Groundwater detection monitoring system design under conditions of uncertainty

dedicated with all my heart

to my beloved husband Ahmed

to my lovely daughter Nur

canım anne ve babama

ve biricik kardeşlerime sevgilerle

Groundwater detection monitoring system design under conditions of uncertainty

Proefschrift

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Summary

Groundwater detection monitoring system design under conditions of uncertainity

Groundwater is an important natural resource for potable, agricultural and industrial purposes. The focus of groundwater investigation has traditionally been on quantification of this resource. However, the upsurge in contamination incidents during the last decades has shifted the focus towards assessment and protection of groundwater quality. Landfills, storage and transportation of commercial materials, mining, agricultural operations, and saltwater intrusion are the major groundwater contamination sources. Among them landfills represent a wide-spread and significant threat to groundwater quality, human health and even some of the ecosystems due to their nature of operation and abundance. In the design of landfills, evaluation of the potential risk associated with groundwater contamination is vital for a groundwater scientist or engineer especially when he/she confronts a sceptical public. Designs of landfill liner systems, detection and assessment of the extent of contaminants in groundwater, and risk assessment for human health and environment are the three main relevant issues. Groundwater quality monitoring systems are the main link among them since they help to determine the likelihood and severity of contamination problems. Therefore, a reliable and efficient monitoring system design is of great importance in the overall design of a landfill. However, more often it is difficult to ensure that a specific monitoring system will detect all of the contaminants released from the landfill due to the numerous and significant uncertainties in both the characterization of the subsurface and the nature of the contaminant source.

For detection monitoring, commonly regulations require at least one background well and three downgradient wells. The position, number (more than the minimum requirement), and depth of the monitoring wells are proposed by the landfill owners or operators and by local authorities. Conventional monitoring program suggested by regulatory agencies requires the monitoring of groundwater quarterly, biannually or annually depending on the type of waste, size and design of landfill and aquifer material for 30 years of post closure monitoring duration. In most cases a quarterly monitoring is undertaken; annual monitoring is undertaken mostly for small landfills located in remote places far away from any groundwater use source. There is no recognition of uncertainty in regulations conversely to reality.

In this thesis a methodology was developed for the design of optimal groundwater monitoring system design at landfill sites under conditions of uncertainty. The reliability assessment of groundwater monitoring systems and the design of the optimal groundwater detection monitoring systems using a multi-objective decision analysis approach under different hydrogeologic scenarios were the main focus of this research.

A simulation model coupling a Monte Carlo framework with a two-dimensional finite difference flow model and a random walk particle-tracking model was used to simulate contaminant plumes. Uncertainties in the hydrogeology and contaminant source were incorporated in the model using Monte Carlo simulations. Spatial variability of the hydraulic conductivity was assumed to be the major contributor to the hydrogeologic uncertainty while uncertainty in the contaminant source was assumed to be limited to the leak location. Detection monitoring systems composed of a single row of wells at different spacing and at different distances from the contaminant sources were considered.

Reliability assessment of monitoring systems at landfill site was performed to evaluate the influence of several parameters including the heterogeneity dispersivity of medium, locations and the number of monitoring wells, threshold concentration, leak size, type of leak and sampling frequency on the efficiency of groundwater detection monitoring systems. The analysis showed that the detection probability increases when the dispersivity of medium increases since the plume gets wider as it travels away from the source. On the other hand, the reliability of monitoring systems decreases as the subsurface heterogeneity increases, mainly due to the fact that the contaminant plumes are more likely to become irregularly shaped in heterogeneous media, and they may go undetected easier because of the variability in the flow field. Another significant outcome of the analysis was that the widely used 3-well monitoring system (minimum regulatory requirement) is not a sufficiently large minimum from the point of view of the detection of the contaminant plume and the prevention of groundwater contamination.

Afterwards, a decision analysis approach was presented for optimal design of groundwater monitoring systems under conditions of uncertainty. The methodology accounts for the multi-objective nature of detection monitoring problem as well. Maximizing the probability of detecting contaminant plumes, minimizing the contaminated area, and the total cost of the monitoring system (i.e., construction, maintenance, and remediation cost, if necessary) were the conflicting objectives incorporated to find the optimal monitoring system in terms of location and number of the wells. It was observed that the most efficient design for detection monitoring should consist of rather a large number of wells located close to contaminant source except for the cases where the unit installation and monitoring cost are very high and/or the unit remediation cost is very cheap.

Then, a new monitoring approach was proposed and implemented to find out how to improve the efficiency of groundwater monitoring systems, particularly the efficiency of 3-well system that fulfils the minimum regulatory requirement and consequently widely applied in practice. To increase the interception of contaminant plumes at early stages by broadening the capture zone of monitoring well(s) simply by continuous pumping from the monitoring well(s) with a small pumping rate is the essence of this approach. The results of analysis showed that the efficiency of the monitoring system improves significantly by the application of the proposed monitoring approach (more than twice even in a heterogeneous highly dispersive medium) Then the former decision model considering the current conventional monitoring approach, was extended by implementing this new monitoring approach in the model. Finally, the applications of both simulation-decision models to Maarsbergen Landfill site (The Netherlands) were presented.

Delft, September 2006

N. Buket Yenigül

SAMENVATTING

HET ONTWERPEN VAN GRONDWATER MONITORING SYSTEMEN ONDER ONZEKERHEID

Grondwater vormt een belangrijke natuurlijke voorraad voor drinkwater en agrarische en industriële doeleinden. Het aandachtsgebied van grondwateronderzoek lag van oudsher op het kwantificeren van deze voorraad. Echter, door de toename van verontreinigingsgevallen gedurende de laatste decennia is de aandacht verschoven naar het vaststellen en beschermen van de grondwaterkwaliteit. Vuilstortplaartsen, opslag en transport van commerciële goederen, exploitatie van mijnen, landbouw en intrusie van zout water zijn de voornaamste bronnen van grondwatervervuiling. Van de genoemde bronnen vormen vuilstortplaatsen een wijd verbreidde en aanzienlijke bedreiging voor de grondwaterkwaliteit, volksgezondheid en zelfs voor sommige ecosystemen door hun aard en veelvoudig voorkomen. Bij het ontwerp van vuilstortplaatsen is evaluatie van het potentiële risico dat samengaat met grondwatervervuiling van groot belang voor grondwaterspecialisten, wetenschappers en ingenieurs, zeker wanneer deze met publieke scepsis worden geconfronteerd. Het ontwerp van stortplaats liner, het signaleren en inschatten van de verspreiding van vervuiling in grondwater en het bepalen van de risico's voor volksgezondheid en milieu zijn de drie voornaamste aandachtsgebieden. Grondwaterkwaliteitmeetnetten vormen de verbinding tussen deze aandachtgebieden aangezien een meetnet de waarschijnlijkheid en ernst van het vervuiling probleem helpt vast te stellen. Derhalve is een betrouwbaar en efficiënt monitoringssysteem van groot belang bij het ontwerp van een stortplaats. Echter, vaak is het moeilijk om te garanderen dat een bepaald monitoringssysteem alle vervuiling zal ontdekken die van een stortplaats vrijkomt vanwege veelvoudige en aanzienlijke onzekerheden in zowel de karakterisatie van de ondergrond als de aard van de vervuilingsbron.

Voor detectie monitoring, vereist de regelgeving in het algemeen tenminste één achtergrond-meetpunt en drie putten stroomafwaarts. De ligging, het aantal (meer dan het minimum vereiste) en de diepte van de monitoringsputten worden voorgesteld door de eigenaren of beheerders van de stortplaats en de lokale autoriteiten. Conventionele monitoringsprogramma's voorgesteld door de regelgevende autoriteiten vereisen monitoring eens per kwartaal, per halfjaar of per jaar, afhankelijk van het soort afval, grootte en ontwerp van de vuilstortplaats en de aard van de aquifer, tot 30 jaar na sluiting van de stortplaats. In de meeste gevallen wordt

ieder kwartaal een monitoring uitgevoerd; jaarlijks monitoring wordt vooral voor kleine stortplaatsen uitgevoerd in afgelegen gebied ver weg van grondwater dat op enige wijze gebruikt wordt. In de regelgeving is geen aandacht voor onzekerheid in tegenstelling tot de realiteit.

In dit proefschrift is een methode ontwikkeld voor het ontwerp van een optimaal grondwater monitoring systeem voor stortplaatsen waarbij onzekerheid een rol speelt. Het testen van de betrouwbaarheid van grondwater monitoring systemen en het ontwerp van het optimale grondwater detectie systeem gebruik makend van multi-criteria besslising analyse in verschillende hydrogeologische scenario's was het belangrijkste doel van dit onderzoek.

Een simulatiemodel dat een Monte Carlo analyse een twee-dimensionale, eindige-differenties grondwaterstromingsmodel en een random walk particle tracking model aan elkaar koppelt, is gebruikt om de vervuilingspluimen te simuleren. Onderzekerheden in de hydrogeologie en bronnen van vervuiling zijn in het model opgenomen door middel van Monte Carlo simulatie. Ruimtelijk variabiliteit van waterdoorlatendheid werd verondersteld de belangrijkste bijdrage te leveren aan de hydrogeologische onzekerheid, onzekerheid in de vervuilingsbron werd verondersteld zich te beperken tot de locatie van het lek. Detectie monitoring systemen die bestaan uit een enkele rij van onttrekkingputten op verschillende afstand van elkaar en tot de bron van vervuiling worden in beschouwing genomen.

Studie naar de betrouwbaarheid van monitoring systemen voor stortplaatsen is uitgevoerd ter evaluatie van de invloed van verschillende parameters waaronder de dispersiviteit van het heterogene medium, de ligging van en het aantal meetpunten, de drempel-concentratie, de omvang van een lek, het soort lek en de meetfrequentie, op de efficiëntie van het monitoring systeem. De analyse toonde aan dat de detectiekans toeneemt indien de dispersiviteit van het medium toeneemt aangezien de pluim breder wordt naarmate de gereisde afstand vanaf de bron groter wordt. Daarentegen, neemt de betrouwbaarheid van het monitoringssysteem af naarmate de heterogeniteit van de ondergrond toeneemt, voornamelijk doordat vervuilingspluimen met grotere waarschijnlijkheid een onregelmatig vorm krijgen in het heterogene medium en makkelijker onopgemerkt kunnen blijven door de variabiliteit van het stromingsveld. Een andere belangrijke uitkomst van de analyse was dat het veel gebruikte 3-meetpunten monitoringssysteem (een minimum vereiste volgens de regelgeving) niet voldoet als minimum wat betreft de detectie van de vervuilingspluim en de preventie van grondwatervervuiling.

