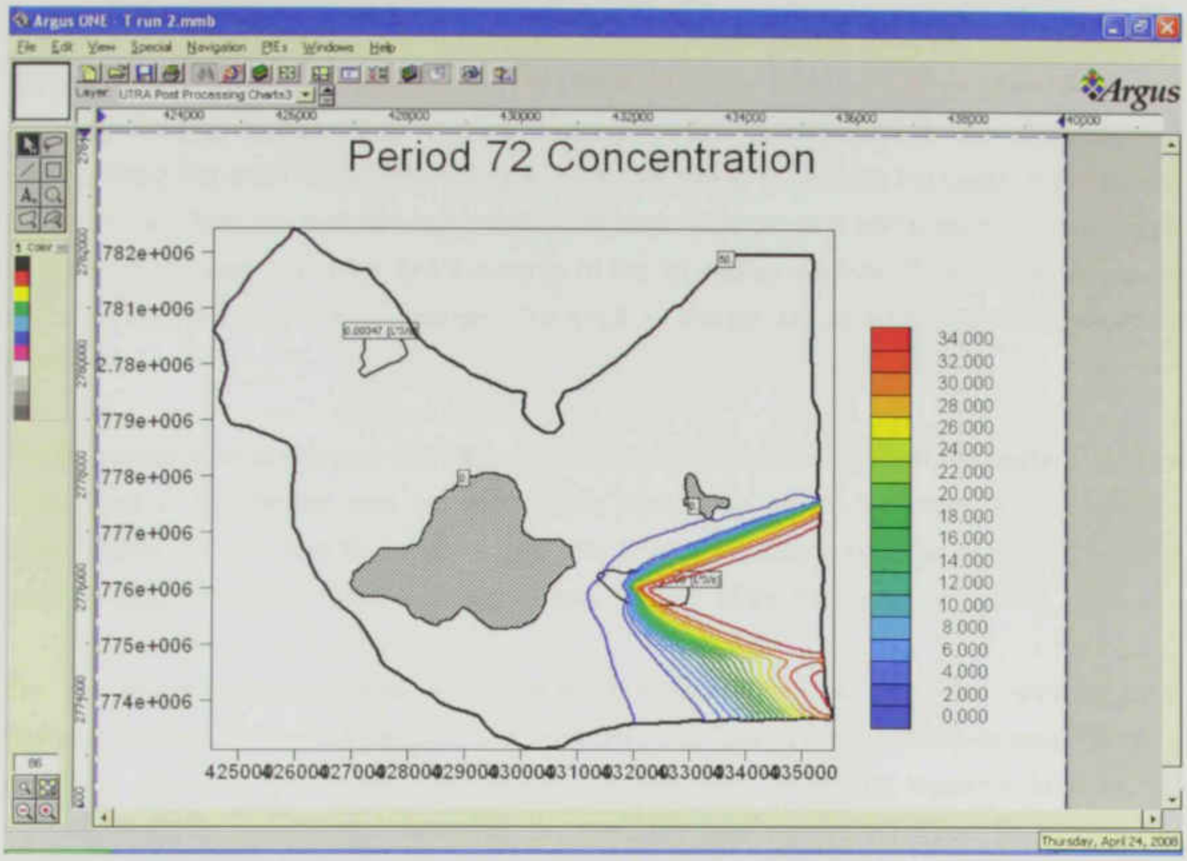

Figure 5.16. Equiconcentration lines after 6 years of simulation.

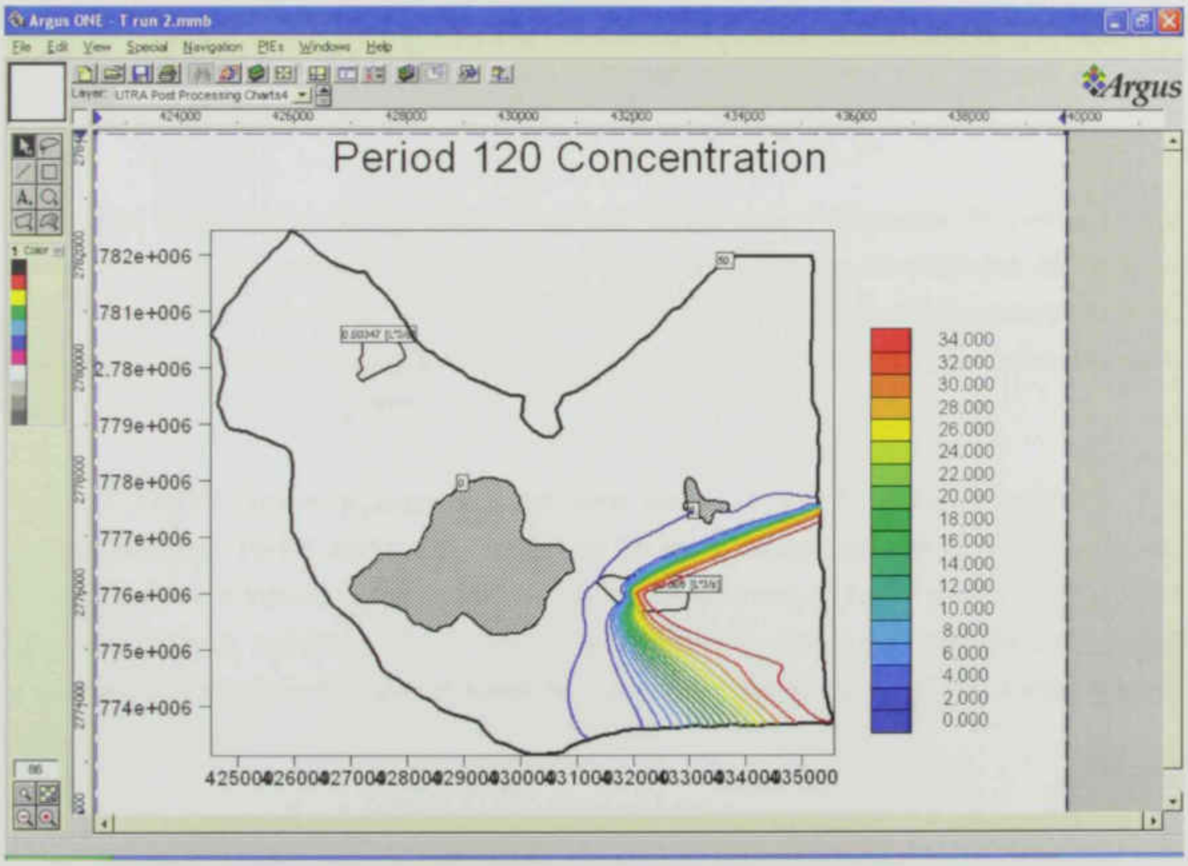

Figure 5.17. Equiconcentration lines after 10 years of simulation.

5.3 Vertical Simulation

SUTRA was employed to simulate the seawater intrusion in a vertical cross-section under unsteady-state conditions. Thus, a vertical cross section perpendicular to the shoreline and running along the main wadi was considered. The geometric shape and the depth of the aquifer were deduced from the available surface contour maps (topography) and boreholes. The length of the study domain was 9 km and the depth of the aquifer varied from 75 m at the seaside to about 25 m at the land side. In between, the depth of the aquifer varied based on the available boreholes.

The free water level at the land side was set as 45 m above the sea level. At the seaside, the free water level in the aquifer was set at the sea level. The bottom boundary was considered impermeable; i.e., no flow for water or salt ions is allowed. Other hydrogeological parameters were considered as given in the basic run (Group 1, Run 1 of the horizontal simulation).

The resulting equipotential lines after one month of simulation are presented, with an equal interval (Figure 5.18) Vertical equipotential lines were encountered in the study domain indicating horizontal flow field (horizontal streamlines). On the other hand, the distance between the equipotential lines near the land side was less than that near the sea boundary. This indicates a steep slope of the water table at the land side and a mild slope of the water table at the seaside. This is also very much consistent with the slope of the ground surface. The groundwater flow velocities at the landside were relatively high. Equipotential lines (and the velocity field) achieved steady-state conditions within one year and exhibited minor changes after the first month of simulation.

Figure 5.19 shows the equiconcentration lines after one month of simulation. The intrusion was limited to a short distance from the sea boundary and the maximum concentration, at the lower point of the seaside boundary, was 9 kg/m^3. The upper part of the seaside boundary remained fresh, indicating that the intrusion starts from the bottom of the aquifer and then moves inward to the land side.

Figures 5.20-5.22 present equiconcentration lines after 1, 5 and 10 years of simulation. The intrusion occupied the full depth of the aquifer as the equipotential lines were mostly vertical. On the other hand, the maximum concentration near the seaside was 22 kg/m^3 after the first year, 30 kg/m^3 after 5 years, and 33 kg/m^3 after 10 years of simulation. Unlike the flow field, the transport of salts and the seawater intrusion process did not achieve steady-state conditions even after 10 years of simulation.

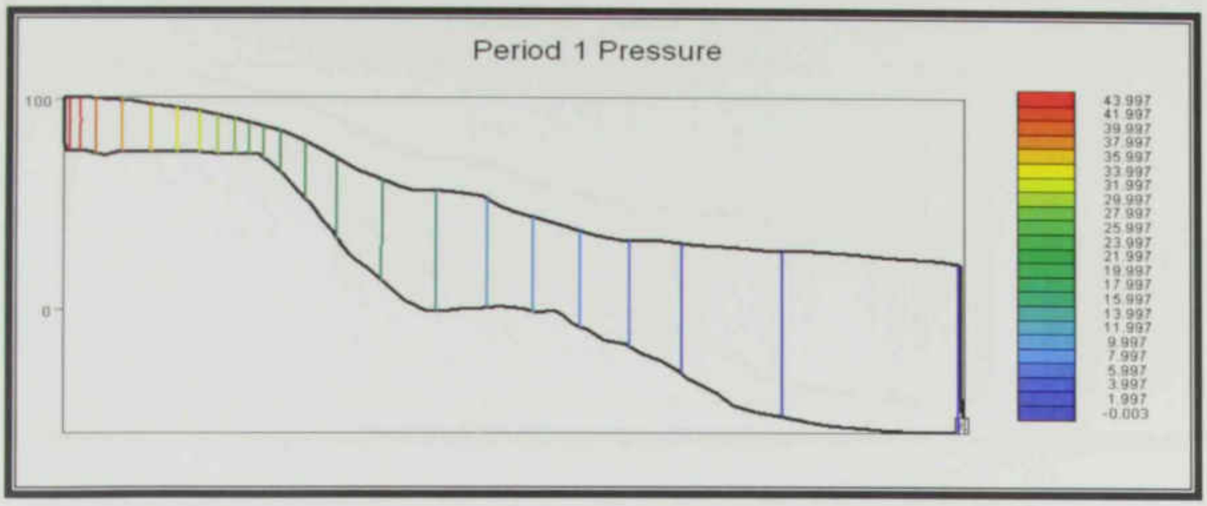

Figure 5.18. Equipotential lines after one month of simulation.

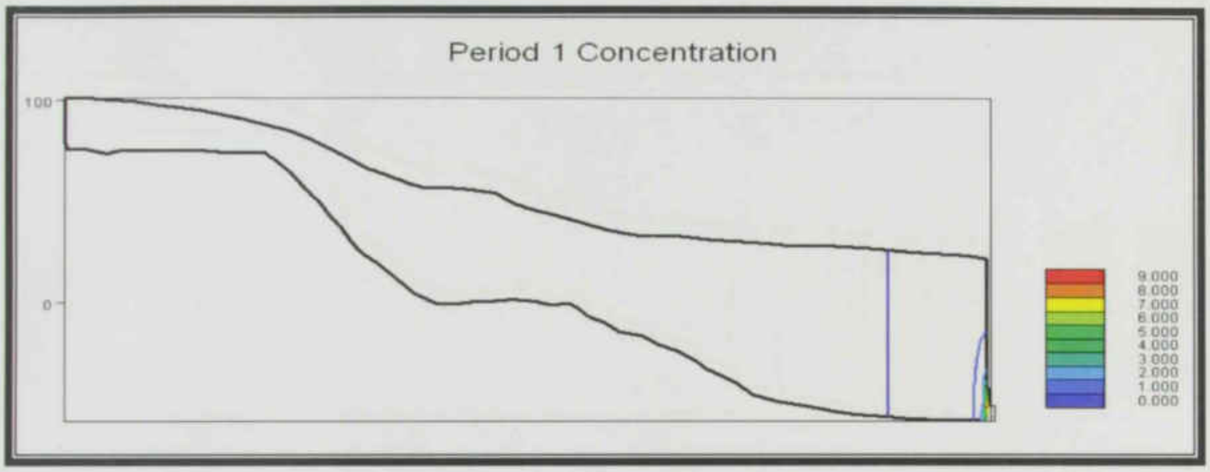

Figure 5.19. Equiconcentration lines after one month of simulation.

