

Department of Chemical Engineering

**Scenario Modeling for Prediction of Contaminant Transport in
Perth Unconfined Aquifer**

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Declaration

To the best of my knowledge and belief this thesis contains no material previously published by any other person except where due acknowledgment has been made.

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ABSTRACT

Rapid development and growth of industrialization has brought immense enrichments in living standards of humans, however, improper planned development also brings along several environmental problems such as pollution of environment and excessive consumption of natural resources. Among all the others, uncontrolled utilization of water poses a severe threat to the coming generations. Past decades have witnessed water shortage in various countries of the world. Although about 80% of the earth's surface is covered with water, around 97.2% of water is salty making it inappropriate for general usage. Among the rest of the 2.8%, which is present as fresh water on surface, a large proportion of it has been found to be severely polluted. The increasing demand of fresh water both for industrial and domestic usage adds great demand on the available groundwater. Moreover, the severe pollution of fresh water on the surface adds more stress on the available groundwater. In Australia, approximately 20% of water supply is from groundwater and in the case of Western Australia groundwater provides two thirds of its water supply needs. Thus, it is important to manage groundwater sources in Western Australia to achieve the optimum water utilization and maintain the water table and it is also essential to decide on an appropriate water budget. Groundwater flow modelling is an effective tool to get appropriate water distribution and, to examine effects from pumping on water levels and direction of groundwater flow paths, thereby helping in its proper management and utilization. Apart from monitoring the flow and utilization, groundwater flow modeling is also vital to keep the track of pollutant in the groundwater. Increasing surface pollution and landfill sites tend to pollute the groundwater due to leaching.

The above mentioned aspects formed the basis of the present research. A groundwater flow model was developed in Visual MODFLOW Premium to study the effect of three different types of soil in and around Perth region. This study also shows the hypothetical contaminated site model for benzene, toluene, ethylbenzene and xylene (BTEX) transport in Perth Superficial unconfined aquifer which includes three major aquifer sediments namely Bassendean Sand, Safety Bay Sand and Tamala Limestone. Among the four different contaminants it was observed that

benzene is able to migrate quickly as compared to the other contaminants due to its smaller distribution coefficient.

This study also explored the major soil parameters such as effect of sorption, effective porosity and hydraulic conductivity on contaminant plume configuration and contaminants concentration for the three types of aquifer sediments. A critical comparison of the behaviour of the three different types of soils was also conducted.

Simulation results of sensitivity analysis have shown that sorption and hydraulic conductivity greatly affected the contaminant plume length and concentration of contaminants with much lesser effect shown by the effective porosity. The simulated results also showed that the movement of contaminant in Tamala Limestone is most rapid by comparing these three types of aquifer sediments together. Thus, it can be said that contaminated sites found in Tamala Limestone needs immediate remediation of contaminants to bring down the contaminants concentration in groundwater.

In brief, the thesis explores the current groundwater scenario in and around Perth region. Based on the information a hypothetical scenario simulation has critically analyzed the various parameters affecting the water and contaminant flow for the various soil parameters. The study is considered as a building block for further research on developing a remediation technique for groundwater contaminant treatment.

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NOMENCLATURE

A	cross-sectional area perpendicular to groundwater flow
b	aquifer thickness
C	concentration of contaminant
C_d	dissolved concentration
C_s	sorbed concentration
D'	mechanical dispersion
dh	change in head
dh/dl	hydraulic gradient
D_L'	longitudinal dispersion
D_m	molecular diffusion
D_{TH}'	horizontal transverse dispersion
D_{TV}'	vertical transverse dispersion
D_x	hydrodynamic dispersion in x-direction
D_y	hydrodynamic dispersion in y-direction
D_z	hydrodynamic dispersion in z-direction
f	forcing function associated with recharge, biological and chemical activities
F_{oc}	fraction of organic carbon matter in soil
g	gravitational acceleration
h	hydraulic head
h_p	pressure head
h_v	velocity head
i	hydraulic gradient
K	hydraulic conductivity

k	the intrinsic permeability of porous medium
K_d	distribution coefficient
K_{oc}	organic carbon partition coefficient
K_x	component of hydraulic conductivity in x-direction
K_y	component of hydraulic conductivity in y-direction
K_z	component of hydraulic conductivity in z-direction
L	average travel distance of the contaminant plume
P	fluid pressure
P_e	effective porosity
P_t	total porosity
Q	rate of groundwater flow
q_s	fluid sink/source term
R	retardation coefficient
S	storativity or storage coefficient
S_s	specific storage
S_y	specific yield
t	time
$t_{1/2}$	half life of contaminants
V_a	volume of aquifer
V_c	contaminant transport velocity
V_g	linear groundwater velocity
V_w	volume of water
V_x	average linear groundwater velocity in x-direction
V_y	average linear groundwater velocity in y-direction
V_z	average linear groundwater velocity in z-direction

x	distance along the flowpath
z	elevation head
$\frac{\partial C}{\partial t}$	change in concentration with time
$\frac{\partial h}{\partial x}$	component of hydraulic gradient in x-direction
$\frac{\partial h}{\partial y}$	component of hydraulic gradient in y-direction
$\frac{\partial h}{\partial z}$	component of hydraulic gradient in z-direction
∇	derivative operator
μ	viscosity of the fluid
α_L	longitudinal dispersivity
α_{TH}	horizontal transverse dispersivity
α_{TV}	vertical transverse dispersivity
α_x	dispersivity
λ	rate of biodegradation
ρ	density of the fluid
ρ_b	aquifer bulk density

Abbreviations

ADWG	Australian Drinking Water Guidelines
ANZECC	Australian and New Zealand Environment and Conservation Council
AQUIFEM	Multi-layered Finite Element Model
ARMCANZ	Agriculture and Resource Management Council of Australia and New Zealand
ASTM	American Society for Testing and Materials
BTEX	Benzene, Toluene, Ethylbenzene, Xylene
DCE	Dichloroethene

DEC	Department of Environment and Conservation
EPA	Environment Protection Authority
GIS	Geographic Information System
GUI	Graphical User Interface
ISCO	In Situ Chemical Oxidation
MCL	Maximum Contaminant Level
MIN3P	Multi-component Reactive Transport Model in Three Dimensions for Variably Saturated Porous Media
MODFLOW	Modular Three Dimensional Finite Difference Model
MODPATH	Particle Tracking Model for MODFLOW
MT3D	Modular Three Dimensional Single Species Transport Model
MT3DMS	Modular Three Dimensional Multispecies Transport Model
NHMRC	National Health and Medical Research Council
PCE	Perchloroethylene
PHAST	Multi-component Reactive Transport in Three-dimensional Saturated Groundwater flow Systems.
PHREEQC	Computer Program for Speciation, Batch-Reaction, One-Dimensional Transport, and Inverse Geochemical Calculations
PHT3D	Multi-component Reactive Transport Model in Three Dimensions
PLASM	Prickett Lonquist Aquifer Simulation Model
PMPATH	Particle Tracking Model
PRB	Permeable Reactive Barrier
RT3D	Reactive Transport Model in Three Dimensions
SOEWA	State of Environment Western Australia
TCA	Trichloroethane
TCE	Trichloroethylene

USGS	United States Geological Survey
VC	Vinyl Chloride
VOC	Volatile Organic Compound
WA	Western Australia
WHI	Waterloo Hydrogeologic Inc.
WinPEST	Nonlinear Parameter Estimation and Predictive Analysis Tool
WRC	Water and River Commission
ZVI	Zero Valent Iron

CHAPTER 1

INTRODUCTION

1.1 Background

Surface water is a limited resource in the south-west region of Western Australia. Climate variability of rainfall and growing populations have increased demands for the limited water supply in Western Australia, and this has led to an increasing dependence on groundwater resources as a water supply (Scatena and Williamson 1999; Wu 2003). Therefore, groundwater is an important source of water supply in Western Australia (WRC 2000). Perth city in Western Australia overlies extensive unconfined and confined aquifers that provide about 40% of the city's potable water supply and about 70% of all the water used in the Perth region (Appleyard 1996; Davidson 1995; Blair and Turner 2004).

In Western Australia, groundwater is the major source of water supplies for the agricultural activity, domestic and industrial water supply in many areas, and is tapped by household bores for watering gardens (Blair and Turner 2004). Therefore, it is essential to maintain the optimum level of the groundwater table to obtain the appropriate water supply. The factors affecting the stability of the water table are: flooding from the rivers, excessive rainfall, drought which is the natural factor and others like excessive withdrawal of the groundwater from industry, irrigation etc. To achieve the optimum water utilization and maintain the water table it is essential to decide on an appropriate water budget. Thus, groundwater flow modeling is an important tool to get appropriate water distribution, to examine effects from pumping on water levels and direction of groundwater flow paths.

As in the rest of the world, drought has recently brought the message to West Australians that fresh water is an important resource that requires careful management (WRC 2000; Blair and Turner 2004). In the south-west of Western Australia, there has been variability in the climate condition over the last 27 years resulting in 50% less run-off into Perth's surface water supply catchments. In the last 7 years the average runoff has declined even further to about 30% of the previous

average (Blair and Turner 2004). This decline and the increasing demand of water in Perth are providing challenges for the Water Corporation (Blair and Turner 2004).

The rapid growth of population and urban areas in Perth has further affected the groundwater quality due to over exploitation of resources and improper waste disposal practices (Appleyard 1996). However, effluents from agricultural and residential activities, industrial areas such as accidental spills and underground storage tank leakage etc. are main sources of groundwater contamination and these contaminants from the sources are discharged into the ground, percolate and may alter the quality of groundwater (Fetter 1999; Jameel and Sirajudeen 2006). When such contaminants are detected in groundwater, it is essential to approximate the amount and level of contamination so as to take necessary actions that can reduce further risks to human health and the hydrogeological environment (Appleyard 1996; Fetter 1999; Villholth 2006). Thus, it is essential to develop a contaminant transport model for protection and management of the groundwater quality, to estimate the future content and concentration of contaminants in groundwater.

Prediction of the movement and degradation of the contaminants in groundwater can be solved by numerical models such as Visual MODFLOW-MT3DMS etc. This study describes the steps that are most-often involved in numerical modelling of contaminated sites and state some of the most commonly used modelling tools. Prediction of spatial and temporal movement of contaminant plumes by using a three-dimensional groundwater and solute transport model is an important tool for the remediation technology such as pump and treat method, permeable reactive barrier (PRB). These types of remediation technology are useful for removal of the contaminants from the groundwater. Pump and treat method and permeable reactive barrier remediation methods are the most commonly used for removal of the contaminants from the groundwater. However, several studies show that removal of contaminants from groundwater by conventional pump and treat method is more expensive and less effective than using permeable reactive barrier (Gavaskar 1999; Day, O'Hannesin and Marsden 1999; Simon and Meggyes 2000; Thiruvengkatachari, Vigneswaran and Naidu 2008). The concept of permeable reactive barrier is relatively simple. Reactive material is placed in the subsurface to intercept a contaminated groundwater plume flowing under a natural gradient. As the

contaminant moves through the material, reactions occur and the contaminants are either immobilized or chemically transformed to a more desirable (e.g., less toxic more readily biodegradable, etc.) state (Roehl 2005; Thiruvengkatachari, Vigneswaran and Naidu 2008). Therefore, a PRB is a barrier to contaminants, but not to groundwater flow (Thiruvengkatachari, Vigneswaran and Naidu 2008).

1.2 Objectives and Significance

1. Site Characterization

Site characterization is an important step to develop a groundwater flow and contaminant transport model. Site characterization is very useful to provide information about the geological information on the type and distribution of subsurface materials or geological formation of the aquifer, hydrogeologic parameters such as hydraulic conductivity, hydraulic gradients, and groundwater geochemical data. The primary objectives of this study are: (1) to gain an understanding of the geological and hydrogeological properties of the Perth region aquifer, (2) to identify the contaminated site characteristics, potential groundwater contaminants and their sources in the Perth region.

The data measured from site characterization is essential to simulate groundwater and transport of contaminants in subsurface. This simulation predicts the future extent and concentration of contaminant plume.

2. Simulation of Groundwater Flow and Contaminant Transport Model:

The secondary objective of this study is: to predict the future extent and concentrations of a dissolved contaminant plume by simulating the combined effects of dispersion, advection, sorption by using scenario modeling of the Perth unconfined Superficial aquifer.

This study represents the sensitivity analysis of the fate and solute transport model. Sensitivity analysis is important to determine the impact of different input parameter values on the model output. The sensitivity analysis is very useful to determine which parameter has the greatest influence on the contaminant transport model. This allows the prediction of worst-case scenarios.

The solute fate and contaminant transport model is an important tool in assessing the need for remedial design at contaminated sites and predicting the spatial and temporal migration of contaminant plumes and concentration of a dissolved contaminant plume.

1.3 Thesis Outline

A brief outline, on a chapter by chapter basis of the thesis is given below:

Chapter 1 presents some background information on groundwater flow and contaminant transport modeling. This includes the importance of groundwater flow and contaminant transport modeling followed by objectives and significance of the research work.

Chapter 2 discusses the Western Australia contaminated site characterization which includes fundamentals of the groundwater flow and contaminant transport processes and their properties, Perth region hydrogeology, groundwater contamination sources and groundwater contaminants of Perth region.

Chapter 3 discusses the numerical models for groundwater flow and contaminant transport study. This includes a comparison of several preprocessing and postprocessing software packages and an overview of Visual MODFLOW Premium and its applications. This chapter also includes the general methodology for development of the groundwater flow and contaminant transport model.

Chapter 4 presents detailed results of groundwater flow and contaminant transport modeling of hypothetical model in Perth region, which shows the contaminant plume behaviour with different scenarios and comparison of pollutant transport in three different types of soil aquifer found in Perth region Superficial unconfined aquifer.

Chapter 5 finally concludes the thesis by providing a summary and conclusions of the project followed by key recommendations.

References and Appendices are given thereafter.

CHAPTER 2

HYDROGEOLOGICAL CHARACTERISTICS AND CONTAMINATED SITES IN WESTERN AUSTRALIA

2.1 Introduction

This chapter provides an insight into the background of the relevant topics and methods for this study. In recent years, groundwater flow and contaminant transport modeling has evolved from a scientific curiosity to widely used design and analysis technology. Groundwater flow modeling focused on the evaluation of groundwater supplies in terms of quantity, but more recent applications have addressed issues of water quality. Groundwater resource issues involving mainly water quantity are largely addressed by groundwater flow models. Groundwater contaminant transport models are often needed when the problem to be addressed involves groundwater quality. A groundwater flow model is a necessary precursor to the development of a groundwater contaminant transport model (Pinder 2002).

2.2 Site Characterization

Site characterization is an important step to develop groundwater flow and contaminant transport model. Site characterization is very useful to provide information about the geological information on the type and distribution of subsurface materials or geological formation of the aquifer, hydrogeologic parameters such as hydraulic conductivity, hydraulic gradients, and groundwater geochemical data.

Therefore, this chapter discusses the data required for groundwater flow and contaminant transport modeling which includes hydrogeology and hydrogeological properties of Perth region, potential groundwater contaminant sources, groundwater contaminants, contaminated site characteristics of Perth region and contaminant transport processes which affect the contaminated groundwater flow.

The data measured from site characterization is essential to simulate groundwater and transport of contaminants in subsurface. This simulation predicts the future extent and concentration of contaminant plume.

2.3 Hydrogeology of the Perth Region

Hydrogeology is the branch of geology that deals with movement and distribution of groundwater in aquifers. The Perth region includes most of the Perth metropolitan area and is surrounded approximately by Gingin Brook to the north, and Dandalup River to the south, the Darling Scarp to the east and Indian Ocean to the west. The area lies almost entirely within the Swan Coastal Plain except for the northeastern part which includes a small portion of the Dandaragan Plateau. The region covers an area about 4000 km² (2600 km² lies to the north of Perth city and 1400 km² to the south) and extends from Perth some 80 km to Guilderton on the north coast, and 70 km south to Mandurah (Davidson 1995).

An aquifer is an underground layer of water bearing permeable rock or unconsolidated materials such as sand, silt, clay and gravel from which groundwater can be extracted using water well (Bear 1972). The geological formations of the Perth region, shown in Figure 2.1, have been grouped mainly into three different aquifers as Superficial aquifer/shallow aquifer, Leederville aquifer and Yarragadee aquifer, each being assigned the name of the major geological unit contributing to the aquifer of the Perth region. The Superficial/shallow aquifers are normally unconfined, the deep/Yarragadee aquifers are confined or semi-confined, the Leederville aquifer is semi-confined, lying between Superficial and Yarragadee aquifers. These aquifers are hydraulically connected; elsewhere they are separated by major confining beds or by the distribution of the geological formation (Davidson 1995).

In the Perth metropolitan area, the groundwater quality is generally good. The average groundwater movement in Superficial aquifers is approximately 30 m year⁻¹; while in deep aquifers, the maximum groundwater movement is less than 1 m year⁻¹ (Davidson 1995; Li et al. 2006). However, the groundwater movement depends mainly on specific site. Both abstraction and injection wells can change groundwater movements in aquifers (Li et al. 2006). The cost of drilling a well into a deep aquifer

is more expensive than a well in a shallow aquifer. Thus, most abstraction wells in the Perth region are drawing groundwater from Superficial aquifers (Davidson 1995). Therefore, it is important to maintain the groundwater quality of Superficial aquifer to match drinking water standards to fulfill human health, safety and environmental regulations.

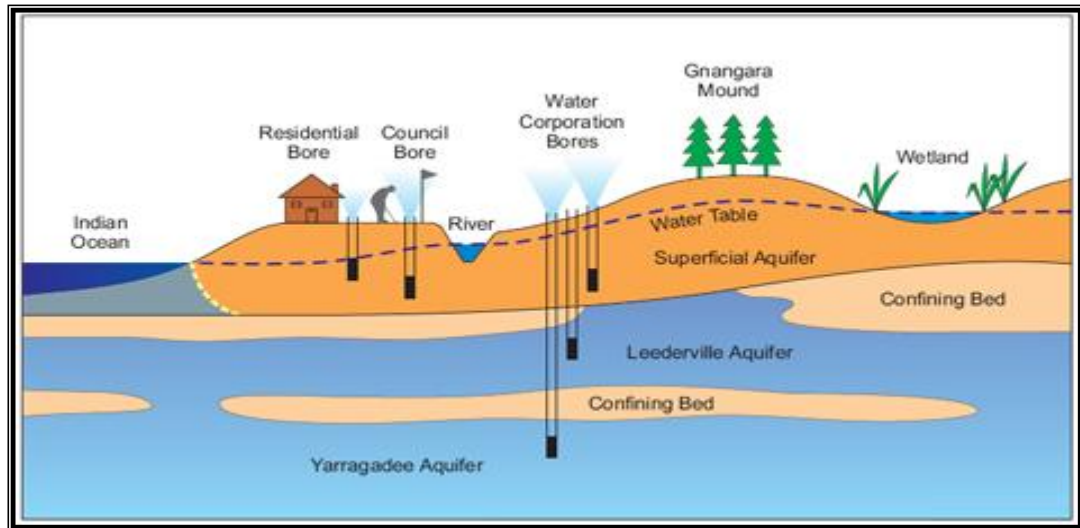


Figure 2.1: Cross Sectional View of Perth Aquifers (Sommer 2006)

2.3.1 Superficial Aquifer

The Superficial aquifer is a major unconfined aquifer found in the Perth region. It is a complex and multilayer aquifer. This type of unconfined aquifers usually receives recharge water directly from the surface, from a surface water body such as river, stream or lake or from precipitation which is in hydraulic connection with it. Natural recharge to unconfined aquifers occurs through lateral groundwater flow or upward discharge from underlying aquifers (Fetter 2001).

The Superficial aquifer is extending throughout the Perth coastal plain, west of the Gingin and Darling Scraps (Davidson 1995). It comprises of a number of different quaternary – tertiary sediments of the coastal plain such as Bassedean Sand, Safety Bay Sand, Tamala Limestone, Gangara Sand, Guildford Clay, Becher Sand and Yoganup Formation as shown in Figure 2.2. It consists of mainly sand, silt, clay and limestone in varying proportions.

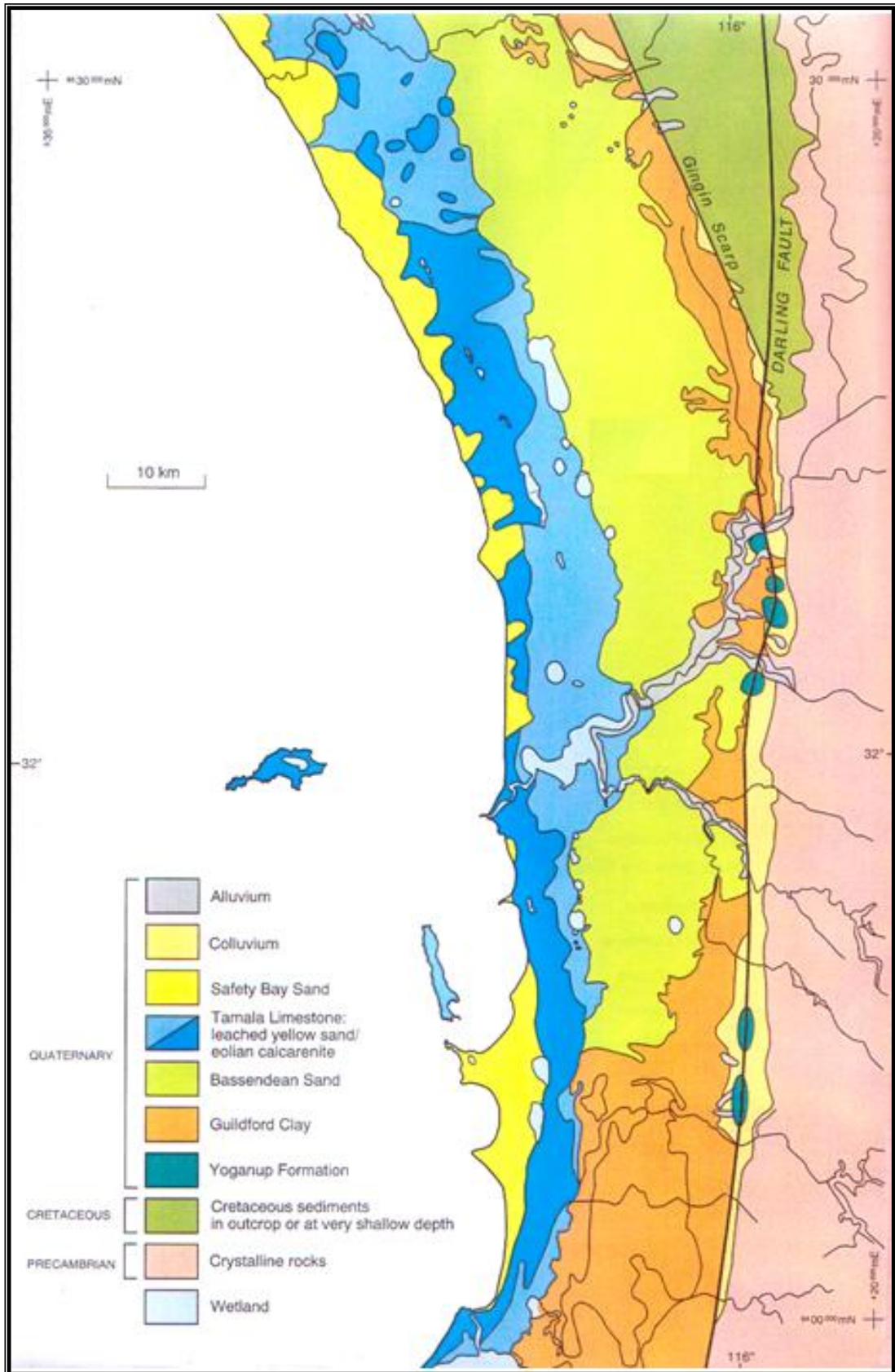


Figure 2.2: Generalized Surface Geology of the Perth Region Superficial Aquifer (Davidson 1995)

The sediments which compose the Superficial aquifer range from Guildford clay in the east adjacent to the darling fault through a sandy succession (Bassendean Sand and Gangara sand) in the central coastal plain area to sand and limestone (Safety Bay Sand, Becher sand and Tamala Limestone) within the coastal belt and they are mainly differentiated by their lithology.

Table 2.1 shows the maximum thickness and lithology of different sediments of the Perth region unconfined Superficial aquifer. It has a maximum thickness of about 110 m, but the average thicknesses of 45 m and 20 m are found in the northern and southern Perth regions respectively (Davidson 1995).

Stratigraphy	Thickness (m)	Lithology	Aquifer
Safety Band Sand	24	Sand and shelly fragments	Superficial aquifer
Becher Sand	20	Sand, silt, clay and shelly fragments	
Tamala Limestone	110	Sand, limestone and minor clay	
Bassendean Sand	80	Sand, subordinate silt and clay	
Gangara Sand	30	Sand, gravel and subordinate silt and clay	
Guildford Clay	35	Clay with subordinate sand and gravel	Confining Bed
Yoganup Formation	10	Sand, silt and clay	Superficial aquifer
Ascot Formation	25	Limestone, sand, shell and clay	

Table 2.1: Lithology of Different Sediments of Superficial Aquifer of Perth Region (Davidson 1995)

2.3.2 Leederville Aquifer

It is a major confined and multilayer aquifer of the Perth region that is filled by slow downward groundwater flow from the overlying Superficial aquifer. The groundwater movement through these aquifers is usually very slow. The Leederville aquifer consists of discontinuous interbedded shales, sandstones and siltstones and comprises of the different sediments such as Leederville Formation, Pinjar Member, Wanneroo Member, and Mariginiup Member. It is separated from Superficial aquifer by Osborne Formation and the Yarragadee aquifer is separated from Leederville aquifer by south Perth shale. Table 2.2 shows the maximum thickness and lithology of different sediments of the Perth region confined Leederville aquifer. Thickness of the aquifer is varying from approximately 50 m to 600 m in Perth region (Davidson 1995).

Stratigraphy	Thickness (m)	Lithology	Aquifer
Leederville Formation	600	Sandstone, siltstone and shale	Leederville Aquifer
Pinjar Member	150	Sandstone, siltstone and shale	
Wanneroo Member	450	Sandstone, siltstone and shale	
Mariginiup Member	250	Sandstone, siltstone and shale	
South Perth Shale	300	Sandstone, siltstone and shale	Confining Bed

Table 2.2: Lithology of Different Sediments of Leederville Aquifer of Perth Region (Davidson 1995)

Sometimes, confining layers are subdivided into aquitard, aquiclude and aquifuges. An aquitard is a low permeability layer that restricts the groundwater flow from one aquifer to another. An aquitard can sometimes, if completely impermeable, be called

aquiclude or aquifuge. The aquitards are composed of layers of either clay or non-porous rock with low hydraulic conductivity.

2.3.3 Yarragadee Aquifer

It is a major confined aquifer underlying the entire Perth region and extending to the north and south within the Perth basin. It is a multi layer aquifer, more than 2000m thick, consisting of interbedded sandstones, siltstones and shales of the Gage Formation, Parmelia Formation, Yarragadee Formation and Cattamarra Coal Measures (Davidson 1995). Table 2.3 shows the maximum thickness and lithology of different sediments of the Perth region confined Yarragadee aquifer.

Stratigraphy	Thickness (m)	Lithology	Aquifer
Gage Formation	350	Sandstone, siltstone and shale	Yarragadee Aquifer
Parmelia Formation	>287	Sandstone, siltstone and shale	
Yarragadee Formation	>2,000	Sandstone, siltstone and shale	
Cattamarra Coal Measures	>500	Sandstone, siltstone and shale	

Table 2.3: Lithology of Different Sediments of Yarragadee Aquifer of Perth Region (Davidson 1995)

2.4 Hydrogeological Properties of Perth Region Aquifer

A groundwater flow and contaminant transport model requires many different types of data to simulate hydrogeological and hydrogeochemical processes influencing the flow of groundwater and migration of dissolved contaminants.

The following section shows the hydrogeological properties of the Perth region aquifer and these properties are important for input data for groundwater flow and contaminant transport modeling studies.

2.4.1 Hydraulic Conductivity

Hydraulic conductivity, K , is a very essential parameter to assess an aquifer's capability to transmit water (Bear 1979) and is also a useful parameter for governing groundwater flow in aquifers which has the units of length over time (Lovanh et al. 2000). It mainly depends on both the porous medium and the moving fluid which includes the soil matrix structure, soil grain size, the comparative quantity of soil fluid available in the soil matrix and type of the soil fluid (Fetter 2001). Thus, the hydraulic conductivity of a porous medium transmitting a given fluid can be determined using the following empirical relationship (Bear 1972):

$$K = \frac{k\rho g}{\mu} \dots\dots\dots \text{Equation 2.1}$$

Where,

k = the intrinsic permeability of porous medium (L^2)

ρ = density of the fluid (ML^{-3})

g = gravitational acceleration (LT^{-2})

μ = viscosity of the fluid ($ML^{-1}T^{-1}$)

K = the hydraulic conductivity (LT^{-1})

Hydraulic conductivity of an aquifer can be measured by the aquifer slug tests and pump tests. Several researchers (Wiedemeier et al. 1995; Lovanh et al. 2000; Zheng and Bennet 2002) observed the values of hydraulic conductivity range over 12 orders of magnitude from 2.5×10^{-12} to 0.05 m/s for different types of soil materials. The expected representative values of hydraulic conductivity for different soil types are presented in Appendix 1. In general, it shows that for the unconsolidated sediments hydraulic conductivity tends to increase with increasing grain size and sorting. The groundwater velocity and dissolved contaminant transport velocity is directly related to the aquifer hydraulic conductivity (Wiedemeier et al. 1995). Subsurface variations in hydraulic conductivity directly influence contaminant fate and transport by providing preferential pathways for contaminant migration. Therefore, site specific hydraulic conductivity values should be used for the groundwater flow and contaminant transport modeling.

The hydraulic conductivity of an aquifer varies with soil type and the following Table shows the typical value of hydraulic conductivity for Superficial aquifer found in Perth region. Table 2.4 shows that the hydraulic conductivity value ranges from 0.4 m/day to 1000 m/day in Superficial aquifer. Therefore, groundwater flow is generally greatest in sediments with a high hydraulic conductivity, such as limestone and sand, and slowest in sediments with a low hydraulic conductivity, such as clay. The hydraulic conductivity of Leederville aquifer generally ranges from 0.2 m/day to 9 m/day and mainly depends upon the sandstone, siltstone and shale proportions and the Yarragadee deep confined aquifer have hydraulic conductivity ranges from 0.7 m/day to 6 m/day in Perth region (Davidson 1995).

2.4.2 Hydraulic Head and Hydraulic Gradient

The total hydraulic head (h) at a given point in the aquifer is composed of pressure head (h_p), velocity head (h_v), and elevation head (z) and is represented by the following equation (Fetter 2001).

$$h = h_p + h_v + Z \dots\dots\dots \text{Equation 2.2}$$

Lithology	Hydraulic conductivity (m/d)
Sand	
Very coarse to gravel	246
Very coarse	204
Coarse	73
Medium to very coarse (moderately sorted)	50
Fine to gravel (poorly sorted)	10
Medium	16.5
Fine to medium	8.2
Fine	4.1
Sand	
Fine to very fine	1.7
Very fine	0.8
Silty	4
Clayey	1
Clay	0.4
Sand and Limestone: Ascot Formation	8
Limestone and Calcarenite: Tamala Limestone	100 to 1000
Eastern Area of Clayey Sediments	Less than 10
Central Sandy Area: Bassendean Sand	10 to 50

Table 2.4: Hydraulic Conductivity Values for Superficial Aquifer (Davidson 1995)

Where, the pressure head (h_p) is correlated to the fluid pressure (P), water density (ρ) and the gravitational acceleration (g) (Lovanh et al. 2000), the velocity head (h_v) is related to the kinetic energy from the motion of the water however the velocity head is mostly ignored in groundwater studies because the velocity of groundwater flowing through porous media is generally very slow and making the velocity term of energy negligible when compared to other groundwater flow terms (Wu 2003; Fetter 2001) and the elevation head (z) is the elevation of the bottom of the piezometer. Thus, for groundwater studies, hydraulic head (h) can be calculated by equation 2.3 (Fetter 2001).

$$h = h_p + z \dots\dots\dots\text{Equation 2.3}$$

The total hydraulic head usually changes across a site (Lovanh et al. 2000). The rate of change in hydraulic head with change in the distance in groundwater flow direction is called hydraulic gradient and is given by the following equation (Fetter 2001).

$$i = \frac{dh}{dl} \dots\dots\dots\text{Equation 2.4}$$

Where, i = hydraulic gradient (L/L) and dh/dl = change in hydraulic head with change in distance (L/L)

The value of hydraulic gradient generally ranges from 0.0001 to 0.05 m/m (Domenico and Schwartz 1990). In this study, it was found that the hydraulic gradient normally ranges from 0.002 m/m to 0.005 m/m for the Perth region aquifers (Thierrim et al. 1993; Davidson 1995; Davis et al. 1999; Trefry et al. 2006). It is also necessary to estimate the accurate value of hydraulic gradient for the specific site because the hydraulic gradients influence the direction and rate of groundwater flow and contaminant migration (Wiedemeier et al. 1995).

2.4.3 Porosity

Porosity is one of the important aquifer properties. Porosity of the aquifer material can be measured as total and effective porosities. The total porosity, denoted by P_t , is defined for a porous medium as the ratio of the volume of voids to the total volume of a representative sample of the medium (Fetter 2001). Effective porosity (P_e) of a porous medium is defined as the ratio of the part of the pore volume where the fluid can flow to the total volume of a medium (Fetter 2001). Thus, the effective porosity value is always less than the total porosity. It is a dimensionless quantity and can be represented either as a percentage or as a fraction.

The porosity of a soil will differ with the arrangement of particles or soil texture because the pore space and the pore spaces arrangement within a soil sample are very complicated and complex to measure (Ohio EPA 2007). It is generally obtained from

the types of aquifer materials and chosen from the literature or it can be estimated by the geotechnical laboratory method or by the tracer test (Lovanh et al. 2000). Appendix 2 represents the accepted literature porosity (total porosity and effective porosity) values for different types of soil texture or aquifer materials and it can be seen that finer grained soils rich in clay will have higher porosities whereas the coarser textured soils rich in sand have lesser porosity values.