Na de hierboven beschreven studie, werd een beslissingsondersteunende methode gepresenteerd voor het optimale ontwerp van grondwatermonitoringssystemen wanneer onzekerheid een rol speelt. De methode neemt tevens de multi-citeria uiteenlopende doelstellingen van monitoringssystemen in beschouwing. De maximale kans op detectie van de vervuilingspluim, het minimaliseren van het vervuilde gebied en de totale kosten van het monitoringssysteem (i.e. constructie, onderhoud en indien noodzakelijk saneringskosten) vormen de tegengestelde doelen die in beschouwing worden genomen om het optimale monitoringssysteem te vinden wat betreft de ligging van en het aantal meetpunten. Er kwam naar voren dat het meest efficiënte ontwerp van een detectiemonitoringssysteem zou moeten bestaan uit een groot aantal putten dichtbij de vervuilingspluim behalve in die gevallen dat een unit installatie- en monitoringskosten erg hoog is en/of een unit sanering erg goedkoop.

Vervolgens is een nieuwe monitoringsaanpak voorgesteld en geïmplementeerd om inzicht te krijgen hoe de efficiëntie van grondwaterkwaliteitsmonitoring verbeterd kan worden, in het bijzonder de efficiëntie van het 3-meetpunten-systeem dat volgens de regelgeving aan de minimum eisen voldoet en als gevolg daarvan wijd in de praktijk verbreid is. De essentie van deze aanpak is het verhogen van de interceptie van vervuilingspluimen in een vroeg stadium door het intrekgebied van de monitoringsputten te verbreden simpelweg door middel van continue bemaling met een klein debiet. De resultaten tonen aan dat de efficiëntie van het monitoringssysteem significant verbetert door de toepassing van de voorgestelde aanpak (meer dan twee maal, zelfs in een heterogeen, zeer dispersief medium). Hierna is het eerder genoemde beslissingsondersteunende model dat uitgaat van de huidige conventionele benadering van monitoring, uitgebreid door de implementatie in het model van deze nieuwe benadering van monitoring. Tot slot wordt de toepassing van beide simulatie-besluit-modellen op de Maarsbergen Stortplaats (Nederland) gepresenteerd.

N. Buket Yenigül

delft, september 2006

CONTENTS

Chapter 1

INTRODUCTION

Groundwater is an important natural resource for potable, agricultural and industrial purposes, since it represents the largest portion of the fresh water supply in the world's hydrologic cycle. Groundwater is also vital for fish production, wildlife habitat, recreational opportunities and other attributes as it nourishes and maintains many ecosystems. The focus of groundwater investigation has traditionally been on quantification of this resource. However, the upsurge in contamination incidents during the last decades has shifted the focus towards assessment and protection of groundwater quality. Landfills, storage and transportation of commercial materials, mining, agricultural operations, and saltwater intrusion are the major groundwater contamination sources. Among them landfills represent a wide-spread and significant threat to groundwater quality, human health and even some of the ecosystems due to their nature of operation and abundance. The tremendous socioeconomic development, the rapid increase in population of urban areas and industrial revolution that has taken place in the last centuries introduced mass production and large scale consumption of goods. The ensuing huge amount of various wastes has induced an enormous increase in the number of all kinds of landfills, leading to inevitable adverse environmental effects. The inherent risks involved plus the growing public awareness and civic involvement in environmental issues present a great challenge to those who are in charge of groundwater protection. In the design of landfills evaluation of the potential risk associated with groundwater contamination is vital for a groundwater scientist or engineer especially when he/she confronts a sceptical public. Therefore monitoring the quality of groundwater is indispensable both to quantify and to limit the exposure risk. Figure 1.1 illustrates the *landfill/groundwater* system.

In this first chapter, a precise definition of the problem addressed in this thesis is followed by a description of the research objectives. Then, the chapter concludes with an outline of the thesis contents.

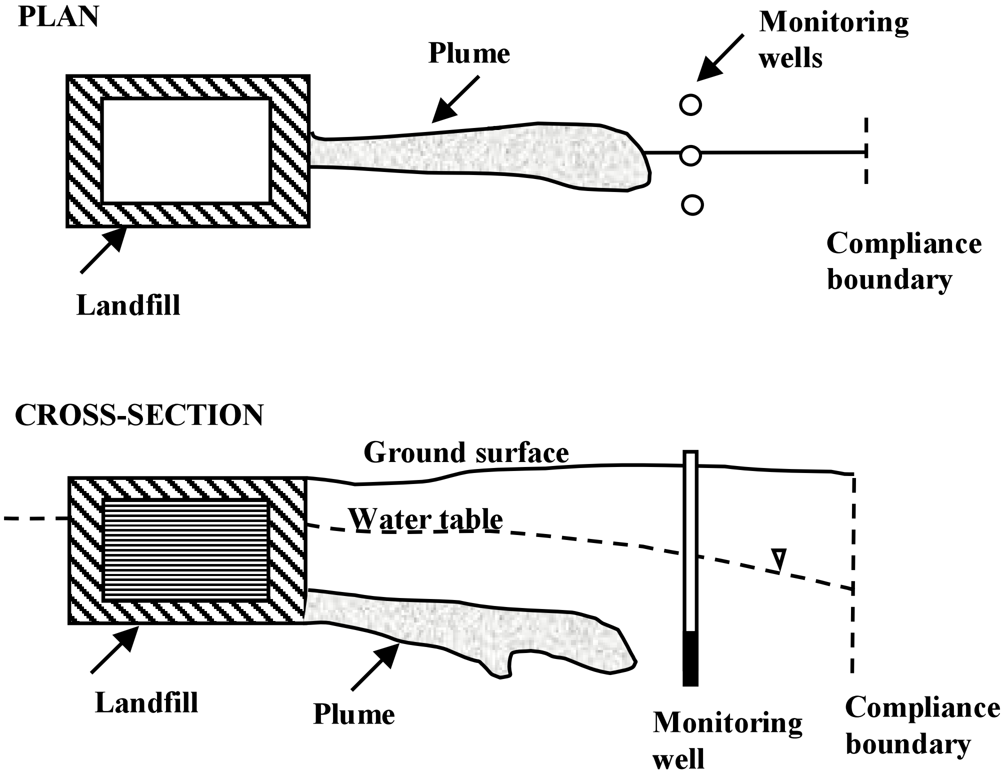


Figure 1.1: Groundwater contamination scenario (after Massmann and Freeze, 1987a).

1.1 Glossary of terms

In communal language *landfill* means waste disposal on land. Over the years the practice has had various names such as 'tips' and 'controlled tipping' in the United Kingdom, 'sanitary landfill' in the United States 'coups' in Scotland, and 'dumps'. However, technically the International Solid Wastes Association defines *landfill* (ISWA, 1992) as "an engineered deposit of waste onto or into land in such a way that pollution or harm to the environment is prevented, and through restoration, land provided which may be used for other purpose". Landfills are classified as municipal waste landfills (non-hazardous) and hazardous waste landfills according to the waste types they include.

Rainwater impinging on waste during the operation phase of a landfill infiltrates into the waste bulk. Together with water contained in the waste it percolates through the waste, and by doing so takes along various substances (organic and inorganic contaminants). These substances are either waste constituents or products of biological degradation and chemical transformation processes. The resulting liquid is called *leachate*. The quantity and composition of the leachate depend on several factors among which the most important ones include the type of the waste, the moisture content, and other phyisco-chemical conditions prevailing in the waste bulk. These conditions, in turn, depend on the climate of the landfill site, on the landfill design, manner of operation, and age. The subsequent movement of the leachate into the surrounding soil, groundwater or surface water can cause considerable contamination problems.

Isolation of waste from its surroundings is usually accomplished by physical barriers or so called *liners*, which are placed on top, bottom, and sides. Liners between the landfill and the natural soil underneath consist of continuous layer(s) of natural or manmade materials that restrict escape of waste or any of its constituents such as leachate or else. Liners are made of materials with very low permeability such as compacted clay or a mixture of bentonite clay powder and sand or synthetic polymeric material, which is called *geomembrane*, with very low permeability.

Leachates pass through a filter layer (usually sand or geotextile), allowing passage of leachate, but retaining the waste. The passed leachate then runs through a drainage layer (usually through a gravel layer incorporating a pipe network, or along a geonet) towards the lowest point in the system, where it can be accessed through a borehole. The filter layer and drainage layer together comprise the *leachate collection and removal system*. If there is another drainage system under the bottom liner (with an other lining layer underneath), it is usually referred to as a *leakage detection system*. The name implies that its principal function is monitoring of the functioning of the bottom liner.

Rainwater falling on a closed and covered landfill either infiltrates into the cover soil layer, or evaporates, or migrates by surface runoff. Subsequently it is collected and removed by a surface water drainage system. Part of this system functions during the operation phase as well; it collects and removes rain water around the active landfill area.