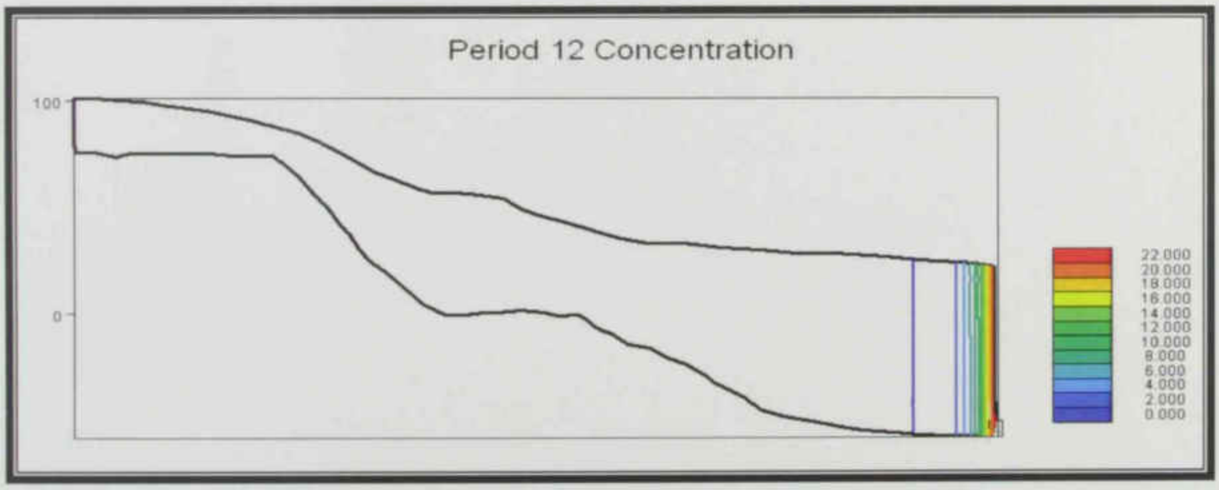

Figure 5.20. Equiconcentration lines after one year of simulation.

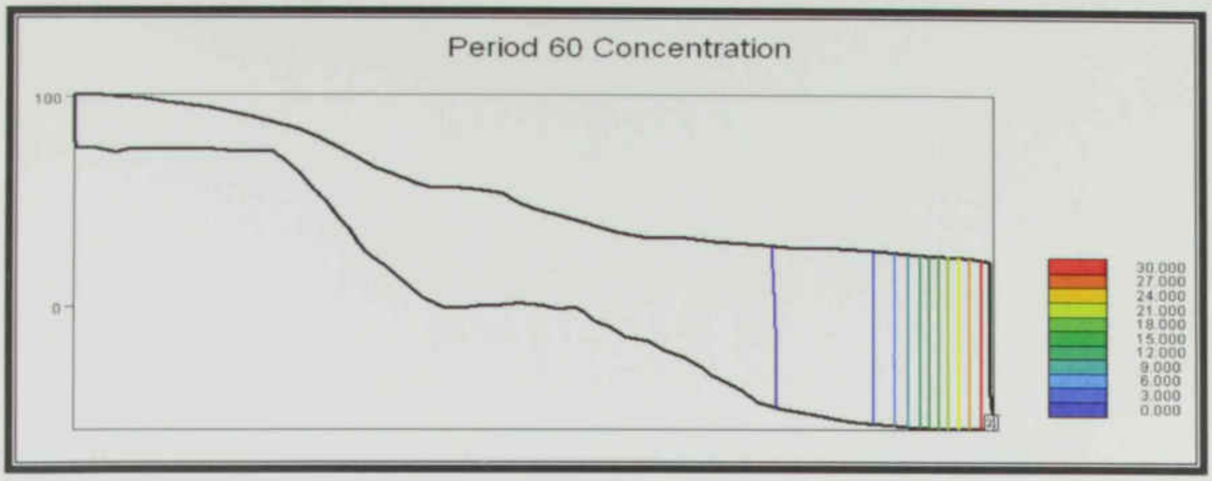

Figure 5.21. Equiconcentration lines after five years of simulation.

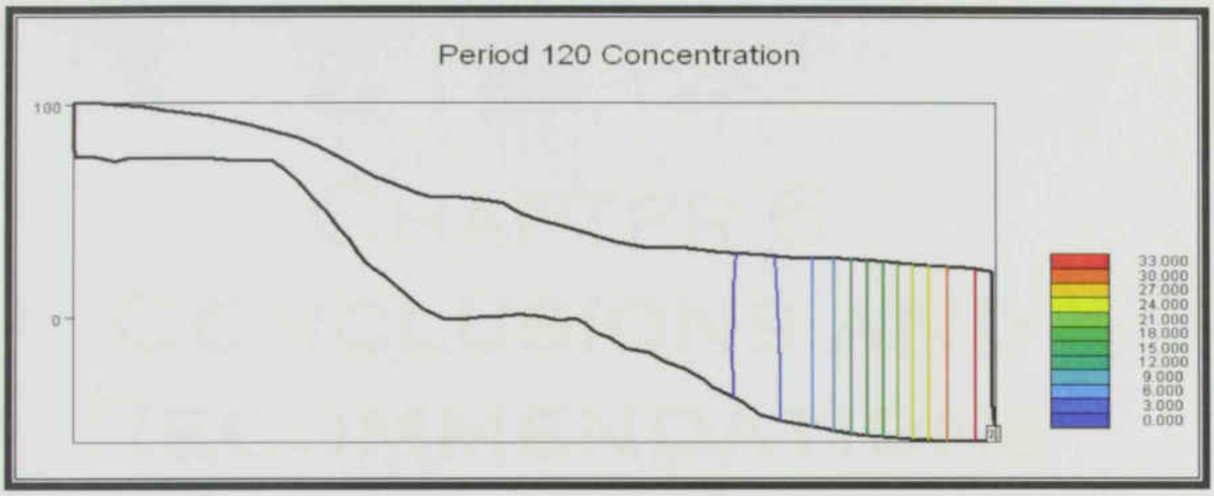

Figure 5.22. Equiconcentration lines after ten years of simulation.

CHAPTER 6
CONCLUSIONS AND RECOMMENDATIONS

Chapter 6. Summary, Conclusions and Recommendations

6.1 Summary

The lack of freshwater has already become commonplace around the globe. The growing population, rising standard of living, and expanding opportunities exert increasing demands for varied needs for water. These needs may be domestic, agricultural, industrial, touristic and others. By the middle of this century many parts of the world, including those where water is plentifully available now, will experience severe water shortages. The problem is more pronounced in arid and semi-arid regions where the lack of freshwater constitutes a major deterrent to their sustainable development. Despite the severe shortages, water continues to be misused, wasted and polluted. This requires, more than ever before, accurate assessment, proper development, improved management, efficient utilization, and increased conservation and protection of the available freshwater resources.

The UAE typifies an arid area and has very limited renewable freshwater resources. Currently, its main source of freshwater is through desalination which is quite expensive. Brackish groundwater is mainly used for agricultural development. Due to the scarcity and randomness of rainfall, surface water resources are quite limited and do not contribute significantly to the water budget not only in the UAE but also in other countries located in the Arabian Gulf Peninsula.

The growth of population in UAE coupled with an increase in human, agricultural, and industrial activities has imposed an increasing demand for freshwater. This increase in demand is often covered by extensive pumping of fresh groundwater, causing subsequent lowering of the water table or piezometric head and upsetting the dynamic balance between the freshwater body and the saline water body. The classical result of such a development is saltwater intrusion. A two to three percent mixing with seawater renders freshwater inadequate for human consumption. A five-percent mixing is enough to abandon the use of a freshwater aquifer.

When dealing with saltwater intrusion problems, two different approaches can be employed. The sharp-interface approach and the dispersion zone approach. The basic concept and even the governing equations are totally different in the two approaches. Under the sharp-interface approach the freshwater and seawater bodies are considered to be immiscible fluids like oil and water. The interface is also considered as an impermeable boundary and hence the water and salt ions are not allowed to cross this boundary. Under the dispersion-zone approach, the two water bodies mix and the density of the mixed fluid varies from that of the seawater near the sea boundary to that of the freshwater near the land boundary. The water flow is mainly under the hydraulic gradient and is thus governed by the Darcy equation, while the transport of the salt ions is mainly under the concentration gradient and is governed by advection, dispersion and adsorption processes.

Under the dispersion-zone approach, two different methods can be implemented; the constant-density and the variable-density methods. In the constant-density method, the density of the fluid in the dispersion zone is not affected by changes in fluid concentrations. In the variable-density method, the density is related to the fluid concentration using a constitutive equation. This study employed the variable-density approach.

The shape and degree of seawater intrusion in a coastal aquifer depend on many factors. These factors are, among others, the type of aquifer (confined, phreatic, leaky, or multi-layer) and its geology, water table and/or piezometric head, seawater concentration and density, natural rate of flow, capacity and duration of water withdrawal or recharge, rainfall intensities and frequencies, physical and geometric characteristics of the aquifer, land use, geometric and hydraulic boundaries, tidal effects, variations in barometric pressure, earth tides, earthquakes, and water wave actions. Some of these factors are natural and related to the hydraulic and geometric characteristics of the hydrogeological system, while others are artificial and related to human activities. The latter can be re-planed to reduce the seawater intrusion encroachment..

This thesis is devoted to the study of the seawater intrusion problem in Wadi Ham aquifer, Fujairah Emirate. The aquifer has been used wisely as a source of freshwater for drinking and irrigation purposes for several decades up to the year 1980. Extensive groundwater pumping was then initiated to meet increasing agricultural demands. Coincidentally, in 1996, the rainfall (main source of groundwater recharge) declined sharply. Groundwater levels dropped and as a result many wells were abandoned. Due to the direct hydraulic contact between the freshwater in the aquifer and saline water of the Gulf of Oman, the saline water encouraged the aquifer and the quality of the groundwater has deteriorated significantly.

SUTRA Argus-One modeling environment was employed to simulate seawater intrusion in the coastal aquifer of Wadi Ham. SUTRA is based on the finite element method and employs the dispersion zone (constant/variable density) approach. Based on the available data, the model was calibrated for the a period of 5 years and was then validated for another period of 11 years. Only groundwater levels were considered in this calibration/validation process as no records were available for the concentration of salts in the groundwater.

The model was then used to simulate the groundwater flow and seawater intrusion in the Wadi Ham aquifer in the horizontal (areal) and vertical (cross sectional) views. For the horizontal simulation both steady and unsteady (transient) conditions were considered and the effect of groundwater pumping, hydraulic conductivity, and dispersivities were considered. In the vertical section, the simulations were conducted under the unsteady-state conditions. Equipotential and equiconcentration lines were presented for the different cases. Recommendations are made to alleviate the seawater intrusion problem in the Wadi Ham.

6.2 Conclusions

Many countries, including United Arab Emirates, suffer from water shortage problems. Due to the lack of other water resources, aquifers have been over exploited to meet increasing demands. The dynamic balance between freshwater and seawater has been disturbed and the quality of groundwater deteriorated. The situation is exacerbated by recent decline in the rainfall. The severity of the problem varies from one aquifer to another depending on many factors. Excessive pumping, regarded as the main factor which accelerates the seawater intrusion process, requires urgent attention.

Based on the current study, the following conclusions are made.

1. The average annual rainfall in UAE has declined sharply during the last decade. The rainfall events are less frequent and drought conditions prevail. In most cases, rainfall intensity and duration are not sufficient to generate surface water runoff. This is fully applicable to the study area in Wadi Ham.

2. Due to the lack of recharge from rainfall and the excessive pumping of groundwater to meet the increasing agricultural demands in the area of Wadi Ham, groundwater water levels have declined significantly and the quality of the groundwater has deteriorated. Many wells have therefore been abandoned.

3. The decline of groundwater levels in the aquifer of Wadi Ham has disturbed the delicate balance between the freshwater in the aquifer and the saline water of the Gulf of Oman. The aquifer has been exposed to a severe seawater intrusion problem. This has caused further deterioration to the quality of the groundwater.

4. Previous geophysical investigations (Sherif et al., 2006) outlined the shape of the dispersion zone in different sections parallel to the shoreline. A fence diagram was also established to elaborate the change of the saline water zone with the distance from the shoreline.

5. Information and data about the hydrogeological conditions and groundwater levels in the Wadi Ham aquifer are available but not complete. Data on the salinity distribution are mostly unavailable.

6. SUTRA model has been successfully calibrated and validated to simulate historical records of groundwater levels in the aquifer of Wadi Ham. The model has not been calibrated to the salinity distribution due to lack of relevant data.

7. The model has been used to simulate the seawater intrusion in the areal view of the Wadi Ham aquifer under steady- and transient-flow conditions. For the vertical simulation, the model was employed under unsteady conditions. All the results were presented in contour maps of equipotential and equiconcentration lines.

8. The effects of pumping from the Khalba well field, longitudinal hydraulic conductivity and dispersivity on the seawater intrusion were investigated under steady-state conditions.

9. The amount of pumping from Khalba well field has the major effect on the overall intrusion migration. Reducing groundwater pumping will help to improve the groundwater quality.

10. Increasing the values of longitudinal hydraulic conductivity allowed more freshwater to travel from the ponding area of the dam Wadi Ham toward the seaside. Therefore, less seawater intrusion was encountered.

11. Reducing the lateral dispersivity from 10 to 1 m limited the impact of seawater intrusion to the area in the proximity of the Khalba field.

12. Vertical simulation scenarios indicated a steep slope of water table near the land side and a mild slope near the seaside. Equipotential lines and hence the flow field achieved the steady state conditions after one year of simulation while equiconcentration lines and hence the solute transport required much more time to achieve semi-steady state conditions.