Porosity does not directly affect the distribution of hydraulic head of an aquifer, but it has a very strong effect on the migration of dissolved contaminants, since it affects groundwater flow velocities through an inverse proportionality relationship (Domenico and Schwartz 1990). Thus, it is an essential parameter to consider for groundwater flow and contaminant transport model. Many researchers estimated the effective porosity of Perth unconfined aquifer ranging from 20 % to 30 % (Thierrim et al. 1993; Davidson 1995; Trefry et al. 2006) and approximately 20 % for the Perth confined aquifer (Davidson 1995).

2.4.4 Specific Storage, Storativity and Specific Yield

Specific storage (S_s), storativity or storage coefficient (S) and specific yield (S_y) are important aquifer properties. Specific storage can be defined as the amount of water released from aquifer storage per unit change in hydraulic head in the aquifer, per unit volume of the aquifer (Freeze and Cherry 1979) and is represented by the following relationship:

$$S_s = \left(\frac{1}{V_a}\right) \left(\frac{V_w}{dh}\right) \dots\dots\dots \text{Equation 2.5}$$

Where,

S_s = specific storage (L^{-1})

V_a = volume of aquifer (L^3)

V_w = volume of water (L^3)

dh = change in hydraulic head (L)

It is an important mechanism for storage in confined aquifers. The value of specific storage of Perth region confined and unconfined aquifer is typically very small, generally 0.0001 1/m or less (Davidson 1995).

Storativity or storage coefficient is the volume of water adsorbed or released from storage per unit area of the aquifer per unit change in aquifer hydraulic head (Freeze and Cherry 1979). Storativity is the vertically incorporated specific storage value for the aquitard or confined aquifer. Storativity, for a confined homogeneous aquifer or aquitard, is expressed by the following relationship (Fetter 2001):

$$S = S_s b \dots\dots\dots \text{Equation 2.6}$$

Where,

S_s = specific storage (L^{-1})

S = storativity or storage coefficient (Dimensionless)

b = aquifer thickness (L)

It is a dimensionless quantity and normally ranges from 0 to the effective porosity value of the aquifer. However, storativity value is much less than 0.01 for confined aquifer or aquitard (Johnson 1967).

For an unconfined aquifer, the storativity value is approximately equal to the specific yield (Fetter 2001). Specific yield can be defined as the volume of water that an unconfined aquifer releases from storage per unit surface area of aquifer per unit decline in the water Table (Wu 2003). Not all the water stored in pore spaces becomes part of flowing or moving groundwater. Water clings to soil particles due to surface tension, cohesion and adhesion. It forms a thin film around a particle.

It is generally used for unconfined aquifer because specific storage component of unconfined aquifer is very small and has a negligible contribution (Fetter 2001). Specific yield value is generally less than or equal to effective porosity (Johnson 1967). Appendix 3 represents the accepted literature specific yield (S_y) values for different types of soil texture or aquifer materials.

Davidson (1995) estimated the specific yield of 5% for the eastern clayey (Guildford clay) region, 20% for central sandy (Bassendean Sand and Gangara sand) region and 30% for western (Tamala Limestone and Safety Bay Sand) region of the Perth Superficial aquifer and he also estimates the specific yield of the Perth confined aquifer, Leederville aquifer and Yarragadee aquifer, approximately as 20%.

2.4.5 Bulk Density

Bulk density of a soil (ρ_b), as used in groundwater model, is defined as the mass of the dry solid divided by the bulk volume of the soil (Fetter 2001). It is used to calculate the retardation factor for the contaminants and is mainly affected by the soil texture. Thus, the bulk density will differ within certain limits for different soil types e.g. the value of bulk density can be as low as 1.1 gm/cm^3 in aggregated loams and clayey soils. However, the value of bulk density can be quite high up to 1.81 gm/cm^3 for sandy soils (Domenico and Schwartz 1998). The literature values of bulk density for different types of aquifer materials can be found from Domenico and Schwartz (1990). The average literature value of bulk density of the Perth unconfined and confined aquifer is approximately 1.7 gm/cm^3 (Davidson 1995; Salama et al. 2005).

2.5 Groundwater Flow Equation

To effectively model the contaminant transport in the groundwater, it is essential that the flow is well defined. The transport of the pollutants is a major factor of the groundwater velocity apart from the affinity to the subsurface solid materials. The flow of groundwater can be mimicked as the flow through a porous medium. The simplest visualization of the groundwater flow can be the flow through a porous medium, as shown below.

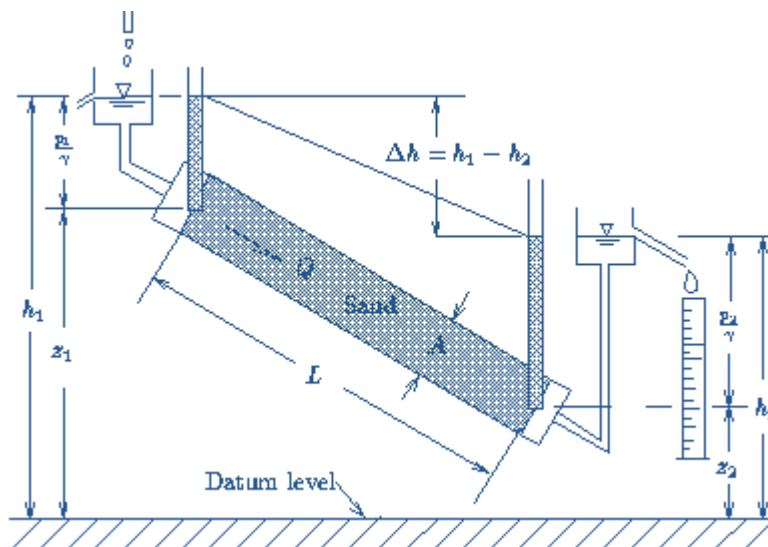


Figure 2.3: Groundwater Flow through Porous Media (Zheng and Bennet 2002)

However, unfortunately, nature is much more complex than the simple porous medium flow. Every layer in the ground has different properties, such as hydraulic conductivity, porosity etc. Apart from that, there are several regions of nonconductive layers known as aquitard. To model a complex region it is necessary to divide the region in small sections, to conduct the mass balance.

The flow through the porous medium in one dimension is described by Darcy's law given as follows:

$$Q = -K A \frac{dh}{dl} \dots\dots\dots \text{Equation 2.7}$$

Where,

Q , is the rate of groundwater flow in volume of water per unit time (L^3T^{-1})

K , is a coefficient of proportionality called hydraulic conductivity (LT^{-1})

dh , is the height difference, measured from a fixed level (L)

A , is the cross-sectional area perpendicular to groundwater flow (L^2)

For the simulation purpose, a sea level can be assumed as the fixed datum. The laws of mass and energy conservation are united with Darcy's law to develop the differential equations describing the flow through porous media. The equations depict flow using the self-governing spatial coordinates, x , y and z , and time, t (Fetter 2001). Thus, for three dimensional groundwater flows in aquifer, on the small section doing the mass balance we get:

(Flow in – Flow out)_x + (Flow in – Flow out)_y + (Flow in – Flow out)_z + mass gain or lost = Accumulation

$$\frac{\partial}{\partial x} \left(K_x \frac{\partial h}{\partial x} \right) + \frac{\partial}{\partial y} \left(K_y \frac{\partial h}{\partial y} \right) + \frac{\partial}{\partial z} \left(K_z \frac{\partial h}{\partial z} \right) + q_s = S_s \frac{\partial h}{\partial t} \dots \text{Equation 2.8}$$

Where,

K_x , K_y and K_z represent the components of hydraulic conductivity in the respective coordinate directions (L/T)

S_s is the specific storage, or volume of water released from storage in a unit volume of aquifer per unit decline in head (L^{-1})

q_s is the fluid sink/source term, or volumetric rate at which water added to or removed from the system per unit volume of aquifer (L^3/T)

$\frac{\partial h}{\partial x}$, $\frac{\partial h}{\partial y}$, $\frac{\partial h}{\partial z}$ represent the components of hydraulic gradient in x , y and z coordinate direction, respectively

It is essential to appreciate the fact that, the height 'h' is related both to the elevation above the datum line and the pressure acting on the water. It is also related to the density of the groundwater, which may vary depending on the presence of the solvents in the water.

The three dimensional groundwater flow equation (equation 2.8) would have several boundary conditions depending on the system. The most simplistic approach would be to define a fixed pressure drop across the end boundary. However, in several cases the constant mass input or output acts as the boundary conditions depending on the presence of the source or sinks (such as river, sea or lake) at the ends. A complex system may also demand a source input at the upper layer of the system due to the precipitation, however, such a system would demand a time variant approach.

Solution of equation 2.8 along with the appropriate boundary condition results in the velocity vector at each element of the groundwater system.

2.6 Groundwater Contamination Sources in Perth Region

Contamination of groundwater occurs when contaminants alter the chemical or biological characteristics of the groundwater and degrade the quality of water and that humans, animals or plants using the water are affected (WRC 1998).

In general, the quality of the shallow groundwater is still very good in Perth region but there are instances of localized groundwater contamination. The potential sources of groundwater contamination fallout in Perth region from leakage of underground petroleum storage tanks, agricultural activities or widespread use of fertilizers on garden, improper waste disposal practice, localized spills of chemicals etc. as point source contamination or diffuse source contamination (WRC 1998). Based on this contamination sources, Department of Environment and Conservation (DEC) of Western Australia identified different contaminated sites in Western Australia. Most of the contaminated sites are caused by heavy industry, service or fuel stations, landfill sites, agricultural activities, power stations, gasworks, mine sites and chemical industries.

Figure 2.4 depicts the percentage of identified Western Australia contaminated sites by sector wise. It shows that around two-thirds of new contaminated sites are identified as service industries and the oil, petroleum and energy sectors. A high number of contaminated sites related to the service sector can be attributed to land redevelopment and new regulations for the Contaminated Sites Act 2003 became operational in December 2006, and are expected to result in a dramatic increase in the number of contaminated sites recorded in 2007. In March 2007, approximately 1358 contaminated sites had been formally reported to the Department of Environment and Conservation of Western Australia (EPA 2007). Based on new regulation of the contaminated sites in Western Australia, they have been classified into three categories (DEC 2006):

1. Contaminated sites with remediation required
2. Contaminated sites with restricted use
3. Remediated contaminated sites for restricted use

Appendix 4 shows the list of the Western Australia contaminated sites with their source of contamination and the type of contaminants. This study also found that most of the contaminated sites in Perth region are historically used as service stations and investigation identified the presence of petroleum hydrocarbon e.g. benzene, toluene, ethylbenzene and xylene, and heavy metal such as lead in groundwater at contaminants concentration levels exceeding Australian Drinking Water Guidelines and the major source of contamination identified at these types of contaminated sites are leakage of underground fuel storage tanks (DEC 2006).

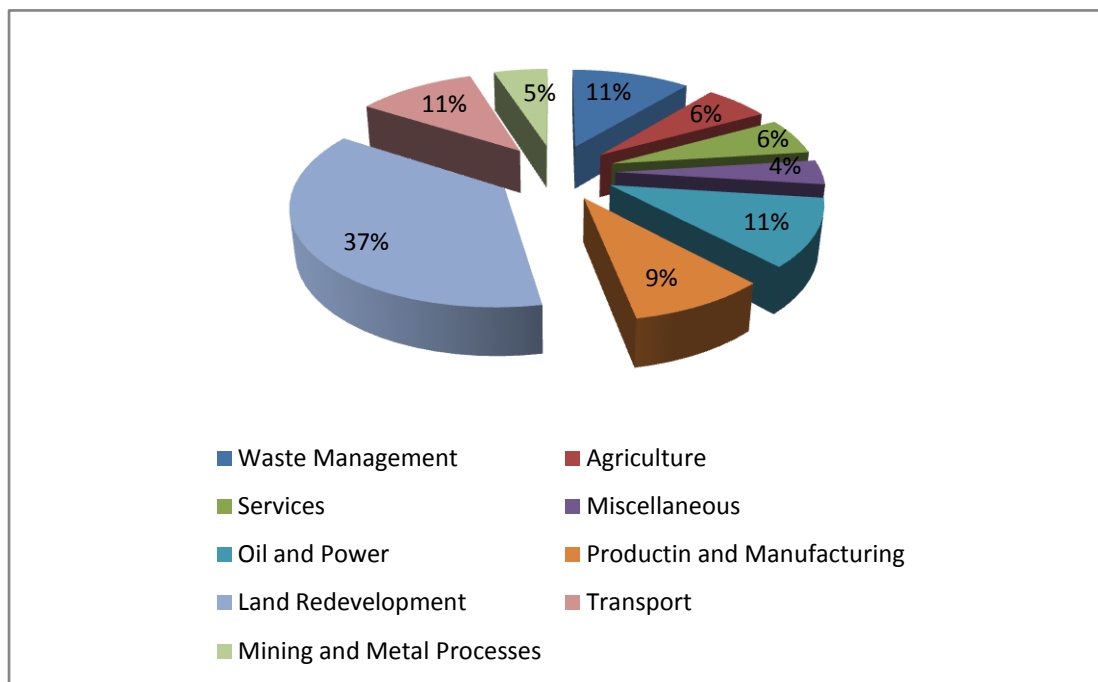


Figure 2.4: Percentage of the Contaminated Sites in Western Australia by Sector (EPA 2007)

The second type of contaminated sites identified in various areas of Perth has historically been used as a fertilizer manufacturing plant, a land use that can cause soil and groundwater contamination (DEC 2006). Groundwater beneath the sites are contaminated with heavy metals including arsenic, lead, chromium, copper and fluoride.

The third type of contaminated site found in Perth region was historically used as railway marshalling yards and associated commercial activities, a land use that has the potential to cause contamination (DEC 2006). The investigations identified

hydrocarbons including fuel and lubricating oils, organic contamination, pesticides and heavy metal such as lead, chromium and copper contamination in soils and underlying groundwater at concentration levels of contaminants exceeding Australian Drinking Water Guidelines (DEC 2006).

The other type of contaminated sites recognized in Perth was historically used as a landfill facility accepting domestic, industrial and quarantine waste. Many types of solid wastes were disposed of at the landfill including municipal waste, industrial waste and hazardous waste. Liquid wastes disposed of at the landfill included effluent, oils and chemicals. The investigation identified hydrocarbons and heavy metals were present in soils and groundwater plume at levels exceeding Australian Drinking Water Guidelines (DEC 2006).

2.7 Groundwater Contaminants

A wide range of materials or chemical substances have been recognized as groundwater contaminants in Perth region. They are inorganic compounds, organic and synthetic compounds, such as pesticides, and other contaminants. As drinking water systems get their water from groundwater and surface water sources, once the source becomes contaminated, the drinking water can also become contaminated.

Based on above mentioned contaminated sites of Perth Region, the following classes of contaminants have been identified in Perth groundwater.

In various areas of Perth, the major concern is contamination of soil and groundwater by benzene, toluene, ethylbenzene, xylene (BTEX) compounds resulting from spills or leaks of petroleum hydrocarbons (Prommer, Barry and Davis 1999). Through its extensive use, petroleum hydrocarbons have turned out to be the most important source of groundwater contamination worldwide. Petroleum hydrocarbons typically are spilled from underground fuel storage tanks or pipelines at airports, refineries, and service stations. It was also found that approximately 20% of the underground storage tanks showed signs of petroleum hydrocarbons leakage in Perth (Trefry et al. 2006). Some of these types of contaminants are found to be carcinogenic.

Chlorinated aliphatic hydrocarbons or other halogenated hydrocarbons such as trichloroethylene (TCE or trichloroethene) and tetrachloroethene (PCE or perchlorethylene) are the most commonly found organic contaminants in Perth groundwater (Benker et al. 1994). These chemicals are mostly used as a degreasing agent for machinery and metal parts in industry and as a solvent in paints (Trefry et al. 2006). They have been widely used in Perth industrial area and small quantities of these contaminants may have considerable effect on groundwater quality (Benker et al. 1994). These types of contaminants are suspected to be carcinogenic and drinking or breathing high levels of these compounds may cause nervous system effects such as lung or liver damage (Li et al. 2006).

A primary assessment in Perth region has discovered elevated heavy metal concentrations in either surface water or groundwater in areas impacted by disturbed acid sulphate soils (Hinwood et al. 2006). A large group of metals have been found at

concentrations which have the potential to impact on both ecological and human health (Appleyard et al. 2004) and these consist of high concentrations of heavy metals such as aluminum, arsenic, cadmium, iron, lead, mercury and selenium, well in excess of national guidelines for both drinking water and recreational water quality and in some cases irrigation standards (ANZECC and ARMCANZ 2000; NHMRC 2004).

In Perth region, aquifers contamination by nutrients (e.g. nitrogen, ammonia and phosphorous), from sources such as septic effluent tanks, garden fertilizers, has caused an increase in the number of pathogens present in groundwater (Trefry et al. 2006). This type of contamination has resulted in a restricted use of unprocessed groundwater due to health problems. The contamination of Perth's water bodies is further exacerbated because the major soil types of the Perth metropolitan area (especially the most prevalent Bassendean Sands) are limited in their capacity to retard the progress of microbes through filtration of water passing through to the aquifer. They also have a very poor capacity to remove chemicals, particularly nutrients, nitrogen and phosphorous, which are the most commonly found pollutants from sanitary waste disposal (Nixon 1996). These types of contaminants mainly adsorb on to the mineral phases in soil and those effects a potential hazard in Perth region aquifers (Trefry et al. 2006).

The other common types of contaminants found in Perth region are pesticides such as diazinon and atrazine (Appleyard 1995; Patterson et al. 2000). They are widespread soil and groundwater contaminants around the Perth region (Patterson et al. 2000) and, even at little concentrations, are a concern due to their toxicity. They are mainly used on parklands, by agriculture and horticulture, and by local householders (Trefry et al. 2006). Once pesticides are released (spilled or leaked) from the source then they may accumulate on top of the soil, or be leached through the soil into the underlying groundwater and cause a risk to human health or potable water supplies. Because of the potential for pesticides to impact quality of groundwater, there have been numerous field studies conducted to evaluate the transport of pesticides from ground surface to groundwater.

The Maximum Contaminant Level (MCL) is set for above mentioned contaminants: that is the maximum allowable level of a contaminant in water that is delivered to

any user of a public water system. Basing on scientific research, higher concentrations could cause health problems in humans (Fetter 1999).

Generally, these types of contaminants interact with the moving groundwater and the soil, and spread out to form a contaminant plume moving in the same direction as the groundwater. The resulting groundwater contamination plume may extend several hundred metres or even further away from the source of pollution.

The following section discusses the contaminant transport processes which mainly affect the contaminant movement in groundwater.

2.8 Contaminant Transport Processes

The contaminant in the groundwater moves along the velocity vector determined from the groundwater equation. The contaminant transport processes are important for measuring the movement and chemical alteration of dissolved contaminants into groundwater. There are mainly two processes that affect the contaminant migration in groundwater:

- (1) Physical processes such as advection, hydrodynamic dispersion , and
- (2) Chemical processes such as sorption and chemical reaction

This study involves BTEX migration in Perth Superficial, Bassendean Sand, Tamala Limestone and Safety Bay Sand, unconfined aquifer. The following sections describe the physical and chemical processes that influence the dissolved contaminants migration.

2.8.1 Advection

Advection is defined as the migration of contaminant due to the bulk movement of groundwater (Wiedemeier et al. 1995): as the contaminant is dissolved into the groundwater and is carried along by groundwater flow (Zheng and Bennett 2002). It is the most important process driving contaminant movement in the subsurface (Wiedemeier et al. 1995). The groundwater linear velocity in the parallel direction to groundwater flow caused by advection is specified by following equation (Zheng and Bennett 2002):

$$V_g = \frac{K}{P_e} \frac{dh}{dl} \dots\dots\dots \text{Equation 2.9}$$

Where, V_g = linear groundwater velocity (L/T),

K = hydraulic conductivity,

P_e = effective porosity (L^3/L^3)

dh/dl = hydraulic gradient (L/L)

The advection process depends mainly on aquifer properties such as hydraulic conductivity, effective porosity and hydraulic gradient and independent of contaminant properties. This study revealed that Bassendean Sand aquifer, Safety Bay Sand aquifer and Tamala Limestone aquifer are permeable and the hydraulic properties of these three aquifers are very different. Bassendean Sand has estimated hydraulic conductivity ranges from 10 m/day to 100 m/day (Davidson 1995), the Safety Bay Sand has a hydraulic conductivity ranging from 1 m/day to 50 m/day (Davidson 1995), and the Tamala Limestone has estimated hydraulic conductivity ranges from 100 to 1000 m/day (Davidson 1995; Smith et al. 2003). From these hydraulic conductivity values estimation and those for porosity and hydraulic gradients, groundwater velocities was estimated by the above equation 2.9 and it was found that Tamala Limestone has much higher groundwater velocities than the other two types of soil. Therefore, it can be clearly seen that the migration of the contaminants in Tamala Limestone can be more rapid than either the Safety Bay Sand or the Bassendean Sand (Trefry et al. 2006).

Thus, the one dimensional advective, dispersive and sorptive equation with only advection component is described by the following partial differential equation 2.10.

$$\frac{\partial C}{\partial t} = -V_x \frac{\partial C}{\partial x} \dots\dots\dots \text{Equation 2.10}$$

Where, C = concentration of contaminant (M/L³)

t = time (T)

V_x = average linear groundwater velocity in x-direction (L/T)

x = distance along the flowpath (L)

The above equation 2.10 is only considering the advective solute transport. Advective solute transport is the main process for the contaminant migration in some cases. However, because of other physical and chemical processes, this equation must be modified to get the accurate mathematical description of the contaminant transport.

2.8.2 Hydrodynamic Dispersion and Dispersivity

Hydrodynamic dispersion is the mechanism of the contaminant transport process whereby a contaminant plume spreads out in longitudinal and transverse directions of the contaminant plume movement (Wiedemeier et al. 1995; Lovanh et al. 2000). Dispersion is important as it dilutes the contaminant plume by uncontaminated groundwater (Lovanh et al. 2000). Dispersion is traditionally called hydrodynamic dispersion in groundwater hydrology (Bear 1972) which has two main components: mechanical dispersion and molecular diffusion (Fetter 1999). Hydrodynamic dispersion, denoted by D , is the summation of molecular diffusion and mechanical dispersion.

Mechanical dispersion, denoted by D' , is the principal mechanism causing hydrodynamic dispersion at mean groundwater velocities and it is given by mechanical dispersion coefficient. Mechanical dispersion is described by the following equation (Fetter 1999):

$$D' = \alpha_x V_x \dots\dots\dots \text{Equation 2.11}$$

Where,

V_x = mean groundwater velocity (L/T)

α_x = dispersivity (L)

There are two fundamental types of mechanical dispersion: longitudinal dispersion (horizontal) and transverse (vertical) dispersion. Longitudinal and transverse dispersions are dependent upon horizontal and vertical variations in permeability. The magnitude of dispersion increases with increasing aquifer anisotropy and heterogeneity (Vance 1997; Prommer et al. 2003).

Longitudinal dispersion is the contaminant spreading in a parallel direction to the direction of groundwater flow. Longitudinal dispersion (α_L) is caused by the variations in pore size, tortuosity, and friction in the pore throat (Wiedemeier et al. 1995). Transverse dispersion is the scattering of a solute in perpendicular directions to the direction of groundwater flow. Transverse dispersion (α_T) (horizontal

transverse dispersion (α_{TH}) and vertical transverse dispersion (α_{TV}) occurs because of the tortuosity of the permeable medium which causes flow paths to branch out from the contaminant plume centerline (Wiedemeier et al. 1995; Vance 1997).

The mechanical dispersion coefficient has three components along three perpendicular coordinates (x, y, z), which are assumed to be proportional to three groundwater velocity components V_x , V_y and V_z and is given by the following equations (Zheng and Bennet 2002):

$$D_L' = \alpha_L V_x \dots\dots\dots \text{Equation 2.12a}$$

$$D_{TH}' = \alpha_{TH} V_y \dots\dots\dots \text{Equation 2.12b}$$

$$D_{TV}' = \alpha_{TV} V_z \dots\dots\dots \text{Equation 2.12c}$$

Where, α_L , α_{TH} , and α_{TV} are the longitudinal dispersivity, horizontal transverse dispersivity and vertical transverse dispersivity, respectively.

Dispersion parameter are mostly estimated from tracer test, or less reliably, at the larger (>250m) scale, by model fitting to the existing contaminant plume (Rivett et al. n.d.). There are several other empirical relationships to estimate the longitudinal dispersivity at field scale. The first method is developed by Gelhar (1993) and is given by 10 percentage of mean contaminant plume travel distance and represented by following equation. However, this empirical relationship gives an approximate value of dispersivity.

$$\alpha_L = 0.1 L \dots\dots\dots \text{Equation 2.13}$$

Where, L is the average travel distance of the contaminant plume.

Neuman (1990) anticipated a superior scale-dependent, empirical method for estimating longitudinal dispersivity which is given by:

$$\alpha_L = 0.0175 L^{1.46} \quad (\text{For } L < 100\text{m}) \dots\dots\dots \text{Equation 2.14a}$$

$$\alpha_L = 0.32 L^{0.83} \quad (\text{For } L > 100\text{m}) \dots\dots\dots \text{Equation 2.14b}$$

This method (Neuman 1990) gives more accurate measurement of α_L (longitudinal dispersivity) than that of Gelhar (1993). There are other methods that have been proposed for measurement of α_L . However, some of these methods require extensive data, and the additional cost coupled to obtain these data is not always reasonable in terms of improved simulation accuracy (Lovanh et al. 2000).

In most of the cases, the horizontal transverse dispersivity is set to be 10% to 30% of the longitudinal dispersivity and the vertical transverse dispersivity is set to be 1% to 5% of the longitudinal dispersivity (ASTM 1994; Vance 1997; Lovanh et al. 2000).

Molecular diffusion can turn out to be the leading mechanism of hydrodynamic dispersion at extremely low groundwater velocities and it is caused by the difference in concentration and can be denoted by D_m . However, molecular diffusion coefficients are small in groundwater which shows that the time required for diffusion to equilibrate the concentrations for small travel distances may be decades or centuries (Trefry et al. 2006). Thus, the effects of molecular diffusion are generally neglected for groundwater studies over mechanical dispersion (Lovanh et al. 2000; Wiedemeier et al. 1995; Davis et al. 1993).

By considering advection and dispersion, one dimensional contaminant transport equation represented by:

$$\frac{\partial C}{\partial t} = D_x \frac{\partial^2 C}{\partial x^2} - V_x \frac{\partial C}{\partial x} \dots\dots\dots \text{Equation 2.15}$$

Where,

D_x = hydrodynamic dispersion in x-direction (L^2/T)

To summarize this further, estimation of a dispersion coefficient for contaminant in a heterogeneous aquifer is a difficult process and it is mainly affected by variations in groundwater velocity at different scales which is caused by the variation in hydraulic conductivity of the aquifer (Lovanh et al. 2000). Therefore, it is difficult to calculate the accurate value of dispersion coefficient without detailed knowledge of these

variations and Lovanh et al. (2000) recommended that Neuman's (1990) method is useful to estimate the dispersion coefficient when the limited hydraulic conductivity data is available.

In the near-homogeneous sand (Bassendean Sand and Safety Bay Sand) unconfined Superficial aquifer of the Perth, dispersion has been estimated to be small with dispersivities in millimeters or centimeters (Thierrin, Davis and Barber 1995; Prommer et al. 2003; Trefry et al. 2006). However, it is extremely difficult to measure a dispersion coefficient in the Tamala Limestone of Perth unconfined Superficial aquifer because of heterogeneity of the aquifer matrix (Trefry et al. 2006).

2.8.3 Sorption

Dissolve contaminants removed from the groundwater by sorption process can adhere on to the soil and/or aquifer matrix (Fetter 2001). Normally, organic contaminants may be sorbed into the organic matter of the aquifer matrix which may provide coating to the soil grains (Wiedemeier et al. 1995; Trefry et al. 2006). Metals and other chemical contaminants may not attach to the organic carbon contents but may adsorb or exchange with mineral phases present in the aquifer matrix (Trefry et al. 2006). Thus, these processes lead to slow down of the contaminants flow relative to the average advective groundwater velocity and reduce the concentration of the dissolved contaminants concentration in groundwater (Wiedemeier et al. 1995; Zheng and Bennett 2002; Trefry et al. 2006). Sorption of dissolved contaminants is mainly caused by the covalent bonding between the chemicals and aquifer matrix and hydrophobic bonding which may control the fate of many contaminants (Wiedemeier et al. 1995). Organic carbon content and clay minerals are the two main components of an aquifer which have the greatest influence on the sorption effect (Wiedemeier et al. 1995).

Apart from of the sorption process, it is also important to estimate the amount of sorption of the dissolved contaminants on to the given aquifer matrix and it is presented by sorption isotherms (Lovanh et al. 2000). There are mainly three types of sorption isotherm: linear isotherm, the Freundlich isotherm and the Langmuir isotherm (Zheng and Wang 1999).

The linear sorption isotherm is the simplest equilibrium isotherm and it is applicable for a dissolved species that is present at a concentration less than one half of its solubility (Lyman, Reidy and Levy 1992). The linear isotherm was used in this study because this is a suitable assumption for BTEX contaminants partitioning from fuel mixtures into the groundwater (Wiedemeier et al. 1995). The linear sorption isotherm is represented by following equation 2.16:

$$C_s = K_d C_d \dots\dots\dots\text{Equation 2.16}$$

Where,

C_s = sorbed concentration is represented by mass of contaminants to mass of the soil, in $\mu\text{g/g}$

K_d = distribution coefficient, slope of the linear isotherm, in ml/g

C_d = dissolved concentration is represented by mass of contaminants to volume of the solution, in $\mu\text{g/ml}$

As discussed earlier, sorption process leads to slow down of the dissolved contaminants transport velocity in groundwater. The retardation coefficient is used to measure the retarded contaminant transport velocity. The retardation coefficient for linear sorption process is estimated by the following relationship (Thorbjarnarson, Inami and Girty 2002).

$$R = 1 + \frac{\rho_b K_d}{P_e} \dots\dots\dots\text{Equation 2.17}$$

Where,

R = retardation coefficient (dimensionless)

ρ_b = aquifer bulk density (M/L^3)

K_d = distribution coefficient (L^3/M)

P_e = aquifer effective porosity (dimensionless)

Thus, the retarded contaminant transport velocity is determined by the following equation:

$$R = \frac{V_g}{V_c} \dots\dots\dots \text{Equation 2.18}$$

Where,

R = retardation coefficient (dimensionless)

V_g = groundwater velocity (L/T)

V_c = contaminant transport velocity (L/T)

Wiedemeier (1995) described two methods to estimate the amount of sorption (and retardation) and quantified the distribution coefficient for a given aquifer/contaminant system which are described in the following section.

In the first method, the distribution coefficient can be estimated by using the fraction of organic carbon content of the aquifer matrix (F_{oc}) and organic chemical partition coefficient for the contaminants (K_{oc}). In the second method, the distribution coefficient can be determined by conducting batch and column experiments. K_{oc} values for contaminants and fraction of organic carbon content for aquifer matrix can be found out from literature.

Thus, the value of K_{oc} and F_{oc} must be estimated from one of the above method. Typical ranges of F_{oc} for Perth region, Bassendean Sand of the Superficial unconfined aquifer contains high percentage of organic carbon matter approximately about 0.6% to 0.08% (Thierrin et al. 1993), whereas Tamala Limestone and Safety Bay Sand contains approximately 0.01 to 0.03% of organic carbon contents (Johnston et al. 1998).

From known values of organic chemical partition coefficient (K_{oc}) and fraction of organic carbon matter present in the aquifer matrix (F_{oc}), the distribution coefficient can be found out by the following relationship (Karickhoff, Brown and Scott 1979).

$$K_d = K_{oc} F_{oc} \dots\dots\dots \text{Equation 2.19}$$

Where,

K_{oc} = organic carbon partition coefficient (L^3/M)

F_{oc} = fraction of organic carbon matter in soil (mg of organic carbon/mg of soil)

The distribution coefficient is used to measure the sorption capacity of organic contaminants to be sorbed to the aquifer and it is mainly controlled by two parameters: the dissolved contaminants hydrophobicity and available surface area of the aquifer for sorption. Therefore, the distribution coefficient will be different for the same contaminants with the composition of the aquifer matrix.

Thus, the one dimensional contaminant transport equation with advection, dispersion and sorption (linear equilibrium sorption isotherm) is given by following expression.

$$R \frac{\partial C}{\partial t} = D_x \frac{\partial^2 C}{\partial x^2} - V_x \frac{\partial C}{\partial x} \dots\dots\dots \text{Equation 2.20}$$

2.8.4 Chemical Reactions

Chemical reactions may influence the migration and toxicity of the contaminants in groundwater system. The chemical reactions of contaminants in groundwater are classified as the abiotic or biotic chemical reaction. Abiotic chemical reactions occur without any need of microbial activity and they may consist of reaction processes such as hydrolysis, precipitation, elimination, complexation, substitution etc. that transform contaminants to other contaminants and potentially change their state or phase (Rivett et al. n.d.). Biotic chemical reaction or biodegradation is mediated by microbial activity. Biodegradation of the contaminants in groundwater is the process which transformed the contaminants to other less toxic chemicals e.g., biodegradation of BTEX is transformed in to the water, carbon dioxide and methane (Wiedemeier et al. 1995). Biodegradation of the contaminants (generally organic contaminants) in groundwater may reduce the contaminant concentration and slow down the contaminant transport velocity (Wiedemeier et al. 1995; Trefry et al. 2006). Biodegradation of the organic contaminants mainly include Redox reactions which involves transfer and electron acceptance between two chemicals.