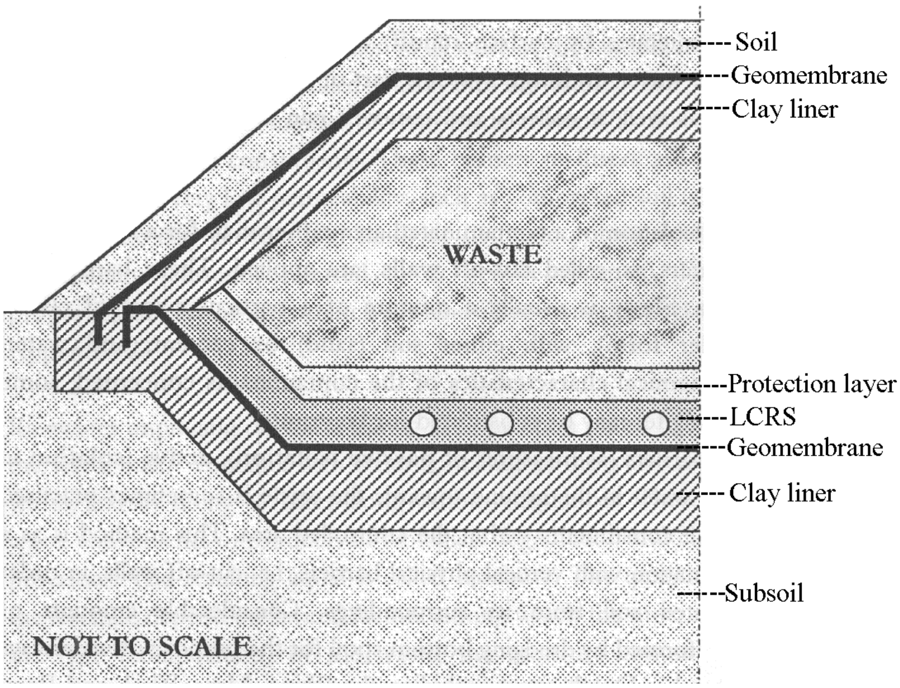


Figure 1.2: Cross-section of a landfill.

If liners are used for isolation of a contaminated area rather than for a new landfill, vertical physical barriers, so-called *cut-off walls,* are applied. These barriers are made as slurry walls. Slurry is usually a mixture of a particular sort of clay-bentonite, with either sand or cement and water. Figure 1.2 demonstrates a cross section of a landfill example and its components.

1.2 Regulatory requirements

The European Community and the U.S. Environmental Protection Agency (USEPA) regulations are widely recognized and applied in many countries. The current regulations for landfill design are included under Resource Conservation and Recovery Act (RCRA), subtitle D in the U.S.A and included under European Landfill Directive (CEC, 1991) in Europe. Under the regulatory requirements all new landfills and extension of landfills must:

with regard to positioning:

- avoid fresh and saltwater wetlands
- be outside a 100 year flood plain
- avoid sole source aquifers, unstable areas, seismic impact zones and faults
- be located at a distance of more than 200 m from any dwelling

with regard to liquid control:

- provide a cap for the landfill which will prevent infiltration of precipitation
- provide a composite liner
- provide a leachate collection system designed to quickly remove leachate without allowing leachate depth (over the liner system) to exceed 30.5 cm
- provide appropriate treatment for leachate removed from landfill
- provide a monitoring program for measuring groundwater near the landfill

with regard to gas control:

- provide a system for recovering landfill gas and prevent its migration
- monitor gas migration

with regard to long term liabilities:

- must provide a trust fund sufficient to allow maintenance and aftercare of the site for a period of at least 30 years, since the total cost of long term maintenance and monitoring of a closed landfill could be higher than the cost of construction of a landfill.

Both the European Landfill Directive and RCRA subtitle D in the USA require 30 years post-closure monitoring period, which also includes a monitoring program for measuring groundwater quality near the landfill. Three major monitoring programs are defined for landfills. Detection monitoring requires at least one background well (hydraulically upgradient from a potential source) and three downgradient wells. The purpose of detection monitoring is early detection of a release to groundwater based on comparison of downgradient well data to background data for a limited number of water quality parameters. Compliance monitoring is implemented if detection monitoring indicates a statistically significant likelihood of release. For municipal waste landfills the compliance point may be located up to 150 meters away from the downgradient boundary of the landfill. Compliance monitoring samples for an expanded suite of hazardous constituents, and requires establishment of concentration limits (compliance or cleanup standards), should any of these constituents be detected. Downgradient well data are compared to concentration limits for each well on a periodic basis. The purpose of compliance monitoring is to determine if the release to groundwater is significant enough to warrant corrective action. Corrective action monitoring is typically implemented if compliance monitoring indicates a statistically significant groundwater impact. Corrective action typically requires a more extensive characterization program and remedial measures. The purpose of the corrective action monitoring is to document the effectiveness of remediation and fulfilment of cleanup goals. The issue of monitoring system design for detection monitoring is addressed in this thesis.

Furthermore regulations require monitoring at regular frequencies to judge the change in quality of the groundwater downgradient of the landfill. In general, groundwater should be monitored quarterly, biannually or annually depending on the type of waste, size and design of landfill, aquifer material, and so on. In most cases a quarterly monitoring is required, however annual monitoring can be undertaken for small landfills located in remote places far away from any groundwater use source. Collection, preservation, and testing of the groundwater sample are important to obtain representative data. Before taking a sample to monitor the groundwater quality, water level of each monitoring well is purged by removing four well volumes (internal radius of the well × the height of water column in the well) of water using a bailer or a pump.

1.3 Problem definition

Despite all counter measures aiming to minimize the chance of leakage, the risk of contamination due to landfill leakages cannot be completely eliminated. The adverse impacts of landfill leachates on adjacent groundwater have prompted a great number of studies (e.g, Gonzalez et al., 1990; Massing, 1994; Eiswirth et al., 1995; Kjeldsen et al., 1995; Chen, P.H. and Wang, C.Y., 1997; Mikac et al., 1998; Riediker et al., 2000; Tatsi and Zouboulis, 2002; Koliopolous, 2003; Chofqi et al., 2004; Yousef, 2005). The U.S. Environmental Protection Agency USEPA has estimated that approximately 75% of landfills in USA are polluting the groundwater (Jones-lee and Lee, 1993). Although certain figures are not available for the time being, landfill leachate is recognized as a quite serious problem within Europe as well. Therefore, the design of a reliable and efficient groundwater monitoring system is of great importance for groundwater protection policy as it helps to determine the likelihood, and severity of contamination problems. However, because of the numerous and significant uncertainties involved, more often it is difficult to ensure that a specific monitoring system will detect all of the contaminants released from the land-fill. Reliable groundwater detection monitoring system design entails various challenges due to the nature of the problem. Among them are the following that inspired the research presented in this thesis:

- The contamination source within a landfill is usually very hard to determine, due to the scarce data on the characteristics of the contamination source such as location and size of a leak, quantity and concentration of the contaminant, and time and duration of a release. Hence, given the occurrence of a contamination, a reliable procedure to determine the characteristics of the source is required. Therefore, *where should wells be placed to detect contaminant plumes early, as well as give information in relation to the likely location of the contaminant source?*
- Correct hydrogeological characterization of the site may be complicated not only due to the inherent variability of the subsurface but also due to incomplete knowledge from site exploration in most cases. Although an iterative procedure for subsurface characterization can be used to minimize hydrogeological uncertainty, it is not possible to know the profile characteristics at every point. The most likely conditions at boring locations are estimated by using judgement and interpolation. Given that this is a site-specific task, *how can the level of uncertainty in the subsurface characterization be quantified to design an accurate and reliable monitoring system?*
- Spatial variability of the subsurface has a great influence on the extent and the characteristics of a contaminant plume. Nevertheless, a significant amount of research has been devoted in the last two decades to the comprehension of the effects of natural heterogeneity on solute transport and to the development of modelling techniques that explicitly account for natural heterogeneity. The question is, *how far can an analytical model that is based on homogenization of heterogeneous aquifer conditions be used in a groundwater monitoring system design while incorporating the effect of various uncertainties on contaminant transport? And how accurate can the likelihood of detecting a contaminant plume be determined using such a model?*
- A minimum number of wells is required to meet the regulations. Yet, it is not clear how many wells are required to achieve a certain level of confidence that leaks will be detected before damage to human health and/or to the environment occurs. Therefore, the number of wells to be located is a primary decision variable in the design of detection monitoring systems. A large number of wells maximizes the detection probability of contaminant plumes, whereas a small number of wells would be cheaper to install and to monitor but would not be as reliable. Therefore, *how many monitoring wells should be placed to meet the desired objectives?*
- Mostly more than 3-wells are required for an optimal system that will enable satisfactorily high detection probability while minimizing the expected contaminated area of the aquifer and/or the total cost of the system. On the other hand, a 3-well monitoring system is quite common in practice since it fulfils the minimum regulatory requirements. Hence, *how can the reliability and efficiency of a 3-well monitoring system be improved?*

- The locations of the monitoring wells are at least as important as the number of the monitoring wells, in determining the likelihood of detecting contaminant plumes. Subsurface characteristics affect the contaminant paths, chemical reactions, rate of transport and the extent of contamination. Therefore *where should the wells be located to maximize the likelihood of detection?*
- The likelihood of detecting a contaminant increases as the plume size increases but still the smallest possible plume size is desired from a remediation perspective. Intuitively, as wells are located far away from the contaminant source the detection probability will be high, but the associated plume size will be large. On the other hand, wells located close to the contaminant source may have a smaller detection probability as well as a smaller plume size. Given the multi-objective nature of the problem, *what could be a consistent and systematic way to evaluate and compare alternative monitoring systems considering these conflicting criteria?*
- In the case that a monitoring system does not detect a contaminant plume before the plume reaches a compliance boundary at the end of the monitoring period, *what are the consequences in relation to future extent of contamination, site remediation and associated costs?*

The design of groundwater monitoring systems has been subject of considerable research. Different approaches have been used in groundwater monitoring system design, including purely deterministic (i.e., no uncertainty in the hydrology or in the parameters involved is explicitly considered), geostatistical (involving kriging and related techniques), optimization based (incorporating also uncertainty), simulation based and probability based approaches. Loaiciga et al. (1992) presented a thorough review of monitoring system design considerations, and noted that one of the main deficiencies of most design approaches is oversimplification of the subsurface. Although statistical methods are not totally exempt from this problem, it was still considered that these methods provide a means through which uncertainties from inherent heterogeneity of the involved variables, and simplifications and errors both in the modelling stage and the numerical/analytical solution phase, can be incorporated into the analysis in a systematic manner. Therefore, owing to the numerous uncertainties due to subsurface and contaminant source characteristics and because decisions must be made in the presence of significant uncertainties, statistical methods should be incorporated in the solution of the issues presented above.