13. In the cross-sectional simulation, the resulting equipotential lines were mostly vertical indicating that the flow velocities are mostly horizontal. On the other hand, the hydraulic gradient near the land side is relatively steep and near the shore boundary is relatively mild.

14. Due to lack of data and unknown levels of anisotropy and heterogeneity of the porous medium in the aquifer, the results should be considered in a qualitative manner rather than quantitative.

6.3 Recommendations

The following recommendations are made.

1. Groundwater pumping should be monitored, controlled and reduced in the coastal aquifers of UAE to prevent the possibility of further deterioration of the groundwater quality. Flow meters should be fixed to measure the groundwater pumping in all farms.

2. Contour maps for groundwater levels should be developed annually to assess the trends and revise the pumping policy. Long-term sustainability of the groundwater resources, with specific reference, to coastal aquifers, should be ensured.

3. Drilling of new pumping wells in coastal areas should be fully assessed and reviewed by pertinent authorities to ensure that these new wells will not accelerate the seawater intrusion process. Location and pumping rates should also be investigated to minimize any possible adverse impacts.

4. A comprehensive Geographical Information Database for the hydrological and hydrogeological systems of UAE, encompassing all related data should be developed and be accessible to researchers and professionals in the area of water resources development and management. This database will also support the decision makers.
5. A national program for public awareness regarding the importance of groundwater resources in UAE and the possible means for water conservation should be launched. This will ensure the long-term sustainability of the water resources in the country.

6. Artificial recharge of groundwater resources should be implemented at a larger scale. Treated wastewater and surface water that might be generated from rainfall events should be fully utilized to recharge the depleted aquifers. This will help to restore both the levels and quality of the groundwater and decelerate the seawater intrusion process.

References

Abdulrazzak, M. 1995. Water Supplies Versus Demand in Countries of Arabian Peninsula. Water Resources Planning and Management, 121,227-234.

Ackerer, P., Youn`es, A., Mos´e, R. 1999. Modeling Variable Density Flow and Solute Transport in Porous Medium: I. Numerical Model and Verification. Transport Porous Med. 35, 345–373.

Ahlfeld, D.P., Heidari, M. 1994. Applications of Optimal Hydraulic Control to Ground-water Systems. Water Resources Planning and Management. 120 (3), 350–375.

Al-Hajj, Y. A. M. 1995. Integrated Groundwater Resources Management in Qatar, Sixth Regional Meeting of the IHP National Committees of the Arab Region, Amman, Jordan.

Al-Muzaini, S.M. and Ismail, A.A. 1994, Irrigation Water and the National Plan for Greening Kuwait, WSTA 2nd Gulf Water Conference: Towards an Integrated Water Resources Management, State of Bahrain (in Arabic).

Al-Rashed, M.F and Sherif, M.M. 2000. Water Resources in the GCC Countries: an overview. Water Resources Management, 14:59-75

Al-Saati, A.J. 1995. Domestic Wastewater Reuse in the GCC, Attaawun Newspaper 10, 33-46.

Al-Shahi F. 2002. Assessment of Groundwater Resources in Selected Areas of Al Ain in the UAE. Master Thesis. United Arab Emirates University.

Al-Sharhan, A. S. and A. E. Nairn. 1995. Tertiary of the Arabian Gulf: Sedimentology and Hydrocarbon Potential, Palaeogeography, Palaeoclimatology, Palaeoecology, 114, pp. 369-380.

Al-Zubari, W.K. 1997. Towards the Establishment of a Total Water Cycle Management and Reuse Program in the GCC Countries, Proc. 3rd Gulf Water Conference, Muscat, Sultanate of Oman.

Andersen, P.F., Mercer, J.W., and White, H.O. 1988. Numerical Modeling of Saltwater Intrusion at Hallandale, Florida". Ground Water, 26(5), 619-630.

Anderson, M.P., and Woesner, W.W. 1992. Applied Groundwater Modeling -Simulation of Flow and Advective Transport. Academic Press, INC. UK.

Appelo C.A.J. and Postma D. 1993. Geochemistry Groundwater and Pollution. A.A Balkema, P.O.Box 1675, 3000 Br Rotterdam, Netherlands, ISBN 90 5410 1067

Appelo C. A. J. and Willemsen A. 1987. Geochemical Calculations and Observations on Saltwater Intrusion, I. A combined Geochemical/mixing cell model. Journal of Hydrology, 94, pp. 313-330.

Appelo C.A.J., Willemsen A., Beekman H.E. and Griffioen J. 1990. Geochemical Calculations and Observations on Saltwater Intrusion. II. Validation of a Geochemical Model with Laboratory Experiments. Journal of Hydrology, 120.

Arunakumaren, N.J., McMahon, G.A., Bajracharya, K. 2000.Water Management in the Lower Burdek: Groundwater Model Conceptualization. Dept. of Natural Resources, Brisbane.

Assouline, S., Shavit, U. 2004. Effects of Management Policies, including Artificial Recharge, on Salinization in a Sloping aquifer: The Israeli Coastal Aquifer case. Water Resources. 40, W04101, doi: 10.1029/2003WR002290

Ataie-Ashtiani, B., Volker, R.E., Lockington, D.A., 1999a. Numerical and Experimental Study of Seepage in Unconfined Aquifers with a Periodic Boundary Condition. Journal of Hydrology. 222, 165–184.

Ataie-Ashtiani, B., Volker, R.E., Lockington, D.A. 1999b. Tidal Effects on Sea-water Intrusion in Unconfined Aquifers. Journal of Hydrology 216 (1e2), 17-31.

Ataie-Ashtiani, B., Volker, R.E., Lockington, D.A. 2001. Tidal Effects on Groundwater Dynamics in Unconfined Aquifers. Hydrological Processes 15 (4), 655-669.

Ataie-Ashtiani, B., Volker, R.E., Lockington, D.A., 2002. Contaminant Transport in the Aquifers Influenced by Tide. Australian Civil Engineering Transactions, Institution of Engineers, Australia, CE43, 1-11.

Ayers, J.F., and Vacher, H.L. 1983. A Numerical Model Describing Unsteady Flow in a Freshwater Lens. Water Resources. Bull., 19(5), 785-792.

Bachu, S. 1995. Flow of Variable Density Formation Water in Deep Sloping Aquifers: Review of Methods of Representation with Case Studies. Journal of Hydrology. 164, 19–38.

Bachu, S., Karsten, M. 2002. Flow of Variable-Density Formation Water in Deep Sloping Aquifers: Minimizing the Error in Representation and Analysis when Using Hydraulic-Head Distributions. Journal of Hydrology. 259, 49–65.

Badon-Ghyben, W., "Nota in verband met de vooregenomen putboring nabij Amsterdam (Notes on the probable results of well drilling near Amsterdam)," Tijdschrift van het Koninklijk Instituut van Ingenierus, The Hague, 1888/9, 8-22, 1888.

Baghdady, A.R., Abu-Zeid, M.M. 2002. Petrography, Facies and Framework of Sedimentation of the Upper Cretaceous–Lower Tertiary sediments in Jabal Malaqet, west of the Northern Oman Mountains, United Arab Emirates. Middle East Research Center. Earth Science Series, vol. 16. Ain Shams University, pp. 1–15.

Baird AJ, Horn DP. 1996. Monitoring and Modeling Groundwater Behavior in Sandy Beaches. Journal of Coastal Research 12(3):630–640.

Barker Andrew P., Newton Robert J. and Botrell Simon H. 1998. Processes Affecting Groundwater Chemistry in the Zone of Saline Intrusion into an Urban Sandstone Aquifer. Applied Geochemistry. 13(6): 735-749.

Barlow, P.M. 2003. Groundwater in Freshwater-Saltwater Environments of the Atlantic Coast. U.S. Geological Survey Circular. 1262. 113.

Bear, J. 1979. Hydraulics of Groundwater. McGraw-Hill, Inc., New York, N.Y.

Bear, J. and Dagan, G. 1964a. Some Exact Solutions of Interface Problems by Means of the Hodograph method. Journal of Geophys. Res., 69, 1563-1572.

Bear, J. and Verruijt, A.1987. Modeling Groundwater Flow and Pollution, D. Reidel Publ. Co. Netherlands. 408.

Bear, J., Cheng, A.H.-D., Sorek, S., Ouazar, D. and Herrera, I. (Eds.). 1999. Seawater Intrusion in Coastal Aquifers - Concepts, Methods and Practices, Kluwer Academic Publishers. The Netherlands. 625.

Bear, J., Cheng, A.H.-D., Sorek, S., Ouazar, D., Herrera, I. 1999. *Seawater Intrusion in Coastal Aquifers e Concepts, Methods and Practices*. Kluwer Academic Publishers.

Ben Homouda M. F., Tarhouni J., Zouari K., and Leduce C. 2008. Geochemical and Isotopic Study of the Origin of the Stalinization in an Unconfined Coastal aquifer of Cap Bon (Tunisia). 20th Salt Water Intrusion Meeting. June 23-27, 2008. Naples, Florida, USA.

Benkabbour, B., Toto, E., Fakir, Y. 2004. Using DC Resistivity Method to Characterize the Geometry and the Salinity of the Plioquaternary Consolidated Coastal Aquifer of the Mamora plain, Morocco. Environ. Geol. 45, 518–526, doi:10.1007/s00254–003–0906–y.

Benntt, G.D., Mundraft, and A,S. Hussain. 1969. Electric Analog Studies of Brain Coning beneath Freshwater Wells in Punjab Region, Pakistan. USGS Wat. Supp. Paper 1608-J, J1-J31.

Bentley H.W., Phillips F.M. and Davis S.N. 1986a. Chlorine-36 in the terrestrial Environment. In: P. Fritz and J.C. Fontes, Editors, Handbook of Environmental Isotope Geochemistry vol. 2-B.

Bentley H.W., Phillips F.M. and Davis S.N. 1986b. Habermehl, P.L. Airey, G.E. Calf, D. Elmore, H.E. Gove and T. Torgersen, Chlorine-36 dating of very old groundwater 1—the great Artesian basin, Australia. Water Resources. Res. 22 (13): 1991–2001.

Bird J.R., Calf G.E., Davie R.F., Fifield L.K., Ophel T.R., Evans W.R. ,. Kellett J.R and. Habermehl M.A. 1989. The role of ^{36}Cl and ^{14}C measurements in Australian groundwater studies. Radiocarbon. 31 (3): 877–883.

Bruggman, G.A. 1990. The Use of pressure Generators in Solving Three Dimensional Salt/Fresh Groundwater Problem. Proc. Of SWIM selected papers.

Brunke and Schhelkes. 1999. H.P. Brunke and K. Schhelkes, A study on salt-water intrusion in coastal areas of Western Asia: model development and applications. Natuurwet. Tijdschr. Proc. SWIM 15, Ghent. 79: 241–248.

Bush, P.W., 1988. Simulation of Saltwater Movement in the Floridan Aquifer System, Hilton Head Island, South Carolina, U.S. Geological Survey. Water Supply Paper 2331.

Carlson C.A, Phillips F.M., Elmore D. and Bentley H.W .1990.Chlorine-36 tracing of salinity sources in the Dry Valleys of Victoria Land, Antarctica. Geochim. Cosmochim. Acta 54:311–318.

CDWR- California dept. Water Resources. 1958. Seawater Intrusion in California: Sacramento, California Dept. Water Res. Bull. 63.

Cheng, A.H.-D., Halhal, D., Naji, A., Ouazar, D. 2000. Pumping Optimization in Saltwater in Saltwater-Intruded Aquifers. Water Resources. Res. 36 (8), 2155–2165.

Choudhury, K., D. K. Saha, P. Chakraborty. 2001, Geophysical study for saline water intrusion in a coastal alluvial terrain. Applied Geophysics. 46: 189-200

Church, T.M., (1996). An Underground Route for the Water Cycle, Nature, 380:579-580.

Commander D.P. , Fifield L.K. , Thorpe P.M, Davie R.F. , Bird J.R. and Turner J.V. 1994. Chlorine-36 and Carbon-14 Measurements on Hypersaline Groundwater in Tertiary Palaeochannels Near Kalgoorlie, Western Australia. Report—Geological Survey of Western Australia. vol. 37

Contractor, D.N. 1983. Numerical Modeling of Saltwater Intrusion in the Northern Guam Lens. Water Resources. Bull., 19(5), 745-751.

Cooper, H.H. Jr. 1959. A Hypothesis Concerning the Dynamic Balance of Fresh and Saltwater in a Costal Aquifer. Geophysical. Research, Vol. 64 No.4

Custodio, E. 1987. Salt- Freshwater Interrelationship Under Natural Condition (Ch. 3) Effect of Human Activities on Salt Freshwater Relationships in Coastal Aquifer (Ch.4) In Groundwater Problems in Coastal Areas. UNESCO-IHP. 14-1123.

Custodio, E. 1986. Ground Problem in Coastal Areas. Studies and Report in Hydrology (UNESCO).

Custodio, E. 1988. Present State of Coastal Aquifer Modeling: short review. In: Custodio, E., Gurgui, A., Lobo Ferreira, J.P. (Eds.), Groundwater Flow and Quality Modeling. NATO ASI Series, vol. 224. Reidel, Dordrecht.785-810.