The degradation of contaminants in groundwater can be expressed as a first order decay rate (Ohio EPA 2007) with respect to the contaminant concentration and it is given by following expression:

$$\frac{dC}{dt} = -\lambda C \dots\dots\dots \text{Equation 2.21}$$

Where,

λ = rate of biodegradation (T^{-1})

dC/dt = change in concentration with time

Biodegradation rate of contaminants for first order model can be estimated by the contaminants half life ($t_{1/2}$) which is described as the time required to reduce the contaminant concentration by one half and is given by (Lovanh et al. 2000):

$$t_{1/2} = \frac{\ln 2}{\lambda} \dots\dots\dots \text{Equation 2.22}$$

Where, $t_{1/2}$ = half life of contaminants (T)

The biodegradation rate is mainly dependent upon the chemical type, aquifer redox conditions, microbial populations of present and their activity towards contaminants present (Rivett 2003), etc. Thus, the values for biodegradation rate are not necessarily constant. Several researchers show that the biodegradation rate (λ) values have been observed for benzene at different sites, ranging over several orders of magnitude from 0 to 0.087 day^{-1} (Alvarez, Anid and Vogel 1991, Alvarez, Cronkhite and Hunt 1998; Rifai et al. 1995; Howard 1991). Therefore, λ should not be extrapolated from the literature. Rather, considerable care must be exercised in its determination to avoid over-predicting or under-predicting actual biodegradation rates and plume behaviour (Lovanh et al. 2000). Due to high variability in biodegradation rate, this study neglected the effect of biodegradation of BTEX compound in contaminant transport model.

Thus, the one dimensional contaminant transport equation with advection, dispersion and sorption (linear equilibrium sorption isotherm) with biodegradation is given by following expression.

$$\frac{\partial C}{\partial t} = \frac{D_x}{R} \frac{\partial^2 C}{\partial x^2} - \frac{V_x}{R} \frac{\partial C}{\partial x} - \lambda C \dots\dots\dots \text{Equation 2.23}$$

Equation 2.23 demonstrates how the processes of advection, dispersion, sorption, and biodegradation are incorporated to explain the movement of solute in the saturated zone in one dimension.

The three dimensional contaminant transport equation were derived from the continuity (conservation of mass) equation, which states that the rate of change of contaminant mass within a unit volume of porous media is equal to the flux of contaminant into the unit volume minus the flux out of the unit volume (Freeze and Cherry 1979).

Processes governing flux into the unit volume include advection and hydrodynamic dispersion (including mechanical dispersion and diffusion). Processes governing flux out of the unit volume include advection, hydrodynamic dispersion, sorption, and chemical reactions. Stated mathematically, the change in solute concentration is given as follow.

$$\text{Change in solute concentration} = \text{Flux in} - \text{Flux out} \pm \text{Reaction}$$

The initial condition of the system can be assumed as $C=C_{\text{initial}}$ for the source of the contaminant, $C=0$ for the remaining subsurface region.

Whereas the boundary conditions at each section of the subsurface region can be assumed as Dankwarts boundary condition at the inlet of the section and Zero Rate change of the concentration at the exit of the section (Zheng and Bennet 2002).

Therefore, the mathematical expression for contaminant fate and transport in groundwater can be rewritten conveniently in the three dimensional form as follow (Domenico and Schwartz 1998; Ewing et al. 2001):

$$R \frac{\partial C}{\partial t} = \nabla(D\nabla C) - \nabla(VC) + f \dots\dots\dots \text{Equation 2.24}$$

In this partial differential equation, $C(x, y, z; t)$ is the dissolved contaminant concentration at a particular point in the aquifer, R is the retardation coefficient, $D = D(x, y, z; t; C)$ is the dispersion coefficient tensor, $V = V(x, y, z; t; C)$ is the groundwater velocity at a given point in the aquifer, $f = f(x, y, z; t; C)$ is a forcing function associated with recharge, biological and chemical activities and, $\nabla = \frac{\partial}{\partial x} + \frac{\partial}{\partial y} + \frac{\partial}{\partial z}$, is the derivative operator in three dimension cartesian coordinates. The 3-D contaminant transport in groundwater equation 2.24 states that the change in concentration of contaminant with time is a function of the amount of contaminant plume spreading in every direction (first term on the right hand side first term is a dispersion term), the amount of contaminant that moves with bulk groundwater flow (second term as advection), and the quantity of contaminant degradation (last term on the right, sink term).

This type of partial differential equation of groundwater flow and contaminant transport in groundwater can be solved by analytical model or numerical model which is discussed in chapter 3.

CHAPTER 3

NUMERICAL MODELS FOR GROUNDWATER FLOW AND CONTAMINANT TRANSPORT MODELING

3.1 Introduction

The complex three dimension partial differential equations 2.8 and 2.24 describing the groundwater flow and contaminant transport can be solved mathematically using either analytical or numerical methods (Mohan and Muthukumaran 2004). The advantages of an analytical method are that it generally provides a precise solution of the governing equation. This method is efficient and simple to get the solution of the governing equation. Many analytical methods have been developed for the groundwater flow and solute transport equations. To obtain the exact analytical solution of the partial differential equations require that the input properties and boundaries of the groundwater flow and contaminant transport system be highly and unrealistically ideal (Konikow and Reilly 1999). For most field problems, the mathematical benefits of obtaining an exact analytical solution are probably over-predicted or under-predicted by the errors introduced by the simplifying assumptions of the complex field environment that are required to apply the analytical model (Mohan and Muthukumaran 2004). On the other hand, for problems where the simplified analytical models are inadequate, the solution of partial differential equations of the groundwater flow and contaminant transport can be approximated by numerical models.

In numerical methods, the continuous hydrogeologic domain is replaced by a discretized domain consisting of an array of nodes and associated grid blocks or elements. Thus, the continuous differential equation, which defines aquifer hydraulic parameters or contaminant parameters everywhere in the system, is replaced by a finite number of algebraic equations that define the aquifer hydraulic parameters or contaminant parameters at specific points. Matrix techniques are generally used for

solving this system of algebraic equations. This approach constitutes a numerical model.

There are mainly five numerical methods described by Anderson and Woessner (1992) for solving groundwater flow and contaminant transport problems. These methods are finite difference method, finite element method, integrated finite difference method, the boundary integral equation and analytical elements methods. The boundary integral equation and analytical elements are relatively new techniques and are not widely used for groundwater flow and contaminant transport modeling, the integrated finite difference methods are closely related to finite element methods. However, the finite difference and finite element methods are the most popular numerical techniques for solving groundwater flow and contaminant transport problems (Anderson and Woessner 1992; Konikow and Reilly 1999; Mohan and Muthukumaran 2004). The main function of both types of the numerical techniques are that they subdivide the area of interest into a grid to a small number of subareas such as cells or elements that are associated with node points (either at the centres of peripheries of the subareas).

Analytical solution of the complex three dimensional partial differential equations 2.8 and 2.24 for groundwater flow and contaminant transport exists only for relatively simple cases. Thus, for solving more complex and realistic conditions, e.g., involving aquifers heterogeneity, transient flow and contaminant transport boundary conditions, etc., numerical techniques such as the finite difference and the finite element methods are required (Prommer et al. 2003; Lovanh et al. 2000). Another advantage of numerical models over other methods is that most codes are capable of simulating contaminant sources that vary over time, allowing simulation of scenarios including source reduction through weathering or through engineered solutions such as remediation technology. Thus, the numerical models would be the most appropriate tool for predicting contaminant fate and transport in groundwater and are also useful for the groundwater management studies.

The choice between the finite element and the finite difference methods generally depends on the problem to be solved and the following section briefly describes both numerical methods.

3.2 Finite Difference and Finite Element Methods

Both these numerical methods use the discretization of the time and space dimensions and allow the continuous boundary value problem for the solution of the partial differential equation to be reduced to the simultaneous solution of a set of algebraic equations and can be solved by direct matrix methods or iterative methods (Konikow and Reilly 1999).

The finite difference methods are conceptually and mathematically very simple and are easy to program for a computer. In general, they are keyed to a relatively uncomplicated, rectangular grid. Finite difference methods solve the groundwater flow and solute transport equation by dividing the problem area into rectangular grid and are widely accepted by the groundwater community (Pinder 2002; Anderson and Woessner 1992).

Finite element methods are mathematically abstract and more difficult to code. The finite element methods could accommodate triangular and deformed rectangular meshes (Pinder 2002). A major advantage of the finite-element methods is the flexibility of the finite-element grid, which allows a close spatial approximation of irregular boundaries of the aquifer and of parameter zones within the aquifer when they are considered. However, the construction and specification of an input data set is much more difficult for an irregular finite-element grid than for a regular rectangular finite-difference grid (Konikow and Reilly 1999). Therefore, the finite difference method is widely used over finite element method for construction of most of the groundwater flow and contaminant transport model. Based on finite difference and finite element methods, three main types of groundwater flow codes have been developed which are represented in Table 3.1.

Name	Type
MODFLOW	3D Finite Difference
PLASM	2D Finite Difference
AQUIFEM-1	2D Finite Element

Table 3.1: Numerical Groundwater Flow Models (Anderson and Woessner 1992)

MODFLOW and PLASM are widely used codes for groundwater flow modelling. AQUIFEM-1 has not gained wide acceptance because it is more difficult to code (Anderson and Woessner 1992). In recent years, most of the groundwater flow models are based on MODFLOW and it is a modular three dimensional finite difference model which is published by the U.S. Geological Survey (McDonald and Harbaugh 1988). MODFLOW has turned out to be the worldwide standard groundwater flow model because of its ability to simulate a wide variety of hydrogeologic systems and its extensive publicly available documentation. MODFLOW is generally used to simulate systems for groundwater supply, containment remediation and mine dewatering.

MODFLOW is a computer model which can simulate saturated three dimensional groundwater flows through an aquifer, including confined, confining and unconfined layers (Harbaugh et al. 2000). It numerically solves the three dimensional groundwater flow equations using a block centered finite difference approach. Steady-state and transient flow can be simulated, including a variety of processes in the model, such as flow to wells, recharge, flow through riverbeds, flow to drains and evapotranspiration (Harbaugh et al. 2000).

This type of numerical flow model can be used for the delineation of catchments, capture zones (e.g., of extraction wells) and for the development of remediation measures. Add-on packages of flow models that simulate purely advective transport can be used for the prediction of contaminants flow path and to estimate how fast the leading edge of a contamination would migrate in a non-reactive case without the occurrence of biodegradation and sorption processes.

Modelling packages, PMPATH (Chiang and Kinzelbach 2001) or MODPATH (Pollock 1994), allow predictions of the contaminant flow path and travel times of non-reactive contaminants using particle-tracking algorithms.

3.3 Reactive and Biodegradation Transport Model

A number of reactive and biodegradation models, such as MT3DMS and RT3D, are widely used to simulate the contaminated groundwater transport and they are described in the following section.

MT3DMS, Modular Three Dimensional Multispecies Transport Model developed by Zheng and Wang (1999), uses the finite difference method to solve the 3D advective, dispersive and reactive transport model. MT3DMS is the new version of MT3D (Modular Three Dimensional Single Species Transport Model) which is developed to provide better numerical efficiency to the model.

MT3DMS is capable of modeling advection in complex steady-state and transient flow fields, anisotropic dispersion, first-order and zero-order irreversible decay and production reactions, and linear and nonlinear sorption. It can also handle Monod reactions, and daughter products. This enables MT3DMS to simulate coupled hydrological transport of multiple chemical species and the chemical reactions among these species and assess the natural attenuation within a contaminant plume (Zheng and Wang 1999).

RT3D, Reactive Transport in Three Dimensions, is a finite-difference numerical code for simulation of three-dimensional, multi-species reactive transport through saturated aquifer (Clement 2001). It comes with a number of pre-programmed reaction kinetics modules, such as biodegradation of oxidisable contaminants consuming one or more electron acceptors, but also has the flexibility to utilize reaction mechanisms of any complexity that are defined by the user in order to adapt the numerical model to a site-specific conceptual hydrochemical model (Clement 2001; WHI 2006).

RT3D can be applied to scenarios involving contaminants such as heavy metals, petroleum hydrocarbons such as BTEX, and/or chlorinated solvents such as PCE, TCE etc. It can also be used to simulate active groundwater treatment processes (remediation) including air-sparging, bio-sparging, chemical oxidation and permeable reactive barriers (Clement 2001). With the growing popularity of natural

attenuation solutions for contaminated sites around the world, RT3D can also be used to predict fate and contaminant transport of groundwater plumes.

3.4 Biogeochemical Modeling

A range of three dimensional multispecies contaminant transport numerical models, e.g., PHT3D (Prommer 2002), PHAST (Parkhurst et al. 2004), MIN3P (Mayer 1999) etc., is available that can simultaneously account for both biodegradation and geochemical reactions. However, PHT3D is widely accepted than the other biogeochemical numerical models because it comes with wide range of kinetically and equilibrium controlled biochemical processes.

PHT3D is a multi-component transport model for three-dimensional reactive transport in saturated porous media. It couples the two existing and widely used numerical models, MT3DMS for the simulation of three-dimensional advective-dispersive, reactive multispecies transport (Zheng and Wang 1999) and the geochemical model PHREEQC (Parkhurst and Appelo 1999) for the quantification of reactive processes. It is applied to simulate the geochemical evolution within the aquifers as well as their remediation which includes the natural and enhanced attenuation and remediation of organic and inorganic contaminants. It can also be used to simulate a wide range of equilibrium and kinetically controlled biochemical processes, such as Redox reaction, ion exchange reaction, mineral precipitation and aqueous complexation which are briefly discussed by Prommer, Barry and Zheng (2003). Applications of this type of numerical models are: trace metal remediation by in situ reactive zones, remediation of the contaminated sites with aromatic and chlorinated hydrocarbon spills (Prommer et al. 2003), remediation of chlorinated and brominated contaminated groundwater modeling (Cohen et al. 2008), modeling the fate of oxidisable organic contaminants such as petroleum hydrocarbons (Prommer, Barry and Davis 2002, Barry et al. 2002) etc.

3.5 Preprocessing and Postprocessing Tools for Groundwater Flow and Contaminant Transport Modeling Study

The process of preparing and assembling input data for a computer based groundwater flow and contaminant transport numerical model is termed as preprocessing, whereas the process of examining and presenting simulation results is referred to as postprocessing.

For regular model applications, preprocessing and postprocessing of model input and output data might comprise the majority of effort for a modeling exercise. Geographic Information System or GIS is one of these tools which is useful for storage and display of data characterized by geographic coordinates. Over the past few years significant advances have been made in incorporating GIS features into the most commonly used graphical user interfaces (GUIs), as well as with respect to rapid three dimensional visualization of the simulation results (Zheng and Bennet 2002). Most commonly used preprocessing and postprocessing software packages for MODFLOW based groundwater flow and contaminant transport modelings are: Visual MODFLOW Premium (WHI 2006), Groundwater Vistas (Rumbaugh and Rumbaugh 2004), Processing MODFLOW (Chiang and Kinzelbach 2001). Selection of the software packages for MODFLOW based groundwater flow and transport modeling depends on a range of factors. However, Visual MODFLOW Premium is used in this study as a preprocessing and postprocessing software package for building of the groundwater flow and contaminant transport model because it is the most comprehensive and user friendly tool for modeling environment for practical applications in three dimensional groundwater flow and contaminant transport model simulation. One of the major benefits of the Visual MODFLOW Premium over other preprocessing and postprocessing software packages is that it is not limited to consistent units for length and time for all model parameters.

Visual MODFLOW Premium comes with a professional graphical preprocessor and postprocessor, three dimensional finite difference groundwater flow models which support all versions of MODFLOW, such as MODFLOW-88, MODFLOW-96 and MODFLOW 2000. It also includes the solute transport models MT3D which includes reactive and biodegradation transport models and biogeochemical models,

such as MT3Dv1.5, MT3DMS, MT3D99, RT3Dv1.0, RT3D 2.5 and PHT3D and SEAWAT-2000, the particle tracking model MODPATH that is useful for estimation of the travel time of the contaminated groundwater from its sources to discharge points (Pollock 1994) and Zone Budget for sub regional water budget calculation. Visual MODFLOW Premium comes with Visual MODFLOW 3D Explorer which is an advanced 3D visualization and animation component for displaying and presenting groundwater flow, pathlines, contaminant transport modeling data, and the simulation results using a variety of 3D graphical formats that includes 3D volumetric shapes for representing model property zones, boundary conditions and contaminant plumes, output times are synchronized to animate the combine transient flow and transport results, animation sequence to record in an .AVI file etc.(WHI 2006). It also includes parameter estimation tool WinPEST. Thus, Visual MODFLOW Premium software package gives the most complete and powerful graphical modeling environment for groundwater flow and contaminant transport model.

3.6 Application of Visual MODFLOW Premium

Visual MODFLOW Premium is widely used for various applications in construction of the three dimensional groundwater and contaminants fate and transport model. The following section illustrates the general application of Visual MODFLOW Premium software package.

Visual MODFLOW Premium can be used to analyse and predict the risk associated with using natural attenuation of the contaminated groundwater as a remediation scheme (Prommer, Barry and Davis 2002; WHI 2006).

It can be used to characterize the groundwater condition and to estimate the impact of potential groundwater contaminants on surface water body, such as lakes and rivers.

It can be applied to perform simulations of prospective ISCO (In Situ Chemical Oxidation) injection scenarios to support the engineering design of full scale contaminated groundwater remediation (Thomas, Brynes and Williams 2006).

It can be widely used to evaluate groundwater remediation techniques such as conventional pump and treat method, permeable reactive barrier etc. and are also useful for estimation of the reductive dechlorination of the chlorinated aliphatic hydrocarbons such as PCE, DCE and TCE in groundwater (WHI 2006).

Visual MODFLOW Premium can predict impacts from saltwater intrusion due to over pumping in coastal regions on groundwater and also useful for groundwater and surface water source management, e.g. delineate well capture zones for municipal drinking water supplies, design and optimize pumping well locations for dewatering projects.

The following section describes two examples of groundwater flow and contaminant transport modelling study by using Visual MODFLOW Premium.

The first PRB installed in Italy (Molfetta and Sethi 2005) for the remediation of chlorinated hydrocarbons such as PCE, TCE, and DCE from the groundwater, promises to reduce the contaminants concentration under the required limitations.

The preliminary laboratory column and field tests suggest a half life of 2 to 25 hrs for various chlorinated hydrocarbon compounds found in groundwater. Based on the preliminary information a Zero Valent Iron (ZVI) based PRB of 120.37*13.80 m was installed in the contaminated area of concern. The dimensioning and position of the PRB was simulated using Visual MODFLOW. The simulated and the laboratory results gave a positive indication, however the actual test results were not published by the authors.

Another study conducted by Scott and Folkes (2000) points about the deficient design of the Permeable Reactive Barrier resulting in improper treatment of the contaminated ground water. They further suggested that modelling the groundwater hydraulic and the contaminants transport helps in understanding the underground scenario, helping in a better design of the barrier. They reported a simulated barrier application for the Groundwater treatment in a site located in Denver, within the region of Denver Basin. The groundwater plume contaminated with DCE, TCE and TCA was treated as it flows through the gate of the Funnel and Gate system. The simulated result of a single Gate system showed approximately 31% of the contaminated plume bypassing the side walls due to the increased head by the installation of the barrier. To overcome the problem, the simulation was redone using 2 and 3 gate system which showed promising results and it is presented in Figure 3.1. The bypass from the reactive walls was around 2.9 m³/day for a single gate system which reduced to 0.9 m³/day for the 2 gate system and further reduced to 0.65 m³/day for the 3 gate system.

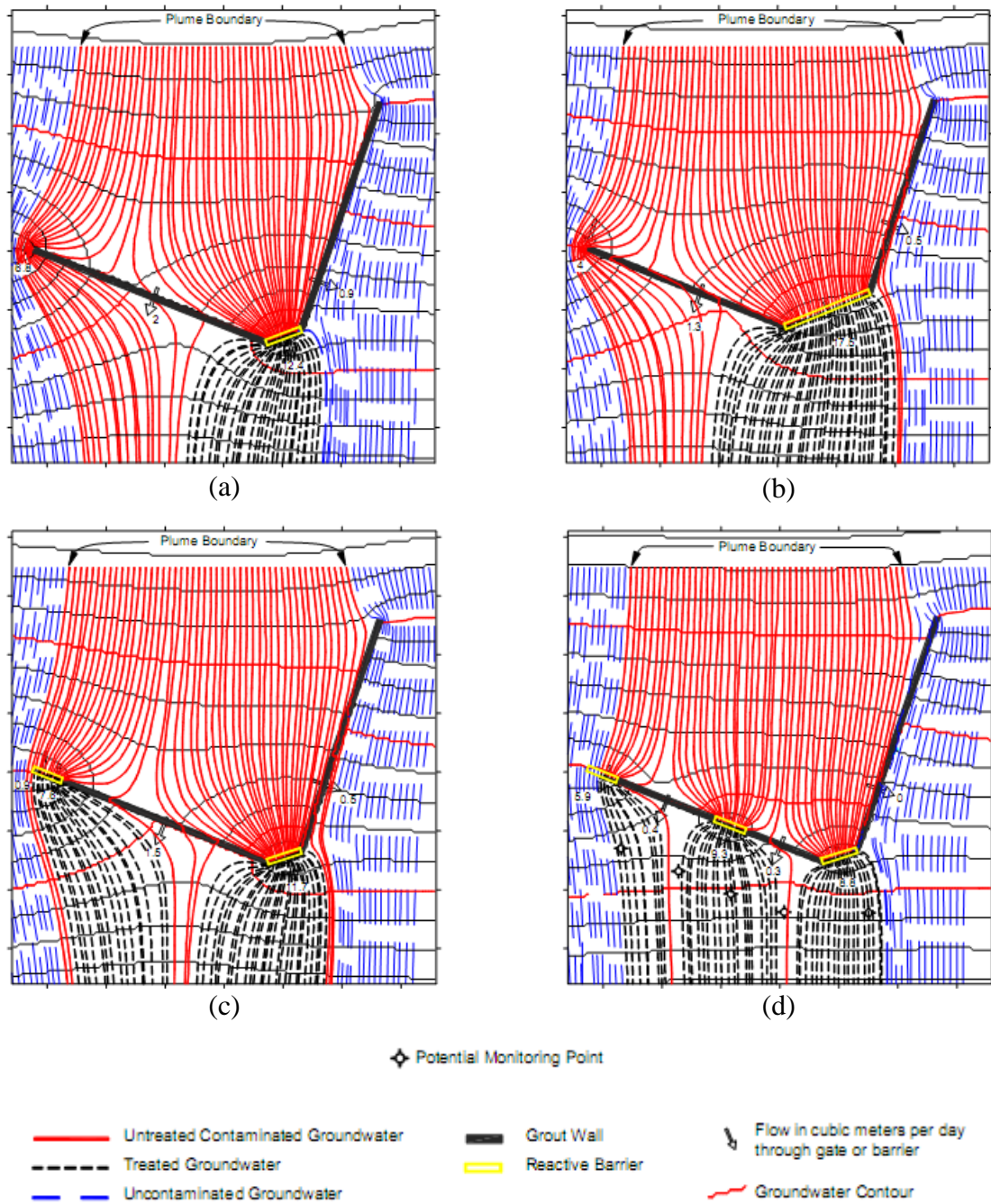


Figure 3.1: Selected Simulation Results Based on (a) One Gate Permeable Reactive Barrier System with Total Gate Length of 15 m (b) One Gate Permeable Reactive Barrier System with Total Gate Length of 45 m (c) Two Gate Permeable Reactive Barrier System with Total G Gate Length of 30 m (d) Three Gate Permeable Reactive Barrier System with Total Gate Length of 45 m (Scott and Folkes 2000)

3.7 General Methodology for Development of Groundwater Flow and Contaminant Transport Model

This section presents an overview of the steps necessary for development of a groundwater flow and solute transport model.

3.7.1 Model Conceptualization

Conceptual model is the first step in the modeling process for development of the groundwater flow and contaminant transport model. It is a 3D representation of the groundwater flow and contaminant transport model based on available geological, hydrogeological and geochemical data for the contaminated sites. Successful conceptual model development involves the collection of contaminated site specific database such as aquifer hydrogeological properties, groundwater flow properties, contaminant source and contaminant transport properties etc. The purpose of the conceptual model is the integration of available site specific data into a coherent representation of the system to be modeled (Zheng and Bennett 2002). After development of the conceptual model, it is helpful for model selection and to build up the numerical contaminant transport model.

3.7.2 Model Calibration

A groundwater flow and contaminant transport model is proficient for accurately predicting the future extent and concentration of a contaminant plume. The model input parameters such as hydraulic conductivity, hydraulic head gradient, effective porosity, recharge, boundary conditions, dispersivity, soil sorption coefficient, and biodegradation rate are required to calibrate the model which can be used for comparisons of the simulated output to observed data (Kumar n.d.).

3.7.3 Model Sensitivity Analysis

Any groundwater model is influenced by uncertainty due to the inability to define the exact spatial and temporal distribution of aquifer hydrogeological and chemical parameters at the actual contaminated sites (Konikow and Reilly 1999). Thus, the sensitivity analysis should be performed by varying model input parameters over

reasonable ranges to obtain the effect of uncertainties on the model output. Sensitivity analysis should also be performed on all models to evaluate the reasonableness of model predictions.

3.7.4 Model Validation

A calibrated model uses chosen values of model input parameters such as hydrogeological and contaminant transport parameters, sources and sinks and boundary conditions to match historical site conditions (Zheng and Bennett 2002). After the model has successfully reproduced measured changes in field conditions, it is ready for predictive simulations.

3.7.5 Model Prediction

After the groundwater flow and solute transport model have been calibrated and the sensitivity analyses have been performed, the model is useful for prediction of the fate and transport of contaminants. Finally, this model can be used to assess different remediation techniques (Zheng and Bennett 2002).

3.7.6 Model Performance Monitoring

Groundwater flow and contaminant transport models are helpful for the prediction of the movement and concentrations of contaminants in groundwater. Model performance monitoring is required to evaluate future contaminated site conditions with model predictions because small errors in the model can result in changes to model output (Konikow and Reilly 1999).

CHAPTER 4

SCENARIO MODELING OF BTEX TRANSPORT IN BASSENDEAN SAND, SAFETY BAY SAND AND TAMALA LIMESTONE

4.1 Introduction

The objective of this study is to build a groundwater flow and contaminant transport model for Perth unconfined (Superficial) aquifer. This study covers three major types of soil found in the Perth region i.e. Bassendean Sand, Safety Bay Sand and Tamala Limestone from which most of the groundwater is extracted (Davidson 1995). The characteristics of the Superficial aquifer, focusing on the Bassendean Sand, Safety Bay Sand and Tamala Limestone, will be described in the following sections. This model shows the potential impact of the fuel (e.g. BTEX) contamination on the supply wells.

The majority of the research work was involved in exploring and finding the specific site hydrogeology and contaminant data. CSIRO, Department of Environment and Conservation (DEC) of WA and published journal articles were used for data collection. Given the fact that, the majority of the site specific data are well classified due to confidentiality issues, a hypothetical model which closely resembles the actual scenario was selected for scenario modeling.

4.2 Model Description

A hypothetical model was selected in this study, where petrol has leaked to groundwater from a subsurface fuel storage tank at a service station. Petrol contains various types of organic hydrocarbons however in this study we considered mainly four types of contaminants: Benzene, Toluene, Ethylbenzene and Xylene (BTEX). These compounds have been found to cause significant impacts on the underlying groundwater quality.

A 3-dimensional representation of the site was created in Visual MODFLOW. The model domain was created for this study as a 1000 x 1000 x 6 meters in the X, Y and Z (corresponding to East-West, North-South and Vertical, respectively) directions, with a general uniform grid spacing of 12 m between grid nodes, refined to 1 m uniform spacing between nodes in the area of the contaminant source and a supply well field. A numerical model consisting of 147 rows, 140 columns, and 1 layer was constructed for this hypothetical model. The relevant site features consist of a contaminant source and a water supply well field. These features are illustrated in the Figure 4.1.

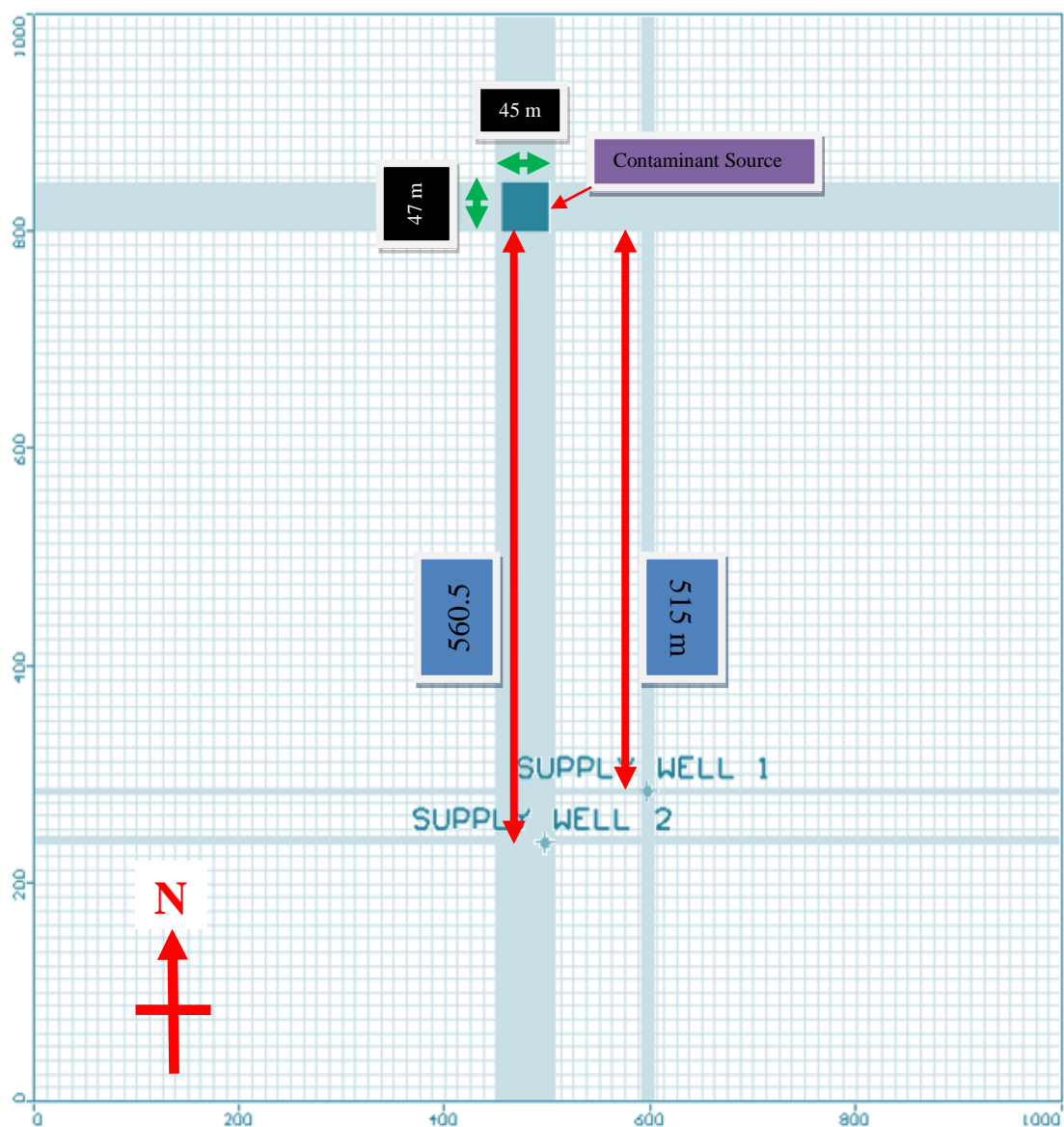


Figure 4.1: Model Domain

The supply well field consists of two wells: the east supply well (supply well 1) and the west supply well (supply well 2). The contaminant source width (47 m) and length (45 m) was assumed in this model. The distance between contaminant source and east supply well is 515 m and the distance between contaminant source and west supply well is 560.5 m.

To get an actual aquifer layout different geological formation tests are essential to be performed. Aquifer layer was adapted to create well tested hypothetical site from the previous research work of Thierrim et al. (1993). A three dimensional view of the layer is given in Figure 4.2.

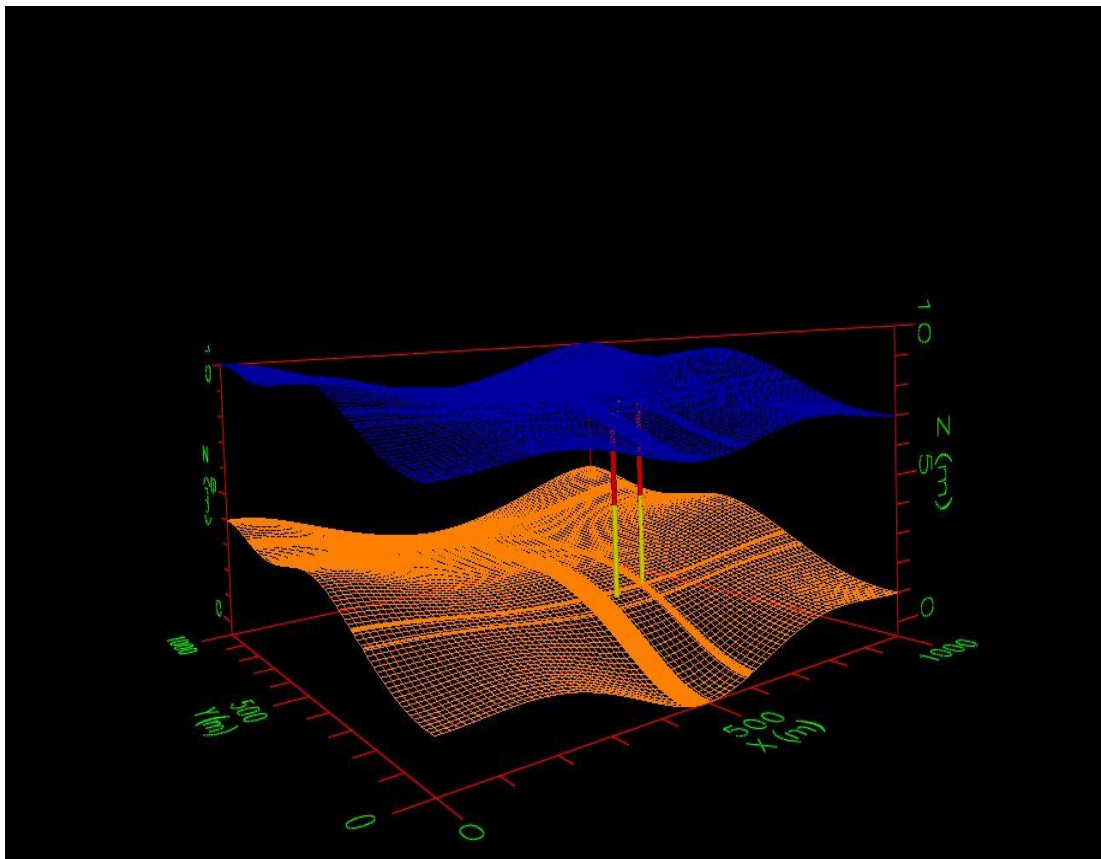


Figure 4.2: 3-D View of Model Layer

When discussing the model, in plan view, the left side was designated as west, the right side as east, the top of the site was designated as north and the bottom of the site as south. Groundwater flow is from north to south in a one layer model system consisting of a Superficial unconfined aquifer. The thickness of the unconfined layer was assumed as an average 6 m.