1.4 Research objectives

The purpose of this study is to formulate a methodology for the design of an optimal groundwater detection monitoring system at landfill sites, under conditions of uncertainty due to subsurface heterogeneity and contaminant source location. The specific objectives of the research presented in this thesis follow from the problem definition given in Section1.3 and are defined as follows:

- to analyze the effects of the spatial heterogeneity of the subsurface and the uncertainties related to contaminant source (leak) location within the landfill on the reliability and efficiency of the monitoring systems,
- to analyze the influence of factors (e.g. dispersivity of medium, threshold concentration , leak size, type of leak and sampling frequency) controlling the size of the contaminant plume on the reliability and efficiency of the monitoring systems,
- to analyze the influence of locations and the number of monitoring wells on the detection probability and the cost of the systems,
- to establish the trade-off among the detection probability, early detection and cost,
- to present a methodology that maximizes the probability of detection while minimizing the contaminated area and total cost,
- to give insight into the effects of broadening the capture zone of monitoring well(s) on the efficiency of monitoring systems by continuous pumping from the monitoring well(s) with a small pumping rate,
- to apply and illustrate the presented methodology for a real landfill site.

1.5 Thesis outline

This thesis comprises eight chapters, which describe the objectives and results obtained in this study. Chapter 2 describes the characteristics of the simulation model to simulate the contaminant plumes originating from landfill leakage. The numerical methods of the solution of governing flow and contaminant transport equations are described in this chapter with special attention to the concept of subsurface heterogeneity and uncertainty modelling.

In Chapter 3, the detection probability of a contaminant plume released from a landfill has been investigated by means of both the simulation described in Chapter 2 and an analytical model for both homogeneous and heterogeneous aquifer conditions. The results of the two models are compared for homogeneous aquifer conditions to illustrate the errors that might be encountered with the simulation model. Moreover, modelling of contaminant transport by an analytical model using effective (macro) dispersivities is presented in this chapter and the results are compared with those obtained by the simulation model in order to investigate how far an analytical model can be applied for groundwater detection monitoring design in heterogeneous aquifer conditions.

Chapter 4 uses the models described in Chapter 2 to study the reliability of groundwater monitoring system in case of an instantaneous contaminant leak. Furthermore, the results of sensitivity analysis with respect to model parameters are discussed.

Chapter 5 presents the development of a multi-objective decision model (called *MONIDAM*), which links a classic decision analysis approach with models described in Chapter 2. The reliability of the groundwater monitoring system in case of a continuous contaminant leak, the application of *MONIDAM* to a hypothetical example, and the sensitivity analysis for the model parameters are also given in this chapter.

Implementation of a new monitoring approach has been introduced in Chapter 6 to design a highly efficient cost-effective 3-well system. In this new approach the main idea is to increase the interception of contaminant plumes at early stages by broadening the capture zone of monitoring well(s) simply by continuous pumping from the monitoring well(s) with a small pumping rate. The results are presented for conventional and proposed monitoring approaches in order to obtain insight into the influence of the proposed new monitoring approach on the efficiency of common monitoring systems.

Chapter 7 presents a decision model (called *MONIDAM-P*), which is an extension of the model described in Chapter 5, by implementing the new monitoring approach described in Chapter 6. Furthermore, the chapter includes the applications of both *MONIDAM* and *MONIDAM-P* to an actual site (Maarsbergen Landfill site). The goal of this application is to determine the optimal groundwater detection monitoring system at the site, to evaluate the efficiency of the existing monitoring system, to compare the efficiency of the estimated optimal monitoring systems with the efficiency of the existing one, and to augment the existing system, if necessary.

Finally, Chapter 8 concludes the thesis with a summary of the main results and conclusions, and recommendations for practical application and future research.

Chapter 2

Characteristics of the Simulation Model

Solution of the governing equations for groundwater flow and contaminant transport is necessary for groundwater monitoring design. A simulation model (adapted from Elfeki, 1996) coupling a Monte-Carlo framework with a two-dimensional finite difference flow model and a random walk particle-tracking model is used to simulate contaminant plumes. The object of this chapter is to describe the characteristics of the simulation model used in this research.

The first section of the chapter emphasizes the sources of uncertainties that have influence on the efficiency of groundwater monitoring systems and the approaches to model them. The properties of heterogeneity of hydraulic conductivity, some properties of the probability distribution that is often adopted to describe hydraulic conductivity–the natural-lognormal distribution– and the description of the generation of random fields to model heterogeneous hydraulic conductivity and the method of uncertainty analysis (Monte Carlo approach) are discussed. The next section describes the model discretization. Finally, the last two sections of the chapter describe the groundwater flow model and particle tracking model for contaminant transport.

2.1 Modelling uncertainty

The level of precision in contaminant plume characterization has a great influence on the success of a groundwater monitoring system. In other words, detection of a contaminant plume by a given monitoring system depends primarily on the concentration of contaminants at monitoring wells. Concentration of contaminants at specified well locations is influenced by many uncertain factors. Hydrodynamic dispersion and the variability of hydrogeological characteristics, such as hydraulic conductivity, the regional groundwater gradient direction and magnitude are usually considered as the major source of uncertainty in characterization of contaminant plumes as they affect the development of the plume in time and space. However, the nature of the contaminant source is also often highly uncertain. Source size and location, and type and rate of release of contaminants are the uncertainties due to the nature of the contaminants and the landfill characteristics that affect the initial characteristics of a contaminant plume.

In this research, a Monte Carlo (*MC*) approach is used for uncertainty analysis.The two main sources of uncertainty are treated in the simulations, due to the high computational expenses. The spatial variability of hydraulic conductivity and the contaminant source (leak) location within the landfill are the uncertainties taken into account in the analysis presented in the following chapters. Furthermore the influence of other parameter is investigated with a sensitivity analysis.

2.1.1 Uncertainty due to subsurface heterogeneity

Hydraulic conductivity, which is a measure of how easily a fluid can move through a porous material, is a particular contributor to uncertainty in contaminant transport. An analytical transport model typically requires the assumption that hydraulic conductivity is homogeneous. However, the transport of contaminants in groundwater is greatly affected by the manner in which the hydraulic conductivity varies in space (e.g., Gelhar, 1986; Freeze et al., 1987; Meyer et al., 1994). Areas of low hydraulic conductivity may slow the flow and reduce the spreading of a plume, whereas high conductivity zones may cause channelling of the plume and abrupt changes in contaminant concentrations.

The available field data indicate that the hydraulic conductivity can be appropriately modelled as a stochastic process (e.g., Smith, 1981; Hoeksema and Kitanidis, 1985; Sudicky, 1986; Hess et al., 1992). That is, the hydraulic conductivity a location may be modelled as a random variable following a specified probability distribution that is statistically correlated with hydraulic conductivity values at nearby locations. The earliest work (Warren and Price, 1961) of modelling the hydraulic conductivity as a random variable suggested that it follows a logarithmic normal distribution. Subsequent works by several researchers (e.g., Freeze, 1975; Sudicky, 1986; Hess et al.; 1992) confirmed this hypothesis. Hence, the function

$$Y = \ln X \tag{2.1}$$

is normally distributed, where X can be hydraulic conductivity K, or transmissivity T. The probability density function $f_X(x)$ of the log-normal distribution can be derived from the normal distribution and has the following expression

$$f_X(x) = \frac{1}{x\sigma_Y\sqrt{2\pi}}\exp\left[-\frac{(\ln x - \mu_Y)^2}{2\sigma_Y^2}\right] \tag{2.2}$$

where μ_Y and σ_Y are the mean and standard deviation of Y. The mathematical expectation or arithmetic mean of X is

$$X_A = \mu_X = E(X) = \exp(\mu_Y + 0.5\sigma_Y^2) \tag{2.3}$$

Two other mean values of X are the geometric mean X_G and the harmonic mean and the harmonic mean X_H. The former reads

$$X_G = \exp\left(\mu_Y\right) \tag{2.4}$$

and the latter reads

$$X_H = \frac{1}{E\left(X^{-1}\right)} = \exp\left(\mu_Y - 0.5\sigma_Y^2\right) \tag{2.5}$$

The variance of X can be written as a function of μ_Y and σ_Y and is given by

$$\sigma_X^2 = \left[\exp\left(\sigma_Y^2\right) - 1\right]\left[\exp\left(2\mu_Y + \sigma_Y^2\right)\right] \tag{2.6}$$

The concept of a random hydraulic conductivity can be extended to random fields (e.g., Marsily, 1986; Gelhar 1993; Meyer et al., 1994; Tompson et al., 1989) since the hydraulic conductivity of a geological formation also varies spatially. Furthermore, Hoeksema and Kitanidis (1985) studied considerable data and concluded that the use of a lognormal distribution and an exponential correlation function to model the spatial variability were reasonable.

Random hydraulic conductivity fields

A description of the main characteristics of random fields is presented in this section. A random field is an extension of the random variable concept in that it is assumed that the parameter modelled exhibits correlation in space in addition to its local variability. For instance, the hydraulic conductivity throughout a geological formation is generally unknown due to the impossibility of measuring conductivity at every location. In addition to the uncertainty due to the insufficient measurements, the hydraulic conductivity varies from one location to another throughout the formation. Two points located near to each other will be more likely to exhibit similar values than two points separated by a large distance. Therefore in addition to a mean value and standard deviation of the hydraulic conductivity, a parameter that measures the degree of correlation between the points is useful. In this regard, the concept of random field is of use. Using the mean, variance and correlation length the spatial variability of hydraulic conductivity can be modelled. In this way the subsurface heterogeneity as it relates to hydraulic conductivity can be modelled probabilistically in a consistent way.

Random fields are considered first order stationary when the mean value is constant in space, and they are considered non-stationary if the mean value varies in space. They are called isotropic when the correlation characteristics are the same in all directions. When the correlation length is very large in relation to the area of interest, the spatial variation pattern will be very smooth. If a very small variance exists, the field is practically deterministic with values equal to the expected value throughout the space, independent of correlation characteristics. In general, geological formations exhibit less variability in the direction of their bedding (mostly horizontally) than the direction perpendicular to it. Therefore, unless a very large amount of data is available to confirm the presence of buried channels or stream beds, generally isotropic conditions are assumed in the bedding direction. Hydraulic conductivity fields are thus represented by stationary random fields (e.g., Marsily, 1986; Gelhar, 1993; Meyer et al., 1994; Storck, et al., 1997; Montas et al., 2000).