Custodio, E., Bruggemen, G.A., Cotecchia, V. 1987. Groundwater Problems in Coastal Areas, Studies and Reports in Hydrology, vol. 35. UNESCO, Paris.

Dam, J.C. van. 1993. Impact of Sea level Rise on Saltwater Intrusion in Estuaries and Aquifers, Keynote lecture delivered at the International Work Shop. Tidal Waters Division (Presently RIKZ) of the Directorate General fro Public Works and Water management. The Beterlands.

Dagan G. and Bear J. 1968. Solving the Problem of Local Interface Upconing in a Coastal aquifer by the Method of Small Perturbation . Journal of Hydrology Res. 29: 198-203.

Davis, S.N., Wittemore, D.O. and Febryka-Martin, J. 1998. Use of Chloride/bromide Ratios in Studies of Potable Water. Ground water, 338-350.

Dawoud M.A. 2005. The Role of Desalination in Augmentation of Water Supply in GCC Countries. Desalination 186: 187-198

Dazy J., Drogue C., Charmandis Ph., Darlet Ch. 1997. The Influence of Marine Inflows on the Chemical Composition of the Groundwater in Small Island the examples of the Cyclades (Greece). Nvironmental Geology 31 (3/4):33-141

Diersch, H.-J. G., Kolditz, O., 1998. Coupled Groundwater Flow and Transport: 2. Thermohaline and 3D convection systems. Adv. Water Resources 21 (5)401–425.

Diersch, H-J., D. Prochnow, and M. Thiele. 1984. Finite Element Analysis of Dispersion Affected Saltwater below A Pumping Well. App. Math. Modeling8.

Domenico, P.A. and Schwartz F.W. 1990.*Physical and Chemical Hydrogeology*, John Wiley & Sons.

Dominick TF, Wilkins B, Roberts H. 1971. Mathematical Model for Beach Groundwater Fluctuations. Water Resources Research 7:1626–1635.

Eeman S., Leijse A., and Van der Zee S. E. A. T. M., 2008. Dispersive Behavior of the Mixing Zone between a Shallow freshwater Lens and Upward Seeping Saline Groundwater. 20 th Salt Water Intrusion Meeting. Naples, Florida, USA.

El Nashar A.M. 2004.The role of desalination in water resources management in Gulf Region, IV Iberian Congress on Management and Planning of the Water Science, Technique and Citizenship: Keys for a Sustainable Management of the Water, Spain.

Electrowatt Engineering Services Ltd, 1981. Wadi Ham Dam II and Groundwater Recharge Facilities. Design **1** (1981) 67.

Elschami F. 1990. Hydrochemical Classification of the Groundwater of the Wadi Dibba in the Northeastern part of the United Arab Emirates. J. Fac. Sci., UAE Univ. **2** 1 (1990), 7–29.

Emch, P.G., and Yeh, W.W.G. 1998. Management model for conjunctive use of coastal surface water and groundwater". J. Water Resources Planning and Management, 124(3), 129-139.

Entec Europe limited (ENTEC).1996. Survey on Ground Water Recharge and Flow in Wadi Ham. Volume 11, Ministry of Agriculture and Fisheries, UAE.

Entec, Europe limited (ENTEC).1995.Survey on Groundwater Recharge and Flow in wadi Ham: Ham and Wadi Wurrayah, Vol.1- Wadi Ham: Ministry of agriculture and Fisheries. UAE.

Environment & Agriculture information center – UAE. 007. Soil and water. Online available:[http://www.fedfin.gov.ae/uaeagricent/directory.asp].

Essaid, H.I. 1986. A Comparison of the Coupled Freshwater-Saltwater Flow and the Ghyben-Herzbergh Sharp Interface Approaches to Modeling of Transient Behavior in Coastal Aquifer Systems. Journal of Hydrology, 86, 169-193.

Essaid, H.I. 1990a. The Computer Model SHARP, a Quasi-Three Dimensional Finite-Difference Model to Simulate Freshwater and Saltwater Flow in Layered Coastal Aquifer System: Model Development and Applications". Rep. No. 90- 4130, U.S. Geological Survey Water Resources. Investigation.

Essaid, H.I. (1990b). "A multilayered sharp interface model of coupled freshwater and saltwater flow in coastal systems: Model development and applications". Water Resources. Res., 26(7):1431-1454.

Essink, G.H.P.O. 1996. Impact of Sea level Rise on Groundwater Flow Regimes, a Sensitive Analysis for the Netherlands. PhD Dissertation Thesis, volume 7, Delft studies in Integrated Water Management, Delft University of Technology. The Netherlands.

Essink, G.H.P.O. 1998. Simulating Density Dependent Groundwater Flow: the Adapted MOC3D. In: Proc. 15th Salt Water Intrusion Meeting. Ghent, Belgium, 69–79.

Essink, G.H.P.O. 2001. Improving Fresh Groundwater Supply— Problems and Solutions. Ocean Coastal Manage. 44, 429–449.

Essink, G.H.P.O., 2003.In: Mathematical Models and Their Application to Salt Water Intrusion Problems, TIAC' 03, Coastal Aquifers Intrusion Technology, Mediterranean Countries, Alicante, Spain. 57–77.

Fang CS, Wang SN, Harrison W. 1972. Groundwater Flow in a Sandy Tidal Beach: Two Dimensional Finite Element Analysis. Water Resources Research 8: 121–128.

Fetter, C.W. 1994. Applied Hydrogeology, 3rd edn, Prentice Hall

Finney, B.A., Samsuhadi and Willis, R. 1992. Quasi-Three Dimensional Optimization Model of Jakarta Basin". J. Water Resour. Plng. and Mgmt., ASCE, 118(1), 18-31.

Flores-M´arquez, E., Campos-Enr´ıquez, J., Ch´avez-Segura, R., Castro-Garc´ıa, J. 1998. Saltwater intrusion of the Costa de Hermosillo aquifer, Sonora, Mexico: A numerical simulation. Geof´ısica Internacional 37 (3), 131–151.

Forsythe, G. E. and Wasow, W. R. 1960. Finite Difference Method for Partial Differential Equation, Wiley, New York.

Fox, L., 1969. Numerical solution of Ordinary and Partial Differential Equation. Pergamon Press, New York.

Freeze, R.A., Cherry, J.A. 1979. Groundwater. Prentice Hall, New York.

Frenkel H. and J. D. Rhoades 1978 "effects of dispersion and swelling on soil hydraulic conductivity", American Society for testing and materials.

Frind, E.O. 1982. Seawater Intrusion in Continuous Coastal Aquifer-Aquitard Systems. Advances Water Resources 5, 89-97.

Gambolati, G., Putti, M., Paniconi, C. 1999. Three-Dimensional Model of Coupled Density-Dependent Flow and Miscible Salt Transport in Groundwater. In: Bear, J. and Cheng, A.H.-D. et al. (Ed.), Seawater Intrusion in Coastal Aquifers - Concepts, Methods and Practices-. Kluwer Academic, Dordrecht, The Netherlands, Ch. 10, pp. 315–545 5–362.

Ghassemi, F., Chen, T.H., Jakeman, A.J., Jacobson, G. 1993. Two and Three-Dimensional Simulation of Seawater Intrusion: Performances of the "SUTRA" and "HST3D" models. AGSO J. Aust. Geol. Geophys. 14 (2–3), 219–226.

Ghassemi, F., Jakeman, A.J., Jacobson, G. 1990. Mathematical Modeling of Seawater Intrusion, Nauru Island. Hydrol. Process. 4, 269–281.

Gingerich Stephen B. 2008. Management of the Iao and Waihee Aquifer Areas with the Aid of 3-D Numerical SUTRA Model, Maui Hawaii. 20 th Salt Water Intrusion Meeting. June 23-27, 2008. Naples, Florida, USA

Gingerich, S., Voss, C. 2002. Three-Dimensional Variable-Density Flow Simulation of a Coastal Aquifer in Southern Oahu, Hawaii, USA. In: Boekelman, R. e. a. (Ed.), Proceedings SWIM17 Delft 2002. Delft University of Technology, Delft.93–103.

Gleick, P.H., (ed.), Water in Crisis- A Guide to the World's Fresh Water Resources, Oxford Univ. Press, 1993

Global Water Intelligence (GWI). Publishing, London, October 2000.

Glover, R.E. 1959. The Pattern of Fresh-water Flow in a Coastal Aquifer". J. Geophys. Res., 64(4), 457-459.

Goldenberg L.C. 1985.Decrease of Hydraulic Conductivity in Sand at the Interface Between Seawater and Dilute Clay Suspension. Journal of hydrology, 78 . 183-199.

Gomis V., Boluda N. and Ruiz F. 1996. Application of a Mode for Simulation Transport of Reactive Multispecies Component to the Study of the Hydrochemistry of Salt water Intrusion. Journal of Contaminant Hydrology.22, 67-81.

Gomis V., Boluda N. and Ruiz F. 1997. Column Displacement Experiments to Validate Hydrochemical Models of Seawater Intrusions, Journal of Contaminant Hydrology 29, 81-91.

Gordon, E.E., Shamir, U., Bensabat, J. 2000. Optimal Management of a Regional Aquifer under Salinisation Conditions. Water Resources. Res. 36 (11), 3193–3203.

Gorelick, S.M., Voss, C.I., Gill, P.E., Murray, W., Saunders, M.A., and Wright, M.H. 1984. Aquifer Reclamation Design: the Use of Contaminant Transport Simulation Combined with Nonlinear programming. Water Resources. Res., 20(4), 415-427.

Gotovac, H., R. Andrecevic, and M. Vranjes. 2001. Effects of Aquifer heterogeneity on the intrusion of seawater, Proceeding: first international Conference on saltwater Intrusion and Coastal Aquifers-Monitoring, Modeling and management, Essaouira, Morocco.

Grant US. 1948. Influence of the water table on beach aggradations and degradation. Journal of Marine Research 7: 655–660.

Harremoes Poul, Anders Malmgern –Hasen" Vandforurening ", polyteknisk forlag, 1 udagave, 1 oplag 1990.

Hem, J.D. 1985. Study and Interpretation of the Chemical Characteristics of Natural Water, U.S. Geol. Survey Water-Supply Paper 2254.

Henry, H.R. 1959. Salt Intrusion into Fresh-water Aquifers". J. Geophys. Res., 64(11)1911-1919.

Henry, H.R. 1964. Effect of Dispersion on Saltwater Encroachment in Costal Aquifer. USGS, Paper 1613-C,C70-C84.

Hillier, J.R. 1993. Management of Saltwater Intrusion in Coastal Aquifers in Queensland, Australia. AGSO J. Aust. Geol. Geophys. 14 (2–3), 213–217.

Hiroshiro, Y., Jinno, K., Berndtsson, R. 2006. Hydrochemical properties of a salinity affected coastal aquifer in western Japan. Hydrol. Process. 20 (6), 1425–1435.

Housein I. and Jalludin J. 1996. The Salinity of Djibouti's Aquifer. Journal of African Earth science. 22(4): 409-414.

Howari F.M. , A. Baghdady , P.C. Goodell. 2006. Mineralogical and Gemorphological Characterization of Sand dunes in the Eastern Part of United Arab Emirates Using Orbital Remote Sensing Integrated with Field Investigations. Geomorphology 83: 67–81.

Huyakorn, P.S., Anderson, P.F., Mercer, J.W. and White, H.O. 1987. Salt Intrusion in Aquifer: Development and Testing of a Three Dimensional Finite Element Model, Water Resources. Res., 23, 293-319.

IWACO.1986. Ground Water Study Drilling of Deep Water at Various Locations in UAE.

Izbicki, J.A. 1996. Seawater Intrusion in a Coastal California Aquifer.U.S. Geol. Survey Fact Sheet 125-96.

Jacob CE. 1950. Flow of Groundwater. In Engineering Hydraulics, Rouse H (ed.). Wiley: New York; 321–386.

Jenkins, S. R. and R. K. Moore. 1984. Effect of Saltwater Intrusion on Soil Erodibility of Alabama Marshlands. Water Resources Research Institute, Auburn University, Auburn.

Johannsen, K., Kinzelbach, W., Oswald, S., Wittum, G. 2002. The Saltpool Benchmark Problem - Numerical Simulation of Saltwater Upconing in a Porous Medium. Adv. Water Resources. 25 (3), 335–348.

Jones and Marrei. 1982. G.P. Jones and S.H. Marrei, Groundwater resources in the United Arab Emirates. Middle East Water Sci. 6 (1):41-51

Jones, B. F., Vengosh, A., Rosenthal, E., and Yechieli, Y. 1999. Geochemical investigations. In: Bear et al. (eds). Seawater Intrusion in Costal Aquifer- Concepts, Methods and Practices. 51-72.

Kacimov, A.R., Sherif, M.M., Perret, J.S., and Al-Mushikhi. 2008. Seawater Intrusion in an Unconfined Coastal Aquifer on a Catchment Scale: Modeling and Field Studies. accepted, Hydrogeology Journal.

Kantorovich, L.V. and Krylov, V. I. 1964. Approximate Methods of Higher Analysis, translated by C.D. Benster, Interscience Publishers. New York.

Kempers, M., and H. Haas. 1994. The Dispersion Zone between Fluids with Different Density and Viscosity in A Heterogenous Porous Medium. J. Fluid Mech. Vol. 267.