The model grid is bounded by no-flow boundaries on the east and west sides. The north side is a constant-head boundary with a uniform head of 10 m. The south side is a constant head boundary with the uniform head equal to 8 m. Hydraulic gradient was set to be 0.002 m/m in all scenarios. The average annual rainfall is about 900 mm/year in Perth Region (Davidson 1995). A value between 5% and 20% of annual rainfall (precipitation) is recommended as an estimate of recharge when other data are not available (WHI 2006). Based on this guidance, recharge was set equal to 11% of annual rainfall (precipitation) (100 mm/yr). This model assumes a higher recharge of 250 mm/yr at the contaminant source where the fuel has been leaked.

The source of contamination was designated at the contaminant source with an initial concentration of 150 mg/l of each contaminant during the simulation time. The modeling assumed no future spills. The same flow and contaminant transport boundary conditions were assumed for all scenarios.

As discussed in chapter 2 the contaminant transport modeling depends on groundwater flow (e.g. hydraulic conductivity, storage coefficient, specific yield, total porosity, effective porosity etc.) and contaminant transport (e.g. advection, dispersion, reaction or biodegradation rates, diffusion and sorption) properties. Reaction effects are not considered in this study. It was also concluded from the previous research (Wiedemeier et al. 1995; Lovanh et al. 2000) that diffusion is usually a small scale process than the hydrodynamic dispersion and is caused by mass transfer along concentration gradients which leads to equalization of chemical concentration. However, diffusion coefficients in groundwater are negligible, which implies that the timescales for diffusion to equilibrate concentrations over relatively small travel distances may be decades or centuries (Johnston et al. 1998). For this reason, the effects of diffusion are often neglected in comparison to other effects. Moreover, for reaction effect, the first order rate constant is a critical parameter as an input in contaminant transport model which depends on the half life of contaminants and has a wide range. For example, half life value of benzene is estimated from 10 to 720 days for different sites as per Howard (1991). Therefore, the rate constant should not be extrapolated from the literature. Rather, a considerable care must be exercised in its determination to avoid over-predicting or under-predicting actual biodegradation rates and plume behaviour (Lovanh et al. 2000).

All other groundwater flow and contaminant properties for major type of soils found in the Perth region were considered in this study and discussed in the following sections. The main focus in this scenario modeling was to see the effect of advection, dispersion and sorption on the contaminant transport.

Advection is defined as the migration of contaminant due to the bulk movement of groundwater (Wiedemeier et al. 1995). Advection term dominates over other terms for most of the contaminant transport field scale problems.

Dispersion is the mechanism of the contaminant transport process whereby a contaminant plume spreads out in longitudinal and transverse directions of the contaminant plume movement (Wiedemeier et al. 1995; Becker et al. 2004). Dispersion is important to consider because it allows the solute plume to be diluted by uncontaminated groundwater (Lovanh et al. 2000).

Sorption is the process between the contaminants sorbed on the solid phase or porous medium and the dissolved contaminants in groundwater (aqueous phase) (Zheng and Wang 1999). Sorption depends on K_d (distribution coefficient) which depends on the K_{oc} (octanol-water partition coefficient) and F_{oc} (fraction of organic carbon contents in soil). K_{oc} values for BTEX contaminants were taken from Johnston et al. (1998) and given in the following Table:

Contaminants	Log K_{oc}	K_{oc}
Benzene	1.9	79.43
Toluene	2.1	158.49
Ethylbenzene	2.2	125.89
Xylene	2.4	251.19

Table 4.1: K_{oc} (octanol - water partition coefficients) for BTEX (Johnston et al. 1998)

In this work, three different kinds of soil type were taken for the simulation: they are Bassendean Sand, Safety Bay Sand and Tamala Limestone. These types of soil were chosen to represent the major soil types found in Perth. In the given scenario multilayer scenario was avoided to reduce the complexity of the model. Moreover

the region in and around Perth had an impervious clay beneath the first surface layer which has very low permeability. The scenario with each of the soil layer is discussed below.

4.3 Model Assumptions

Steady state conditions were assumed for all the scenario simulations because this study neglected the changing climate conditions. Furthermore, a constant recharge rate was assumed for the simulation.

The groundwater flow and contaminant transport properties vary from zone to zone, however zonal variations in all parameters, in combination with the poorly constrained recharge estimates, create the potential for a non-unique solution (Yeniguel et al. 2006). Thus for ease of simulation, it was assumed to remain constant in the modeled scenario, thus suggesting a homogeneous aquifer.

Diffusion and reaction or biodegradation effects were neglected during the scenario modeling. Sorption is represented by linear equilibrium partitioning between the aqueous and solid phases in this model.

In this model, it was also assumed that the groundwater flows from north to south and has no flow boundary condition on west and east sides. It was also assumed that the initial concentration was 150 mg/l for each contaminant and there was no future spill of BTEX contaminants.

As per Australian Drinking Water Guidelines, the MCL (Maximum Contaminant Level) is 0.001 mg/l, 0.8 mg/l, 0.3 mg/l and 0.6 mg/l for benzene, toluene, ethylbenzene and xylene, respectively (ADWG 2004). In this model, the supply well was specified to be contaminated when the upper limit of BTEX contaminant concentrations reach 0.5 mg/l in the well for each of the contaminants.

4.4 Bassendean Sand Hydrogeology

Bassendean Sand is present over most of the central coastal plain area of Perth region. It is pale-grey to white, fine to coarse but mainly medium grained, moderately sorted, sub rounded to rounded quartz sand. It has an upward fining progression in grain size. Fine-grained, black, heavy minerals are commonly spotted, and may occur as lenses. The Bassendean Sand unit varies in known thickness to a maximum of about 80 m. It conformably overlies Gangara Sand and un-conformably overlies Tamala Limestone (Davidson 1995).

Hydrogeological characteristics of the Bassendean Sand are summarized in Table 4.2. In the central sandy (Bassendean Sand) area of Perth region, hydraulic conductivities range from 10 m/d (1.16×10^{-4} m/s) to more than 50 m/d (5.78×10^{-4} m/s) (Davidson 1995). Thierrin et al. (1993) and Davidson (1995) determined effective porosities and specific yields ranging from 20% to 30%, F_{oc} (Fraction of organic carbon content in Bassendean Sand) ranging from 0.08 % to 0.6 % and α_L (longitudinal dispersivity) ranging from 0.025 to 0.027 by using bromide tracer test.

Flow Properties	
Hydraulic Conductivity (K)	10 to 50 m/day
Effective Porosity (P_e)	20% to 30%
Specific Yield (S_y)	20% to 30%
Thickness (b)	Maximum 80 m
Contaminant Transport Properties	
Organic Carbon Content (F_{oc})	0.08 to 0.6 %
Longitudinal Dispersivity (α_L)	0.025 to 0.027 m

Table 4.2: Flow and Contaminant Transport Properties of Bassendean Sand

4.4.1 Flow and Contaminant Transport Properties for Bassendean Sand Model

Table 4.3 shows the typical flow and contaminant transport properties which was used for the simulation. The ratio of vertical hydraulic conductivity (K_z) to horizontal hydraulic conductivity (K_x) was assumed to be 0.1. Specific storage was assumed to be 1×10^{-5} 1/m in all simulations. Based on Gelhar, Welty and Rehfeldt (1992), horizontal transverse dispersivity (α_{TH}) is typically of the order of 10 percent of the longitudinal dispersivity (α_L) and vertical transverse dispersivity (α_{TV}) is typically of the order of 1 percent of the longitudinal dispersivity (α_L). So, α_{TH}/α_L was set as 0.1 and α_{TV}/α_L was set as 0.01 in all simulations. Bulk density of the aquifer material was assumed to be 1700 kg/m^3 .

Flow Properties	
Horizontal Hydraulic Conductivity (K_x, K_y)	$3 \times 10^{-4} \text{ m/s}$
Vertical Hydraulic Conductivity (K_z)	$3 \times 10^{-5} \text{ m/s}$
Effective Porosity (P_e)	28 %
Total Porosity (P_T)	35 %
Specific Yield (S_y)	28 %
Specific Storage (S_s)	$1 \times 10^{-5} \text{ 1/m}$
Layer Thickness (b)	Maximum 6 m
Bulk Density (ρ_b)	1700 kg/m^3
Contaminant Transport Properties	
Organic Carbon Content (F_{oc})	0.08 to 0.6 %
Longitudinal Dispersivity (α_L)	0.026 m
Horizontal Transverse Dispersivity (α_{TH})/Longitudinal Dispersivity (α_L)	0.1
Vertical Transverse Dispersivity (α_{TV})/Longitudinal Dispersivity (α_L)	0.01

Table 4.3: Typical Values of Flow and Transport Properties of Bassendean Sand Model

4.4.1.1 Effect of Sorption on Contaminant Plume

The simulation was carried out on a site with Bassendean Sand. For this scenario groundwater velocity is 68 m/year which is calculated by equation 4.3. The first attempt was made to see the effect of sorption on contaminant transport with scenario modeling. Contaminant sorption depends on the organic carbon content of the aquifer matrix and the organic carbon partitioning coefficient (K_{oc}) for each contaminant. K_{oc} values are obtained from Johnston et al. (1998) for this study. F_{oc} is considered to be 0.6% in Bassendean Sand. Distribution coefficients (K_d) normalized to fraction of total organic carbon content are expressed as K_{oc} . The following equation 4.1 gives the expression relating K_d to K_{oc} and F_{oc} and the values of K_d are reported in Table 4.4.

$$K_d = K_{oc} F_{oc} \dots\dots\dots \text{Equation 4.1}$$

Contaminants	Log K_{oc}	K_{oc} (L/Kg)	F_{oc} (%)	K_d (L/mg)
Benzene	1.9	79.43	0.6	4.77×10^{-7}
Toluene	2.1	158.49	0.6	9.51×10^{-7}
Ethylbenzene	2.2	125.89	0.6	7.55×10^{-7}
Xylene	2.4	251.19	0.6	1.51×10^{-6}

Table 4.4: Distribution Coefficients (Kd) for $F_{oc} = 0.6\%$

For the first case, fraction of organic carbon content of Bassendean Sand was assumed to be 0.6%. Initial concentration is 150 mg/l of each contaminant. All other data are shown in Table 4.3. The contaminant plumes observed are shown in Figure 4.3 with results tabulated in Table 4.5.

Contaminants	Time (days)	Plume Length (m)	Plume Width (m)
Benzene	7400	626	53
Toluene	13100		
Ethylbenzene	10900		
Xylene	20100		

Table 4.5: Simulation Results at $F_{oc} = 0.6\%$

It was observed from the simulations that benzene reached most rapidly to supply well compared to toluene, ethylbenzene or xylene. For any contaminant, the time to reach the supply well is highly dependant on distribution coefficient. The higher the distribution coefficient, the greater the sorption to the aquifer matrix and thus the higher will be the time for the contaminants to reach to the supply well. To conclude, a retardation coefficient term was studied in depth from the literature (Wiedemeier et al. 1995; Fetter 2001; Zheng an Bennet 2002). The retardation coefficient, R, describes the extent to which the movement of dissolved contaminants can be slowed down by sorption to the aquifer matrix (Lovanh et al. 2000). The degree of retardation depends on both aquifer properties and contaminant transport properties. The retardation coefficient is the ratio of the groundwater velocity (V_g) to the average velocity of a migrating contaminant (V_c). The retardation coefficient for equilibrium linear sorption is determined from the distribution coefficient using Equation 4.2 (Freeze and Cherry 1979). Sorption is the process between the dissolved contaminants in aqueous phase and the sorbed contaminants on the solid phase (Zheng and Wang 1999). It is generally assumed that equilibrium conditions exist between the solid-phase and the aqueous-phase concentrations and that the sorption reaction is quick enough, comparative to the groundwater velocity, to be treated as instantaneous which will allow contaminant to adsorb first rather than to flow with groundwater. Therefore, the higher the K_d the longer will be the time for the contaminant to reach the supply well from the contaminant source.

$$R = \frac{V_g}{V_c} = 1 + \frac{\rho_b \times K_d}{P_e} \dots\dots\dots \text{Equation 4.2}$$

Figures 4.3 and 4.4 represent the typical benzene, toluene, ethylbenzene and xylene plume at $F_{oc} = 0.6\%$ and 0.08% , respectively. The results show that there was no significant effect of F_{oc} on the plume shape for all contaminants. The results also demonstrated that BTEX concentration reached approximately 4.5×10^{-31} mg/l in the supply well 1 for 25000 days which is far below than the MCLs and it can be totally neglected and results are shown in Figures 4.5 and 4.6.

Figure 4.7 depicts the simulation results for concentration of BTEX along the centerline of contaminant plume at different F_{oc} at 1000 days. The results show that there is a significant effect of F_{oc} on contaminant plume concentration along the

centerline of the plume. It is observed that decrease in F_{oc} BTEX contaminants covers short distance before reaching the concentration level of 0.5 mg/l at supply well 2.

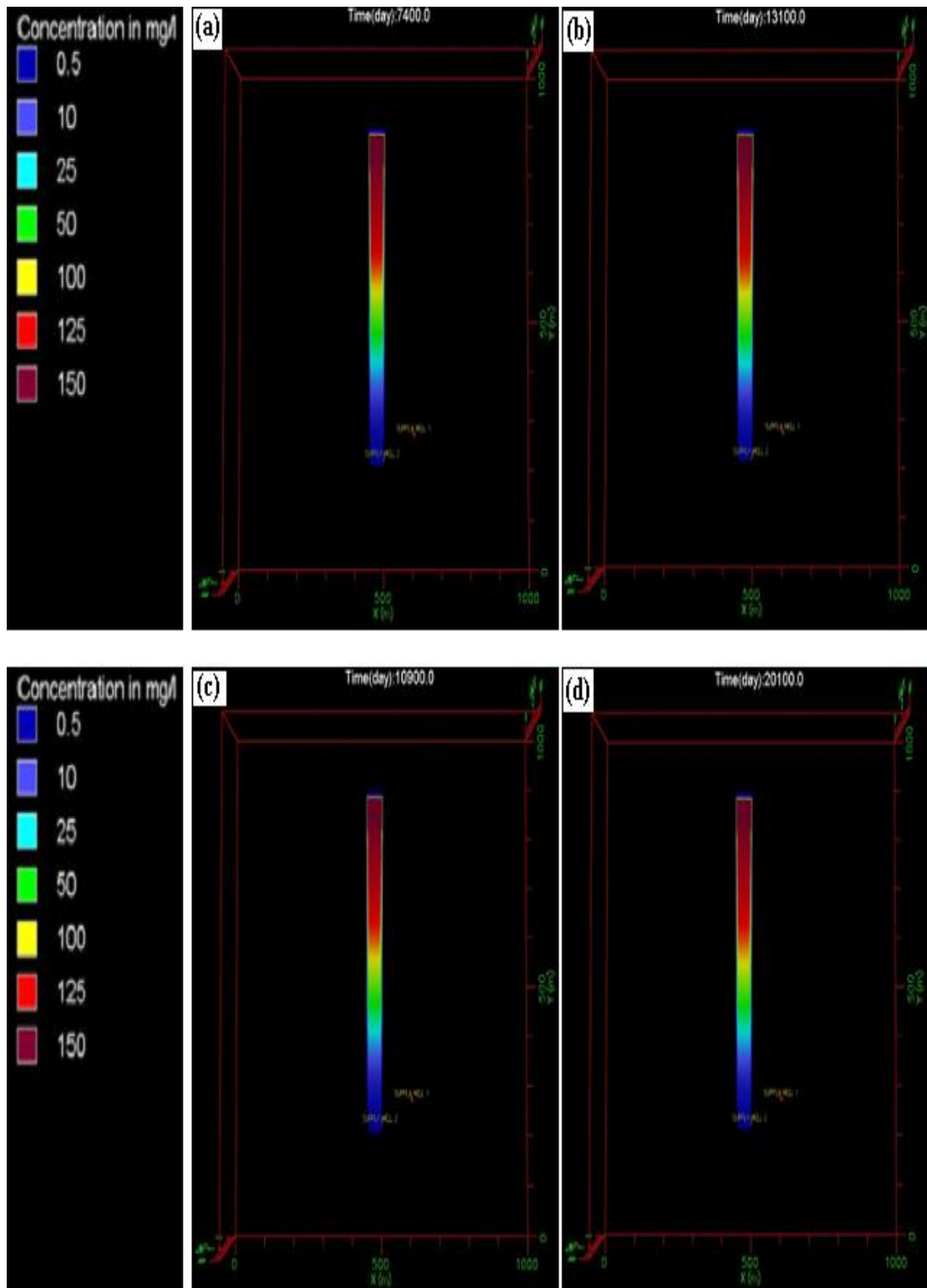


Figure 4.3: Contaminant Plume at $F_{oc} = 0.6\%$ for (a) Benzene (b) Toluene (c) Ethylbenzene (d) Xylene

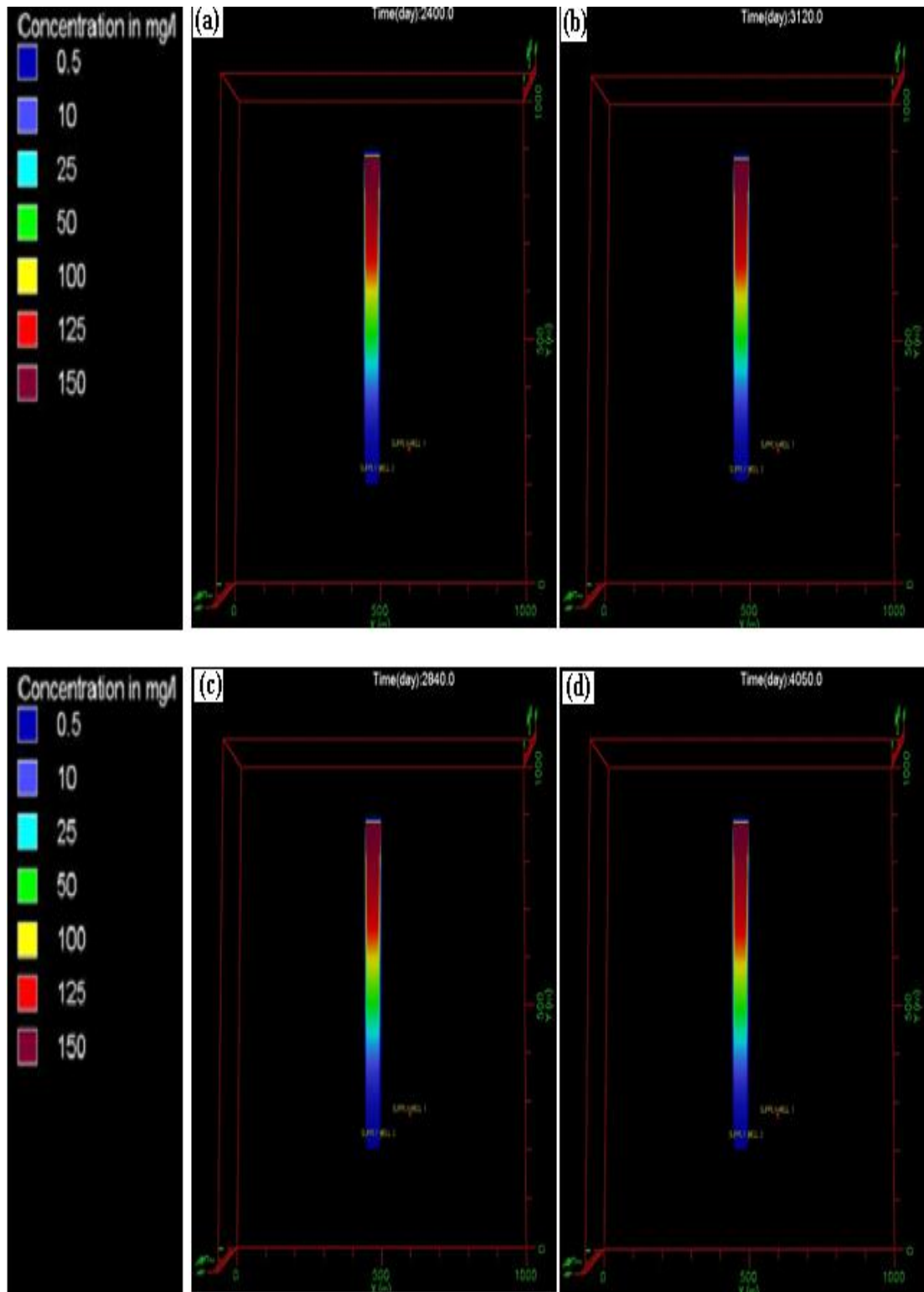


Figure 4.4: Contaminant Plume at $F_{oc} = 0.08\%$ for (a) Benzene (b) Toluene (c) Ethylbenzene (d) Xylene

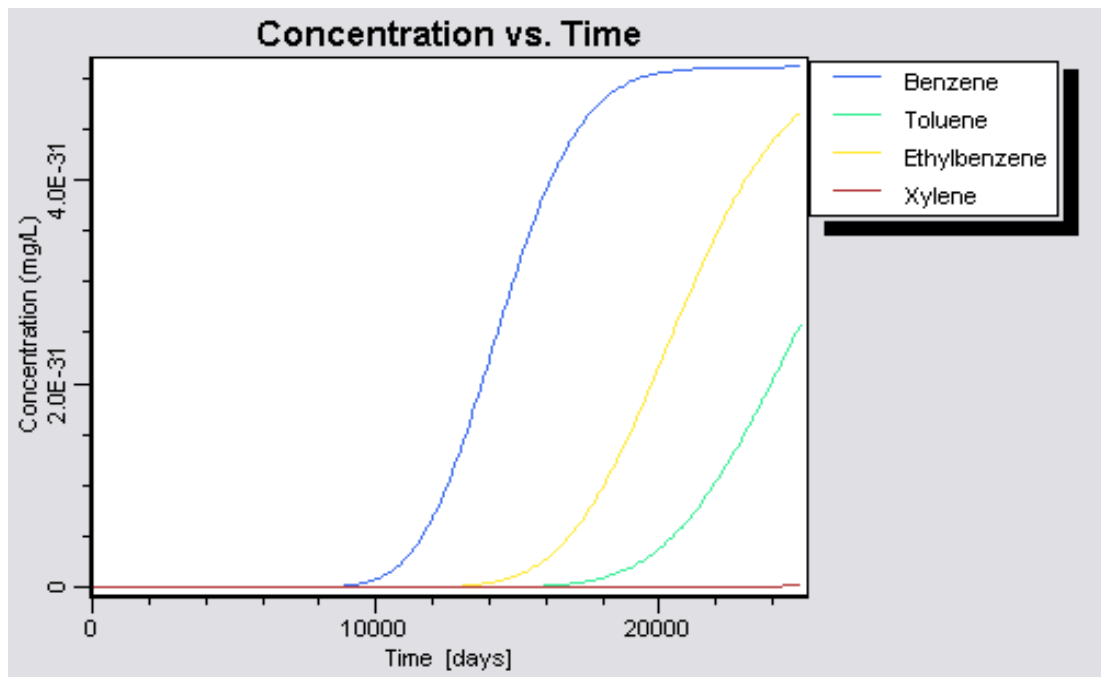


Figure 4.5: BTEX Concentration versus Time at Supply Well 1 downgradient of the Contaminant Source for $F_{oc} = 0.6\%$

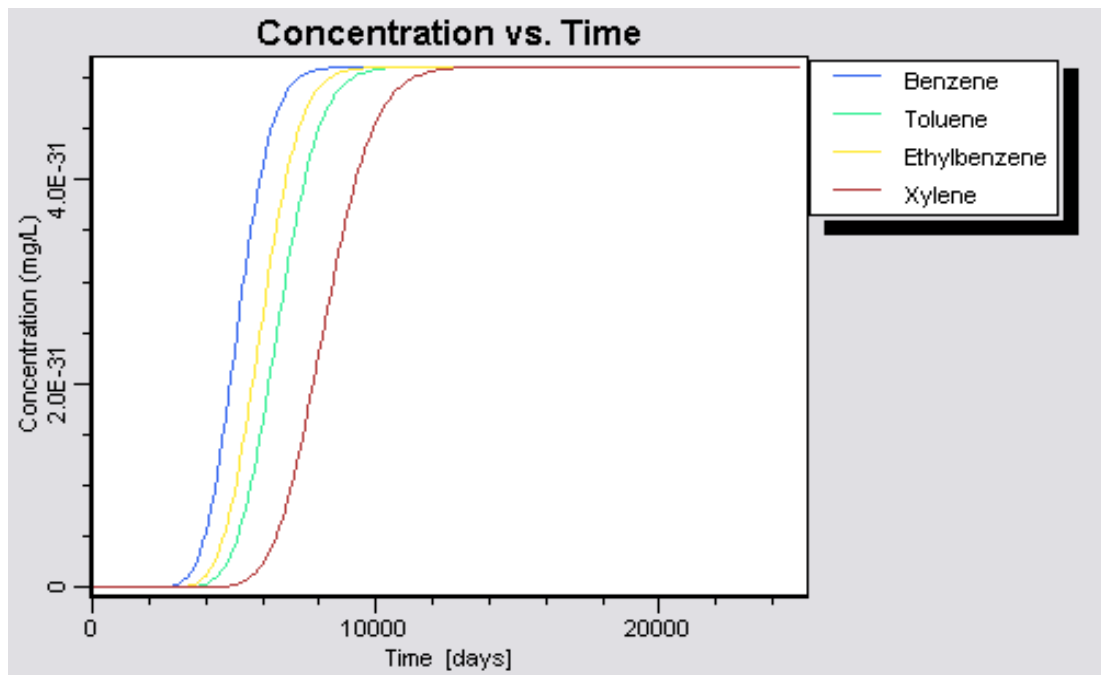


Figure 4.6: BTEX Concentration versus Time at Supply Well 1 downgradient of the Contaminant Source for $F_{oc} = 0.08\%$

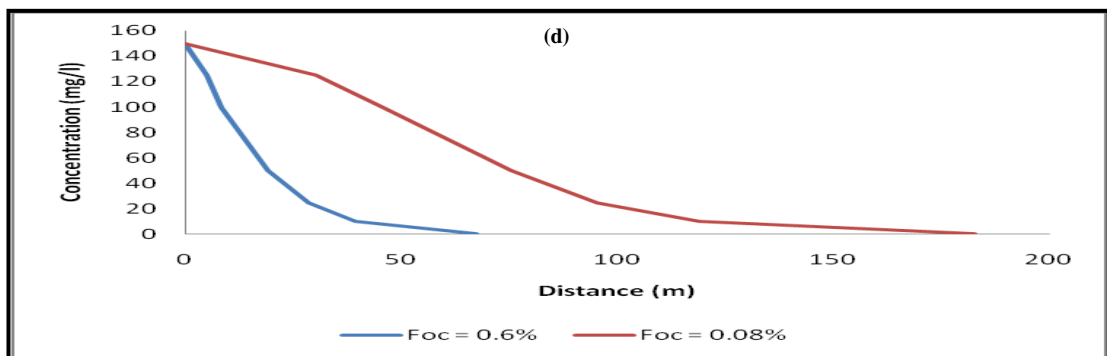
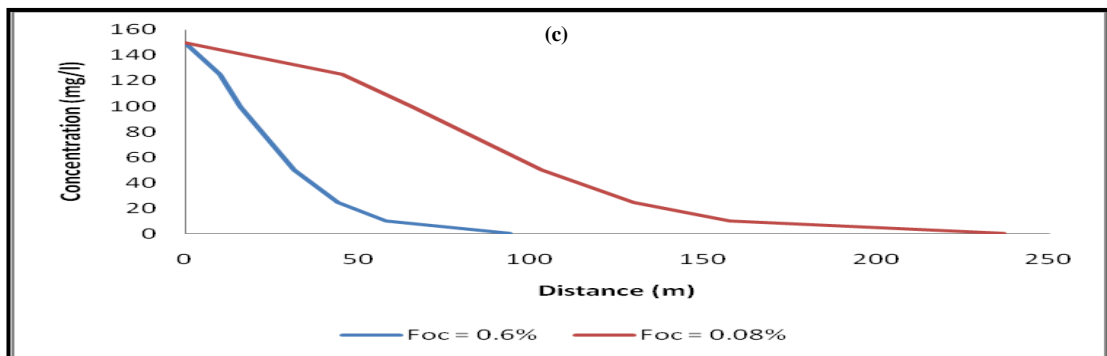
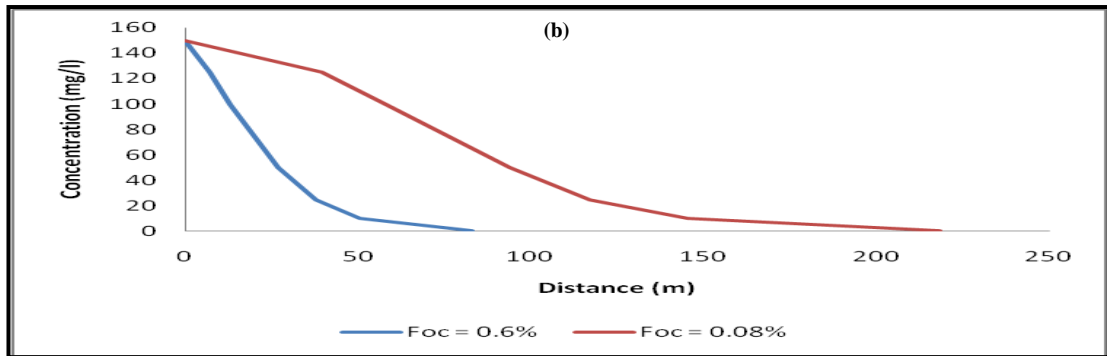
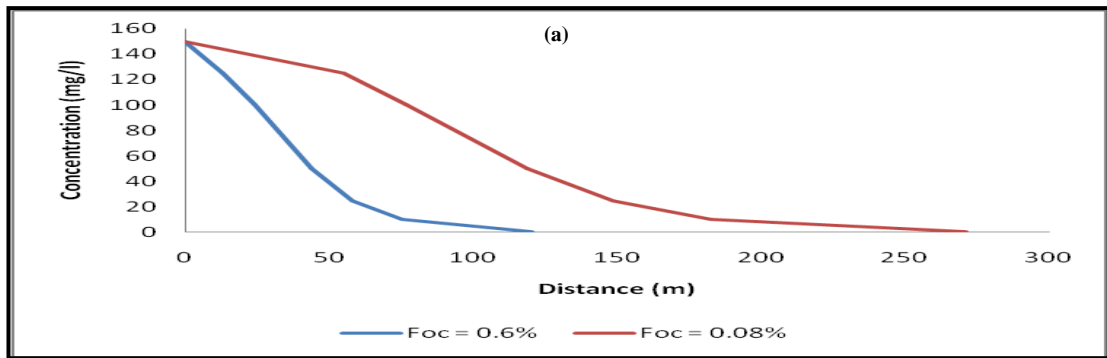


Figure 4.7: Concentration of Contaminants (BTEX) on Supply Well 2 along the Centerline of the Plume at 1000 days for (a) Benzene (b) Toluene (c) Ethylbenzene (d) Xylene

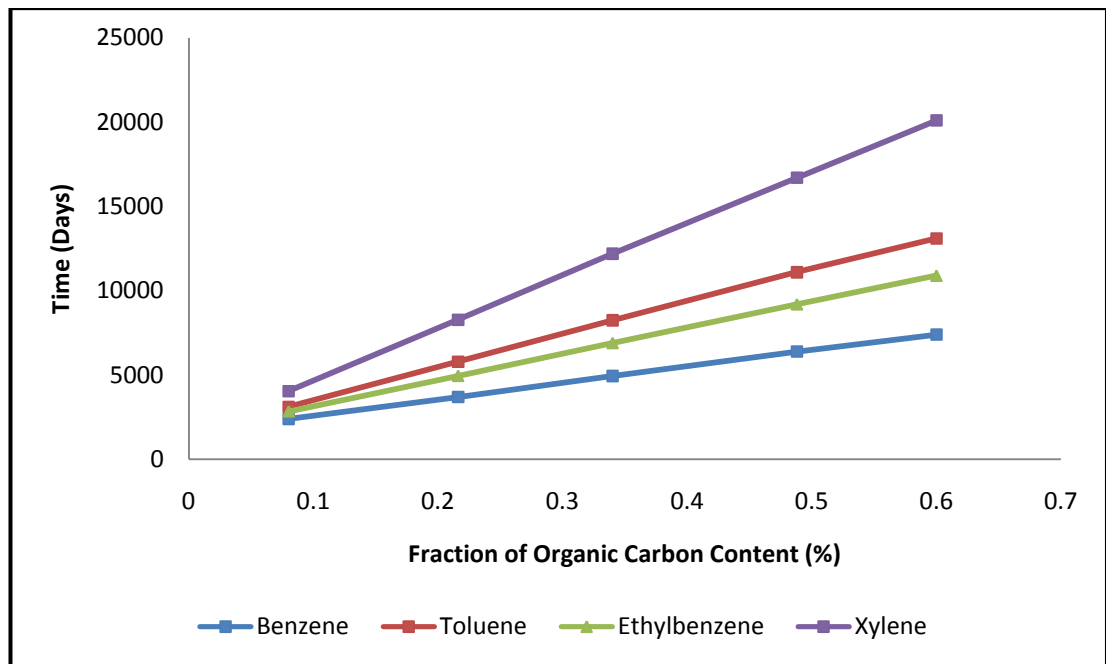


Figure 4.8: BTEX Contaminants travel time to reach the Supply Well 2 from the Contaminant Source for F_{oc} ranging from 0.08% to 0.6%

As per the general properties of the Bassendean Sand, F_{oc} has a wide range. Different simulations were carried out for Benzene, Toluene, Ethylbenzene and Xylene (BTEX) with a different F_{oc} ranging from 0.08% to 0.6%. The results are plotted in Figure 4.8.