Hence in this research, subsurface heterogeneity is represented probabilistically by a lognormal random hydraulic conductivity field. The random hydraulic conductivity field is modelled as an isotropic stationary Gaussian field with a given mean μ_Y, variance$\sigma_Y{}^2$ and correlation length λ (see e.g., Gelhar, 1986). In other words, the mean and the covariance of the hydraulic conductivity are independent of location and the correlation distance is independent of direction. An exponential form is considered for the correlation characteristics. Gelhar (1993) summarized the wide-ranging data on variance and correlation length of the natural logarithm of the hydraulic conductivity from several different sites. The values chosen in this thesis reflect a range of field observations presented in the literature from investigations carried out on a scale corresponding to that of the model presented here.

Generation of random hydraulic conductivity fields

There are several methods to generate Gaussian random fields. The Turning Bands Method, Lower-Upper decomposition, Sequential Gaussian Simulation, Nearest Neighbour Method are commonly used in the literature. Not only because an existing code was available and modifications could be easily incorporated when needed, but also being the most widely used technique in subsurface hydrology, the turning bands method, proposed originally by Matheron (1971) and adopted for two dimensions by Mantoglou and Wilson (1982), was used in this study. The algorithm generates a random field by superposition of a series of one-dimensional simulations along several lines radiating outward from an arbitrarily chosen origin in space.

The Turning Bands Method is a repetition of a two-step procedure. First a realization of a random process with zero mean and a prescribed covariance function is generated on one line. Second, each point in the simulated random field is orthogonally projected onto the generated line. The two steps are repeated for a given number of lines and then a final value is assigned to each grid point in the field by taking a weighted average over the total number of lines. For further reading on the turning bands method the reader is referred to Mantoglou and Wilson (1982) and Tompson et al. (1989).

2.1.2 Uncertainty due to contaminant source location

The specific location and size of the source contaminants (i.e. the location and size of the leak) within a landfill is one of the major sources of uncertainty even though the potential area of contamination, defined by the landfill boundary, is well defined. The reason is that the actual release of contaminants may occur over a small portion of the landfill. The likelihood of a release of contaminants to the environment is related to the ability of the liner system of the landfill to prevent leaks from occurring. Although liner systems are highly redundant, there are still areas in the landfill where the liner may not be as reliable in preventing leaks. The location and size of a leak will depend upon the type of liner system (e.g., earthen or synthetic, single or composite) and the construction and the quality of the liner. Flexible membrane liners generally fail either along poorly bonded seams or through punctures. Compacted clay liners may fail due to preferential flow through localized zones of relatively highly hydraulic conductivity. They are likely to fail near confining walls along the liner boundary (Bagchi, 2004).

There appears to be no detailed information available on the probabilities of the spatial configuration of liner failures. It is thus reasonable to assume that it is possible to identify a portion of the liner that represents all potential leak locations. This portion might be the seam of a flexible membrane liner or it could be the entire areal extent of the liner. In this thesis, liner failure is modelled as point source that occurs at random from a set of potential leak locations. The assumption of a small leak is conservative, as it is more difficult to detect such type of leaks than to detect large leaks or simultaneous, multiple leaks from several locations. The random leak location is assumed to follow a uniform probability distribution. This choice is mainly due to the absence of specific data.

2.1.3 Monte Carlo approach

The groundwater detection monitoring system design methodology developed and presented in this study uses a Monte Carlo (*MC*) approach in the analysis of the uncertainties mentioned above. The *MC* approach was chosen in this research since it is conceptually straightforward and has also been used extensively to deal with groundwater flow and/ or contaminant transport problems in heterogeneous media (e.g. Gelhar, 1986; Meyer et al., 1994; Storck et al., 1997; Montas et. al, 2000). The *MC* approach is a powerful tool for simulations of stochastic phenomena and requires not many assumptions. It does not attempt to solve the problem as a stochastic differential equation but rather computes deterministic solutions for a number of numerically generated realizations, and analyzes the ensemble of realizations statistically to estimate means, variances and probability density functions. Therefore, it is easy to implement even in case of multiple uncertain input parameters. On the other hand, a large number of realizations, which makes the approach computationally quite expensive, is required in order to obtain output values accurately. However, this limitation becomes less of concern with the recent developments of computers.

In this research, the *MC* approach is based mainly on generating a random field of the hydraulic conductivity to represent the subsurface heterogeneity, to generate the leak location and the particles. As mentioned in Section 2.1.1 and Section 2.1.2, a log-normal probability distribution is assumed for the hydraulic conductivity and a uniform probability distribution is assumed for the leak location. Using a random number generator several realizations for these parameters are generated. Then the groundwater flow and/or transport equations that are described in the following sections are solved numerically to obtain the concentration distribution at each point in time at each node in the grid model. The specific *MC* simulation procedure used for the analysis presented in the following chapters is accounted for in the text.

2.2 Model domain and model discretization

One can legitimately argue that two-dimensional simulations are poor approximations to natural three-dimensional systems. However, for regional scale problems, where the planar dimension of an aquifer is much larger than its thickness, two-dimensional models give results with minor deviations from the reality (Dagan, 1986; Rubin, 1990; Boggs et al., 1992). In a two-dimensional model, formation properties are averaged over the depth and regarded as a function of horizontal dimension only. Using the fully penetrating monitoring wells in the calculations, the possible density effects due to higher concentrations of a leachate, which is kept outside the scope of this thesis, could also be compensated as well as the integration of the quality over the entire depth of the aquifer in proportion to the vertical conductivity distribution. Furthermore, the sampling procedure itself, which requires the extraction of a given amount of water from them before taking the sample, might be another rationale for the 2D analysis. This sampling procedure provides a water sample originating from the entire depth of the aquifer while ensuring the removal of stagnant water. However, possible effects of short-circuiting inside long piezometer screens due to vertical gradients may cause uncertainty with respect to the origin of a given sample. Although this factor is regarded beyond the scope of this thesis, it will be prevented by the new monitoring approach (based on continuous low level pumping) introduced in Chapter 6. Moreover, Freyberg (1986), found that the motion of a plume and its centre of mass are essentially horizontal. Moltyaner et al. (1993), investigated the effect of dimensionality on transport at Twin Lakes using a natural gradient test. They found that over the first 40 m along the mean flow path, the three-dimensional model does not reproduce the plume migration any better than a two-dimensional model. Therefore, considering the computational cost required for three-dimensional transport modelling, a two-dimensional confined aquifer model is considered in this study.

A rectangular model domain with a length of L_x, width of L_y and a computational grid spacing Δx and Δy in the x- and y-directions has been used in the analyses throughout the thesis. Choosing a domain size and a discretization level for the model domain is the trickiest issue in numerical analysis. In general, the higher the discretization level of flow and transport problems, the better will be the subsequent solution of flow and transport equations. However, the higher the discretization level, the greater the computational effort required. Therefore a balance must be found between the level of discretization and the computational expense. Ababou et al. (1989) suggested that a correlation length four times greater or equal to the domain discretization and smaller or equal to one twenty-fifth of the domain dimension is required for statistically meaningful results from replicates of a stationary hydraulic conductivity field. They proposed as a rule of thumb the following ratio between the grid cell size Δx and correlation length λ with respect to the degree of heterogeneity:

$$\frac{\lambda}{\Delta x} \geq 1 + \sigma_Y^2 \tag{2.7}$$

On the other hand, Bellin et al. (1992) found that a ratio of $\lambda / \Delta x = 4$ in the range of $\sigma_Y^2 \leq 1.6$ provided satisfactory accuracy and convergence of computations. The principal ratio applied in this thesis is $\lambda / \Delta x = 7.5$, which also satisfies the ratio suggested by Ababou et al. (1989). Therefore, this discretization level is considered to be sufficient for all cases tested and whenever a different ratio is used, this is accounted for in the text.

2.3 Groundwater flow model

Transport of contaminants in groundwater is dependent on the nature of the flow systems since the contaminant migration follows the path lines. In a steady state flow system the velocity field is kept constant during contaminant transport simulations, whereas in a transient system the velocity field changes in time. This variability in time requires multiple solutions of the model at successive times over the period of interest. Formulating the transient conditions in a *MC* framework as described above, is computationally very demanding and it may not be feasible for practical applications. Therefore in this study a two-dimensional steady-state saturated groundwater flow in an isotropic heterogeneous aquifer in a horizontal plane is assumed. This choice is mainly to simplify the hydrogeological environment and to understand thoroughly the influence of subsurface heterogeneity, dispersivity of medium and other parameters regarding the contaminant source characteristics on efficiency of groundwater monitoring systems at a reasonable computational effort.

The general equation for two dimensional steady state groundwater flow in a horizontal plane is (Kinzelbach, 1986):

$$div\left[T\, grad\, h\right] = 0 \tag{2.8}$$

where $T = KB$ is the transmissivity tensor $\left[L^2/T\right]$, K = hydraulic conductivity $\left[L/T\right]$, B = aquifer thickness $\left[L\right]$, h = the hydraulic head$\left[L\right]$. The transmissivity tensor is written as

$$T = \begin{bmatrix} T_{xx} & T_{xy} \\ T_{yx} & T_{yy} \end{bmatrix} \tag{2.9}$$

It must be noted that T is symmetric, hence, $T_{xy} = T_{yx}$. In simplified form when the major axes are aligned with the main direction of flow, the off diagonal terms disappear (i.e. $T_{xy} = T_{yx}$=0). The two dimensional model of steady-state saturated groundwater flow in an isotropic heterogeneous aquifer with a constant thickness is applied on a rectangular domain of dimension $(0 \le x \le L_x,\ 0 \le y \le L_y)$ in this research. Hence the equation to be solved is:

$$\frac{\partial}{\partial x}\left(K_{xx}\frac{\partial h}{\partial x}\right) + \frac{\partial}{\partial y}\left(K_{yy}\frac{\partial h}{\partial y}\right) = 0 \tag{2.10}$$

where K_{xx} is the hydraulic conductivity in the x- direction and K_{yy} is the hydraulic conductivity in the y- direction. The solution of this equation with regard to boundary conditions results in the hydraulic head h, as a function of x and y for saturated heterogeneous media under steady state conditions.