Khouri, J. and Deroubi, A. 1990. Water Resources in the Arab World, UNESCO/ACSAD, Damascus. 166.

Kim, G., Lee, K.-K., Park, K.-S., Hwang, D.-W. and Yang, H.-S. 2003. Large submarine groundwater discharge (SGD) from a volcanic island, Geophys. Res. Lett., 30(21): 2098, doi:10.1029/2003GL018378.

Kipp, K. J., 1986. HST3D. A Computer Code for Simulation of Heat and Solute Transport in Three-Dimensional Groundwater Flow Systems. Water-Resources Investigations 86–4095, IGWMC, International Ground Water Modeling Center. USGS.

Knight JH. 1981. Steady Periodic Flow through a Rectangular Dam. Water Resources Research 17(4): 1222–1224.

Kohout, F. A. 1985. The Flow of Freshwater and Saltwater in the Biscayne Aquifer of the Miami Area, Florida. Sea water in Coastal Aquifers. U.S. Geological Survey, Water Supply Paper No. 1613-C.

Kohout F. A. 1965. A Hypotheses Concerning Cyclic Flow of Saltwater related in Biscayne Aquifer of the Miami Area, Floridian Aquifer. Trans. New York Acas. Sci. II, 28(2):240-271.

Kolditz, O., Habbar, A., Kaiser, R., Schulze-Ruhfus, M., Thorenz, C. 1998. ROCKFLOW - Theory and Users Manual. Release 3.2. Institut fu¨rStr¨omungsmechanik und Elektronisches Rechnen im Bauwesen, Universit¨at Hannover.

Konikow, L.F., and Bredehoeft, J.D. 1978. Computer Model of Two Dimensional Solute Transport and Dispersion in Groundwater. U.S.G.S. Techniques of Water Resources Investigations, Book 7, Chapter C2, p. 90.

Konikow, L.F., Arevalo, J.R., 1993. Advection and Diffusion in a Variable-Salinity Confining layer. Water Resources Research 29 (8), 2747-2761.

Korsbech, Kristian L. 1991. To-Dimensional Modeling of Salt and Sindtrængning. Master of Science Thesis, ISVA, DTU.

Kumar A. Narayan, Carsten Schleeberger , Keith L. 2007. Bristow Modeling Seawater Intrusion in the Burdekin Delta Irrigation Area, North Queensland, Australia. Agriculture Water Management 8 9, 2 1 7 – 2 2 8

Lahm, T., Bair, E., VanderKwaak, J. 1998. The Role of Salinity-572 Derived Variable-Density Fluid Flow in the Displacement of Brine From Shallow, Regionally Extensive Aquifers. Water Resources. Res. 34 (6), 1469–1480.

Larabi A., Faouzi M., and Cheng A. H. D. 2008. Assessment of Groundwater Resources in Rmel Coastal Aquifer (Morocco) by SEAWAT. 20 th Salt Water Intrusion Meeting. June 23-27,. Naples, Florida, USA.

Lebbe, L. & Essink, G. 1999. Section 12.11. MOC DENSITY/ MOCDENS3D-code. P. 434-439, in Chapter 12. Survey of Computer Codes and Case Histories, Eds. Sorek, S. & Pinder, G. F. in: Seawater Intrusion in Coastal Aquifer, Concepts, Methods and Practices. Eds. Bear, J., H-D, Herrera, I., Sorek, S. and Ouazar D. Kluwers Academic Publishers.

Ledoux, E., Sauvagnac, S., Rivera, A. 1990. A Compatible Single Phase/ Two phase Numerical Model: 1. Modeling the Transient Salt-water/fresh-Water Interface Motion. Ground Water 28 (1), 215-223.

Lebbe, L. Jonckheere S., and Vandenbohede A. 2008. Modeling of Historical Evolution of Salt-Water Distribution in the Phreatic Aquifer in and Around the Silted up Zwin Estuary Mouth (Flanders, Belgium). 20 th Salt Water Intrusion Meeting. Naples, Florida, USA.

Lehmann B.E., Loosli H.H and Purtschert R. 1995. A comparison of Chloride and Helium Concentrations in Deep Groundwater Proceeding: Isotopes in Water Resources Management, Vienna, 20–24. 3–17.

Lewis, J. B. 1987. Measurements of Groundwater Seepage Flux onto a Coral reef: Spatial and Temporal Variations, Limnol. Oceanogr., 32:1165-1169.

Li L, Barry DA, Pattiaratchi CB. 1997. Numerical Modelling of tidal-Induced Beach Water Table Fluctuations. Coastal Engineering 30: 105–123.

Louis, H. M. 1992. Saltwater Upconing in an Aquifer Overlain by a Leaky Confining Bed. Groundwater Vol.30, No. 2.

Lyons W.B., Welch K.A. and Sharma P. 1998.Chlorine-36 in the Waters of the McMurdo Dry Valley lakes, southern Victoria Land, Antarctica: Revisited. Geochim. Cosmochim. Acta 62 2. 185–191.

Malkovsky, V., Pek, A., Tsang, C., 2002. Dynamic Stabilization of Heat Generating liquid Waste Plume in a Sloping Aquifer. Journal of Hydrology. 258, 69–82.

Mantoglou, A., 2003. Pumping Management of Coastal Aquifers Using Analytical Models of Saltwater Intrusion. Water Resources. Res. 39 (12).

Mantoglou, A., Papantoniou, M., Giannoulopoulos, P. 2004. Management of Coastal Aquifers Based on non-linear Optimization and Evolutionary Algorithms. J. Hydrol. 297, 209–228. agr i c u ltura l water management 8 9 (2 0 0 7) 2 1 7– 2 2 8 227

Manzano, M., Custodio, E. and Jones, B.F. 1990 "Progress in the Understanding of Groundwater Flow Through the Aquitard of the Llobregat Delta, Barcelona, Spain", In: Livro de homenagem a Carlos Romariz. A.C.A. de Matos (prefacer), univ. lisboa. Secc. Geol. Econ. Apl., Lisbon, Pprtugal, 115-126.

Masciopinto, C. 2006. Simulation of Coastal Groundwater Remediation: the Case of Nardo' Fractured Aquifer in Southern Italy. Environmental Modelling& Software 21 (1), 85-97.

Mather, J. R. 1984. Water Resources, Distribution, use, and Management. John Wiley & Sons, Inc. and V.H. Winstzons & Sons, New York, Silver Spring Maryland, 439.

Matthess, G. (1982). The Properties of Groundwater. John Wiley, New York, 406.

Mayer, A.S., Kelley, T., Miller, C.T. 2002. Optimal Design for Problems Involving Flow and Transport Phenomena in Saturated Subsurface Systems. Advance Water Resources. 25, 1233–1256.

Mazor, 1992. E. Mazor, Reinterpretation of 36Cl data: Physical Processes, Hydraulic Interconnections and Age Estimates in Groundwater Systems. Appl. Geochem. 7. 351–360.

Mazor, 1997. E. Mazor. Chemical and Isotopic Groundwater Hydrology: the applied approach (2nd ed.), Marcel Dekker, Inc, New York. 255–269.

Meinardi, C. R. 1976. Characteristic Example of Natural Groundwater Composition in the Netherland. Rijkistitut Voor drink water Voor ziening, Modeling.

Mercer, J.W., Larson, S.P., Faust, C.R. 1980. Simulation of Saltwater Interface motion. Ground Water 18 (4), 374-385.

Milnes, E., Renard, P. 2004. The Problem of Salt recycling and Seawater Intrusion in Coastal Irrigated Plains: an example from the Kiti aquifer (Southern Cyprus). Journal of Hydrology. 288 (3–4), 327–343.

Ministry of Agriculture and Fisheries report, 1995. Soil and water Department, UAE.

Ministry of Agriculture and Fisheries. 1986. Ministry of Agriculture and Fisheries. Drilling of Deep Water Wells at Various Locations in UAE Groundwater Development in the Eastern Agriculture Region vol. 8, Ministry of Agriculture and Fisheries, Dubai, UAE. 107.

Ministry of communication.1996. UAE Climate. Cultural Foundation. Prepared by Ministry of Communication. First edition.

Ministry of Environment and water. 2007. Dams in United Arab Emirates, Online available: [http://www.uae.gov.ae/uaeagricent/wateranddam/dams_en.stm].

Mohamed A.M.O., Maraqa M., Al Handhaly J. 2005, Impact of land Disposal of Reject Brine from Desalination Plants on Soil and Groundwater. Desalination 182: 411-433

Moore, W. S. 1996. Large Groundwater Inputs to Coastal Waters Revealed by 226Ra Enrichments, Nature, 380:612-614.

Morell, I., Medina, J., Pulido, A. And Fenandez-Rubio, R. 1996. The Use of Bromide and Strontium as Indicator of Marine Intrusion in the aquifer of Oropesa-Torreblanca. Proc. 14th Salt Water Intrusion Meeting. Malmo. Sweden. 629-640.

Murphy, S.F., Sorensen, R.C. 2001. Saltwater Intrusion in the Mackay Coastal Plains Aquifer. Proc. Aust. Sugar Cane Technol. 23, 70–76.

Muscat, M. and R.D. Wyckoff. 1935. An a[approximate Theory of Water Coning in Oil Production. Trans. Amer. Inst. Min Metall. Pet. Eng.114.

Nicholson, S. E, 1995. Dry Land Climatology Oxford University Press, Oxford.

Nielsen P. 1990. Tidal Dynamics of the Water Table in Beaches. Water Resources Research 26(9): 2127–2134.

Nielsen P. 1999. Groundwater Dynamics and Salinity in Coastal Barriers. Journal of Coastal Research,V15 (3):732-740.

Oswald, S., Kinzelbach, W. 2004. Three-Dimensional Physical Benchmark Experiments to Test Variable-Density Flow Models. Journal of Hydrology 290 (1–2), 22– 42.

Padilla F. Benavente J., Cruz/Sanjulian J. 1997. Numerical Simulation of the Influence of Management Alternatives of a Projected Reservoir on a Small Alluvial Aquifer Affected by Seawater Intrusion (Almunecar, Spain), Envioronmental Geology 33 (1), pp. 72-80.

Paniconi, C., Khlaifi, I., Lecca, G., Giacomelli, A., Tarhouni, J. 2001. A modeling Study of Seawater Intrusion in the Korba Coastal Plain, Tunisia. Phys. Chem. Earth (B) 26 (4), 345–351.

Parlange JY, Stagnitti F, Starr JL, Braddock RD. 1984. Free Surface Flow in Porous Media and Periodic Solution of the Shallow Flow Approximation. Journal of Hydrology 70: 251–263.

Philip JR. 1973. Periodic non-linear Diffusion: an Integral Relation and its Physical Consequences. Australian Journal of Physics 26: 513–519.

Phillips F.M., 2000. Chlorine-36. In: P. Cook and A.L. Herczeg, Editors, Environmental Tracers in Subsurface Hydrology. 299–337.

Pinder, G.F., and H.H. Cooper. 1970. A Numerical Techniques Position of the Seawater Front. Water Ressources. Res.,3 .875-881

Polo, J.F., Ramis, F.J.R., 1983. Simulation of Salt Water-Fresh Water Interface Motion. Water Resources Research 19 (1), 61-68.

Price, M. 1998. Introduction to Groundwater, 2nd edn, Chapman &Hall.

Raskin, P., E. Hansen, and R. Margdis. 1995. Water and Sustainability: A Global Outlook. Stockholm Environment Institute, Stockholm.

Rectenwald E., and Bennett M., 2008. Utilizing Stable Isotopes (^{2}H, ^{18}O) to Better Identify Different Water Types of the Floridan Aquifer System in Southwest Florida. 20 th Salt Water Intrusion Meeting. June 23-27. Naples, Florida, USA.

Reev, C. 1988. A comparison of Different Methods for Representing the Hydrodynamic Trnsor in Numerical Models. Proc. 10th SWIM, Ghent.

Lewis, J. B. 1987. Measurements of Groundwater Seepage Flux onto a Coral reef: Spatial and Temporal Variations, Limnol. Oceanogr., 32:1165-1169.

Li L, Barry DA, Pattiaratchi CB. 1997. Numerical Modelling of tidal-Induced Beach Water Table Fluctuations. Coastal Engineering 30: 105–123.

Louis, H. M. 1992. Saltwater Upconing in an Aquifer Overlain by a Leaky Confining Bed. Groundwater Vol.30, No. 2.

Lyons W.B., Welch K.A. and Sharma P. 1998.Chlorine-36 in the Waters of the McMurdo Dry Valley lakes, southern Victoria Land, Antarctica: Revisited. Geochim. Cosmochim. Acta 62 2. 185–191.

Malkovsky, V., Pek, A., Tsang, C., 2002. Dynamic Stabilization of Heat Generating liquid Waste Plume in a Sloping Aquifer. Journal of Hydrology. 258, 69–82.

Mantoglou, A., 2003. Pumping Management of Coastal Aquifers Using Analytical Models of Saltwater Intrusion. Water Resources. Res. 39 (12).