It is clear from Figure 4.8 that increase in F_{oc} , results in distribution coefficient (K_d) increases as well as sorption and therefore contaminants will take more time to reach supply well from the contaminant source. From the graph, more or less, a linear relationship is found between the contaminants travel time to reach the supply well from the contaminant source at constant dispersivity.

Figure 4.9 shows the breakthrough profile for Benzene, Toluene, Ethylbenzene and Xylene, respectively for different F_{oc} at supply well 2 downgradient of the contaminant source. It is observed that as F_{oc} increases, plume takes longer time to reach the saturation value. The results show the contaminant plume stabilisation time at different F_{oc} and the maximum concentration of the contaminant at respected time. It is observed that the maximum concentration of the contaminants in supply well 2 is about 53 mg/l which is well above the MCLs as per Australian Drinking Water Guidelines.

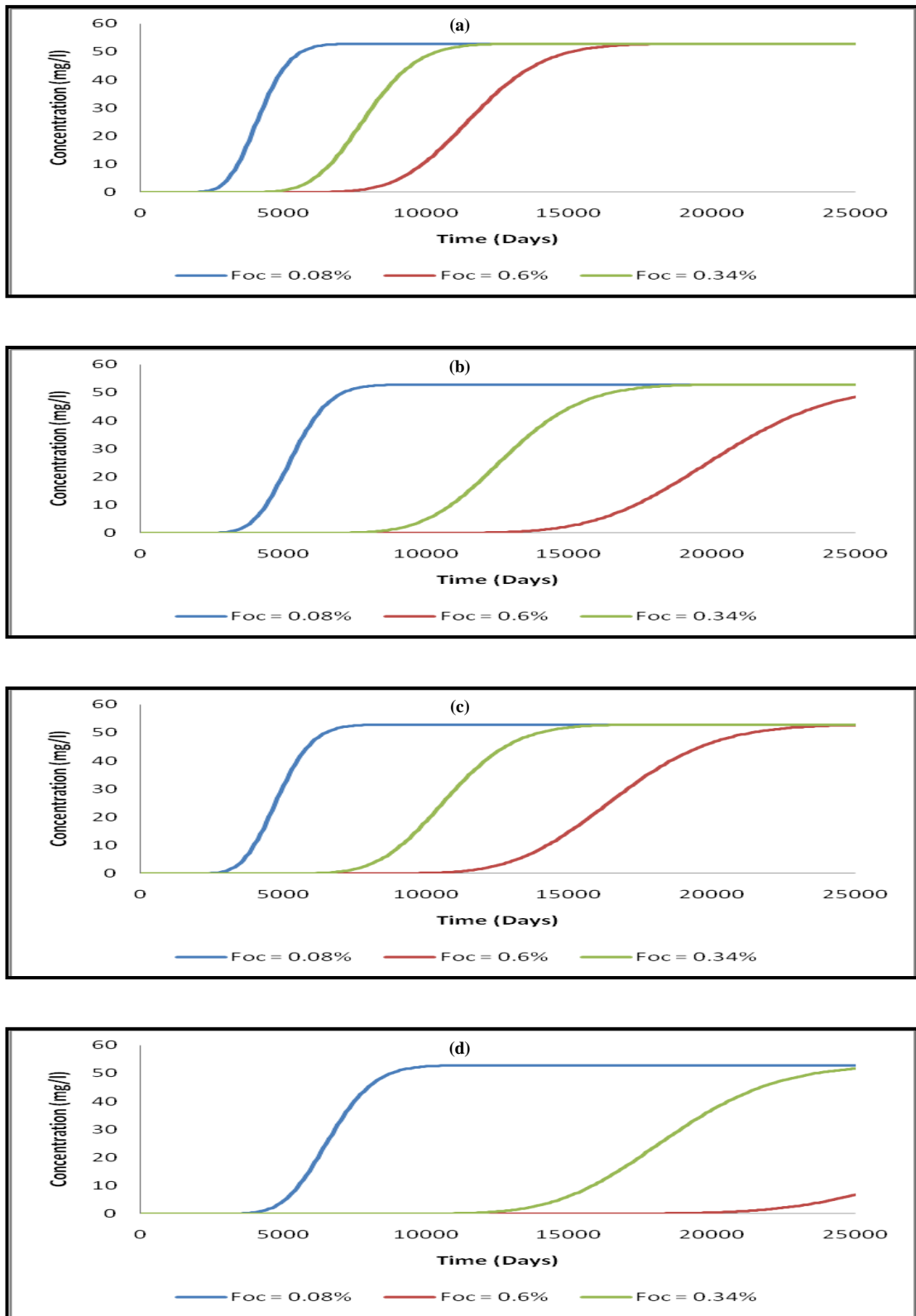


Figure 4.9: BTEX Concentration versus Time at Supply Well 2 downgradient of the Contaminant Source (a) Benzene (b) Toluene (c) Ethylbenzene (d) Xylene

4.4.1.2 Effect of Effective Porosity (P_e) on Contaminant Plume

To study the effect of effective porosity, P_e , on contaminant plume of hypothetically created site with Bassendean Sand three cases were considered. Each case had three different values of P_e . The value of effective porosity ranged from 20% to 30% for Bassendean Sand. For this simulation, groundwater flow and contaminant transport properties were used from Table 4.3. For this scenario, the fraction of organic carbon content, F_{oc} , was set to 0.6%.

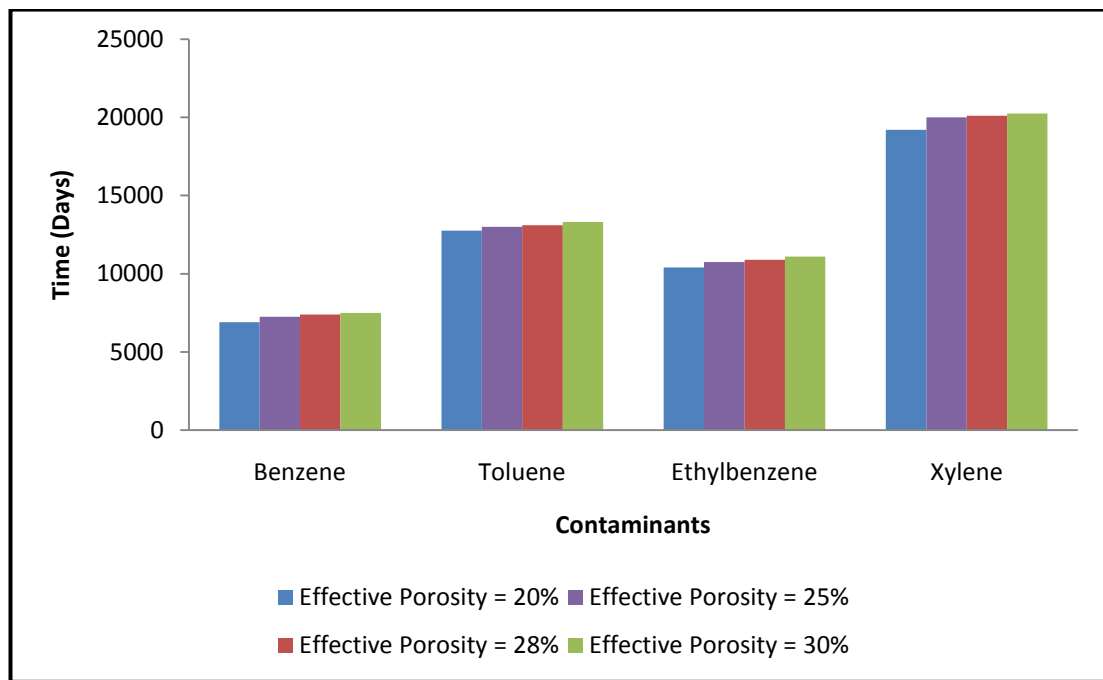


Figure 4.10: BTEX Contaminants time to reach the Supply Well 2 for different values of Effective Porosity

The effective porosity of the subsurface region greatly affects the groundwater flow pattern which in turn demonstrates its effect on contaminant transport velocity. As shown in Figure 4.10, it is clear that with increase in effective porosity (P_e) the time will increase for a contaminant to reach the supply well from the contaminant source. These effects are due to change in groundwater velocity and contaminant transport velocity for each case.

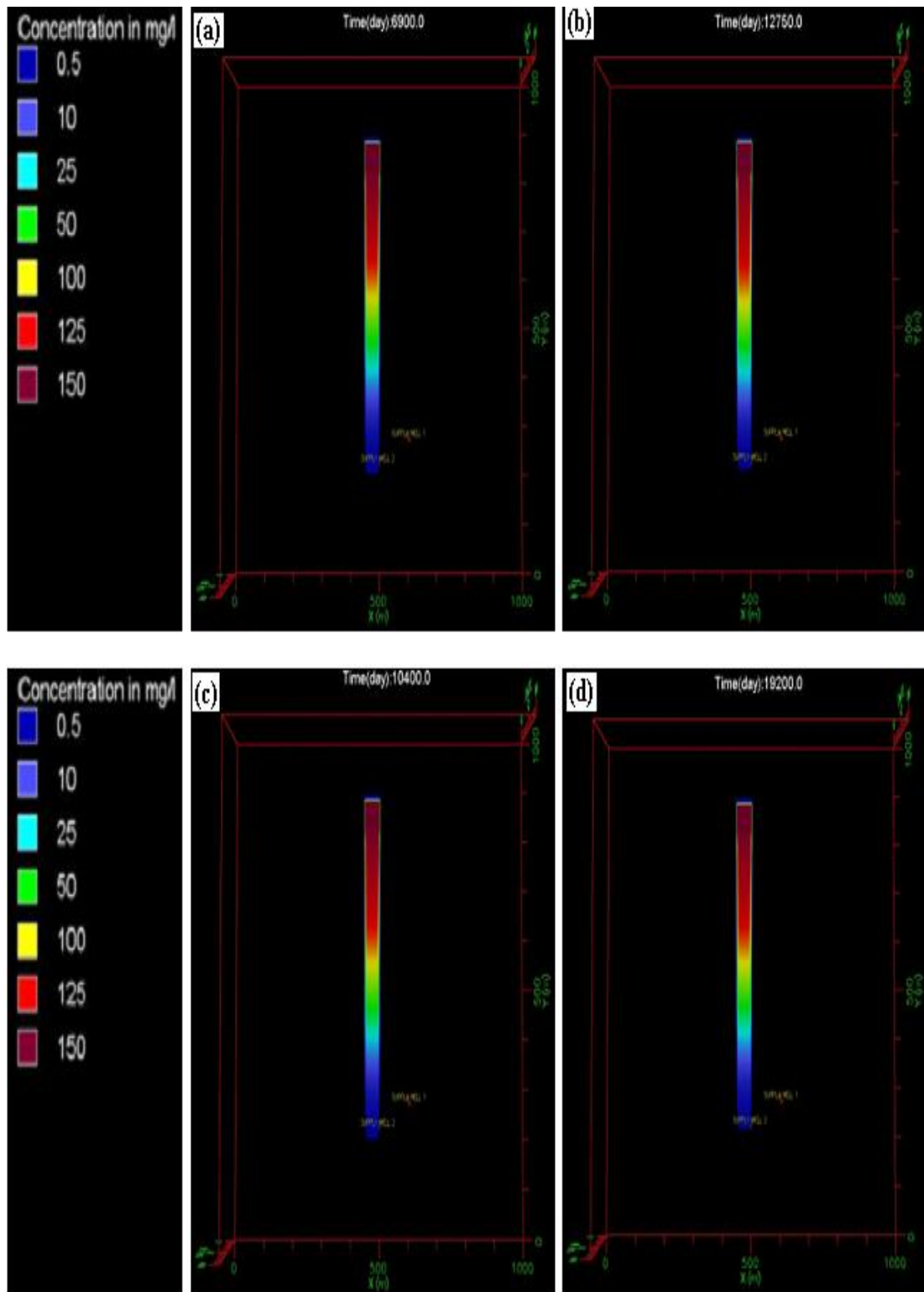


Figure 4.11: Contaminant Plume at Effective Porosity = 20% for (a) Benzene (b) Toluene (c) Ethylbenzene (d) Xylene

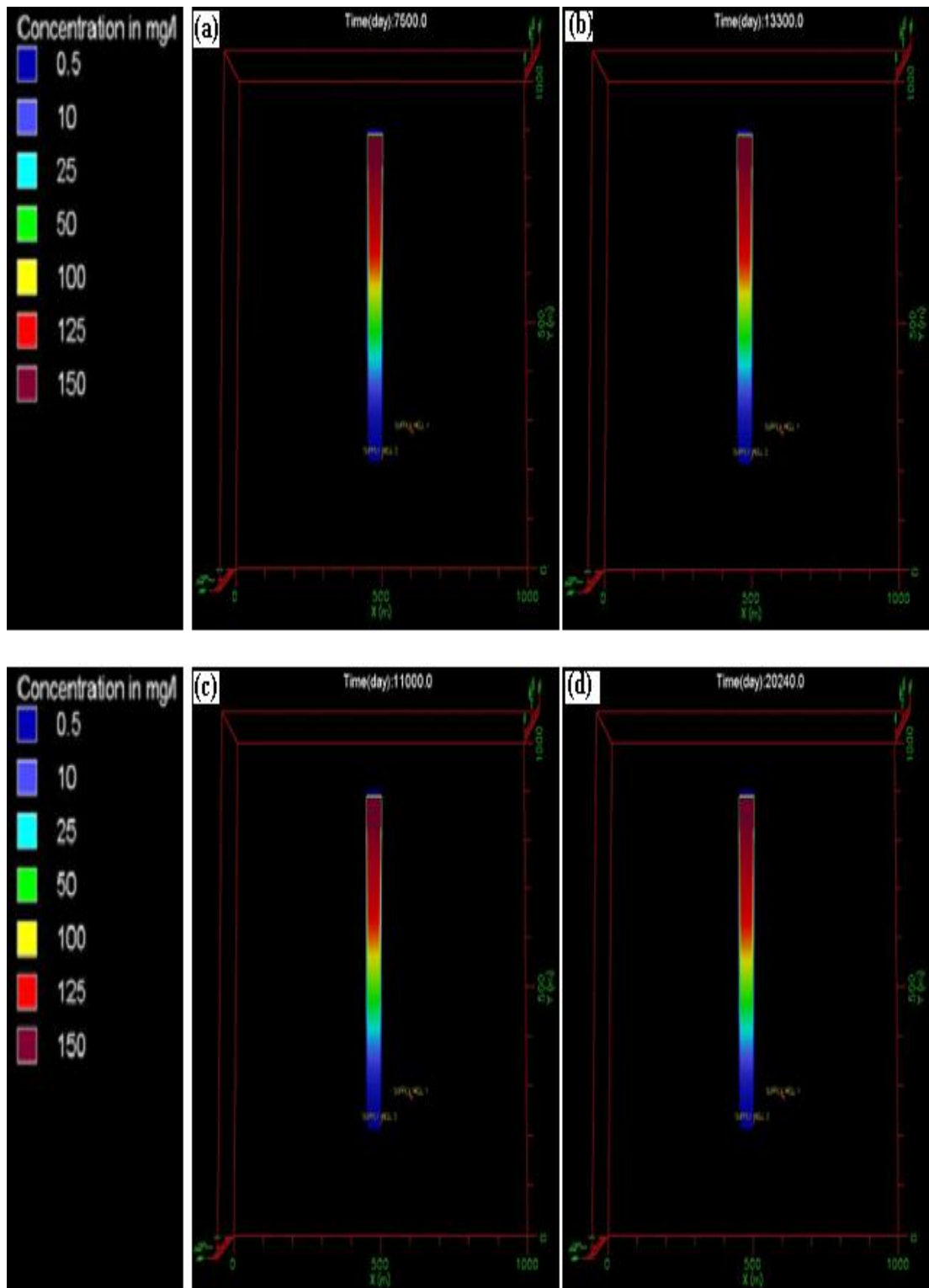


Figure 4.12: Contaminant Plume at Effective Porosity = 30% for (a) Benzene (b) Toluene (c) Ethylbenzene (d) Xylene

Groundwater velocity was found from equation 4.3 and is tabulated in Table 4.6.

$$V_g = \frac{K}{P_e} \frac{dh}{dl} \dots\dots\dots \text{Equation 4.3}$$

Where v_g is groundwater velocity in m/year, P_e is effective porosity, dh/dl is hydraulic gradient in m/m and K is hydraulic conductivity in m/year.

Case	Effective Porosity (%)	Groundwater Velocity (m/year)
1	20	94
2	25	76
3	28	68
4	30	63

Table 4.6: Groundwater Velocity for different Effective Porosity

Figures 4.11 and 4.12 show the BTEX contaminant plume for effective porosities of 20% and 30%, respectively. They show that benzene contaminant took less time to reach the well from contaminant source for 20% effective porosity than the 30% effective porosity. It also shows that there won't be any effect on contaminant plume shape for different effective porosity. The results demonstrate that BTEX concentration approximately 4.5×10^{-31} mg/l in the supply well 1 for 25000 days which is far below than the MCLs and it can be totally neglected and results are shown in Figure 4.13, 4.14a and 4.14b.

Figure 4.15 demonstrates the simulation results for concentration of BTEX along the centerline of contaminant plume at different effective porosity at 1000 days. The results show that there is little effect of effective porosity on contaminant plume concentration along the centerline of the contaminant plume. It was observed that increase in effective porosity BTEX contaminants covered less distance before reaching the concentration level of 0.5 mg/l at supply well 2.

Figure 4.16 shows the breakthrough profile for benzene, toluene, ethylbenzene and xylene, respectively for different effective porosity at supply well 2 downgradient of the contaminant source. It is observed that as effective porosity increases, plume takes longer time to reach the saturation value. The results show the contaminant

plume stabilisation time at different effective porosities and the maximum concentration of the contaminant at respected time. It was observed that the maximum concentration of the contaminants in supply well 2 is about 53 mg/l which is well above the MCLs as per Australian Drinking Water Guidelines.

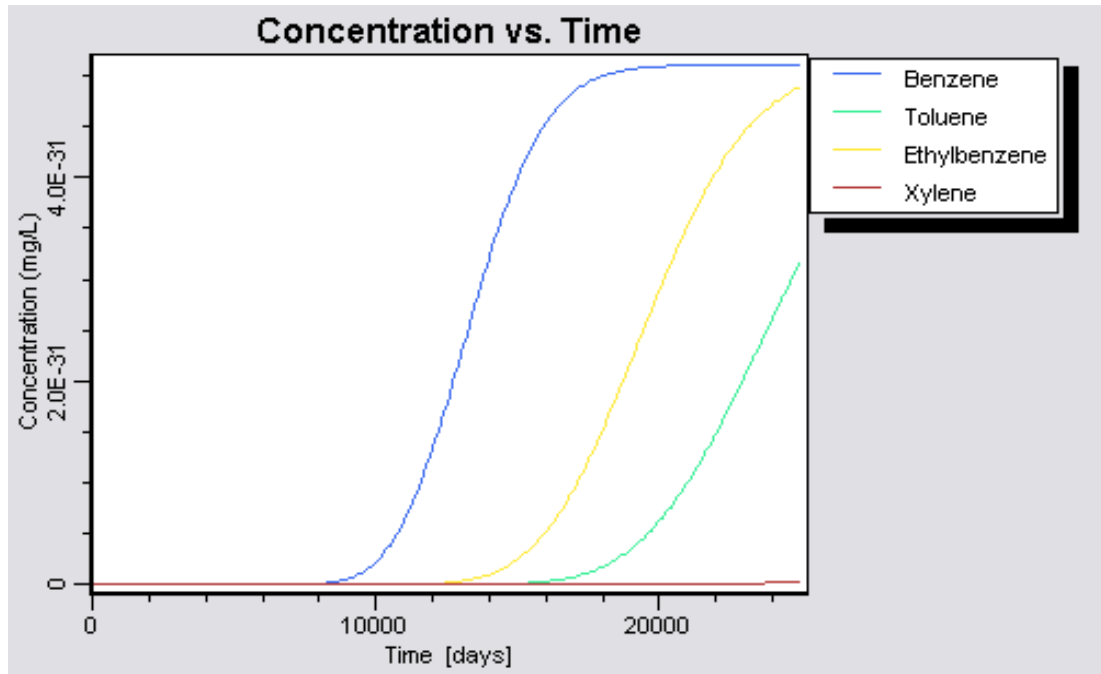


Figure 4.13: BTEX Concentration versus Time at Supply Well 1 downgradient of the Contaminant Source for Effective Porosity = 20%

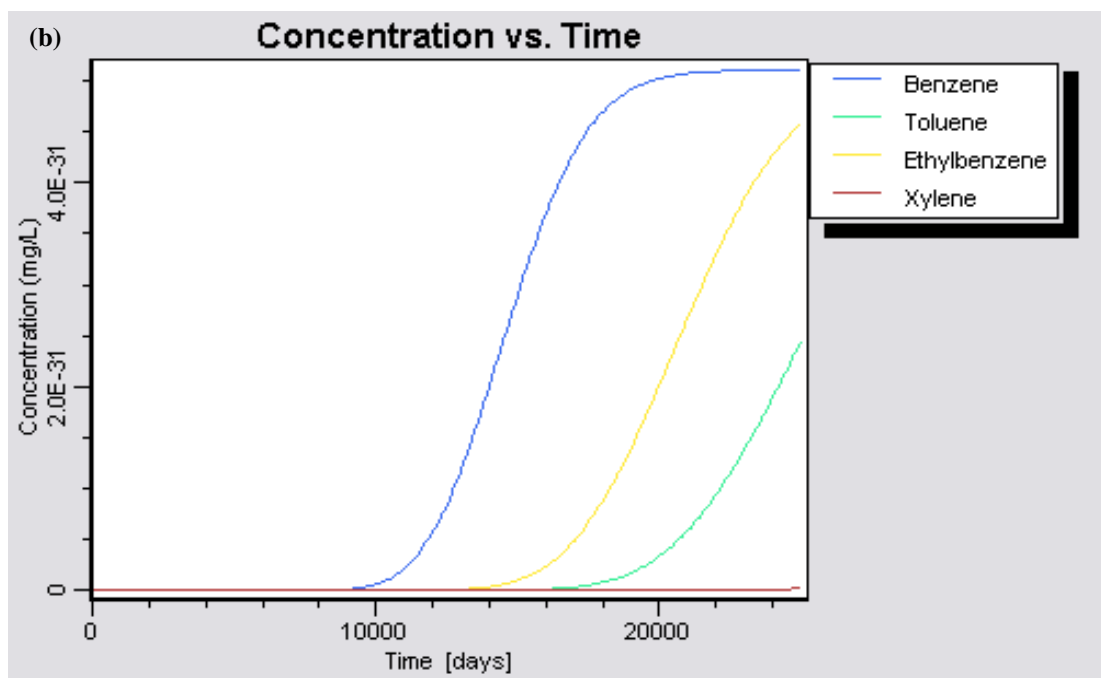
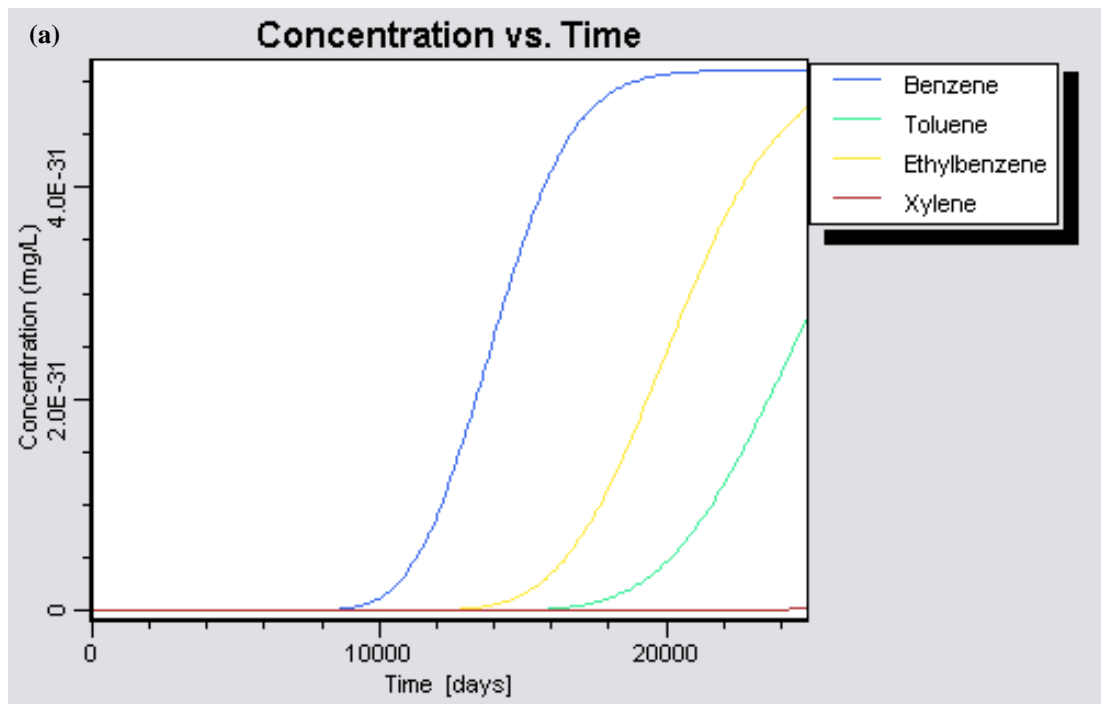


Figure 4.14: BTEX Concentration versus Time at Supply Well 1 downgradient of the Contaminant Source for (a) Effective Porosity = 25% (b) Effective Porosity = 30%

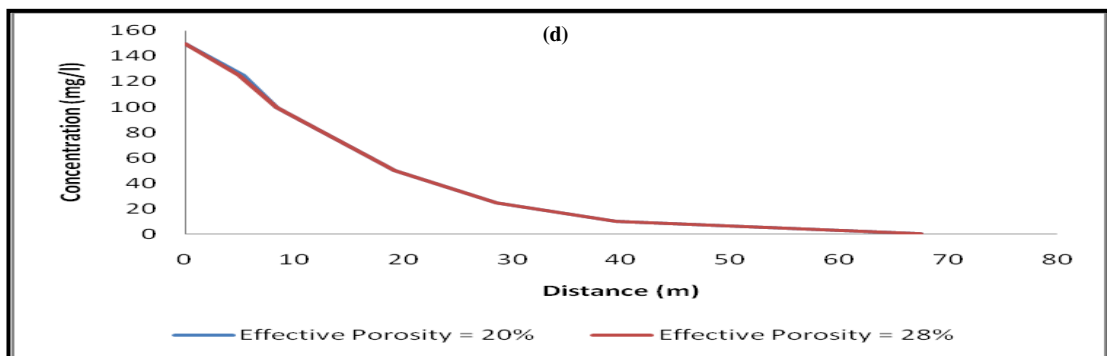
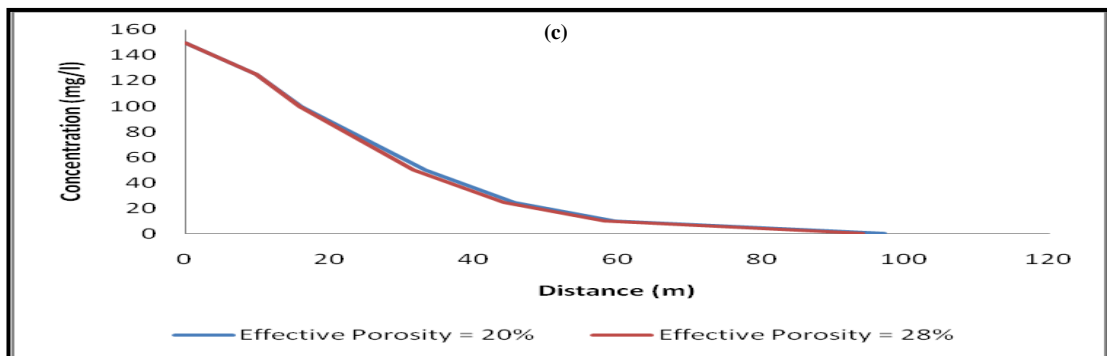
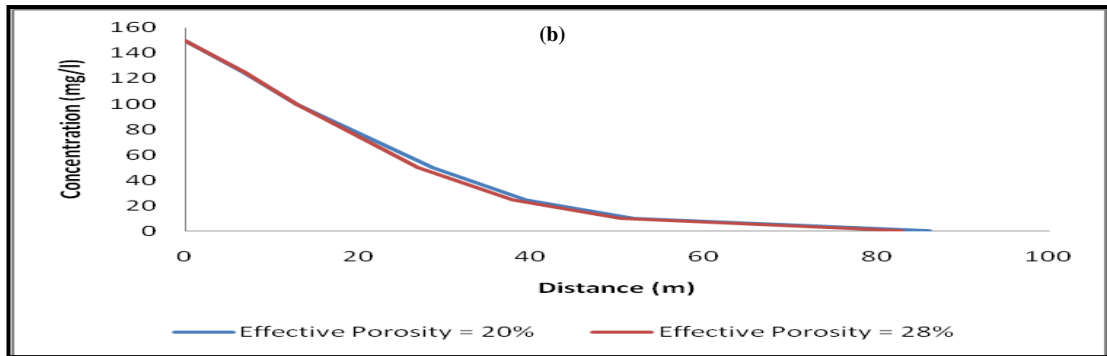
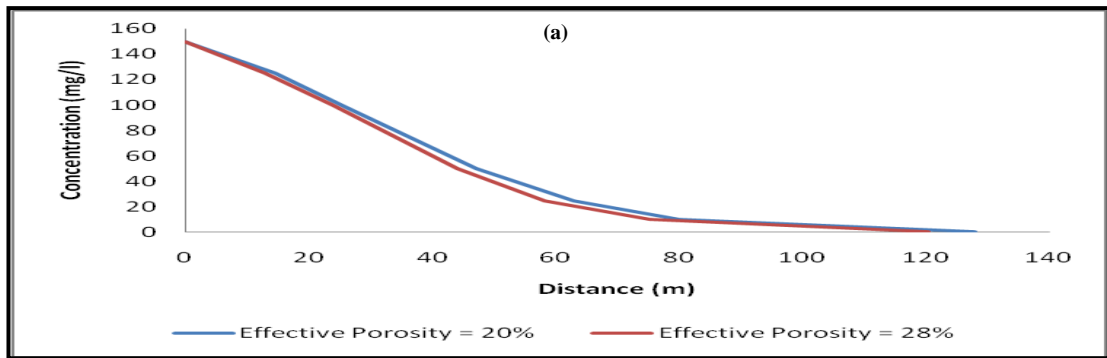


Figure 4.15: Concentration of Contaminants (BTEX) on Supply Well 2 along the Centerline of the Plume at 1000 days for (a) Benzene (b) Toluene (c) Ethylbenzene (d) Xylene

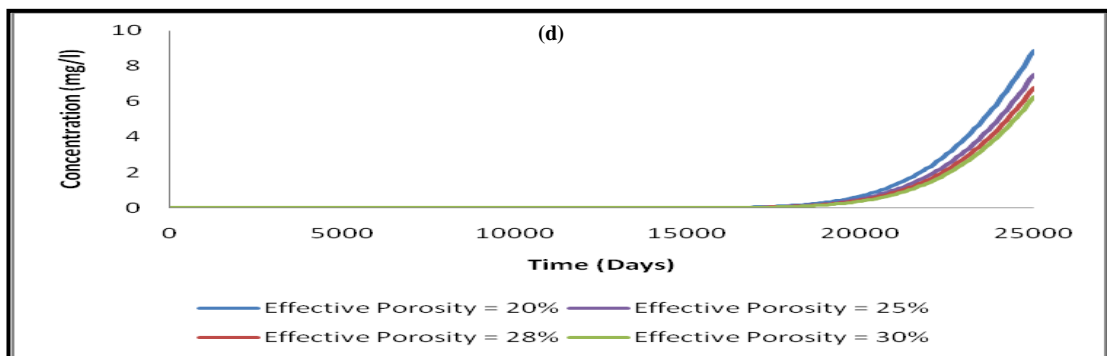
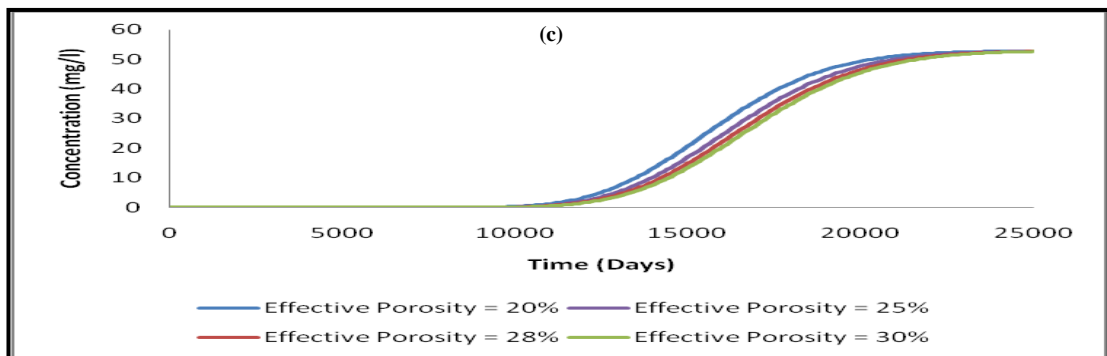
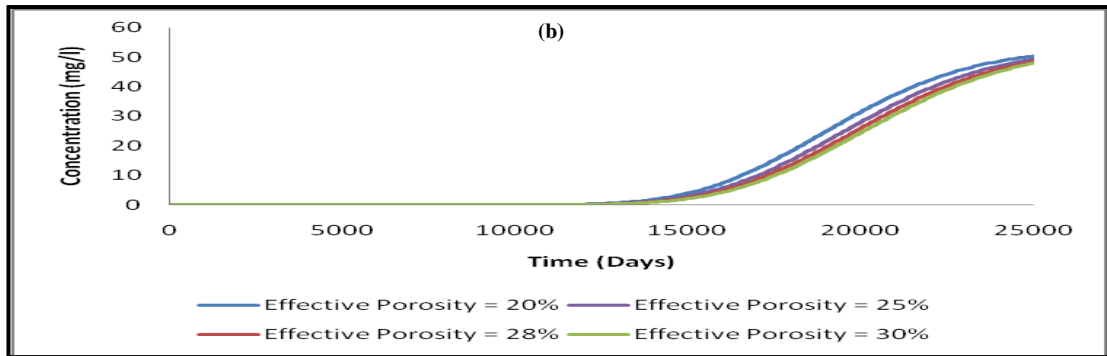
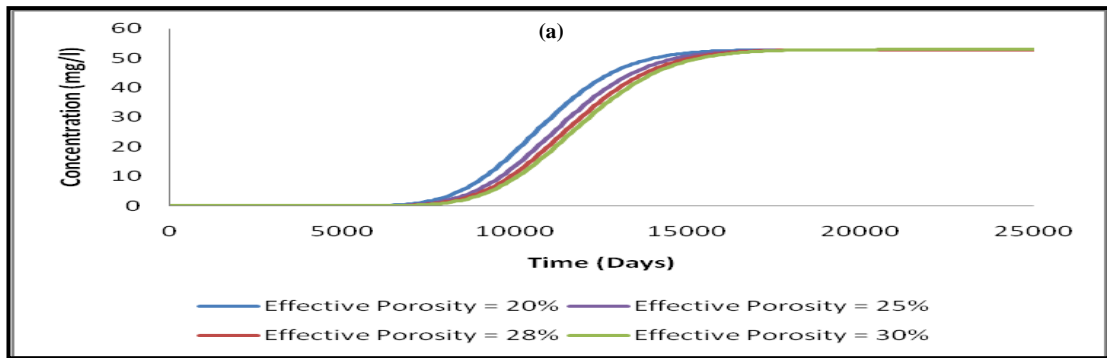


Figure 4.16: BTEX Concentration versus Time at Supply Well 2 downgradient of the Contaminant Source (a) Benzene (b) Toluene (c) Ethylbenzene (d) Xylene

4.4.1.3 Effect of Hydraulic Conductivity on Contaminant Plume

To study the effect of hydraulic conductivity which depends on the soil properties, on contaminant plume of hypothetically created site with Bassendean Sand two cases were considered. The two values of horizontal hydraulic conductivity, 3×10^{-4} m/s (26 m/day) and 5.78×10^{-4} m/s (50 m/day) were selected for Bassendean Sand (Davidson 1995). For this simulation, groundwater flow and contaminant transport properties were taken from Table 4.3. For this simulation, fraction of organic carbon content (F_{oc}) was set to 0.6%.