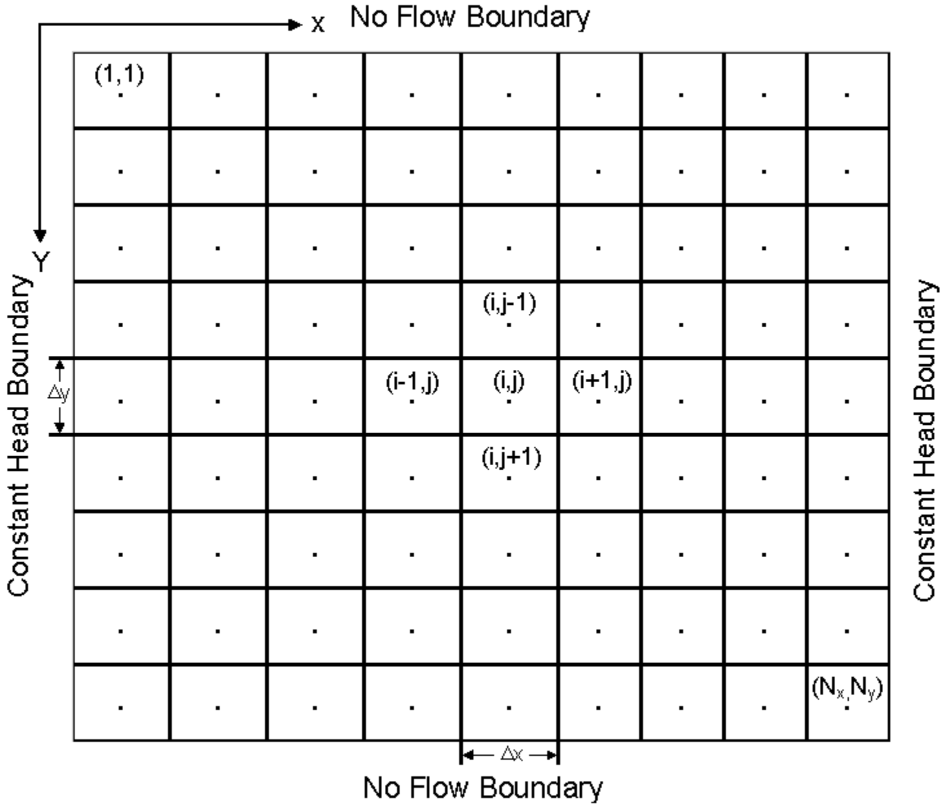


Figure 2.1: 2D finite difference grid discretization with flow boundary conditions (adapted from Kinzelbach, 1986).

A block-centred five-point finite difference method is used to discretize Equation (2.10). For the details on block-centred five-point finite difference method reader is referred to Kinzelbach (1986) and Bear and Verruijt (1987). Figure 2.1 schematizes the block-centred five-point (central point plus 4 neighbouring points) finite difference method used in this study. The domain of interest is discretized into a number of cells with dimensions of Δx and Δy, in the x- and the y- directions respectively. The variable of interest is defined at cell centroids, which are known as grid points or nodes. The groundwater flow is approximated by;

$$K_{xx}\frac{\partial h}{\partial x} \approx K_{xx}(i+\frac{1}{2},j)\left[\frac{h(i+1,j)-h(i,j)}{\Delta x}\right] \tag{2.11}$$

here, $K_{xx}(i+\frac{1}{2},j)$ is the interface hydraulic conductivity between node (*i+1, j*) and node (*i, j*). The hydraulic conductivity estimated with the harmonic mean of the neighbouring nodes in x- direction is given by,

$$K_{xx}(i+\frac{1}{2},j)=\frac{2K_{xx}(i+1,j)K_{xx}(i,j)}{K_{xx}(i,j)+K_{xx}(i+1,j)} \tag{2.12}$$

Similarly in y- direction,

$$K_{yy}\frac{\partial h}{\partial y} \approx K_{yy}(i, j+\frac{1}{2})\left[\frac{h(i, j+1)-h(i, j)}{\Delta y}\right] \tag{2.13}$$

where,

$$K_{yy}(i, j+\frac{1}{2}) = \frac{2K_{yy}(i, j+1)K_{yy}(i, j)}{K_{yy}(i, j+1)+K_{yy}(i, j)} \tag{2.14}$$

Then,

$$\frac{K_{xx}(i+\frac{1}{2}, j)\left[\frac{h(i+1, j)-h(i, j)}{\Delta x}\right] - K_{xx}(i-\frac{1}{2}, j)\left[\frac{h(i, j)-h(i-1, j)}{\Delta x}\right]}{\Delta x} +$$

$$+\frac{K_{yy}(i, j+\frac{1}{2})\left[\frac{h(i, j+1)-h(i, j)}{\Delta y}\right] - K_{yy}(i, j-\frac{1}{2})\left[\frac{h(i, j)-h(i, j-1)}{\Delta y}\right]}{\Delta y} = 0 \tag{2.15}$$

Define the coefficients:

$$A(i, j) = K_{xx}(i+\frac{1}{2}, j)/\Delta x^2 \tag{2.16}$$

$$B(i, j) = K_{yy}(i, j+\frac{1}{2})/\Delta y^2 \tag{2.17}$$

$$C(i, j) = K_{xx}(i, j-\frac{1}{2})/\Delta x^2 \tag{2.18}$$

$$D(i, j) = K_{yy}(i, j-\frac{1}{2})/\Delta y^2 \tag{2.19}$$

$$E(i, j) = A(i, j)+B(i, j)+C(i, j)+D(i, j) \tag{2.20}$$

Then the two-dimensional finite difference approximation to the groundwater flow equation can be written as:

$$A(i, j)h(i+1, j)+B(i, j)h(i, j-1)+C(i, j)h(i-1, j)+D(i, j)h(i, j+1)-E(i, j)h(i, j) = 0 \tag{2.21}$$

Dirichlet and Neumann boundary conditions are used to solve the equations. Dirichlet conditions are related to a known hydraulic head and Neumann conditions are related to known flow values (Bear and Verruijt, 1987). In this study, no flow and constant head are considered on the boundaries of the flow domain. Once the hydraulic heads are obtained from the solution of the groundwater flow equation, the internodal Darcy's velocity components $q_x(i+\frac{1}{2}, j)$, between nodes (*i+1, j*) and (*i,j*), and $q_y(i, j+\frac{1}{2})$, between nodes (*i, j+1*) and (*i, j*) can be computed as:

$$q_x(i+\frac{1}{2},j) = -K_{xx}\frac{\partial h}{\partial x} \approx -K_{xx}(i+\frac{1}{2},j)\left[\frac{h(i+1,j)-h(i,j)}{\Delta x}\right], \tag{2.22}$$

and

$$q_y(i,j+\frac{1}{2}) = -K_{yy}\frac{\partial h}{\partial y} \approx -K_{yy}(i,j+\frac{1}{2})\left[\frac{h(i,j+1)-h(i,j)}{\Delta y}\right] \tag{2.23}$$

The groundwater flow velocities in the x- direction (v_x) and the y- direction (v_y) are calculated by dividing the Darcy velocities by the effective porosity of the medium. The average groundwater velocities are used as a part of input to the contaminant transport model. The conjugate gradient method is used to solve the groundwater flow equation for saturated heterogeneous media. The method involves an iterative procedure in which the estimate of the hydraulic head at each cell is updated at every iteration until a convergence criterion is fulfilled.

2.4 Particle tracking model for contaminant transport

In this study the movement of contaminants in the subsurface is represented by the advection-dispersion equation.

The contaminant is assumed to be conservative and to have no interaction with the solid matrix. The rationale is to simplify the parameter sensitivity analysis in order to investigate the influence of the nature of the transport environment, mainly the dispersivity and heterogeneity of the medium, on monitoring system design in a simple and straightforward manner, unencumbered by the complications of biological and chemical interactions such as retardation, decay and microbiological transformation. In the design of capture and containment systems in heterogeneous medium, advection and dispersion are the most important transport mechanisms.

However, the biological processes usually leads to the reduction of the concentration of particular organic contaminants but in general do not ensure a reduction in toxicity. On the other hand chemical interactions such as adsorption/desorption or decay can significantly slow the rate of the contaminant transport (Gorelick et al., 1993). The spatial concentration distribution curve will be steeper at the plume front and flatter at the plume tail, when retardation is taken into account (Bear and Buchlin, 1987). Ultimately, a transient plume migration in a steady state flow domain is considered in this study and the two-dimensional advection-dispersion equation for this case can be written as (Bear, 1972):

$$\frac{\partial C}{\partial t} + v_x\frac{\partial C}{\partial x} + v_y\frac{\partial C}{\partial y} - \frac{\partial}{\partial x}\left[D_{xx}\frac{\partial C}{\partial x} + D_{xy}\frac{\partial C}{\partial y}\right] - \frac{\partial}{\partial y}\left[D_{yx}\frac{\partial C}{\partial x} + D_{yy}\frac{\partial C}{\partial y}\right] = 0 \tag{2.24}$$

where C is the concentration of the contaminant at time t at location (x, y), v_x and v_y are the average groundwater flow velocity components in the x- and y- direction respectively, and D_{xx}, D_{xy}, D_{yx}, D_{yy} are the components of the pore scale hydrodynamic dispersion tensor (Bear, 1979),

$$D_{ij} = \left(\alpha_T |v| + D_m\right)\delta_{ij} + \left(\alpha_L - \alpha_T\right)\frac{v_i v_j}{|v|}, \tag{2.25}$$

where δ_{ij} is the Kronecker delta (δ_{ij} =1 for $i = j$ and δ_{ij} =0 otherwise), α_L is the longitudinal dispersivity, α_T is the longitudinal dispersivity, D_m is the molecular diffusion coefficient and $|v|$ is the mean groundwater velocity given by

$$|v| = \sqrt{v_x^2 + v_y^2} \tag{2.26}$$

Having obtained the velocity field for each realization of the hydraulic conductivity field, the solution of the transport equation and the spatio-temporal evolution of the concentration field are obtained by employing a random walk particle model. It is assumed that $C(x, y, 0) = 0$ for $0 \le x \le L_x$, $0 \le y \le L_y$. The boundary condition $\partial C / \partial y\ (x, 0, t) = 0$, $\partial C / \partial y\ (x, L_y, t) = 0$ for $t \ge 0$ is imposed.