Mantoglou, A., Papantoniou, M., Giannoulopoulos, P. 2004. Management of Coastal Aquifers Based on non-linear Optimization and Evolutionary Algorithms. J. Hydrol. 297, 209–228. agr i c u ltura l water management 8 9 (2 0 0 7) 2 1 7– 2 2 8 227

Manzano, M., Custodio, E. and Jones, B.F. 1990 "Progress in the Understanding of Groundwater Flow Through the Aquitard of the Llobregat Delta, Barcelona, Spain", In: Livro de homenagem a Carlos Romariz. A.C.A. de Matos (prefacer), univ. lisboa. Secc. Geol. Econ. Apl., Lisbon, Pprtugal, 115-126.

Masciopinto, C. 2006. Simulation of Coastal Groundwater Remediation: the Case of Nardo' Fractured Aquifer in Southern Italy. Environmental Modelling& Software 21 (1), 85-97.

Mather, J. R. 1984. Water Resources, Distribution, use, and Management. John Wiley & Sons, Inc. and V.H. Winstzons & Sons, New York, Silver Spring Maryland, 439.

Matthess, G. (1982). The Properties of Groundwater. John Wiley, New York, 406.

Mayer, A.S., Kelley, T., Miller, C.T. 2002. Optimal Design for Problems Involving Flow and Transport Phenomena in Saturated Subsurface Systems. Advance Water Resources. 25, 1233–1256.

Mazor, 1992. E. Mazor, Reinterpretation of 36Cl data: Physical Processes, Hydraulic Interconnections and Age Estimates in Groundwater Systems. Appl. Geochem. 7. 351–360.

Mazor, 1997. E. Mazor. Chemical and Isotopic Groundwater Hydrology: the applied approach (2nd ed.), Marcel Dekker, Inc, New York. 255–269.

Meinardi, C. R. 1976. Characteristic Example of Natural Groundwater Composition in the Netherland. Rijkistitut Voor drink water Voor ziening, Modeling.

Mercer, J.W., Larson, S.P., Faust, C.R. 1980. Simulation of Saltwater Interface motion. Ground Water 18 (4), 374-385.

Milnes, E., Renard, P. 2004. The Problem of Salt recycling and Seawater Intrusion in Coastal Irrigated Plains: an example from the Kiti aquifer (Southern Cyprus). Journal of Hydrology. 288 (3–4), 327–343.

Ministry of Agriculture and Fisheries report, 1995. Soil and water Department, UAE.

Ministry of Agriculture and Fisheries. 1986. Ministry of Agriculture and Fisheries. Drilling of Deep Water Wells at Various Locations in UAE Groundwater Development in the Eastern Agriculture Region vol. 8, Ministry of Agriculture and Fisheries, Dubai, UAE. 107.

Ministry of communication.1996. UAE Climate. Cultural Foundation. Prepared by Ministry of Communication. First edition.

Lewis, J. B. 1987. Measurements of Groundwater Seepage Flux onto a Coral reef: Spatial and Temporal Variations, Limnol. Oceanogr., 32:1165-1169.

Li L, Barry DA, Pattiaratchi CB. 1997. Numerical Modelling of tidal-Induced Beach Water Table Fluctuations. Coastal Engineering 30: 105–123.

Louis, H. M. 1992. Saltwater Upconing in an Aquifer Overlain by a Leaky Confining Bed. Groundwater Vol.30, No. 2.

Lyons W.B., Welch K.A. and Sharma P. 1998.Chlorine-36 in the Waters of the McMurdo Dry Valley lakes, southern Victoria Land, Antarctica: Revisited. Geochim. Cosmochim. Acta 62 2. 185–191.

Malkovsky, V., Pek, A., Tsang, C., 2002. Dynamic Stabilization of Heat Generating liquid Waste Plume in a Sloping Aquifer. Journal of Hydrology. 258, 69–82.

Mantoglou, A., 2003. Pumping Management of Coastal Aquifers Using Analytical Models of Saltwater Intrusion. Water Resources. Res. 39 (12).

Mantoglou, A., Papantoniou, M., Giannoulopoulos, P. 2004. Management of Coastal Aquifers Based on non-linear Optimization and Evolutionary Algorithms. J. Hydrol. 297, 209–228. agr i c u ltura l water management 8 9 (2 0 0 7) 2 1 7– 2 2 8 227

Manzano, M., Custodio, E. and Jones, B.F. 1990 "Progress in the Understanding of Groundwater Flow Through the Aquitard of the Llobregat Delta, Barcelona, Spain", In: Livro de homenagem a Carlos Romariz. A.C.A. de Matos (prefacer), univ. lisboa. Secc. Geol. Econ. Apl., Lisbon, Pprtugal, 115-126.

Masciopinto, C. 2006. Simulation of Coastal Groundwater Remediation: the Case of Nardo' Fractured Aquifer in Southern Italy. Environmental Modelling& Software 21 (1), 85-97.

Mather, J. R. 1984. Water Resources, Distribution, use, and Management. John Wiley & Sons, Inc. and V.H. Winstzons & Sons, New York, Silver Spring Maryland, 439.

Matthess, G. (1982). The Properties of Groundwater. John Wiley, New York, 406.

Mayer, A.S., Kelley, T., Miller, C.T. 2002. Optimal Design for Problems Involving Flow and Transport Phenomena in Saturated Subsurface Systems. Advance Water Resources. 25, 1233–1256.

Mazor, 1992. E. Mazor, Reinterpretation of 36Cl data: Physical Processes, Hydraulic Interconnections and Age Estimates in Groundwater Systems. Appl. Geochem. 7. 351–360.

Mazor, 1997. E. Mazor. Chemical and Isotopic Groundwater Hydrology: the applied approach (2nd ed.), Marcel Dekker, Inc, New York. 255–269.

Meinardi, C. R. 1976. Characteristic Example of Natural Groundwater Composition in the Netherland. Rijkistitut Voor drink water Voor ziening, Modeling.

Mercer, J.W., Larson, S.P., Faust, C.R. 1980. Simulation of Saltwater Interface motion. Ground Water 18 (4), 374-385.

Milnes, E., Renard, P. 2004. The Problem of Salt recycling and Seawater Intrusion in Coastal Irrigated Plains: an example from the Kiti aquifer (Southern Cyprus). Journal of Hydrology. 288 (3–4), 327–343.

Ministry of Agriculture and Fisheries report, 1995. Soil and water Department, UAE.

Ministry of Agriculture and Fisheries. 1986. Ministry of Agriculture and Fisheries. Drilling of Deep Water Wells at Various Locations in UAE Groundwater Development in the Eastern Agriculture Region vol. 8, Ministry of Agriculture and Fisheries, Dubai, UAE. 107.

Ministry of communication.1996. UAE Climate. Cultural Foundation. Prepared by Ministry of Communication. First edition.

Ministry of Environment and water. 2007. Dams in United Arab Emirates, Online available: [http://www.uae.gov.ae/uaeagricent/wateranddam/dams_en.stm].

Mohamed A.M.O., Maraqa M., Al Handhaly J. 2005, Impact of land Disposal of Reject Brine from Desalination Plants on Soil and Groundwater. Desalination 182: 411-433

Moore, W. S. 1996. Large Groundwater Inputs to Coastal Waters Revealed by 226Ra Enrichments, Nature, 380:612-614.

Morell, I., Medina, J., Pulido, A. And Fenandez-Rubio, R. 1996. The Use of Bromide and Strontium as Indicator of Marine Intrusion in the aquifer of Oropesa-Torreblanca. Proc. 14th Salt Water Intrusion Meeting. Malmo. Sweden. 629-640.

Murphy, S.F., Sorensen, R.C. 2001. Saltwater Intrusion in the Mackay Coastal Plains Aquifer. Proc. Aust. Sugar Cane Technol. 23, 70–76.

Muscat, M. and R.D. Wyckoff. 1935. An a[approximate Theory of Water Coning in Oil Production. Trans. Amer. Inst. Min Metall. Pet. Eng.114.

Nicholson, S. E, 1995. Dry Land Climatology Oxford University Press, Oxford.

Nielsen P. 1990. Tidal Dynamics of the Water Table in Beaches. Water Resources Research 26(9): 2127–2134.

Nielsen P. 1999. Groundwater Dynamics and Salinity in Coastal Barriers. Journal of Coastal Research,V15 (3):732-740.

Oswald, S., Kinzelbach, W. 2004. Three-Dimensional Physical Benchmark Experiments to Test Variable-Density Flow Models. Journal of Hydrology 290 (1–2), 22– 42.

Padilla F. Benavente J., Cruz/Sanjulian J. 1997. Numerical Simulation of the Influence of Management Alternatives of a Projected Reservoir on a Small Alluvial Aquifer Affected by Seawater Intrusion (Almunecar, Spain), Envioronmental Geology 33 (1), pp. 72-80.

Paniconi, C., Khlaifi, I., Lecca, G., Giacomelli, A., Tarhouni, J. 2001. A modeling Study of Seawater Intrusion in the Korba Coastal Plain, Tunisia. Phys. Chem. Earth (B) 26 (4), 345–351.

Parlange JY, Stagnitti F, Starr JL, Braddock RD. 1984. Free Surface Flow in Porous Media and Periodic Solution of the Shallow Flow Approximation. Journal of Hydrology 70: 251–263.

Philip JR. 1973. Periodic non-linear Diffusion: an Integral Relation and its Physical Consequences. Australian Journal of Physics 26: 513–519.

Phillips F.M., 2000. Chlorine-36. In: P. Cook and A.L. Herczeg, Editors, Environmental Tracers in Subsurface Hydrology. 299–337.

Pinder, G.F., and H.H. Cooper. 1970. A Numerical Techniques Position of the Seawater Front. Water Ressources. Res.,3 .875-881

Polo, J.F., Ramis, F.J.R., 1983. Simulation of Salt Water-Fresh Water Interface Motion. Water Resources Research 19 (1), 61-68.

Price, M. 1998. Introduction to Groundwater, 2nd edn, Chapman &Hall.

Raskin, P., E. Hansen, and R. Margdis. 1995. Water and Sustainability: A Global Outlook. Stockholm Environment Institute, Stockholm.

Rectenwald E., and Bennett M., 2008. Utilizing Stable Isotopes (^{2}H, ^{18}O) to Better Identify Different Water Types of the Floridan Aquifer System in Southwest Florida. 20th Salt Water Intrusion Meeting. June 23-27. Naples, Florida, USA.

Reev, C. 1988. A comparison of Different Methods for Representing the Hydrodynamic Trnsor in Numerical Models. Proc. 10th SWIM, Ghent.

Reilly, T. E. And Goodman, A.S., "Quantitative Analysis of Saltwater- Freshwater Relationships in Ground-Water Systems- a historical perspective", J. Hydrol., 80.125-160.1985.

Reilly, T.E., Goodman, A.S., 1987. Analysis of Saltwater Upcoming Beneath a Pumping Well. J. Hydrol. 89, 169–204.

Rivera, A., and E. Ledox. (1988). Modelling of a Coastal Aquifer in Maxico. Proc. Of SWIM 10.

Rizk Z.S and H.A. El-Etr. 1997. Hydrogeology and Hydrogeochemistry of Some Springs in the United Arab Emirates. Symposium on Desert Studies in the Kingdom of Saudi Arabia, Extant and Implementation, 27–29 Rabi-II, 1415 H., 2–4 October, 1994. Arabian J. Sci. Engng 22 1C .

Rizk Z.S., Alsharhan A.S. and Shindo S.S. 1997. Evaluation of Groundwater Resources of United Arab Emirates Proceedings of the Third Gulf Water Conference, Muscat, Sultanate of Oman vol. 1 95–122.

Rizk, Z. S., Hassan K. Garamoon and Hassan A. El-Etr, 1995, Measurement of the Hydraulic Properties of Dune and Interdune Areas Around Al-Ain City, United Arab Emirate: The International Symposium on the Quaternary Desert and Climate Changes, UAE University, 16.

Rizk, Z. S. 1999. Water Resources in United Arab Emirates, A Review Article, Menoufla University, Egypt

Rogers, P., and P. Lydon (eds.). 1995. Water in the Arab World: Perspectives and Prognosis. Harvard University Press, Cambridge, Mass.

Rubin, H., and G. Pinder. 1977. Approximate Analysis of Up Coning. Adv. Wat. Resour. 1(2)

Rushton, K. R. 1979. Estimation Transmissivity and Storage Coefficient for Abstraction Well Data. Groundwater Journal 16.

Rushton, K.R., and S.C. Redshaw, 1979, Seepage and Groundwater Flow, John Wiley and Sons, NY., 339.

Sadeg, S.A., Karahanoglu, N., 2001. Numerical Assessment of Seawater Intrusion in the Tripoli Region, Libya. Environ. Geol. 40 (9), 1151–1168.

Sandberg, P., T. Lang and A. Andersen. 1998. Recurrent studies of Chloride Ingress in Uncracked Marine Concrete at Various Exposure Times and Elevation, Cement and Concrete Research, Vol. 28,(10): 1489-1503, 1998, Pergamon.

Sauter, F., Leijnse, A., Beusen, A. 1993. METROPOL. User's Guide. Tech. Rep. 725205.003, National Institute of Public Health and Environmental Protection, Bilthoven, the Netherlands.

Scheller, H.,Geochemie des eaux souterraines1956. Application Aux Eaux Des Gisements De Petrole, Soc. Ed. technip, Paris, 213.