It is clear from Figure 4.17 that the higher the hydraulic conductivity, the contaminants tended to reach the well faster from the contaminant source. The effect is because the velocity of groundwater and dissolved contaminants are unswervingly associated to the hydraulic conductivity of the saturated zone. Variations in hydraulic conductivity directly influence contaminant fate and transport by providing preferential pathways for contaminant migration by keeping hydraulic gradient and effective porosity constant which is represented by equation 4.3.

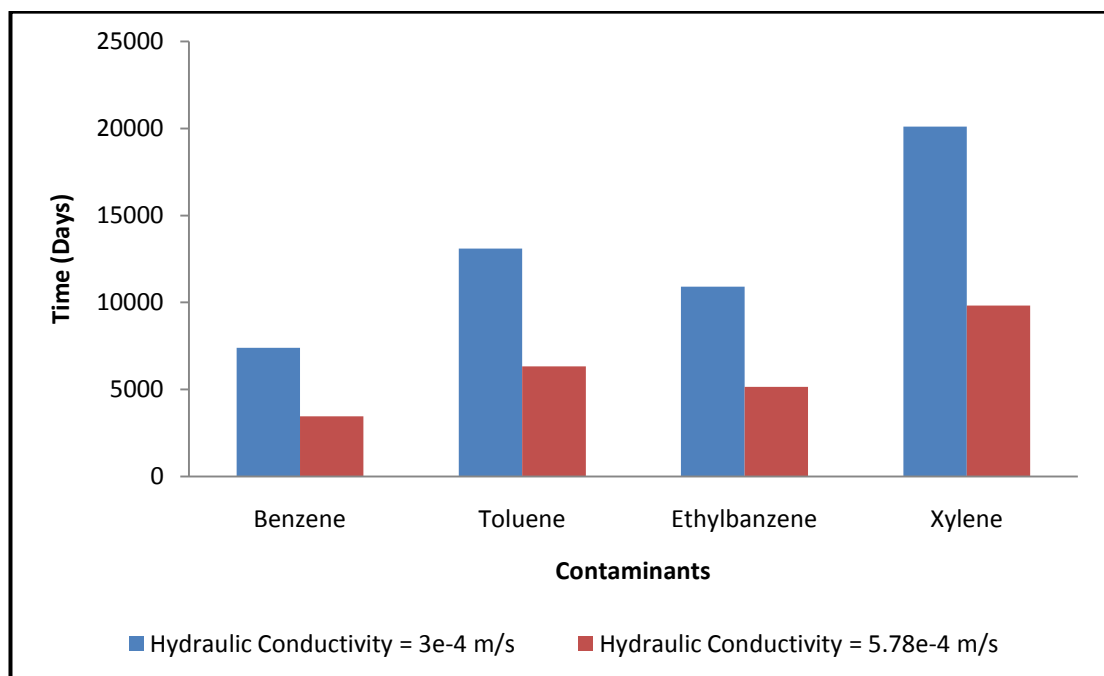


Figure 4.17: BTEX Contaminants time to reach the Supply Well 2 for different values of Hydraulic Conductivity

Figure 4.18 shows the BTEX contaminant plume for the hydraulic conductivity value 5.78×10^{-4} m/s. It represents the time for BTEX contaminant plume reaching the supply well. These results show that, for the value of hydraulic conductivity 5.78×10^{-4} m/s, benzene plume took less time to reach the supply well from contaminant source than that for hydraulic conductivity 3×10^{-4} m/s (which was shown in Figure 4.3), and similar results were also found for toluene, ethylbenzene and xylene. The results demonstrate that BTEX concentration approximately 3.3×10^{-29} mg/l in the supply well 1 for 25000 days which is far below than the MCLs and it can be totally neglected and results are shown in Figure 4.19. It also shows that the contaminant plume size is unaffected by changes in hydraulic conductivity values from 26 m/day to 50 m/day.

Figure 4.20 shows the simulation results for concentration of BTEX along the centerline of plume at different hydraulic conductivity at 1000 days. The results show that there is a remarkable effect of hydraulic conductivity on contaminant plume concentration along the centerline of the plume. It is observed that decrease in hydraulic conductivity BTEX contaminants covers shorter distance before reaching the concentration level of 0.5 mg/l at supply well 2.

Figure 4.21 shows the breakthrough profiles for benzene, toluene, ethylbenzene and xylene, respectively for different hydraulic conductivity at supply well 2 downgradient of the contaminant source. It is observed that as hydraulic conductivity increases, plumes take less time to reach the saturation value. The results show the contaminant plume stabilisation time at different hydraulic conductivity and the maximum concentration of the contaminant at respected time. It is observed that the maximum concentration of the contaminants in supply well 2 is about 53 mg/l for the value of hydraulic conductivity 3×10^{-4} m/s and approximately 109 mg/l in case of hydraulic conductivity 5.78×10^{-4} m/s which is well above the MCLs as per Australian Drinking Water Guidelines.

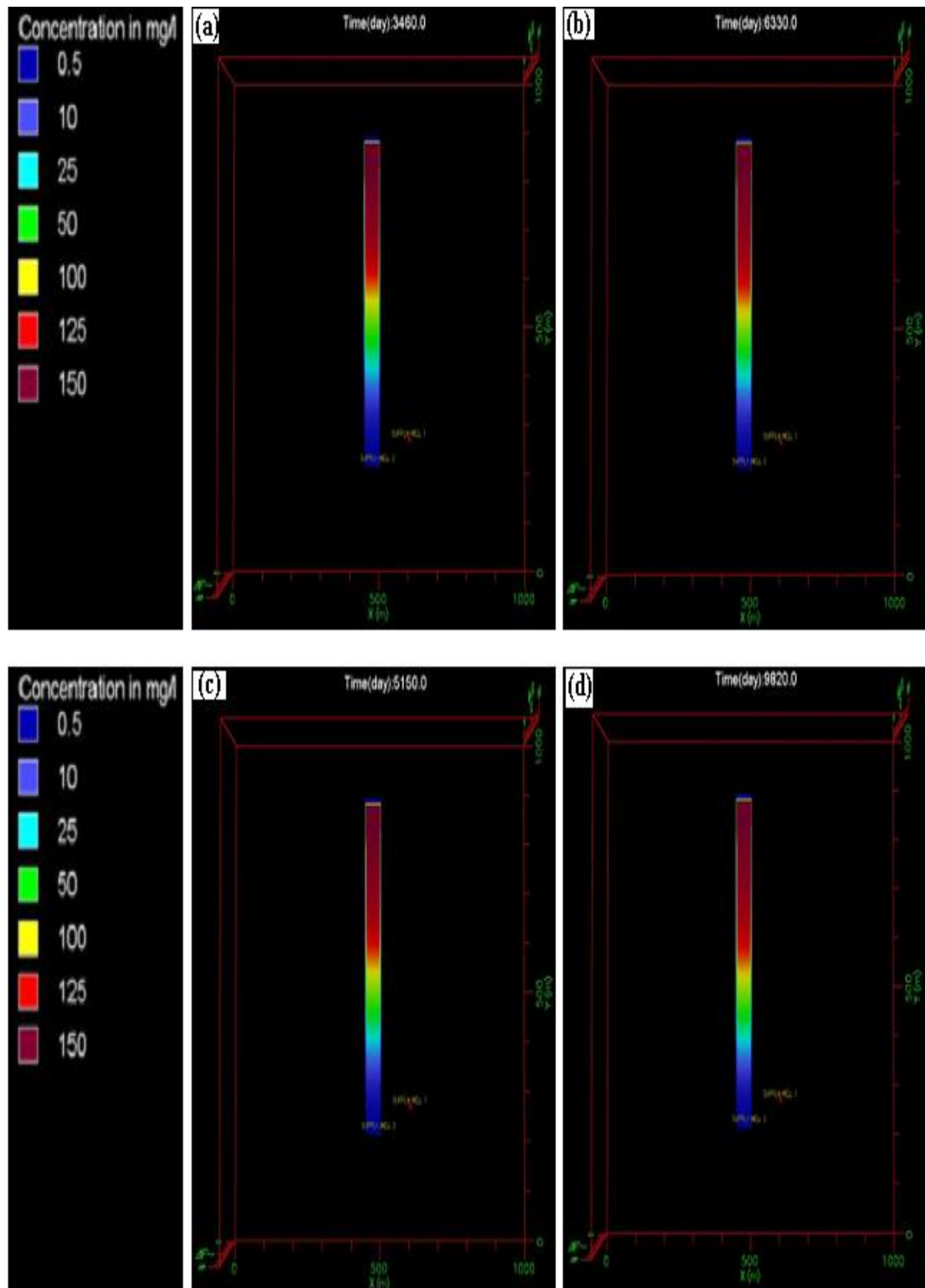


Figure 4.18: Contaminant Plume at Hydraulic Conductivity = 5.78×10^{-4} m/s for (a) Benzene (b) Toluene (c) Ethylbenzene (d) Xylene

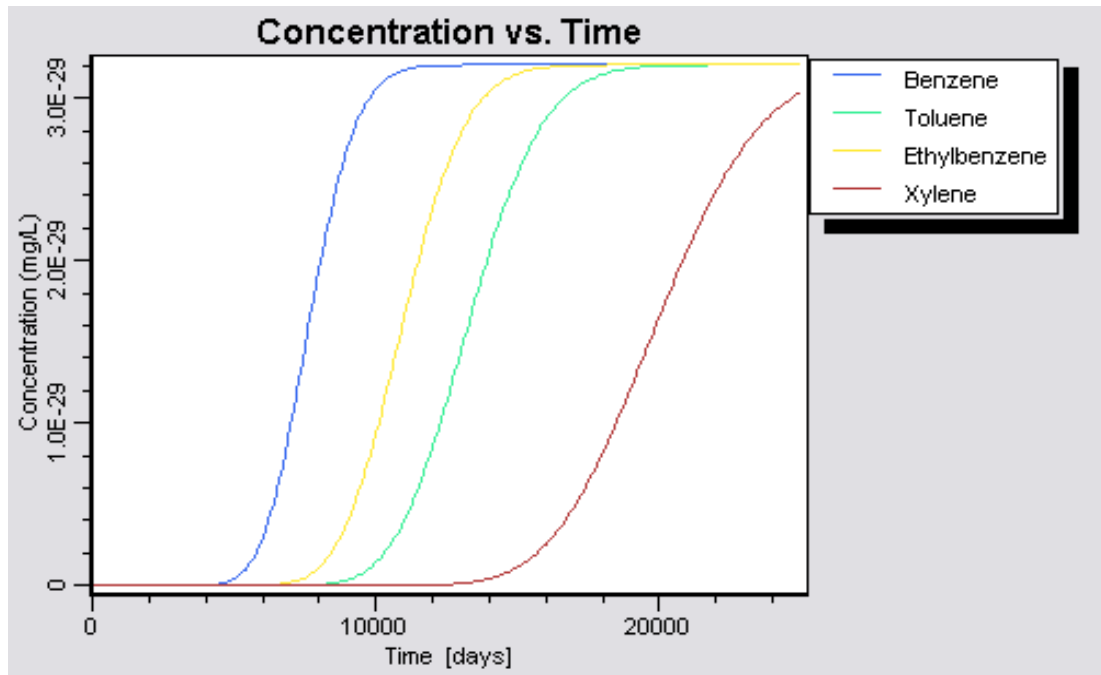


Figure 4.19: BTEX Concentration versus Time at Supply Well 1 downgradient of the Contaminant source for Hydraulic Conductivity = 5.78×10^{-4} m/s

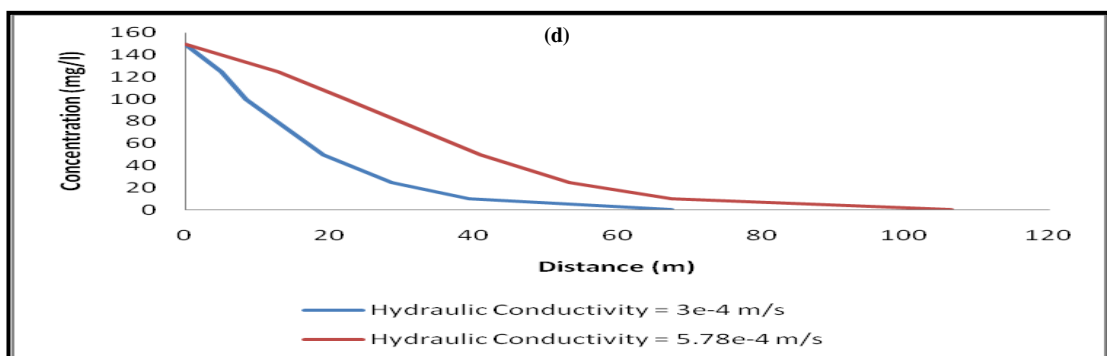
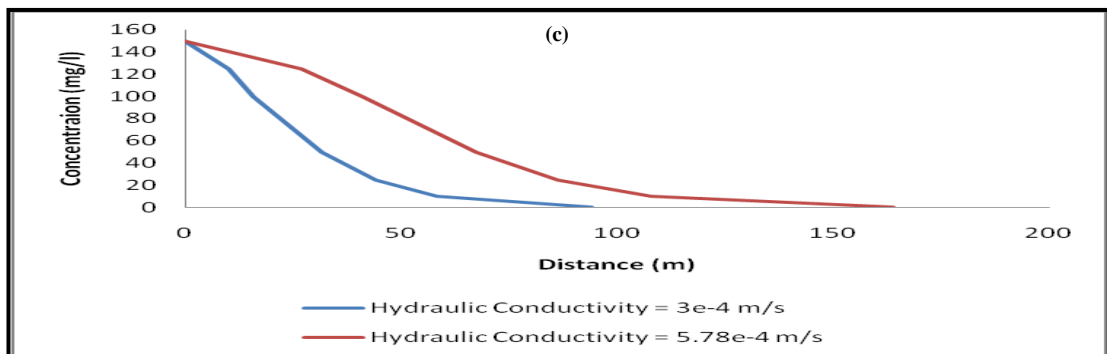
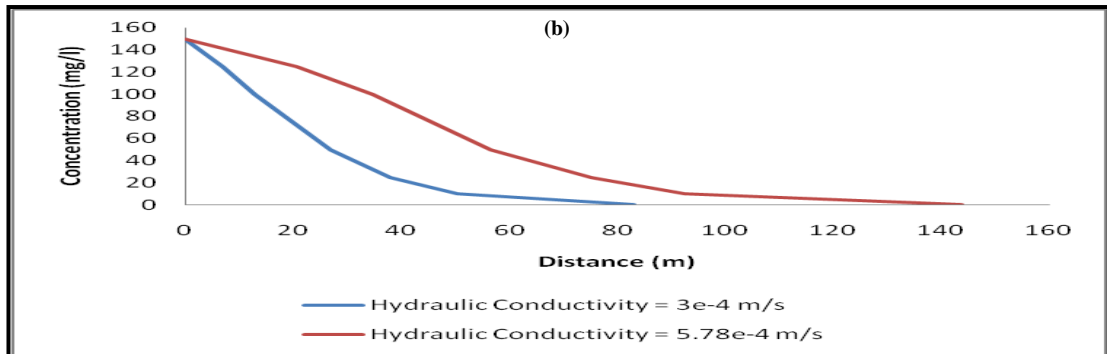
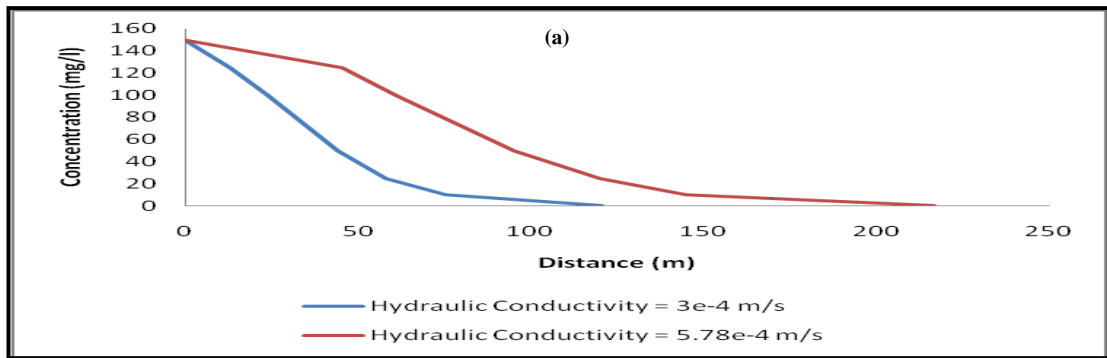


Figure 4.20: Concentration of Contaminants (BTEX) on Supply Well 2 along the Centerline of the Plume at 1000 days for (a) Benzene (b) Toluene (c) Ethylbenzene (d) Xylene

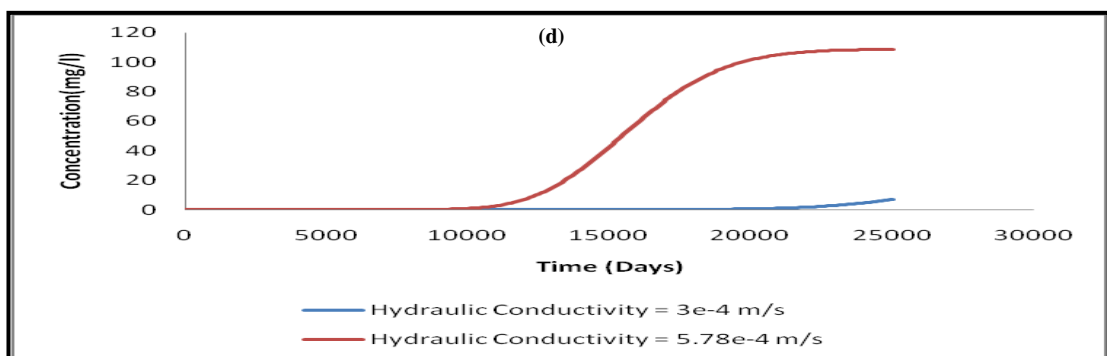
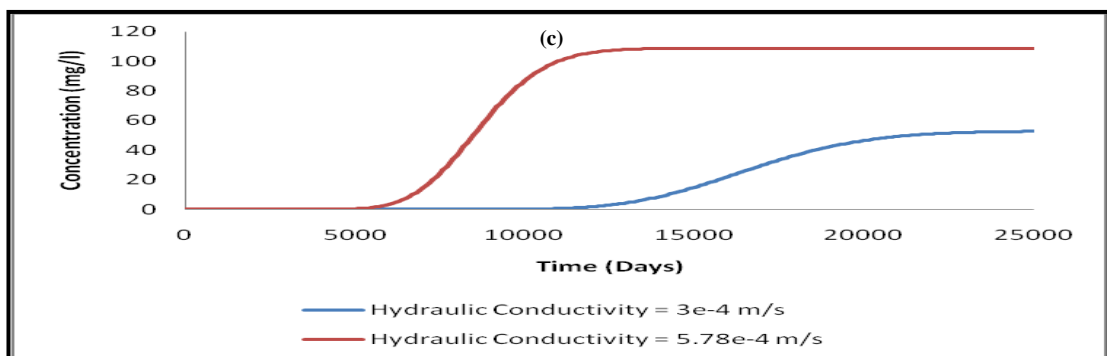
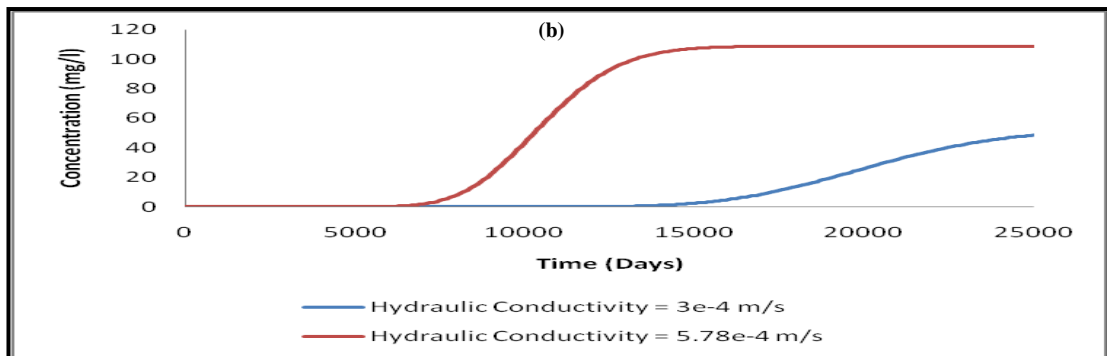
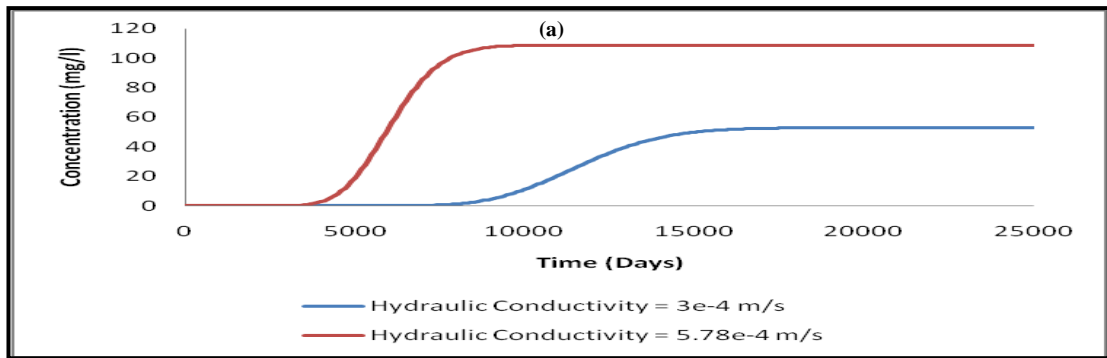


Figure 4.21: BTEX Concentration versus Time at Supply Well 2 downgradient of the Contaminant Source (a) Benzene (b) Toluene (c) Ethylbenzene (d) Xylene

4.5 Safety Bay Sand Hydrogeology

Safety Bay Sand is a surface layer of the Superficial formation that overlies Tamala Limestone and Becher Sand. It consists of white, unconsolidated, fine grained to medium grained quartz sand and shell fragments with traces of fine grained, black, heavy minerals. The Safety Bay Sand unit varies in known thickness to a maximum of about 24 m (Davidson 1995).

Hydrogeological characteristic of Safety Bay Sand are summarized in Table 4.7. In western sandy (Safety Bay Sand) area of Perth coastal plain, hydraulic conductivity value ranges from 1 m/d (1.16×10^{-5} m/s) to more than 50 m/d (5.78×10^{-4} m/s) (Davidson 1995). Smith et al. (2003) determined groundwater velocities ranged from 10 to 150 m/year. Davidson (1995) determined effective porosity and specific yield range from 30% to 40% from aquifer tests. Johnston and Desvignes (2002) assigned F_{oc} (Fraction of organic carbon content in Safety Bay Sand) ranges from 0.01% to 0.03%.

Flow Properties	
Hydraulic Conductivity (K)	1 to 50 m/day
Effective Porosity (P_e)	30% to 40%
Specific Yield (S_y)	30% to 40%
Thickness (b)	Maximum 24 m
Contaminant Transport Properties	
Organic Carbon Content (F_{oc})	0.01 to 0.03 %
Longitudinal Dispersivity (α_L)	0.026 m

Table 4.7: Flow and Contaminant Transport Properties of Safety Bay Sand

4.5.1 Flow and Contaminant Transport Properties for Safety Bay Sand Model

Table 4.8 discussed typical flow and contaminant transport properties which were used for the simulation. The ratio of vertical hydraulic conductivity (K_z) to horizontal hydraulic conductivity (K_x) was assumed to be 0.1. Specific Storage was assumed as 1×10^{-5} 1/m in this simulation. Groundwater velocity found from equation 4.3 and was set to be 122 m/year for this simulation. Based on Gelhar, Welty and Rehfeldt (1992), horizontal transverse dispersivity (α_{TH}) is typically of the order of 10 percent of the longitudinal dispersivity (α_L) and vertical transverse dispersivity (α_{TV}) is typically of the order of 1 percent of the longitudinal dispersivity (α_L). So, α_{TH}/α_L was set as 0.1 and α_{TV}/α_L was set as 0.01 in all simulations. Bulk density of the aquifer material was assumed to be 1700 kg/m^3 .

Flow Properties	
Horizontal Hydraulic Conductivity (K_x, K_y)	$5.78 \times 10^{-4} \text{ m/s}$
Vertical Hydraulic Conductivity (K_z)	$5.78 \times 10^{-5} \text{ m/s}$
Effective Porosity (P_e)	30 %
Total Porosity (P_T)	35 %
Specific Yield (S_y)	30 %
Specific Storage (S_s)	$1 \times 10^{-5} \text{ 1/m}$
Layer Thickness (b)	Maximum 6 m
Contaminant Transport Properties	
Organic Carbon Content (F_{oc})	0.01 to 0.03%
Longitudinal Dispersivity (α_L)	0.026 m
Horizontal Transverse Dispersivity (α_{TH})/Longitudinal Dispersivity (α_L)	0.1
Vertical Transverse Dispersivity (α_{TV})/Longitudinal Dispersivity (α_L)	0.01

Table 4.8: Typical Values of Flow and Transport Properties of Safety Bay Sand Model

Figure 4.22 shows the breakthrough profiles for benzene, toluene, ethylbenzene and xylene, respectively for Safety Bay Sand at supply well 2 downgradient of the contaminant source. It was found that all the contaminants travelled at the same velocity. The figure shows that the maximum concentration of each of the contaminants in supply well 2 reached 109 mg/l after 4200 days from the source and this concentration of contaminant is well above the MCLs as per Australian Drinking Water Guidelines. On the other hand, Figure 4.23 demonstrated that BTEX concentration reached approximately 3.3×10^{-31} mg/l in supply well 1 for 25000 days which is far below than the MCLs.

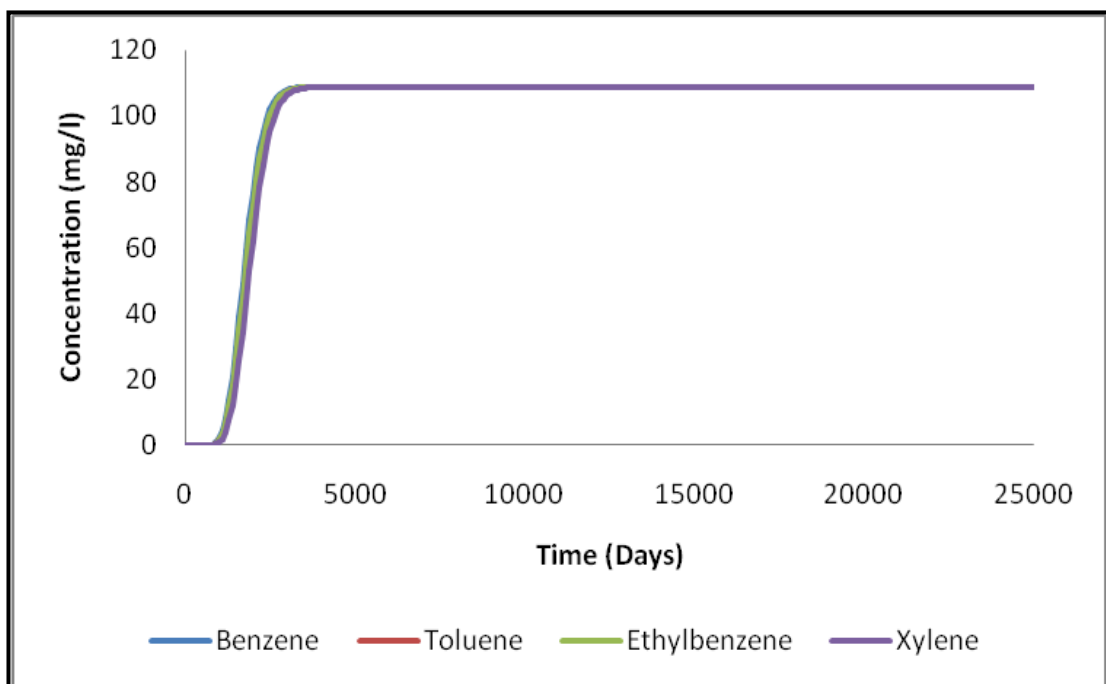


Figure 4.22: BTEX Concentration versus Time at Supply Well 2 downgradient of the Contaminant Source (a) Benzene (b) Toluene (c) Ethylbenzene (d) Xylene

Figure 4.24 shows the contaminant plume for BTEX contaminant to reach the well from contaminant source. This Figure shows that the effect of sorption is negligible because all the contaminants took the same time to reach the well from the contaminant source. Thus, it can be concluded that there is no effect of sorption because of the low organic carbon content ($F_{oc} \leq 0.01\%$) and low clay mineral content observed in this type of soil, retardation of BTEX is not likely to be a significant process affecting solute transport at this site, so the retarded solute transport velocity can be assumed to be equal to the groundwater velocity.