In this study the random walk particle tracking model is used to perform the transport simulations since it facilitates the solution of problems having zero or low dispersivity values (large Peclet numbers), and since it does not exhibit numerical dispersion (Kinzelbach, 1986). The particle tracking model is basically the representation of the spatial distribution of some extensive quantity, such as the mass of a particular chemical constituent, by a large collection of particles. The state of the system at some particular time will be defined by a set of attributes associated with each particle, say, for example, position, mass, or species type.

In this study, the particles represent the mass of a conservative contaminant in the flow field. Each particle is assigned the same fixed amount of contaminant mass. Dispersion is modeled by superimposing a random movement on the convective particle movement, which has the statistical properties that correspond to the properties of the physical dispersive process. A large number of individual random walks of particles form a dispersing particle cloud characterizing a contaminant mass distribution.

In the random walk particle tracking model the concentration distribution at a fixed time has the form of the probability density function of a normal variable with mean value μ and standard deviation σ:

$$f(x) = \frac{1}{\sqrt{2\pi}\sigma} \exp\left[-\frac{1}{2}\left(\frac{x-\mu}{\sigma}\right)^2\right] \tag{2.27}$$

The solution to the advection-dispersion equation in one dimensional form for an instantaneous release of a solute of M_o (g) from location x_o, longitudinal dispersivity α_L, and mean groundwater flow velocity v_x in the x- direction, is:

$$C(x,t) = \frac{C_0}{\sqrt{4\pi\alpha_L v_x t}} \exp\left[-\frac{\left(x - x_0 - v_x t\right)^2}{4\alpha_L v_x t}\right] \tag{2.28}$$

where $C_0 = M_0 / \varepsilon B$, with ε the effective porosity and B the aquifer thickness. Comparing the two equations, it is clear that the mean value and the standard deviation are (Figure 2.2):

$$\mu = x_0 + v_x t \tag{2.29}$$

$$\sigma = \sqrt{2\alpha_L v_x t} \tag{2.30}$$

implying that the position of the centre of the plume moves at the mean groundwater velocity and the plume disperses around this centre with a variance that depends on the dispersion coefficient and increases linearly with time. The same concept can be extended to two-dimensional modelling with longitudinal and transverse dispersion.

Given the analogy between the transport equation (Equation (2.28)) and the Fokker-Planck equation (Uffink, 1990), the two-dimensional particle tracking equations incorporating dispersion can be written as (Kinzelbach, 1986):

$$X_p(t+\Delta t) = X_p(t) + v_x \Delta t + \left(\frac{\partial D_{xx}}{\partial x} + \frac{\partial D_{xy}}{\partial y} \right) \Delta t + \frac{v_x}{|v|} Z \sqrt{2\alpha_L |v| \Delta t} - \frac{v_y}{|v|} Z' \sqrt{2\alpha_T |v| \Delta t} \tag{2.31}$$

$$Y_p(t+\Delta t) = Y_p(t) + v_y \Delta t + \left(\frac{\partial D_{yx}}{\partial x} + \frac{\partial D_{yy}}{\partial y} \right) \Delta t + \frac{v_y}{|v|} Z \sqrt{2\alpha_L |v| \Delta t} + \frac{v_x}{|v|} Z' \sqrt{2\alpha_T |v| \Delta t} \tag{2.32}$$

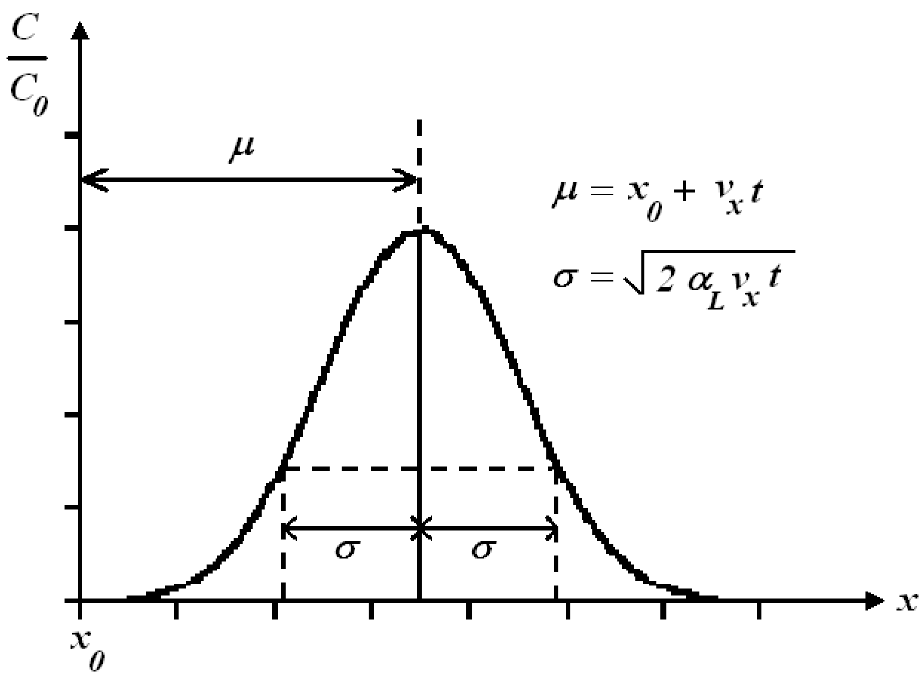


Figure 2.2: Solution of the transport equation in one dimension viewed as normal distribution (after Kinzelbach, 1986).

where $X_p(t)$, $Y_p(t)$ are the *x*- and *y*- coordinates of a particle at time *t*, Δt is the time step used in calculations, *Z*, *Z′* are two independent random numbers drawn from a normal distribution with mean zero and variance one, α_L is the longitudinal, and α_T the transverse dispersivity and *v* is the resultant flow velocity.

On the right hand sides of both Equation (2.31) and (2.32), the first terms correspond to the previous position of the particle, the second terms correspond to the convective displacement, the third terms are the Fokker-Plank term (a counter-term has to be added to correct the unrealistic accumulation of particles at stagnation zones), and the last two terms are the stochastic dispersive displacements projected in the *x*- and the *y*- directions respectively.

The dispersivities of the medium, α_L and α_T depend on many factors including the scale of measurements, numerical properties, model dimensions, and possibly also time and space. Previous studies show that there is a very wide range of dispersivity values (on the order of millimetres to several meters) and the ratio of transverse to longitudinal dispersivity is in general less than 1 and could be considered as a constant (see e.g., Bear, 1972; Gelhar, 1986; Loaiciga, 1989; Smedt and Bronders, 1989; Meyer et al., 1994; Vomvoris and Gelhar, 1990; Storck et al., 1997; Ribeiro, 2000;, Hudak, 2002; Rahman et al., 2005; Cirpka et al., 2006; Shulze-Makuch, 2005). It must be noted that there is an ongoing research in this field and that final agreement has not yet been reached. Therefore dispersivity values used in the analysis presented in the following chapters were chosen based on aforementioned studies and among them, special attention was given to those studies that refer to the adverse affect of landfill leachates on groundwater quality. Moreover, in the numerical experiment presented in the coming chapters D_m is assumed to be zero in order to limit the number of parameters in the model and simplify the analysis.

The solution of the advection-dispersion transport equation by the random walk method provides the discrete particle displacements and not the concentration values. A discretized grid model, similar to the one used in the solution of groundwater flow equations, is superimposed to convert the particle distributions into concentrations. The average concentration at time *t* in a grid cell (*i*, *j*) with dimensions Δx and Δy in (*x*- and *y*-directions respectively), is:

$$C_{ij}\left(t\right)=\frac{M_o n_{ij}(t)}{N\varepsilon b_{ij}\Delta x\Delta y} \tag{2.33}$$

where $C_{ij}(t)$ is the volume averaged concentration in grid cell (*i*, *j*) at time *t*, $n_{ij}(t)$ is the number of particles in grid cell (*i*, *j*) at time *t*, *N* is the total number of particles released, ε is the effective porosity and b_{ij} is the thickness of the grid cell (assumed to be constant unit thickness as a 2D model is considered in this research).

One should be aware that the number of particles used in the model has a great influence on the computation of concentration values. In advection modelling, two particles at the same initial location will follow the same path since it is only determined by the groundwater flow field; hence a small number of particles is needed, which reduces the computational effort. On the other hand, when modelling dispersion, the number of particles used is important. Since spreading of the contaminants is affected by a random component, two particles placed at the same initial location will most likely follow different paths, although on average (due to the law of large numbers) they will follow the advective transport path. A small number of particles may not model the spreading of the plume appropriately, resulting in incorrect estimates of the contaminant concentration.

In addition, the time and release rate of contaminants will influence the concentration characteristics. For the simulation of a continuous leak, new particles start from the source location at every time step Δt. This is computationally very expensive since it leads to the use of a very large number of particles. However, in the case of a stationary flow field and a source of constant strength a continuous source can be simulated by convolution from the solution for an instantaneous pulse of contaminants using a relatively small number of particles (Kinzelbach, 1986). It is assumed that N particles released at time $t+\Delta t$ will follow the same paths as N particles released at t. The concentration distribution in every time step is obtained by adding the N moving particles to the old concentration distribution.