Schmorak, S., and A. Mercado. 1969. Upconing of Freshwater- Saltwater Interface below Pumping Wells, Field Study. Water Ressources Res. .5(6)

Sciabica, M. G., Paniconi, C., Barrocu, G. 1994. Three-Dimensional Model of Saltwater Intrusion in the Capoterra Coastal Aquifer System (Sardinia). In:Barrocu, G. (Ed.), Proceedings of the 13th Salt-Water Intrusion Meeting. Dipartimento di Ingegneria del Territorio, Universit`a di Cagliari, Cagliari,Italy, 77–86.

Sears, R. 1998. The British Nuclear Fuels Drigg low-level Waste Site Characterization Program. In: Robins, N.S. (ed) Groundwater pollution, Aquifer recharge and vulnerability. Geological Society. London Special Publication, 130, 37-46

Segol, G., and G. F. Pinder. 1976. Transient Simulation of Saltwater Intrusion in Southeastern Florida. Wat. Reasou. Res., 12, 65-70.

Segol, G., Pinder, G.F., Gray, W.G. 1975. A Galerkin-Finite Element Technique for Calculating the Transient Position of Saltwater Front. Water Resources Research 11 (2), 343-347.

Sethi, L.N., Panda, S.N., Nayak, M.K. 2006. Optimal Crop Planning and Water Resources Allocation in a Coastal Groundwater Basin, Orissa, India. Agric. Water Manage. 83, 221–232.

Shamir, U., and Dagan, G. 1971. "Motion of the Seawater Interface in Coastal Aquifers: a numerical solution ". Water Resources Res., 7(3), 644-657.

Shamir, U., Bear, J., and Gamliel, A. 1984. Optimal Annual Operation of a Coastal Aquifer. Water Resources Res., 20(4), 435-444.

Sherif M.M, El Mahmoudi A., Garamoon H., Kacimov A., Akram S., Ebraheem A.and Shetty A. 2006.Geoelectrical and Hydrogeochemical Studies for Delineating Seawater Intrusion in the Outlet of Wadi Ham, UAE", Environmental Geology, 49: 536 – 551.

Sherif M. Kacimov A. 2008. Pumping of Brackish and Saline Water in Coastal Aquifers; An Effective Tool Foe Alleviation of Seawater Intrusion. 20 th Salt Water Intrusion Meeting. June 23-27, 2008. Naples, Florida, USA.

Sherif M.M. and Kacimove A. 2006. Effect of Porosity on Unsteady Seawater Intrusion in Confined Coastal Aquifer", International Conference on Challenges in Coastal Hydrology and Water Quality, Baton Rouge, Louisiana.

Sherif, M.M. and Kacimove, A., 2006b "Modeling of Groundwater in two Selected Coastal Aquifers of UAE and Oman as a Precursor for Water Resources Management", Final Report, Research Sectors, UAEU/SQU, UAE.

Sherif, M.M. et al., 2005. "Assessment of the Effectiveness of Al Bih, Al Tawiyean and Ham Dams in Groundwater Recharge Using Numerical Models", Final Report Vol. 1: Main Report, Ministry of Agriculture and Fisheries, Dubai, UAE.

Sherif M.M., 1999. "The Nile Delta Aquifer in Egypt", Chapter 17 in "Seawater Intrusion in Coastal Aquifers: Concepts, Methods and Practices" Bear, J., Cheng, A., Sorek, S., Ouazar, D., and Herrera, A., (eds.)., Book Series: Theory and Application of Transport in Porous Media, Vol. 14, pp. 559-590, Kluwer Academic Publishers, The Netherlands,

Shiklomanov, I. A., and J. C. Rodda. 2003. World Water Resources at the Beginning of the Twenty-First Century. Cambridge University Press, Cambridge, 450.

Smiles DE, Stokes AN. 1976. Periodic Solutions of a Nonlinear Diffusion Equation Used in Groundwater Flow Theory: examination using a Hele–Shaw model. Journal of Hydraulics 31: 27–35.

Sommariva C. and Syambabu V.S.N. 2001. Increase in Water Production in UAE. Desalination 138: 173-179

Southwell, R.V. 1940. Relaxation Methods in Engineering Science, Oxford University Press. London.

Souza, W.R., and Voss, C.I. 1987. Analysis of Anisotropic Coastal Aquifer System Using Variable-Density Flow and Solute Transport Simulation. Journal of Hydrology, 92, 17-41.

Stakebeek, A. 1988. Experiment on the Movement of Brackish Water near Infiltration and Abstraction Wells. Proc. 10 th SWIM, Ghent.

Strack, O., 1995. A Dupuit-Forcheimer Model for three-Dimensional Flow With Variable Density. Water Resources. Res. 31 (12), 3007–3017.

Strarinsky, A., Bielsky, M., Lazar, B., Steinitz, G. and Raab, M. 1983. Strontium Isotope Evidence on the History of Oilfield Brines, Mediterranean Coastal plain, Israel, Geochim. Cosmochim. Acta, 47, 687-695,

Stuyfzand Pieter j. 1995. :The Impact of land Reclaimation on Groundwater Quality and Future Drinking water supply in the Netherlands", Wat. Sci. Tech. Vol.31, No 8, pp. 47-59, 1995.

Taigbenu, A.E., Liggett, J.A., Cheng, A.H.-D., 1984. Boundary integral solution to Seawater Intrusion into Coastal Aquifers. Water Resources Research 20 (8), 1150-1158.

Reilly, T. E. And Goodman, A.S., "Quantitative Analysis of Saltwater- Freshwater Relationships in Ground-Water Systems- a historical perspective", J. Hydrol., 80.125-160.1985.

Reilly, T.E., Goodman, A.S., 1987. Analysis of Saltwater Upcoming Beneath a Pumping Well. J. Hydrol. 89, 169–204.

Rivera, A., and E. Ledox. (1988). Modelling of a Coastal Aquifer in Maxico. Proc. Of SWIM 10.

Rizk Z.S and H.A. El-Etr. 1997. Hydrogeology and Hydrogeochemistry of Some Springs in the United Arab Emirates. Symposium on Desert Studies in the Kingdom of Saudi Arabia, Extant and Implementation, 27–29 Rabi-II, 1415 H., 2–4 October, 1994. Arabian J. Sci. Engng 22 1C .

Rizk Z.S., Alsharhan A.S. and Shindo S.S. 1997. Evaluation of Groundwater Resources of United Arab Emirates Proceedings of the Third Gulf Water Conference, Muscat, Sultanate of Oman vol. 1 95–122.

Rizk, Z. S., Hassan K. Garamoon and Hassan A. El-Etr, 1995, Measurement of the Hydraulic Properties of Dune and Interdune Areas Around Al-Ain City, United Arab Emirate: The International Symposium on the Quaternary Desert and Climate Changes, UAE University, 16.

Rizk, Z. S. 1999. Water Resources in United Arab Emirates, A Review Article, Menoufla University, Egypt

Rogers, P., and P. Lydon (eds.). 1995. Water in the Arab World: Perspectives and Prognosis. Harvard University Press, Cambridge, Mass.

Rubin, H., and G. Pinder. 1977. Approximate Analysis of Up Coning. Adv. Wat. Resour. 1(2)

Rushton, K. R. 1979. Estimation Transmissivity and Storage Coefficient for Abstraction Well Data. Groundwater Journal 16.

Rushton, K.R., and S.C. Redshaw, 1979, Seepage and Groundwater Flow, John Wiley and Sons, NY., 339.

Sadeg, S.A., Karahanoglu, N., 2001. Numerical Assessment of Seawater Intrusion in the Tripoli Region, Libya. Environ. Geol. 40 (9), 1151–1168.

Sandberg, P., T. Lang and A. Andersen. 1998. Recurrent studies of Chloride Ingress in Uncracked Marine Concrete at Various Exposure Times and Elevation, Cement and Concrete Research, Vol. 28,(10): 1489-1503, 1998, Pergamon.

Sauter, F., Leijnse, A., Beusen, A. 1993. METROPOL. User's Guide. Tech. Rep. 725205.003, National Institute of Public Health and Environmental Protection, Bilthoven, the Netherlands.

Scheller, H.,Geochemie des eaux souterraines1956. Application Aux Eaux Des Gisements De Petrole, Soc. Ed. technip, Paris, 213.

Schmorak, S., and A. Mercado. 1969. Upconing of Freshwater- Saltwater Interface below Pumping Wells, Field Study. Water Ressources Res. .5(6)

Sciabica, M. G., Paniconi, C., Barrocu, G. 1994. Three-Dimensional Model of Saltwater Intrusion in the Capoterra Coastal Aquifer System (Sardinia). In:Barrocu, G. (Ed.), Proceedings of the 13th Salt-Water Intrusion Meeting. Dipartimento di Ingegneria del Territorio, Universit`a di Cagliari, Cagliari,Italy, 77–86.

Sears, R. 1998. The British Nuclear Fuels Drigg low-level Waste Site Characterization Program. In: Robins, N.S. (ed) Groundwater pollution, Aquifer recharge and vulnerability. Geological Society. London Special Publication, 130, 37-46

Segol, G., and G. F. Pinder. 1976. Transient Simulation of Saltwater Intrusion in Southeastern Florida. Wat. Reasou. Res., 12, 65-70.

Segol, G., Pinder, G.F., Gray, W.G. 1975. A Galerkin-Finite Element Technique for Calculating the Transient Position of Saltwater Front. Water Resources Research 11 (2), 343-347.

Sethi, L.N., Panda, S.N., Nayak, M.K. 2006. Optimal Crop Planning and Water Resources Allocation in a Coastal Groundwater Basin, Orissa, India. Agric. Water Manage. 83, 221–232.

Shamir, U., and Dagan, G. 1971. "Motion of the Seawater Interface in Coastal Aquifers: a numerical solution ". Water Resources Res., 7(3), 644-657.

Shamir, U., Bear, J., and Gamliel, A. 1984. Optimal Annual Operation of a Coastal Aquifer. Water Resources Res., 20(4), 435-444.

Sherif M.M, El Mahmoudi A., Garamoon H., Kacimov A., Akram S., Ebraheem A.and Shetty A. 2006.Geoelectrical and Hydrogeochemical Studies for Delineating Seawater Intrusion in the Outlet of Wadi Ham, UAE", Environmental Geology, 49: 536 – 551.

Sherif M. Kacimov A. 2008. Pumping of Brackish and Saline Water in Coastal Aquifers; An Effective Tool Foe Alleviation of Seawater Intrusion. 20 th Salt Water Intrusion Meeting. June 23-27, 2008. Naples, Florida, USA.

Sherif M.M. and Kacimove A. 2006. Effect of Porosity on Unsteady Seawater Intrusion in Confined Coastal Aquifer", International Conference on Challenges in Coastal Hydrology and Water Quality, Baton Rouge, Louisiana.

Sherif, M.M. and Kacimove, A., 2006b "Modeling of Groundwater in two Selected Coastal Aquifers of UAE and Oman as a Precursor for Water Resources Management", Final Report, Research Sectors, UAEU/SQU, UAE.

Sherif, M.M. et al., 2005. "Assessment of the Effectiveness of Al Bih, Al Tawiyean and Ham Dams in Groundwater Recharge Using Numerical Models", Final Report Vol. 1: Main Report, Ministry of Agriculture and Fisheries, Dubai, UAE.

Sherif M.M., 1999. "The Nile Delta Aquifer in Egypt", Chapter 17 in "Seawater Intrusion in Coastal Aquifers: Concepts, Methods and Practices" Bear, J., Cheng, A., Sorek, S., Ouazar, D., and Herrera, A., (eds.)., Book Series: Theory and Application of Transport in Porous Media, Vol. 14, pp. 559-590, Kluwer Academic Publishers, The Netherlands,

Shiklomanov, I. A., and J. C. Rodda. 2003. World Water Resources at the Beginning of the Twenty-First Century. Cambridge University Press, Cambridge, 450.

Smiles DE, Stokes AN. 1976. Periodic Solutions of a Nonlinear Diffusion Equation Used in Groundwater Flow Theory: examination using a Hele–Shaw model. Journal of Hydraulics 31: 27–35.

Sommariva C. and Syambabu V.S.N. 2001. Increase in Water Production in UAE. Desalination 138: 173-179

Southwell, R.V. 1940. Relaxation Methods in Engineering Science, Oxford University Press. London.

Souza, W.R., and Voss, C.I. 1987. Analysis of Anisotropic Coastal Aquifer System Using Variable-Density Flow and Solute Transport Simulation. Journal of Hydrology, 92, 17-41.

Stakebeek, A. 1988. Experiment on the Movement of Brackish Water near Infiltration and Abstraction Wells. Proc. 10 th SWIM, Ghent.

Strack, O., 1995. A Dupuit-Forcheimer Model for three-Dimensional Flow With Variable Density. Water Resources. Res. 31 (12), 3007–3017.

Strarinsky, A., Bielsky, M., Lazar, B., Steinitz, G. and Raab, M. 1983. Strontium Isotope Evidence on the History of Oilfield Brines, Mediterranean Coastal plain, Israel, Geochim. Cosmochim. Acta, 47, 687-695,

Stuyfzand Pieter j. 1995. :The Impact of land Reclaimation on Groundwater Quality and Future Drinking water supply in the Netherlands", Wat. Sci. Tech. Vol.31, No 8, pp. 47-59, 1995.