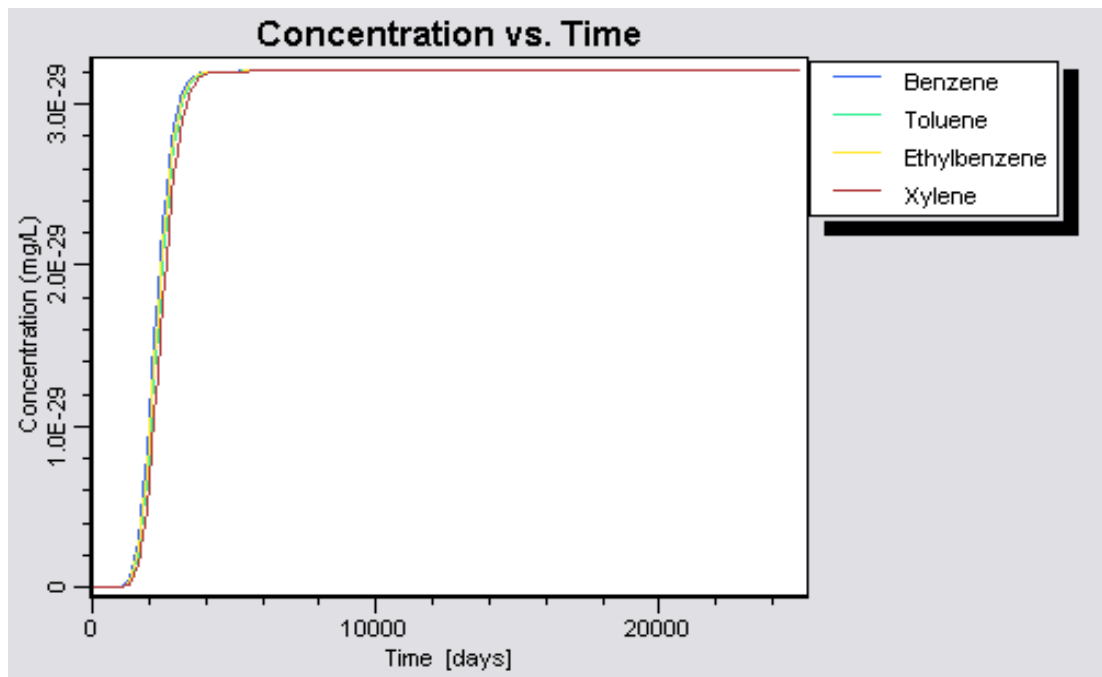


Figure 4.23: BTEX Concentration versus Time at Supply Well 1 downgradient of the Contaminant Source

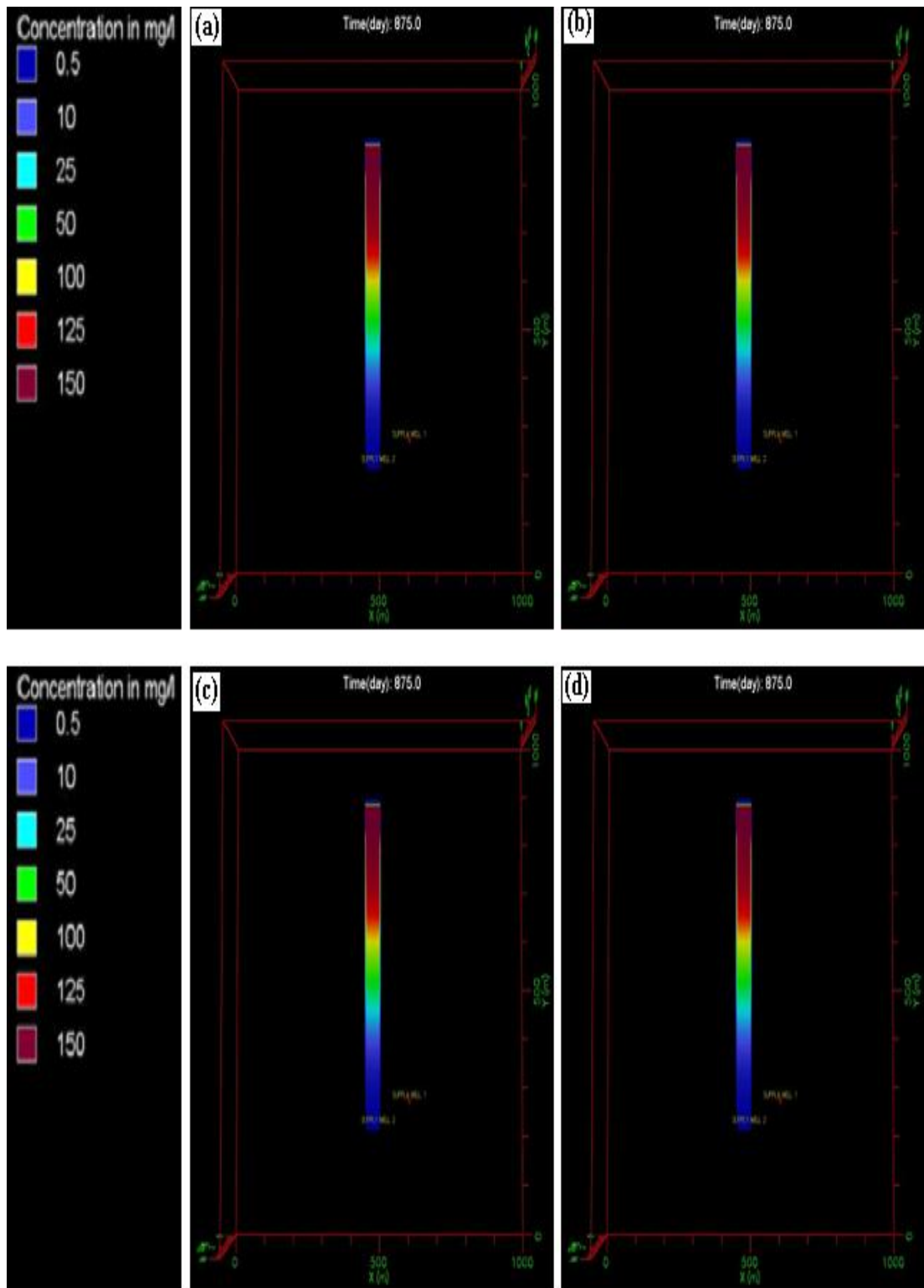


Figure 4.24: Contaminant Plume in Safety Bay Sand for (a) Benzene (b) Toluene (c) Ethylbenzene (d) Xylene

4.6 Tamala Limestone Hydrogeology

Tamala Limestone is a creamy white to yellow; calcareous eolianite (contains greater than 50% of calcium carbonate). It consists of various proportions of quartz sand, fine-grained to medium grained shell fragments and minor clay lenses. The Tamala Limestone unit varies in known thickness to a maximum of about 110 m (Davidson 1995).

Hydrogeological and contaminant transport properties of Tamala Limestone are summarized in Table 4.9. In the eastern margin of the Tamala Limestone, hydraulic conductivities ranged from 100 m/d (1.16×10^{-4} m/s) to 1000 m/d (1.16×10^{-3}) (Davidson 1995). Davidson (1995) determined groundwater velocities range from 200 to more than 2000 m/year, effective porosity and specific yield range from 30% to 40% from pump and slug test. Johnston and Desvignes (2002) found F_{oc} (Fraction of organic carbon content in Safety Bay Sand) approximate 0.01%.

Flow Properties	
Hydraulic Conductivity (K)	100 to 1000 m/day
Effective Porosity (P_e)	30% to 40%
Specific Yield (S_y)	30% to 40%
Thickness (b)	Maximum 110 m
Contaminant Transport Properties	
Organic Carbon Content (F_{oc})	0.01 %
Longitudinal Dispersivity (α_L)	0.026 m

Table 4.9: Flow and Contaminant Transport Properties of Tamala Limestone

4.6.1 Flow and Contaminant Transport Properties for Tamala Limestone Model

Table 4.10 discussed typical flow and contaminant transport properties which were used for this simulation. The ratio of vertical hydraulic conductivity (K_z) to horizontal hydraulic conductivity (K_x) was set to be 0.1. Specific Storage was assumed as 1×10^{-5} 1/m in this simulation. Groundwater velocity found from equation 4.3 and was set to be 2435 m/year for this simulation. Based on Gelhar, Welty and Rehfeldt (1992), horizontal transverse dispersivity (α_{TH}) is typically on the order of 10 percent of the longitudinal dispersivity (α_L) and vertical transverse dispersivity (α_{TV}) is typically on the order of 1 percent of the longitudinal dispersivity (α_L). So, α_{TH}/α_L was set as 0.1 and α_{TV}/α_L was set as 0.01 in all simulations. Bulk density of the aquifer material was assumed to be 1700 kg/m^3 .

Flow Properties	
Horizontal Hydraulic Conductivity (K_x, K_y)	$1.16 \times 10^{-3} \text{ m/s}$
Vertical Hydraulic Conductivity (K_z)	$1.16 \times 10^{-4} \text{ m/s}$
Effective Porosity (P_e)	30 %
Total Porosity (P_T)	35 %
Specific Yield (S_y)	30 %
Specific Storage (S_s)	$1 \times 10^{-5} \text{ 1/m}$
Layer Thickness (b)	Maximum 6 m
Contaminant Transport Properties	
Organic Carbon Content (F_{oc})	0.01%
Longitudinal Dispersivity (α_L)	0.026 m
Horizontal Transverse Dispersivity (α_{TH})/Longitudinal Dispersivity (α_L)	0.1
Vertical Transverse Dispersivity (α_{TV})/Longitudinal Dispersivity (α_L)	0.01

Table 4.10: Typical Values of Flow and Transport Properties of Tamala Limestone Model

Figure 4.25 shows the breakthrough profiles for benzene, toluene, ethylbenzene and xylene, respectively for Tamala Limestone at supply well 2 downgradient of the contaminant source. It is found that all the contaminants travel at the same velocity. The figure shows that the maximum concentration of each of the contaminants in supply well 2 reached 130 mg/l after 3400 days from the source and this concentration of contaminant is well above the MCLs as per Australian Drinking Water Guidelines. On the other hand, Figure 4.26 demonstrated that BTEX concentration approximately 1.5×10^{-28} mg/l in supply well 1 for 25000 days which is far below than the MCLs.

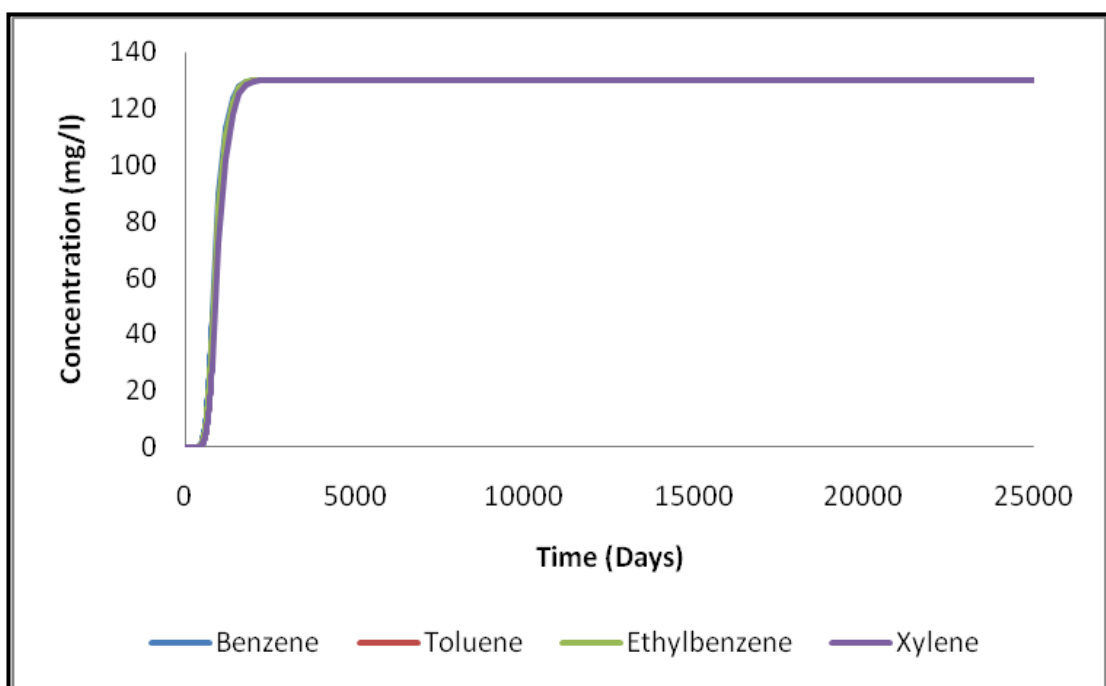


Figure 4.25: BTEX Concentration versus Time at Supply Well 2 downgradient of the Contaminant Source (a) Benzene (b) Toluene (c) Ethylbenzene (d) Xylene

Figure 4.27 shows the contaminant plume for BTEX contaminant time to reach the well from contaminant source. This Figure shows that the sorption is negligible because all the contaminants take the same time to reach the well from the contaminant source. Thus, it can be concluded that there is no effect of sorption because of the low organic carbon content ($F_{oc} \leq 0.01\%$) and low clay mineral content observed in this type of soil, retardation of BTEX is not likely to be a significant process affecting solute transport at this site, so the retarded solute transport velocity can be assumed to be equal to the groundwater velocity.

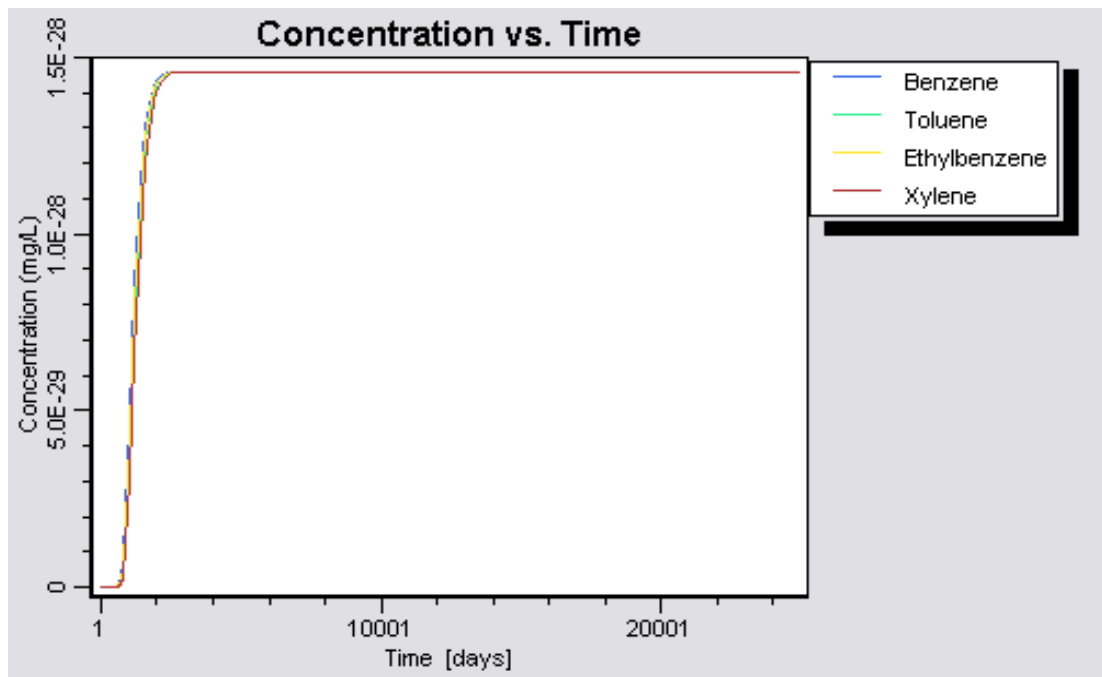


Figure 4.26: BTEX Concentration versus Time at Supply Well 1 downgradient of the Contaminant Source

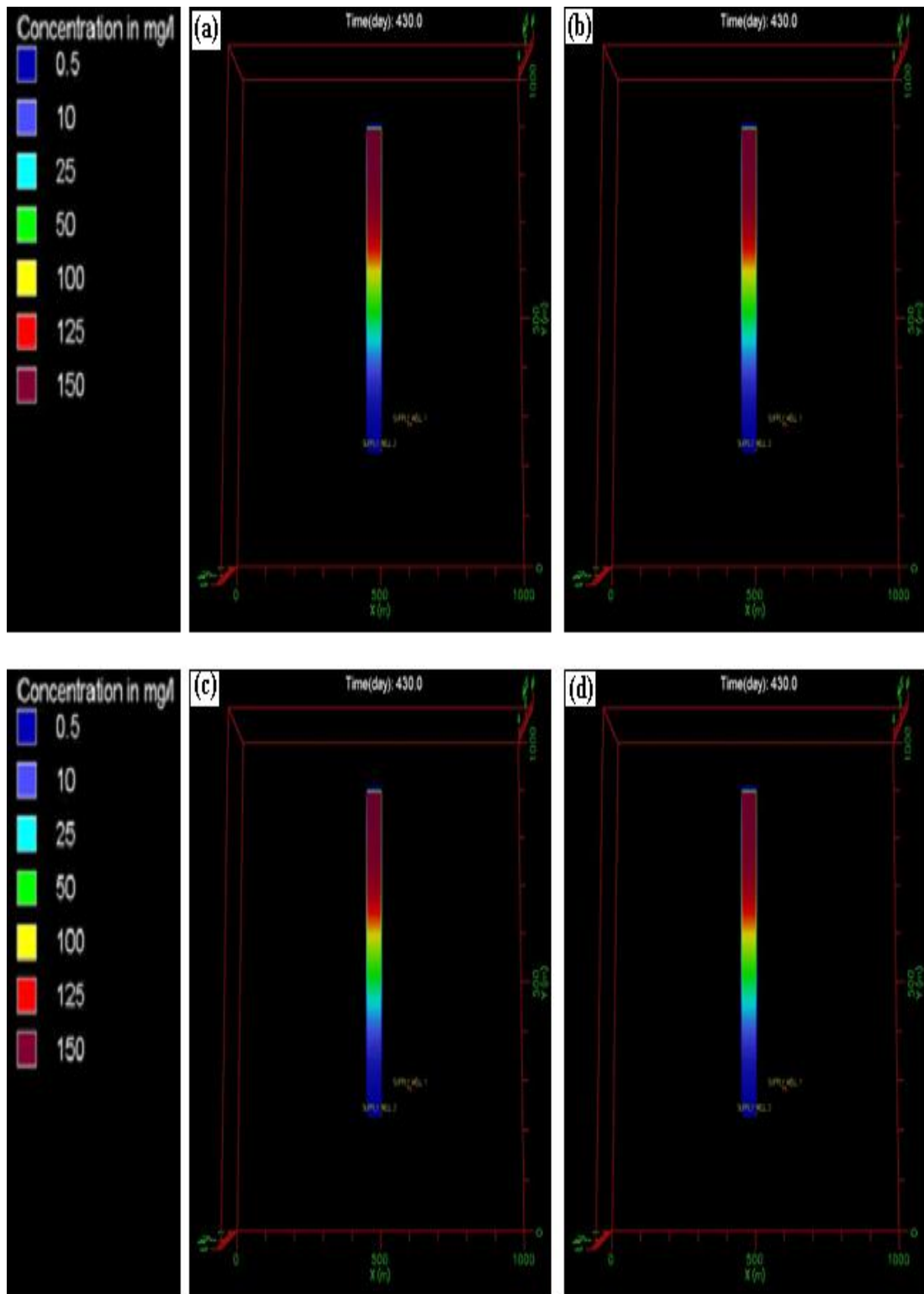


Figure 4.27: Contaminant Plume in Tamala Limestone for (a) Benzene (b) Toluene (c) Ethylbenzene (d) Xylene

4.7 Comparison of BTEX Transport in Three Types of Soil

This study presents the effects on BTEX contaminant plume behaviour for major types of soil found in Perth region: Bassendean Sand, Safety Bay Sand and Tamala Limestone. Data was taken from Table 4.3 for Bassendean Sand, Table 4.8 for Safety Bay Sand and Table 4.10 for Tamala Limestone for the comparison of these three types of soil.

Table 4.11 depicts the comparison of three different types of aquifer sediments found in Perth unconfined Superficial aquifer at 5000 days. The results show the concentration of BTEX at two different locations along the path of the plume. As observed from the data, the concentration in Well 1, which is slightly away from the direct path of the contaminants plume, has a negligible concentration of the pollutant whereas the concentration in observation Well 2 is significantly higher. It points out the effect of the soils axial dispersivity for the contaminants. As also observed from the simulation, the soil with higher F_{oc} , tends to retard the transport of pollutants. The concentration of the pollutants in Well 1 of Bassendean Sand^b shows zero concentration of the pollutants and similarly for other types of aquifer sediments, concentration of contaminants is far below the Australian Drinking Water Guidelines at 5000 days. Thus, it can be said that Well 1 is unaffected by contaminants in all three types of aquifer sediments found in Perth unconfined Superficial Aquifer. Table 4.11 also shows the concentration of BTEX contaminants in Well 2 at 5000 days for three different types of aquifer sediments found in Perth unconfined superficial aquifer. The result demonstrates slow migration of BTEX contaminants in Bassendean Sand^a and Bassendean Sand^b whereas in Tamala Limestone and Safety Bay Sand, contaminants reach concentrations close to source level concentrations above the Australian Drinking Water Standards in less than 1000 days (for Safety Bay Sand, it took 875 days and for Tamala Limestone, it took 430 days). These two later types of soil have low organic carbon content ($F_{oc} \leq 0.01\%$) and low clay mineral content. It also demonstrates the BTEX contaminants concentration in Well 2 at 5000 days is relatively higher than the Australian Drinking Water Guidelines in all three types of sand aquifer found in Perth unconfined superficial aquifer.

Figure 4.28 shows the simulation results of benzene contaminant plume for Bassendean Sand, Safety Bay Sand and Tamala Limestone at 400 days from downgradient distance from the contaminant source. The result demonstrates that for Bassendean Sand the sorption effect is dominant and which is seen by the slow migration of contaminants whereas in Tamala Limestone and Safety Bay Sand it is negligible because these two types of soil have low organic carbon content ($F_{oc} \leq 0.01\%$) and low clay mineral content. Thus, retardation of BTEX is not likely to be a significant process affecting solute transport at assumed hypothetical contaminated site, so the retarded solute transport velocity can be assumed to be equal to the groundwater velocity. Though Safety Bay Sand and Tamala Limestone, it can be notified from the results that the benzene in Safety Bay Sand covers the shorter distance than the Tamala Limestone, this is due to the effect of groundwater velocity. Tamala Limestone has a higher groundwater velocity than the Safety Bay Sand.

Figure 4.7a, Figure 4.22 and Figure 4.25 show the breakthrough curve for benzene contaminants at supply well 2 for Bassendean Sand, Safety Bay Sand and Tamala Limestone respectively. The results revealed that benzene reaches supply well 2 most rapidly in Tamala Limestone than that of either Safety Bay Sand or Bassendean Sand and it also shows that the maximum steady state concentration in supply well 2 is much more higher in Tamala Limestone than the other two types of soil. Similar effects were also found for toluene, ethylbenzene and xylene. Thus, it can be said that contaminated sites found in Tamala Limestone needs immediate remediation of contaminants to bring down the contaminants concentration in groundwater.

Sediments	Benzene Concentration in Supply Well 1 (mg/l)	Toluene Concentration in Supply Well 1 (mg/l)	Ethylbenzene Concentration in Supply Well 1 (mg/l)	Xylene Concentration in Supply Well 1 (mg/l)
Bassendean Sand ^a	2.29 X 10 ⁻³¹	3.91 X 10 ⁻³²	9.32 X 10 ⁻³²	2.37 X 10 ⁻³³
Bassendean Sand ^b	7.50 X 10 ⁻³⁹	0	0	0
Safety Bay Sand	3.20 X 10 ⁻²⁹	3.20 X 10 ⁻²⁹	3.20 X 10 ⁻²⁹	3.20 X 10 ⁻²⁹
Tamala Limestone	1.46 X 10 ⁻²⁸	1.46 X 10 ⁻²⁸	1.46 X 10 ⁻²⁸	1.46 X 10 ⁻²⁸
	Benzene Concentration in Supply Well 2 (mg/l)	Toluene Concentration in Supply Well 2 (mg/l)	Ethylbenzene Concentration in Supply Well 2 (mg/l)	Xylene Concentration in Supply Well 2 (mg/l)
Bassendean Sand ^a	44	21	30	4.5
Bassendean Sand ^b	0.00075	3.61 X 10 ⁻¹⁰	9.58 X 10 ⁻⁸	8.24 X 10 ⁻¹⁶
Safety Bay Sand	109	109	109	109
Tamala Limestone	130	130	130	130

Table 4.11: Comparison of Three different types of Aquifer Sediments found in Perth Unconfined Superficial Aquifer at 5000 days

^a Bassendean Sand at Fraction of Organic Carbon Content = 0.08%, Effective Porosity = 28% and Hydraulic Conductivity = 3 X 10⁻⁴ m/s

^b Bassendean Sand at Fraction of Organic Carbon Content = 0.6%, Effective Porosity = 28% and Hydraulic Conductivity = 3 X 10⁻⁴ m/s

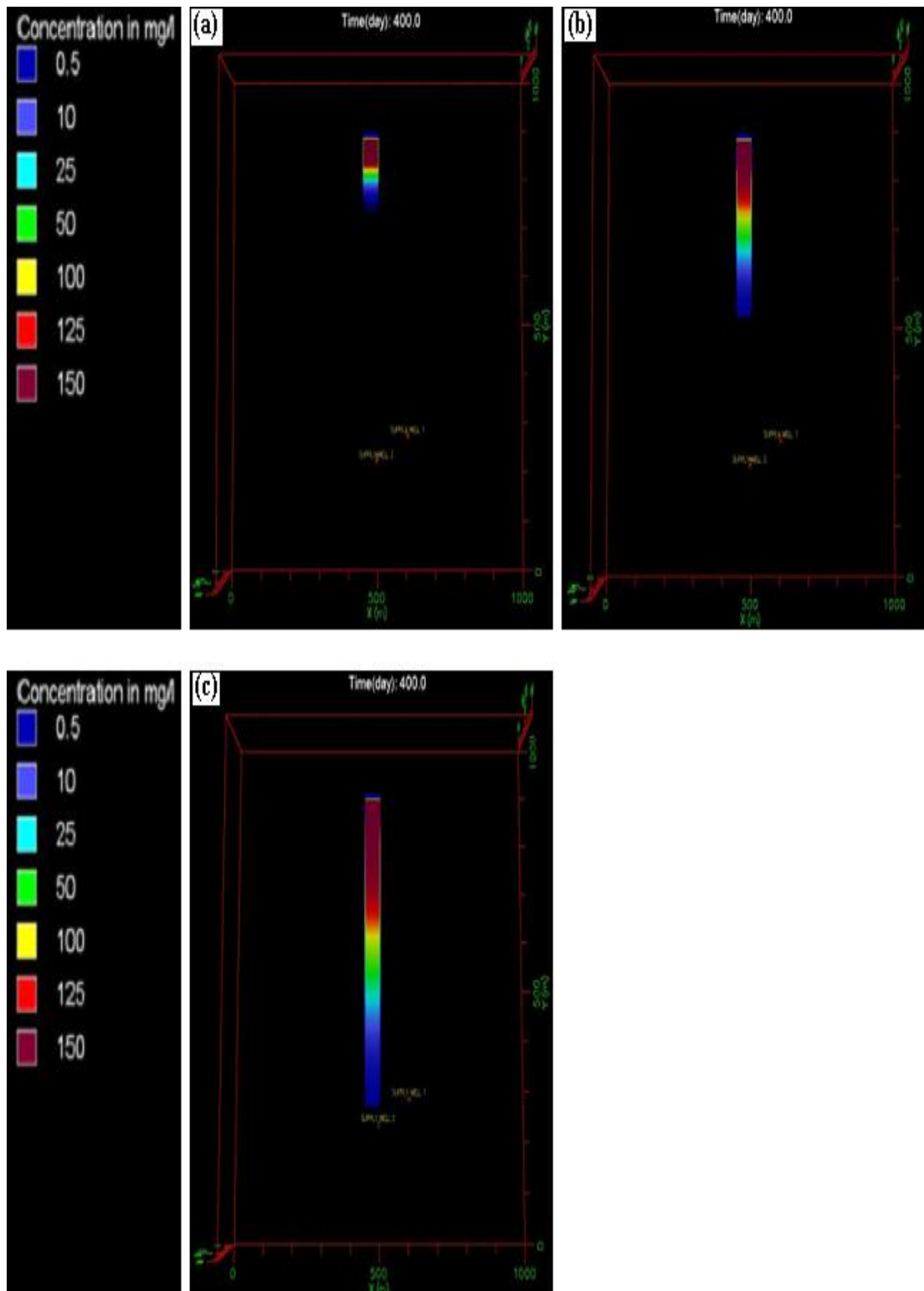


Figure 4.28: Benzene Contaminant Plume at 400 days for (a) Bassendean Sand (b) Safety Bay Sand (c) Tamala Limestone

CHAPTER 5

CONCLUSIONS AND RECOMMENDATIONS FOR FUTURE WORK

5.1 Summary/Conclusions of Contaminant Transport Model Studies

It is clear that the conclusions stemming from the current study are restricted to the hypothetical sites described earlier and should not be applicable in general. This study involves the analysis of the varying model input parameters to determine the impact of different parameter values on the model outputs.

A hypothetical model was selected using the Domenico solution to the advection-dispersion-sorption equation to demonstrate the effect of the different hydrogeological and contaminant transport parameters on the BTEX contaminant plume behaviour at 560.5 m downgradient from the contaminant source of Perth unconfined aquifer.

The simulations were performed using MODFLOW 2000 and MT3DMS. The MODFLOW 2000 was used to simulate the groundwater flow under various site conditions and MT3DMS was used to simulate the BTEX plume migration.

The effect of individual parameters on the contaminant plume is presented below.

Effect of Sorption on Contaminant Plume

Changes in contaminant concentration versus time profiles as a result of sorption were evaluated by plotting the profiles from the source to downgradient locations for different fractions of organic carbon contents for Bassendean Sand. For this analysis, a longitudinal dispersivity of 0.026 m was assumed. The time required to reach steady-state was observed to increase as the fraction of organic carbon content was increased. It was realized that sorption effect did not significantly change the steady-state concentration of the BTEX plume. It can also be concluded that benzene reached the supply well fastest when compared to that of toluene, ethylbenzene and xylene. Thus it can be said that the higher the distribution coefficient, the greater the

sorption to the aquifer matrix and thus the higher will be the travel time for the contaminants.

It can also be concluded that there was no significant effect of sorption because of the low organic carbon content ($F_{oc} \leq 0.01\%$) and low clay mineral content in both the Safety Bay Sand and Tamala Limestone. Retardation of BTEX is not likely to be a significant process affecting solute transport, so the retarded solute transport velocity can be assumed to be equal to the groundwater velocity.

Effect of Effective Porosity on Contaminant Plume

Bassendean Sand has a range of effective porosities; therefore this model was simulated to determine the impact of varying effective porosities on the contaminant plume behaviour.

Breakthrough profiles due to variations in effective porosity were obtained by plotting the profiles from the source to downgradient locations for different values of effective porosity for Bassendean Sand assuming constant dispersivity and organic carbon content. It can be concluded from this study that the time required for BTEX contaminants to reach steady-state increases as the effective porosity increases, e.g., the simulated benzene plume length (defined by 0.5 mg/l contour) decreased by about 7% when the effective porosity increased by 29% (i.e., from 20% to 28%) at 1000 days. This exhibits that model simulations are not very sensitive to small errors in porosity and that small errors are introduced by using effective porosity values from literature. It was also found that the effective porosity did not change the steady-state concentration.

Effect of Hydraulic Conductivity on Contaminant Plume

Observed values of hydraulic conductivity for Bassendean Sand ranged from 10 m/day to 50 m/day; therefore sensitivity analyses have been performed to determine the effect of varying the hydraulic conductivity on the contaminant plume behaviour.

Breakthrough profiles due to variability in hydraulic conductivity were evaluated by plotting the profiles from the source to downgradient locations for different values of hydraulic conductivity for Bassendean Sand assuming constant dispersivity and

organic carbon content. It can be concluded from this study that the BTEX contaminant plume advects faster with increasing hydraulic conductivity, e.g., the simulated benzene plume length (defined by 0.5 mg/l contour) increased approximately 45% when the hydraulic conductivity increased by 52% (i.e., from 26 m/day to 50 m/day) at 1000 days. This exhibits that model simulations are most sensitive to the change in hydraulic conductivity. It can be inferred that hydraulic conductivity has a significant effect on steady state concentration of the plume.

Comparison between Bassendean Sand, Safety Bay Sand and Tamala Limestone

This study considered the three types of soil found in Perth region unconfined Superficial aquifer. By comparing these types of soil together, it can be concluded that the movement of contaminant in Tamala Limestone is most rapid. Therefore, contaminated sites found in Tamala Limestone needs immediate remediation of contaminants to bring down the contaminants concentration in the groundwater.

Finally, it can be concluded that a groundwater flow and solute transport model is useful for predicting the future extent and concentration of a contaminant plume, which can be useful to estimate the potential impacts of contaminants on human health and the environment. Prediction of spatial and temporal movement of contaminant plumes by using a three-dimensional groundwater and solute advective-dispersive-sorptive transport model is an important tool in assessing the need for remedial design at contaminated sites.

5.2 Recommendations for Future Work

This study has identified the following future research directions:

1. It is recommended to collect the site-specific characterization data (hydrogeological and contaminant transport parameters) and calibrate the model to match the given field condition. This will not only improve the reliability of the model but also help to verify the model simulated results to the actual field conditions.
2. Groundwater flow and solute transport model simulations are used to include the properties related to heterogeneity, as geological structures of aquifers are not often homogeneous and uniform. The assumption of homogeneous conditions in groundwater flow problems generally gives an appropriate estimation. In contaminant transport problems however, the heterogeneous aquifer may have a significant influence on the contaminant plume behaviour, as low hydraulic conductivity zones in the aquifer may slow down the flow and reduce the spreading of a contaminant plume, whereas high conductivity zones may cause sudden changes in contaminant plume concentrations. Therefore, it is recommended that heterogeneity in actual model prediction should be considered.
3. It is recommended that the effect of changes in climatic factors such as mean annual rainfall, evapotranspiration rate should be studied on groundwater flow and contaminant transport model.
4. Various remediation technologies to remove the contamination from the groundwater should be evaluated and experimental studies conducted.

REFERENCES

- Alvarez, P. J. J., P. J. Anid, and T. M. Vogel. 1991. Kinetics of Aerobic Biodegradation of Benzene and Toluene in Sandy Aquifer Material. *Biodegradation* 2 (1):43-51.
- Alvarez, P. J. J., L. A. Cronkhite, and C. S. Hunt. 1998. Use of Benzoate to Establish Reactive Buffer Zones for Enhanced Attenuation of BTX Migration: Aquifer Column Experiments. *Environmental Science & Technology* 32 (4):509-515.
- Anderson, M. P., and W. W. Woessner. 1992. *Applied Groundwater Modeling: Simulation of Flow and Advective Transport*. San Diego: Academic Press.
- Appleyard, S. J. 1995. Investigation of Ground Water Contamination by Fenamiphos and Atrazine in a Residential Area: Source and Distribution of Contamination. *Ground Water Monitoring & Remediation* 15 (4):110-113.
- . 1996. Impact of Liquid Waste Disposal on Potable Groundwater Resources near Perth, Western Australia. *Environmental Geology* 28 (2):106-110.
- Appleyard, S., S. Wong, B. Willis-Jones, J. Angeloni, and R. Watkins. 2004. Groundwater Acidification Caused by Urban Development in Perth, Western Australia: Source, Distribution, and Implications for Management. *Australian Journal of Soil Research* 42 (5):579-585.
- American Society for Testing and Materials (ASTM). 1994. 1994 Annual Book of ASTM Standards: Emergency Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites (Designation: ES 38-94). West Conshohocken, PA: American Society for Testing and Materials.
- Australian and New Zealand Environment and Conservation Council (ANZECC) and Agricultural and Resource Management Council of Australia and New Zealand (ARMCANZ). 2000. Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Canberra: National Water Quality Management Strategy.