Taigbenu, A.E., Liggett, J.A., Cheng, A.H.-D., 1984. Boundary integral solution to Seawater Intrusion into Coastal Aquifers. Water Resources Research 20 (8), 1150-1158.

Terry Bengtsson. 2008. Assessing Well Field Impact on Water Quality in the Upper Floridian Aquifer in Southwest Florida. 20 th Salt Water Intrusion Meeting. June 23-27, 2008. Naples, Florida, USA.

Tompson, A. F. B. and W.G. Gray. 1986. A Second Order Approach for the Modeling of Dispersion in Porous Media. Wat. Res. 21.

Toth, J., 1970. A conceptual Model of The Groundwater Regime and the Hydrogeologic Environment. Journal of Hydrology, vol. 10, p. 164-176.

Turner IL, Coates BP, Acworth RI. 1996. The Effects of Tides and Waves on Water-Table Elevations in Coastal Zones. Hydrogeology Journal 4(2): 51–69.

Uffink, G.J.M. 1990. Analysis of Dispersion by the Random Walk Method. Ph.D. dissertation thesis, Delft University of Technology, The Netherlands.

Vengosh, A. and Pankratov, I. 1998. Chloride/ bromide and Chloride/ fluoride Ratios of Domestic Sewage Effuents and Associated Contaminated Ground Water", Ground Water.

Vengosh, A., Heumann, K.G., Juraske, S. and Kasher, R. 1994. Boron Isotope Application For Tracing Sources of Contamination in Groundwater. Environment. Sci. Tech., 28, 1968-1974,.

Volker, R., and Rushton, K. 1982. An Assesment of the Importance of Some Parameters for Seawater Intrusion in Aquifers and a Comparison of Dispersive and Sharp-interface Modeling Approaches. J. Hydrology, 56, 239-250.

Volker, R.E. 1980. Predicting the Movement of Seawater into a Coastal Aquifer. Tech. Pap. 51, Aust. Water Resour. Counc. Dept. of Natl. Devel. Energy, Aust. Gov. Publ. Serv. Canberra.

Voss, C. I., 1984. SUTRA-Saturated Unsaturated Transport. A Finite-Element Simulation Model for Saturated-Unsaturated, Fluid-Density-Dependent Ground-water Flow with Energy Transport or Chemically-Reactive Single-Species Solute Transport, U.S. Geol. Survey. Water-Resource. Invest. Rep. 84-4369, 409 p.

Voss, C. I. Provost, A., 2002. SUTRA, A model for saturated-unsaturated variable-density ground-water flow with solute or energy transport. Water-Resources Investigations 02–4231, U.S. Geological Survey. 6

Voss, C.I. Souza, W.R. 1987. Variable Density Flow and Solute Transport Simulation of Regional Aquifers Containing a Narrow Freshwater-saltwater Transition Zone. Water Resources Research 23 (10), 1851-1866.

Voss. C. I. and W.R. Souza. 1986. Modeling in a Regional Aquifer Containing a Narrow Transitions Zone between Freshwater and saltwater Using Solute Transport Simulation. Proc. Of SWIM9 of the Tertiary Aquifer. Water International, Volume 32, Number 2, Pg. 325-338.

Ward, D. 1991. Data Input for SWIFT/386, version 2.50. Technical report, 619

Werner A. D., Nation E., and Habermehl M. A. 2008. Seawater Intrusion in Australia- A National Perspective of Future Challenges. 20 th Salt Water Intrusion Meeting. June 23-27, 2008. Naples, Florida, USA.

Whitaker, F.F. and Smart, P.L., "Bacterially-Mediated Oxidation of Oxidation of Organic Matter: a Major Control on Groundwater Geochemistry and Porosity Generation in Oceanic Carbonate Terrains, Breakthroughs in Karst Geomicrobiology and Redox Geochemistry", Abstracts and Field – Trip Guide for the symposium held Feb. 16-19, 1994.

Willis, R., and Finney, B.A. 1988. Planning Model For Optimal Control of Saltwater Intrusion. Water Resources Planning and Management. ASCE, 110(3), 333- 347.

Wilson, J., and Sa da Costa, A. 1982. "Finite Element Simulation of a Saltwater/Freshwater Interface with Indirect Toe Tracking". Water Resources Res. 18(4), 1069-1080.

World Health Organization and UNICEF. 2000. Global Water Supply and Sanitation Assessment 2000 report.

Xue, Yuqun, Xie Chunhong, Wu Jichun, Liu Peimin, Wang Jianji and Jiand Qingbo. 1995. A three/dimensional miscible transport model for seawater intrusion in China . Water Resources Research, 31, (o 4) : 903-912, April.

Yager Richard M. and Misut Paul E. 2008 Simulation of Processes Controlling Migration of Saline Water and Brine Above a Flooded Salt Mine in Western New York, USA. 20 th Salt Water Intrusion Meeting. June 23-27, 2008. Naples, Florida, USA.

Yang Jing- Yea. 2008. A Case Study of Finite- Element Numerical Modeling on Salt Water Intrusion for the Ping-Tung Plain. 20 th Salt Water Intrusion Meeting. June 23-27, 2008. Naples, Florida, USA

Yechiel Y., Ronen D. and Kaufman. 1996. The source and Age of Groundwater Brines in the Dead Sea area, as deduced from 36Cl and 14C. Geochim. Cosmochim. Acta 60 (11): 1909–1916.

Yechiel Y. and Sivan O. 2008. Chemical and Isotopic Evidence for Seawater Intrusion- Examples from the Coastal Aquifer of the Mediterranean and the Dead Sea. 20 th Salt Water Intrusion Meeting. June 23-27, 2008. Naples, Florida, USA.

Zekster, I.S. and Everett, L.G. (Eds.), 2004. Groundwater Resources of the World and their Use, IHP-VI, Series on Groundwater No. 6. UNESCO (United Nations Educational, Scientific and Cultural Organization), 342.

Zhang, Q., Volker, R.E., Lockington, D.A., 2004. Numerical Investigation of Seawater Intrusion at Gooburrum, Bundaberg, Queensland, Australia. Hydrogeol. J. 12 (6), 674–687.

Zhou, X., Chen, M., Ju, X., Wang, J., 2000. Numerical Simulation of Sea water Intrusion near Beihai, China. Environ. Geol. 40 (1–2), 223–233.

معتدل قرْيب من جهة شاطيء البحر. وبالتالي خطوطْ equipotential تحت ظروف الحالة الثابتة بعد عام واحد مِنْ المحاكاةِ بينما تطلبت خطوط equiconcentration وقتَ أكثر بكثير للوصول الحالة شبه الثابتة.

الهبوط المستمر لمستويات المياه الجوفيَّةِ في الطبقة الجوفيةِ مِنْ وادي حام خلال العقدان الماضيةَ سبب خلل في التوازن الدقيقَ بين الماء العذبِ في الطبقة الجوفيةِ والماءِ الملحي لخليج عُمان. إنّ الطبقة الجوفية حالياً تتْعرّضُ إلى مشكلة حادّةِ بسبب تداخل ماءِ البحر. وبسبب قلةِ بَعْض البياناتِ المهمةِ، مثل dispersivity والسجلات المتوفرةَ عنْ مستويات الملوحةِ في المناطق قيْد النظر، ومستويات تباين الخواص والتغاير في مسامية الوسط في طبقة المياه الجوفية ، ولذلك فإن النتائج تعتبر نوعية.

خلصت الإطروحة الى مجموعةَ من التوصياتِ لتخفيف مشكلةِ تداخل ماءَ البحر في الطبقة الجوفيةِ الساحليةِ لوادي حام والتي تُساعدُ على استعادة والحفاظ على مصادر المياه الجوفيَّةَ. علاوةَ على التغذية الصناعية للمياه الجوفيَّةِ، وذلك باستخدام المياه المبددة المُعَالجة والماءَ السطحيَ المتاحَ الذيْ يمكن ان يتولد من أحداثِ الامطار والسيول، والتي يتوَجبُ تنفيذها على نطاق اوسع. ان ضخ المياه الجوفية يجب أن يخضع للاشراف والخفض والسيطرةَ من قبل السلطات المعنيةَ. ويتعين بناء قاعدة بيانات شاملة تشمل جميع المعلومات والبيانات ذات الصلة لموارد المياه الجوفية في البلاد. والنماذج الرقمية وينبغي أن تستخدم لدعم عملية صنع القرار لضمان الإدارة السليمة لموارد المياه الجوفية المحدودة.

ملخص الرسالة

تعنى إدارة موارد المياه الجوفية في الطبقات الجوفية الساحلية للامارات العربية المتحدة بتوفير الاحتياجات المائية لضمان النمو المستمر في المناطق الساحليةِ. هذه الطبقات الجوفيةِ يَجِبُ أنْ تُدْرَسَ بعناية لتقييم مستوى تدخل ماءِ البحرِ الحالي وتوقعُ أيّ تدخل آخر في المستقبل تحت ممارساتِ الضخ الحاليةِ. ويَجِبُ الأخذ باعتبار الزيادة المستقبليةِ لاستغلال المياه الجوفيّة، وذلك للزيادةَ في نشاطاتِ الزراعة والسكانَ. ان الضخ المستمر طويل المدى منْ هذه الطبقات الجوفيةِ يَجِبُ أنْ يحددَ بدقةَ ويجب تحديد المكان الامثل لأيّ ضخ إضافي.

تشير السجلات خلال العقدين الماضيين بأنّ مستويات المياه الجوفيّة تقريباً في كُلّ الطبقات الجوفية الساحلية للإمارات العربيةِ المتحدةِ في تتناقصُ تدريجي. هذا التناقص في مستوى المياه الجوفيّة يُرتبط طبيعيا بتدهور ملحوظ في نوعيةِ المياه الجوفيّة، وبشكل رئيسي بسبب مشكلةِ تداخل ماءَ البحرِ وندرة تجدد المياه الجوفيةِ. حتى عام 1996، كان المتوسّط السنوي للامطار في الإمارات العربيةِ المتحدة يقدر بـ 110 ملليمتر. وتشير البيانات المسجلة للأمطار خلال السنوات الـ10 الماضية بأنّ المتوسط السنويَ انخفض إلى أقل منْ 80 ملليمتر.

الكثير منْ الآبار والعديد من المزارع تُركتْ طوال العقد الأخير في الطبقات الجوفيةِ الساحليةِ للإمارات العربيةِ المتحدةَ بسبب التدخل المفرطِ لماءِ البحر في طبقات الماء العذبِ الجوفيةِ. المشكلةَ كَانتْ أكثرَ وضوحاً في الطبقة الجوفيةِ الساحليةِ للفجيرة. من الناحية الأخرى، ابدت المنظماتِ والمستفيدين مخاوفهم وإهتمامَهم لدِراسَة مشكلةِ تداخل ماءَ البحر في منطقةِ وادي حام بسبب أهميتِه التأريخيةِ لتطويرِ المياه الجوفيّةِ والأنشطة الزراعيةِ. لذلك اختيرت الطبقة الجوفية الساحلية للفجيرة لتكون مادةِ بحث هذا المشروع.

يجب تقديراستمرارية الضخ على المدى الطويل منْ الطبقة الجوفيةِ الساحليةِ لوادي حام كما يَجِبُ تحديد الموقعَ الامثل لأيّ ضخ إضافي. في هذه الدراسة إستخدمَ برنامج، SUTRA، لمحاكات تدفق المياه الجوفيّة وتدخلَ ماءِ البحر في طبقة الجوفيةِ لوادي حام. تجرى عمليةِ المحاكاة في ظل الحالةِ الثابتةِ وحالة عدم الاستقرار ونقل الاملاح في المياه الجوفيّةِ. تمت الدراسة باعتبار المقطع الأفقي والمقطع العمودي وقدمت النتائجُ على شكل equipotential equiconcentration lines.و تمت معايرة والتثبت من صحة النموذج على السجلاتِ المتوفرةِ عن المياه الجوفيّةِ. وكذلك تمت دراسة تأثيرات زيادةِ ضخ المياه الجوفيّةِ منْ حقل كلباء.

تُشيرُ النتائجُ إلى ان تداخل ماءَ البحر في طبقةَ وادي حام الجوفيةِ تعتمد إلى حد بعيد على الضخ منْ حقل كلباء. خفض مستوى الضخ منْ حقل كلباء يُساعدُ على تحسين نوعيةِ المياه الجوفيّة. بينما زيادةُ قيَمَة معامل النفاذية الطولي يسمح للماء العذب للانتقال اكثر منْ المنطقةِ التي يتجمع فيها الماء منْ سدّ وادي حام نحو شاطيء البحر. ولذلك، تداخل ماءِ البحر يحدث بصورة اقل. خفض ال dispersivity منْ 10 إلى 1 مل يحد من تأثيرَ تداخل ماءِ البحر إلى المنطقةِ القريبة من حقل كلباء. و أشارتْ سيناريوهات المحاكاةِ العمودية إلى هبوط حاد لمستوى المياه الجوفية بعيدا عن خط الشاطئ و منحدر

جامعة الامارات العربية المتحدة

عمادة الدراسات العليا

برنامج ماجستير علوم موارد المياه

عنوان الرسالة:

تقييم تعرض المياه الجوفية لتداخل مياه البحر في الخزان الجوفي الساحلي بوادي حام، الامارات العربية المتحدة

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