- Australian Drinking Water Guidelines (ADWG). 2004. National Medical and Health Research Council (NHMRC).
- Barry, D. A., H. Prommer, C. T. Miller, P. Engesgaard, A. Brun, and C. Zheng. 2002. Modelling the Fate of Oxidisable Organic Contaminants in Groundwater. *Advances in Water Resources* 25 (8-12):945-983.
- Bear, J. 1972. *Dynamics of Fluids in Porous Media*. New York: Elsevier.
- . 1979. *Hydraulics of Groundwater*. New York: McGraw-Hill Companies.
- Becker, M. W., S. A. Collins, D. W. Metge, R. W. Harvey, and A. M. Shapiro. 2004. Effect of Cell Physicochemical Characteristics and Mobility on Bacterial Transport in Groundwater. *Journal of Contaminant Hydrology* 69 (3-4):195-213.
- Benker, E., G. B. Davis, S. Appleyard, D. A. Barry, and T. R. Power. 1994. Groundwater Contamination by Trichloroethene (TCE) in a Residential Area of Perth: Distribution, Mobility, and Implications for Management.
- . 1996. Trichloroethene (TCE) Contamination in an Unconfined Sand Aquifer Underlying a Residential Area of Perth, Western Australia. *Hydrogeology Journal* 4 (1):20-29.
- Blair, P., and N. Turner. 2004. Groundwater—A Crucial Element of Water Recycling in Perth, Western Australia. Paper read at Meeting the Sustainable Water Challenge, at Bankstown, NSW.
- Chiang, W. H., and W. Kinzelbach. 2001. *3D-Groundwater Modeling with PMWIN*. Berlin, Germany: Springer.
- Clement, T. 2001. RT3Dv2.5—A Modular Computer Code for Simulating Reactive Multi-species Transport in 3-Dimensional Groundwater Systems. Richland, USA: Pacific Northwest National Laboratory, PNNL-SA-11720.

- Cohen, E. L., B. M. Patterson, A. J. McKinley, and H. Prommer. 2008. Zero valent iron remediation of a mixed brominated ethene contaminated groundwater. *Journal of Contaminant Hydrology*. In Press, Corrected Proof.
- Contaminated Sites Act 2003. 2006. Western Australia: Department of Environment and Conservation (DEC).
- Davidson, W. A. 1995. Hydrogeology and Groundwater Resources of the Perth Region, Western Australia. Bulletin 142. Western Australia Geological Survey.
- Davis, G. B., C. Barber, T. R. Power, J. Thierrin, B. M. Patterson, J. L. Rayner, and Qinglong Wu. 1999. The Variability and Intrinsic Remediation of a BTEX Plume in Anaerobic Sulphate-rich Groundwater. *Journal of Contaminant Hydrology* 36 (3-4):265-290.
- Davis, G. B., C. D. Johnston, J. Thierrin, T. R. Power, and B. M. Patterson. 1993. Characterizing the Distribution of Dissolved and Residual NAPL Petroleum Hydrocarbons in Unconfined Aquifers to Effect Remediation. *Journal of Australian Geology and Geophysics* 14 (2/3):243-248.
- Davis, S. N. 1969. Porosity and Permeability of Natural Materials. In *Flow Through Porous Media*. New York: ed. R.J.M. De Wiest, Academic Press.
- Day, S. R., S. F. O'Hannesin, and L. Marsden. 1999. Geotechnical Techniques for the Construction of Reactive Barriers. *Journal of Hazardous Materials* 67 (3):285-297.
- Di Molfetta, A., and R. Sethi. 2005. The First Permeable Reactive Barrier in Italy. Paper read at Consoil 2005 - Proceedings of the 9th International FZK/TNO Conference on Soil Water Systems, 3-5 October, at Bordeaux
- Domenico, P. A., and F. W. Schwartz. 1990. *Physical and Chemical Hydrogeology*. New York: John Wiley & Sons, Inc.
- . 1998. *Physical and Chemical Hydrogeology*. New York: John Wiley & Sons, Inc.

Enviro-Base Pro. 2005. Canada: Waterloo Hydrogeologic Inc.

Environmental Protection Authority (EPA) State of the Environment Report: Western Australia. Land Contamination. 2008. Department of Environment and Conservation 2007 [cited 16th June 2008]. Available from <http://www.soe.wa.gov.au/report/land/land-contamination.html>.

Ewing, R., S. Kang, J. Kim, and T. B. Stauffer. 2001. A Practical Three-Dimensional Estimation Technique for Spatial Distribution of Groundwater Contaminant Concentrations. *J. Korean Math. Soc* 38 (3):523-559.

Fetter, C. W. 1999. *Contaminant Hydrogeology*, Upper Saddle River, New Jersey: Prentice-Hall.

———. 2001. *Applied Hydrogeology*, Upper Saddle River, New Jersey: Prentice-Hall.

Freeze, R. A., and J. A. Cherry. 1979. *Groundwater*. New Jersey: Prentice Hall, Upper Saddle River.

Gavaskar, Arun R. 1999. Design and Construction Techniques for Permeable Reactive Barriers. *Journal of Hazardous Materials* 68 (1-2):41-71.

Gelhar, L. W. 1993. *Stochastic Subsurface Hydrology*. New Jersey: Printice Hall, Englewood Cliffs.

Gelhar, L. W., C. Welty, and K. R. Rehfeldt. 1992. A Critical Review of Data on Field-Scale Dispersion in Aquifers. *Water Resources Research* 28 (7):1955-1974.

Groundwater Pollution. 1998. Water Facts10. Water and River Commissions (WRC).

Harbaugh, A. W., E. R. Banta, M. C. Hill, and M. G. McDonald. 2000. MODFLOW-2000. The U. S. Geological Survey Modular Ground-Water Model-User Guide to Modularization Concepts and the GroundWater Flow Process: U. S. Geological Survey Open File Report.

- Hinwood, A. L., P. Horwitz, S. Appleyard, C. Barton, and M. Wajrak. 2006. Acid Sulphate Soil Disturbance and Metals in Groundwater: Implications for Human Exposure through Home Grown Produce. *Environmental Pollution* 143 (1):100-105.
- Howard, P. H. 1991. *Handbook of Environmental Degradation Rates*: CRC Press.
- Jameel, A. A., and J. Sirajudeen. 2006. Risk Assessment of Physico-Chemical Contaminants in Groundwater of Pettavaithalai Area, Tiruchirappalli, Tamilnadu – India. *Environmental Monitoring and Assessment* 123 (1):299-312.
- Johnson, A. I. 1967. Specific Yield-Compilation of Specific Yields for Various Materials: US Geological Survey Water-Supply Paper 1662-D.
- Johnston, C. D., and A. Desvignes. 2003. Evidence for Biodegradation and Volatilisation of Dissolved Petroleum Hydrocarbons during In Situ Air Sparging in Large Laboratory Columns. *Water, Air, & Soil Pollution: Focus* 3 (3):25-33.
- Johnston, C. D., J. L. Rayner, B. M. Patterson, and G. B. Davis. 1998. Volatilisation and Biodegradation during Air Sparging of Dissolved BTEX Contaminated Groundwater. *Journal of Contaminant Hydrology* 33 (3-4):377-404.
- Thomas, K. E., C. Byrnes, and R. D. Williams. 2006. A Visual MODFLOW and MT3D Model to Support Remediation of a Petroleum Contamination Site using In Situ Chemical Oxidation Paper read at Petroleum Hydrocarbons and Organic Chemicals in Groundwater: Prevention, Assessment, and Remediation, at Texas.
- Karickhoff, S. W., D. S. Brown, and T.A. Scott. 1979. Sorption of Hydrophobic Pollutants on Natural Sediments. *Water Research* 13 (3):241-248.
- Konikow, L. F., and T. E. Reilly. 1999. *Groundwater Modeling: The Handbook of Groundwater Engineering*. Boca Raton: CRC Press.

- Kumar, C. P. Groundwater Flow Models. National Institute of Hydrology. Roorkee. India. [cited 12th April, 2008] Available from <http://www.angelfire.com/nh/cpkumar/hydrology.html>
- Li, Q., B. Harris, C. Aydogan, M. Ang, and M. Tade. 2006. Feasibility of Recharging Reclaimed Wastewater to the Coastal Aquifers of Perth, Western Australia. *Process Safety and Environmental Protection* 84 (4):237-246.
- Lovanh, N., Y. K. Zhang, R. C. Heathcote, and P. J. J. Alvarez. 2000. Guidelines to Determine Site-Specific Parameters for Modeling the Fate and Transport of Monoaromatic Hydrocarbons in Groundwater. In *IDoN Resources (Editor)*.
- Lyman, W. J., P. J. Reidy, and B. Levy. 1992. *Mobility and Degradation of Organic Contaminants in Subsurface Environments*, CK Smoley, Inc., Chelsea Michigan. Chelsea, Michigan: C. K. Smoley, Inc.
- Managing Groundwater Use. 2000. Water Facts 11. Water and River Commissions (WRC).
- Mayer, K. U. 1999. A Numerical Model for Multicomponent Reactive Transport in Variably Saturated Porous Media, University of Waterloo, Ontario, Canada.
- McDonald, M. G., and A. W. Harbaugh. 1988. A Modular 3D Finite Difference Groundwater Flow Model: United States Geological Survey techniques of Water Resources Investigations
- Meggyes, T., and F. G. Simon. 2000. Removal of Organic and Inorganic Pollutants from Groundwater Using Permeable Reactive Barriers. Part 2. Engineering of permeable reactive barriers. *Land Contamination & Reclamation* 8 (3):175-187.
- Mohan, S., and M. Muthukumar. 2004. Modelling of Pollutant Transport in Ground Water. *Journal of the Institution of Engineers (India)* 85:22-32.
- Morris, D. A., and A. I. Johnson. 1967. Summary of Hydrologic and Physical Properties of Rock and Soil Materials as Analyzed by the Hydrologic

Laboratory of the US Geological Survey 1948-1960: USGS Water Supply Paper.

Neuman, S. P. 1990. Universal Scaling of Hydraulic Conductivities and Dispersivities in Geologic Media. *Water Resources Research* 26 (8):1749 - 1758.

Nixon, M.D. 1996. Thirteenth Report of the Standing Committee on Constitutional Affairs and Statutes Revision in Relation to a Petition Concerning Compulsory Connection to the Sewerage System. Western Australia: Legislative Council.

Ohio, EPA. 2007. Ground Water Flow and Fate and Transport Modeling. Division of Drinking and Ground Waters State of Ohio Environmental Protection Agency

Parkhurst, D. L., and C. A. J. Appelo. 1999. User's Guide to PHREEQC—A Computer Program for Speciation, Reaction-Path, 1D-Transport, and Inverse Geochemical Calculations In *Technical Report 99-4259*: US Geological Survey Water-Resources Investigations Report.

Parkhurst, D. L., K. L. Kipp, P. Engesgaard, and S. R. Charlton. 2004. PHAST—A Program for Simulating Groundwater Flow, Solute Transport, and Multicomponent Geochemical Reactions. In *Techniques and Methods 6-A8*: United States Geological Survey.

Patterson, B. M., P. D. Franzmann, J. L. Rayner, and G. B. Davis. 2000. Combining Coring and Suction Cup Data to Improve the Monitoring of Pesticides in Sandy Vadose Zones: A Field-Release Experiment. *Journal of Contaminant Hydrology* 46 (1-2):187-204.

Pinder, G. F. 2002. *Groundwater Modeling Using Geographical Information Systems*. New York: John Wiley & Sons.

Pollock, D. W. 1994. User's Guide for MODPATH/MODPATH-PLOT, Version 3: A Particle Tracking Post-processing Package for MODFLOW. The US

Geological Survey Finite-Difference Groundwater Flow Model. In *US Geological Survey Open-File Report*

- Prommer, H. 2002. PHT3D–A Reactive Multicomponent Transport Model for Saturated Porous Media. User’s Manual Version 1.0. The University of Edinburgh, UK: Contaminated Land Assessment and Remediation Research Centre.
- Prommer, H., D. A. Barry, and G. B. Davis. 2002. Modelling of Physical and Reactive Processes during Biodegradation of a Hydrocarbon Plume under Transient Groundwater Flow Conditions. *Journal of Contaminant Hydrology* 59 (1-2):113-131.
- Prommer, H., D. A. Barry, and C. Zheng. 2003. PHT3D- A MODFLOW/MT3DMS Based Reactive Multicomponent Transport Modeling. *Ground Water* 41 (2):247-257.
- Prommer, H., G. B. Davis, and D. A. Barry. 1999. Geochemical Changes during Biodegradation of Petroleum Hydrocarbons: Field Investigations and Biogeochemical Modelling. *Organic Geochemistry* 30 (6):423-435.
- Prommer, H., G. B. Davis, D. A. Barry, and C. T. Miller. 2003. Modelling the Fate of Petroleum Hydrocarbons in Groundwater. In *Health and Environmental Assessment of Site Contamination*.
- Rifai, H. S., R. C. Borden, J. T. Wilson, and C. H. Ward. 1995. Intrinsic Bioattenuation for Subsurface Restoration. In *Intrinsic Bioremediation*.
- Rivett, M. O., J. Drewes, M. Barrett, J. Chilton, S. Appleyard, H. H. Dieter, D. Wauchope, and J. Fastner. Chemicals: Health Relevance, Transport and Attenuation. www.bvsde.paho.org/bvsacd/cd59/protecting/sect1-4.pdf.
- Rivett, M. O., and R. M. Allen-King. 2003. A Controlled Field Experiment on Groundwater Contamination by a Multicomponent DNAPL: Dissolved Plume Retardation. *Journal of Contaminant Hydrology* 66 (1-2):117-146.

- Roehl, K. E. 2005. *Long-Term Performance of Permeable Reactive Barriers*: Elsevier Science.
- Rumbaugh, J. O., and D. B. Rumbaugh. 2004. Guide to using Groundwater Vistas, Version 4. New York: Environmental Simulations Inc.
- Salama, R. B., R. Silberstein, and D. Pollock. 2005. Soils Characteristics of The Bassendean and Spearwood Sands of the Gangara Mound (Western Australia) and their Controls on Recharge, Water Level Patterns and Solutes of The Superficial Aquifer. *Water, Air, & Soil Pollution: Focus* 5 (1):3-26.
- Scatena, M. C., and D. R. Williamson. 1999. *A Potential Role for Artificial Recharge in the Perth Region: A Pre-feasibility Study*: Centre for Groundwater Studies.
- Scott, K. C., and D. J. Folkes. 2000. Groundwater Modeling of a Permeable Reactive Barrier to Enhance System Performance. Paper read at Hazardous Waste Research, at Denver.
- Simon, F. G., and T. Meggyes. 2000. Removal of Organic and Inorganic Pollutants from Groundwater using Permeable Reactive Barriers. *Land Contamination & Reclamation* 8 (2):103-116.
- Smith, A. J., J. V. Turner, D. E. Herne, and W. P. Hick. 2003. Quantifying Submarine Groundwater Discharge and Nutrient Discharge into Cockburn Sound, Western Australia: Centre for Groundwater Studies Report.
- Sommer, B. 2006. Drying and Re-wetting of Organic Wetland Sediments: Biogeochemistry and Implications for Wetland Management, Faculty of Computing, Health and Science, Edith Cowan University, Perth.
- Thierrin, J., G. B. Davis, and C. Barber. 1995. A Ground-Water Tracer Test with Deuterated Compounds for Monitoring In Situ Biodegradation and Retardation of Aromatic Hydrocarbons. *Ground Water* 33 (3):469-475.
- Thierrin, J., G. B. Davis, C. Barber, B. M. Patterson, F. Pribac, T. R. Power, and M. Lambert. 1993. Natural Degradation Rates of BTEX Compounds and Naphthalene in a Sulphate Reducing Groundwater Environment.

- Thiruvengkatachari, R., S. Vigneswaran, and R. Naidu. 2008. Permeable Reactive Barrier for Groundwater remediation. *Journal of Industrial and Engineering Chemistry* 14 (2):145-156.
- Thorbjarnarson, K. W., J. H. Inami, and G. H. Girty. 2002. Visual Solute Transport: A Computer Code for Use in Hydrogeology Classes *Journal of Geoscience Education* 50 (3):287-291.
- Trefry, M. G., G. B. Davis, C. D. Johnston, A. G. Gardiner, D. W. Pollock, and A. J. Smith. 2006. Status of Groundwater Quality in the Cockburn Sound Catchment: Final Report to the Cockburn Sound Management Council: CSIRO Land and Water.
- User's Manual for Visual MODFLOW Version 4.2 2006. Canada: Waterloo Hydrogeologic Inc. (WHI).
- Vance, D. B. 1997. Transverse Dispersion Drives Natural Attenuation. *Environmental Technology* 7 (1):31-32.
- Villholth, K. G. 2006. Groundwater Assessment and Management: Implications and Opportunities of Globalization. *Hydrogeology Journal* 14 (3):330-339.
- Wiedemeier, T. H., J. T. Wilson, D. H. Kampbell, R. N. Miller, and J. E. Hansen. 1995. Technical Protocol for Implementing Intrinsic Remediation with Long-term Monitoring for Natural Attenuation of Fuel Contamination Dissolved in Groundwater. Texas: Air Force Center for Environmental Excellence, Brooks Air Force Base.
- Wu, Ming Zhi. 2003. Effects of Spatial Variability on the Effectiveness of Artificial Aquifer Recharge in the Perth Basin Department of Environmental Engineering. Centre for Water Research. University of Western Australia.

Yeniguel, N. B., A. T. Hendsbergen, A. M. M. Elfeki, and F. M. Dekking. 2006. Detection of Contaminant Plumes Released from Landfills. *Hydrology and Earth System Sciences Discussions* 3 (3):819-857.

Zheng, C., and G. D. Bennett. 2002. *Applied Contaminant Transport Modelling*. 2nd ed. New York: John Wiley & Sons.

Zheng, C., and P. Wang. 1999. MT3DMS: A Modular Three-Dimensional Multispecies Transport Model for Simulation of Advection, Dispersion, and Chemical Reactions of Contaminants in Groundwater Systems; Documentation and User's Guide. Tuscaloosa, Alabama: University of Alabama.

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APPENDIX – 1

This appendix represents the accepted literature values for hydraulic conductivity for different aquifer matrix which is an important input parameter for groundwater flow and contaminant transport modeling study.

If an insufficient amount of site-specific data exists or the site is more complex than the model can handle, then literature values are often used to support the model. However, models relying on literature data would need to rely on good sensitivity and uncertainty analysis. Thus, the most common technique is to use an accepted literature value for the types of materials making up the aquifer matrix, and then to calibrate a contaminant transport model by adjusting the value of hydraulic conductivity (in conjunction with other input parameters) within the range of accepted literature values until the modeled and observed contaminant distribution patterns match and sensitivity analyses should be performed to determine the effect of varying the hydraulic conductivity of the aquifer materials on numerical model results.

Soil Materials	Average Value of Hydraulic Conductivity (m/s)
Alluvium	0.00133
Alluvium sand and gravel with clay lenses	0.001
Anhydrite	8.9 e^{-11}
Basalt	2.9 e^{-9}
Basalt, fractured	0.01558
Basalt, vesicular	7.1 e^{-5}
Basaltic lava and sediments	0.05450
Chalk	0.00027
Chalk, fractured	0.0022
Clay	3.2 e^{-10}
Clay, unweathered marine	4 e^{-11}
Clayey sand	1 e^{-7}
Clayey silt	8 e^{-8}

Soil Materials	Average Value of Hydraulic Conductivity (m/s)
Clayey slate	$4 e^{-6}$
Coal	$3 e^{-6}$
Dolomite	0.00007
Dolomite and limestone	0.00013
Dolomite and limestone, fractured	0.00265
Dolomite, fractured	0.00012
Dolomite, weathered	$2 e^{-5}$
Gabbro, weathered	$1.5 e^{-6}$
Glacial outwash	0.00071
Glacial till	$4 e^{-9}$
Glacial till and fine sand	$2.7 e^{-5}$
Gneiss, fractured	$1.8 e^{-6}$
Granite	$6 e^{-12}$
Granite, fractured	0.00004
Granite, weathered	0.00001
Gravel	0.02575
Gravel and cobbles	0.0029
Gravel layered with silty sand	0.00355
Gravelly clay	$3 e^{-9}$
Gravelly silt	$3 e^{-7}$
Igneous and metamorphic rocks, fractured	$1.6 e^{-6}$
Igneous and metamorphic rocks, unfractured	$2.5 e^{-12}$
Limestone	0.00003
Limestone, fractured	0.00269
Limestone, karst	0.0505
Loess	$1.4 e^{-7}$
Salt	$1 e^{-11}$
Sand	0.00014
Sand and gravel	0.01458

Soil Materials	Average Value of Hydraulic Conductivity (m/s)
Sand and gravel with clay lenses	0.0009
Sand and gravel, glaciofluvial	0.01257
Sand, silt and clay	0.0005
Sand, coarse	0.00077
Sand, eolian	0.00023
Sand, fine	0.00007
Sand, fluvial	0.0006
Sand, glaciofluvial	0.00005
Sand, gravel and silt	0.0013
Sand, medium	0.00025
Sand, very fine	0.000155
Sandstone	0.00012
Sandstone with sand, silt and clay	2.9×10^{-10}
Sandstone, fine	2.3×10^{-6}
Sandy silt	3×10^{-8}
Schist and gneiss weathered	3.6×10^{-7}
Shale	1.4×10^{-11}
Shale, weathered	3×10^{-6}
Silt	0.00001
Siltstone	3.7×10^{-10}
Silty sand	1×10^{-6}
Tuff	2×10^{-6}

Literature Values of Hydraulic Conductivity for different Soil Materials (Morris and Johnson 1967; Davis 1969; Fetter 2001; EnviroBase Pro 2005)

APPENDIX – 2

This appendix represents the accepted literature values for porosity for different aquifer matrix which is an important input parameter for groundwater flow and contaminant transport modeling study.

The most common method is to use an accepted literature for the types of materials making up the soil materials or aquifer matrix, and then to calibrate the groundwater flow and contaminant transport model by adjusting the value of porosity (in conjunction with other input parameters) within the range of accepted literature values until the modeled and observed contaminant distribution patterns match. Because aquifer materials can have a range of effective porosity, sensitivity analyses should be performed to determine the effect of varying the porosity on numerical model results.

Soil Materials	Total Porosity, P_t		Effective Porosity, P_e	
	Range	Arithmetic Mean	Range	Arithmetic Mean
Alluvium	0.20 - 0.40	0.30	^a -	-
Basalt	0.03 - 0.35	0.17	0.02 - 0.10	0.06
Basalt, fractured	0.15 - 0.28	0.22	0.01	0.01
Basalt, vesicular	0.45 - 0.55	0.50	0.04	0.04
Chalk	0.02 - 0.40	0.22	-	-
Clay	0.34 - 0.57	0.42	0.01 - 0.18	0.06
Dolomite	0.01 - 0.18	0.07	0.01 - 0.14	0.05
Dolomite and limestone	0.04 - 0.08	0.06	-	-
Dolomite and limestone, fractured	0.06 - 0.60	0.33	-	-
Gabbro, weathered	0.42 - 0.45	0.43	-	-
Glacial outwash	0.30 - 0.40	0.35	0.23 - 0.24	0.24
Glacial till	0.30 - 0.35	0.33	0.03 - 0.11	0.07

Soil Materials	Total Porosity, P_t		Effective Porosity, P_e	
	Range	Arithmetic Mean	Range	Arithmetic Mean
Granite	0.00 - 0.04	0.02	-	-
Granite, weathered	0.34 - 0.57	0.45	-	-
Gravel (fine)	0.25 - 0.40	0.34	0.21 - 0.35	0.28
Gravel (medium)	-	-	0.17 - 0.44	0.24
Gravel (coarse)	0.24 - 0.36	0.28	0.13 - 0.25	0.21
Gravel and cobbles	0.20 - 0.24	0.22	-	-
Igneous and metamorphic rocks, fractured	0.00 - 0.10	0.05	0.04	0.04
Igneous and metamorphic rocks, weathered	0.20 - 0.40	0.30	0.10 - 0.20	0.15
Limestone	0.07 - 0.56	0.30	0.00 - 0.36	0.14
Limestone, fractured	0.01 - 0.045	0.03	0.02 - 0.10	0.06
Limestone, karst	0.025 - 0.28	0.15	-	-
Loess	0.45 - 0.50	0.48	0.14 - 0.22	0.18
Sand	0.24 - 0.42	0.35	0.06 - 0.30	0.20
Sand and gravel	0.15 - 0.35	0.25	0.20 - 0.25	0.23
Sand and gravel with clay lenses	0.25 - 0.35	0.30	-	-
Sand and gravel, glaciofluvial	0.07 - 0.40	0.22	-	-
Sand, clay and silt	0.22 - 0.28	0.25	-	-
Sand, coarse	0.31 - 0.46	0.39	0.18 - 0.43	0.30
Sand, eolian	0.40 - 0.45	0.42	0.32 - 0.47	0.38

Soil Materials	Total Porosity, P_t		Effective Porosity, P_e	
	Range	Arithmetic Mean	Range	Arithmetic Mean
Sand, fluvial	0.40 - 0.42	0.41	-	-
Sand, glaciofluvial	0.33 - 0.38	0.36	-	-
Sand, gravel and silt	0.25 - 0.39	0.32	-	-
Sand, medium	0.28 - 0.40	0.34	0.16 - 0.46	0.32
Sand, very fine	0.40 - 0.60	0.50	-	-
Sandstone	0.10 - 0.40	0.25	0.05 - 0.25	0.15
Sandstone, fine	-	-	0.02 - 0.40	0.21
Sandstone, medium	-	-	0.12 - 0.41	0.27
Sandstone with sand, silt and clay	0.23	0.23	-	-
Schist	0.04 - 0.49	0.38	0.22 - 0.33	0.26
Silt	0.35 - 0.50	0.42	0.01 - 0.39	0.20
Siltstone	0.21 - 0.41	0.35	0.01 - 0.33	0.12
Tuff	-	-	0.02 - 0.47	0.21

Literature Values of Porosity for different Soil Materials (Morris and Johnson 1967; Davis 1969; Fetter 2001; EnviroBase Pro 2005)

^a, a hyphen indicates that no data are available.

APPENDIX - 3

This appendix represents the accepted literature values for specific yield and specific storage for different aquifer matrix which is an important input parameter for groundwater flow and contaminant transport modeling study.

The most common method is to use an accepted literature for the types of materials making up the soil materials or aquifer matrix, and then to calibrate the groundwater flow and contaminant transport model by adjusting the value of specific yield and specific storage (in conjunction with other input parameters) within the range of accepted literature values until the modeled and observed contaminant distribution patterns match. Because aquifer materials can have a range of specific yield and specific storage, sensitivity analyses should be performed to determine the effect of varying the porosity on numerical model results.

Soil Materials	Specific Yield			Specific Storage (1/m)
	Minimum	Maximum	Average	
Basalt	0.02	0.10	0.06	6.3×10^{-5}
Basalt, fractured	- ^a	-	0.01	3.8×10^{-6}
Basalt, vesicular	-	-	0.04	-
Basaltic lava and sediments	-	-	0.07	-
Clay	0.00	0.05	0.02	1×10^{-3}
Clayey silt	-	-	-	3×10^{-4}
Coal	-	-	0.01	6×10^{-5}
Dolomite	0.005	0.10	0.06	-
Dolomite and limestone, fractured	0.02	0.05	0.04	-
Glacial till	0.04	0.18	0.07	5.5×10^{-4}
Gneiss	-	-	-	1.2×10^{-5}

Soil Materials	Specific Yield			Specific Storage (1/m)
	Minimum	Maximum	Average	
Granite, fractured	-	-	-	1.1×10^{-4}
Granite, weathered	-	-	0.01	-
Gravel	0.15	0.3	0.22	2.9×10^{-5}
Gravelly sand	0.20	0.35	0.25	-
Gravel, fine	0.21	0.35	0.25	-
Gravel, medium	0.13	0.26	0.23	-
Gravel, coarse	0.12	0.26	0.22	-
Igneous and metamorphic rocks, fractured	0.02	0.05	0.04	1×10^{-6}
Igneous and metamorphic rocks, unfractured	0.00	0.03	0.02	3.2×10^{-5}
Limestone, fractured	-	-	-	1×10^{-7}
Loess	0.15	0.20	0.18	-
Sand	0.10	0.30	0.20	3.2×10^{-4}
Sand and gravel	0.20	0.35	0.25	2.2×10^{-5}
Sand, fine	0.10	0.21	0.28	-
Sand, medium	0.15	0.32	0.26	-
Sand, coarse	0.20	0.35	0.27	-
Sandy clay (mud)	0.03	0.12	0.07	-
Sandstone	0.01	0.20	0.10	-
Sandstone, fine	0.02	0.40	0.21	-
Sandstone, medium	0.21	0.41	0.27	-
Schist	0.22	0.33	0.26	-

Soil Materials	Specific Yield			Specific Storage (1/m)
	Minimum	Maximum	Average	
Silt	0.03	0.19	0.18	-
Siltstone	0.01	0.08	0.04	-
Dune sand	-	-	0.38	-
Tuff	0.02	0.47	0.21	-
Peat	-	-	0.44	-

Literature Values of Specific Yield and Specific Storage for different Soil Materials (Morris and Johnson 1967; Davis 1969; Fetter 2001; EnviroBase Pro 2005)

^a, a hyphen indicates that no data are available.

APPENDIX - 4

This appendix shows some of the Western Australia contaminated sites with that source of contamination and contaminants with their location. It shows that most of the contaminated sites in Perth region are historically used as a service station and investigation identified the presence of petroleum hydrocarbon e.g. benzene, toluene, ethylbenzene and xylene, and heavy metal such as lead in groundwater at contaminants concentration levels exceeding Australian Drinking Water Guidelines and the major source of contamination identified at these types of contaminated sites are leakage of underground fuel storage tanks (DEC 2006).

Location	Classification	Contaminants	Contamination Source
10, Railway Pde, Bayswater, WA	Contaminated remediation required	Heavy metals: arsenic, lead, chromium, and copper	Fertilizer manufacturing
7, Mooney St, Bayswater, WA	Contaminated remediation required	Heavy metals: arsenic, lead, chromium, and copper	Fertilizer manufacturing
364, Railway Pde, Beckenham, WA	Remediated for restricted use	Hydrocarbons (Petrol and Diesel) and lead	Service station
5, Military Rd, Bellevue, WA	Contaminated restricted use	Petroleum hydrocarbons, chlorinated hydrocarbons and brominated hydrocarbons	Former Waste Control site, Bellevue
153, Great Eastern Hwy, Belmont, WA	Contaminated remediation required	Hydrocarbons (Petrol and Diesel) and lead	Service station

Location	Classification	Contaminants	Contamination Source
83, Wheatley Dr, Bullcreek, WA	Remediated for restricted use	Hydrocarbons (Petrol and Diesel) and lead	Service station
141, Twickenham Rd, Burswood, WA	Contaminated remediation required	Hydrocarbons (Petrol and Diesel) and lead	Service station
147, Raleigh St, Carlisle, WA	Contaminated remediation required	Hydrocarbons (Petrol and Diesel) and lead	Service station
124, Guger St, Claremont, WA	Remediated for restricted use	Hydrocarbons (Petrol and Diesel) and lead	Service station
363, Canning Hwy, Como, WA	Contaminated remediation required	Hydrocarbons (Petrol and Diesel) and lead	Service station
360, Grand Prom, Dianella, WA	Remediated for restricted use	Hydrocarbons (Petrol and Diesel) and lead	Service station
307, Great Eastern Hwy, Midland, WA	Contaminated remediation required	Hydrocarbons (Petrol Diesel and oil) and lead	Service station
3001, Walter Pl, North Fremantle, WA	Contaminated remediation required	Petroleum Hydrocarbons and heavy metals	Railway marshalling yards and associated activities
27, Clever Tce, Rivervale, WA	Remediated for restricted use	Hydrocarbons and heavy metals	Storage of fuels/oils and hydraulic fluids
1, Cockburn Rd, South Fremantle, WA	Contaminated remediation required	Hydrocarbons and heavy metals	Landfill facility accepting domestic, industrial and quarantine waste

Location	Classification	Contaminants	Contamination Source
3, Canning Hwy, South Perth, WA	Contaminated remediation required	Hydrocarbons and heavy metals	Service station
9001, Hay St, Subiaco, WA	Contaminated remediation required	Hydrocarbons	Bitumen batching plant
95, Forest Lakes Dr, Thornlie, WA	Contaminated remediation required	Hydrocarbons and heavy metals	Service station
164, Albany Hwy, Victoria Park, WA	Contaminated restricted use	Petroleum and Diesel Hydrocarbons, monocyclic aromatic hydrocarbons and heavy metals	Service station
1, Felspar St, Welshpool, WA	Remediated for restricted use	Polychlorinated Biphenyles (PCBs)	Medical waste incinerator
36, Railway Pde, Welshpool, WA	Remediated for restricted use	Hydrocarbon, DDT, pesticides, arsenic	Warehousing and distribution of pesticides and herbicide
84, Vahland Ave, Willetton, WA	Contaminated remediation required	Hydrocarbons (Petrol and Diesel) and lead	Service station
37, Blythe Ave, Yokine, WA	Remediated for restricted use	Hydrocarbons (Petrol, Diesel and oil)	Service station

Western Australia Contaminated Sites Database (DEC 2006)