On the other hand, Δt should be small enough so that the particles will move continuously from one grid cell to another and the overshoot in the field is avoided, i.e., the particle should not move more than one grid cell during a time step. Considering a simple rule of thumb (Tompson and Gelhar, 1990), in the calculations presented in the following chapters a Δt value is chosen that satisfies the restriction $v_{max}\, \Delta t << \Delta x$, where v_{max} is the maximum velocity in the flow field and Δx is the grid cell dimension in the mean flow direction (x- direction).

The dispersion in the model fulfils two roles: to introduce microdispersion and to mimic dispersion due to heterogeneities on the sub cell scale, which includes those due to the integration over the thickness of the aquifer. In a 2D analysis, any subscale heterogeneity must be captured by modelling dispersion as a mixing process that is subjected to Fick's law. This mixing mechanism is also necessary to allow particles for hopping from one flow line to another and to mimic other effects like 3D rotational fields due to local changes in anisotropy (Hemker and Bakker, 2004). Therefore, without transverse dispersion on the sub-cell scale, macrodispersion due to heterogeneities and 3D rotations will not occur.

Chapter 3

Detection of Contaminant Plumes Released from Landfills

Adapted from Yenigul, N.B., Hensbergen, A.T., Elfeki, A.M.M., and Dekking, F.M., Detection of contaminant plumes released from landfills, submitted to Hydrological Earth Systems and Sciences and published in HESSD, 3, 819-857, 2006.

This chapter presents the application of the simulation model presented in the previous chapter plus an analytical model for both homogeneous and heterogeneous aquifer conditions in order to compute the detection probability of a contaminant plume released from a landfill. The chapter begins with introductory remarks revealing the particular concern of this chapter. The next two sections describe how the simulation model computes the detection probability and the characteristics of the analytical model. Finally an illustrative example is presented.

3.1 Introduction

Analytical models are generally available only for very simplified situations such as homogeneous medium and uniform flow. Simulations are used to incorporate the properties related to heterogeneity, as geologic environments are seldom uniform and homogeneous. The assumption of homogeneous conditions (e.g., hydraulic conductivity constant in space) in groundwater flow problems may yield an appropriate approximation in some situations. In contamination problems however, the extent and characteristics of a contaminant plume may be significantly influenced by the heterogeneous nature of geologic formations. Areas of low hydraulic conductivity may slow the flow and reduce the spreading of a plume, whereas high conductivity zones may cause channelling of the plume and abrupt changes in contaminant concentrations. These types of regimes cannot be appropriately analyzed under assumptions of a homogeneous medium. Still, the significance of analytical models should not be underestimated, as they are important tools to verify the simulations and to obtain a thorough understanding of the phenomena. Hence in the first, homogeneous aquifer conditions are considered. The concentration distribution of contaminants and the detection probability of monitoring wells are determined for both instantaneous and continuous leak cases by simulation and analytical models. The results of the simulation model are compared with those of the analytical model for homogeneous aquifer conditions to illustrate the errors that might be encountered with the simulation model and to investigate the influence of certain parameters.

Secondly, a comparison between results of simulations and results of a particular analytical model in heterogeneous aquifer conditions is presented. Since there is a general agreement that hydraulic conductivity variations play an important role in contaminant transport a very primitive worst-case assumption for homogenization of a heterogeneous medium might be using a large hydraulic conductivity value (although still homogeneous). This may result in over estimation of the velocity and extent of the plume. Consequently, this may result in very conservative and costly monitoring. On the other hand, if a very small value of hydraulic conductivity is used, unconservative designs may result in under estimation of the contaminant plume. In the last two decades a significant amount of research has been devoted to the comprehension of the effects of natural heterogeneity on solute transport and to the development of modelling techniques which explicitly account for natural heterogeneity (e.g., Gelhar et al., 1979; Gelhar and Axness, 1983; Dagan, 1984 and 1986; Vomvoris and Gelhar1990; Thompson and Gelhar, 1990; Rubin, 1990; Kapoor and Gelhar, 1994 a and b; McLaughlin and Ruan, 2001; Hu et. al, 2002). Clearly, modelling of contaminant transport using an advection-dispersion equation with effective (macro) dispersivities is common practice. The effective (macro) dispersion coefficient embodies the effect of unresolved advective heterogeneity on the spatial second moment and can be used to describe the average concentration distribution. In this study, the mean concentration field is determined (e.g., Kapoor and Gelhar, 1994) using the effective dispersion coefficient in the analytical model. Here the effective dispersion coefficient is the summation of the local dispersivities and constant macrodispersivities as computed by Gelhar and Axness (1983) and the detection probability of the contaminant plume is computed for homogenized heterogeneous aquifer conditions. The results of the analysis based on the simulation and analytical model are compared to find the answers to the questions: How far an analytical model can be used in groundwater monitoring system design while incorporating the effects of various heterogeneities on contaminant transport? How accurate can the detection probability of a contaminant plume by a given monitoring well be computed by an analytical model, which uses macrodispersivities to homogenize the heterogeneity? How large will be the discrepancies between the results obtained by the two models?

3.2 Probability of detection by the simulation model

The simulation model presented in the previous chapter is used to compute the detection probability, $P_{d(mw)}$ of a given monitoring well. First, a realization of a random hydraulic conductivity field (for heterogeneous media) is generated. After solving the steady state groundwater flow model to determine the velocity field a random leak location is generated. Then the random walk transport model is solved to determine the concentration field of the contaminant plume. Finally, the model checks whether the concentration value at a given monitoring well location exceeds a given threshold concentration (detection limit) to determine whether a plume is detected or not detected by a given monitoring well.

Detection of a contaminant plume by a monitoring well (*mw*), is defined as the event where the contaminant concentration at the well location, C_{mw} at some time t is equal to or greater than a given threshold concentration C_{TH}. Therefore the probability of detection $P_{d(mw)}$ of a given plume by a given monitoring well is:

$$P_{d(mw)} = P\left(C_{mw} \geq C_{TH}, \text{ at some time } t\right) = \frac{1}{N_{MC}} \sum_{i=1}^{N_{MC}} I_d^{(i)} \tag{3.1}$$

Here, N_{MC} is the total number of simulation runs, i.e., the number of the plumes, $I_d^{(i)}$ is the indicator function of detection by the monitoring system for realization i, i.e. $I_d^{(i)}$ equals 1 if the simulated contaminant plume i is detected by the given monitoring well, and equals zero otherwise.

3.3 Description of the analytical model

3.3.1 Homogeneous aquifer conditions

The concentration at position (x, y) and at time t due to an instantaneous release of contaminant at location (x_0, y_0) is given by (Bear, 1972),

$$C(x, y, t) = \frac{C_0}{\sqrt{4\pi\alpha_L v_x t}\sqrt{4\pi\alpha_T v_x t}} \exp - \left[\frac{(x - x_0 - v_x t)^2}{4\alpha_L v_x t} + \frac{(y - y_0)^2}{4\alpha_T v_x t}\right] \tag{3.2}$$

This is a point wise, concentration whereas in the simulation model the concentration is calculated by means of particles in a grid cell (see Equation (2.33)). Hence one must average the concentration over the grid cells in order to make an equitable comparison between the concentration values calculated by the analytical and the simulation model. Therefore a weighted average of the theoretical concentration with weights corresponding to Simpson's rule for dimension 2 is used in the analytical model. In highly dispersive media and/or far away from the source the averaging does not make much difference since the plumes are already quite spread out in such cases. However for the locations where the plume is very peaked the effect will be very noticeable. But even in the region where the averaging does not matter, the Simpson approximation for the integral over a grid cell will give a small bias.

To find the concentration of a plume resulting from a continuous leak two different approaches can be taken. The first approach is to approximate such a plume by repeated small instantaneous plumes at short time intervals. In fact, taking the intermittent time intervals shorter and shorter, apart from inherent numerical instability around the origin, in this way the exact concentration will be approached better and better. The second approach is to use the approximation of the concentration by the Hantush well function (Kinzelbach, 1986). Calculations with Matlab showed that for wells not too far from the source the two approximations are quite close, but further away the Hantush approximation breaks down. The Hantush function looks like an elegant closed form, but the improper integral it contains limits its numerical application. For large x- values, numerical breakdown occurs as in the Hantush formula a very large number is multiplied with a number close to zero.

3.3.2 Heterogeneous aquifer conditions

Heterogeneity can be dealt with by defining the homogeneous equivalent properties, known as averaging. The advection-dispersion equation that includes the effect of the variations of velocities at the local and regional scale on solute dispersion to describe the (average) solute transport can be written as (Kapoor and Gelhar, 1994):

$$\frac{\partial \bar{C}}{\partial t} + v\frac{\partial \bar{C}}{\partial x_1} - v(A_{ij} + \alpha_{ij})\frac{\partial^2 \bar{C}}{\partial x_i \partial x_j} = 0 \tag{3.3}$$

where $\bar{C}$ is the mean concentration, v is the mean velocity in the x_1 direction, A_{ij} and α_{ij} are the macrodispersivities and local dispersivities, respectively. The mean concentration, governed by Equation (3.3) for an instantaneous release of contaminant is assumed to be Gaussian. Thus in order to include both local and regional dispersion in the analytical model and compute the mean concentration value at position (x, y) and time t due to an instantaneous release of contaminant at location (x_0, y_0) Equation (3.2) is modified as follows:

$$\bar{C}(x,y,t) = \frac{C_0}{\sqrt{4\pi(A_L+\alpha_L)v_x t}\sqrt{4\pi(A_T+\alpha_T)v_x t}}\exp - \left[\frac{(x-x_0-v_x t)^2}{4(A_L+\alpha_L)v_x t} + \frac{(y-y_0)^2}{4(A_T+\alpha_T)v_x t}\right] \tag{3.4}$$

Theoretically derived A_L and A_T values are given by (Gelhar and Axness, 1983),

$$A_L = \sigma_Y^2 \lambda / \gamma^2 \quad \text{and} \quad A_T = \frac{\sigma_Y^2 \alpha_L}{8\gamma^2}\left(1 + 3\frac{\alpha_T}{\alpha_L}\right) \tag{3.5}$$

where λ and σ_Y are the correlation length and standard deviation of the ln K field. γ is a flow factor, which for the isotropic case is $\gamma = 1 + \sigma_Y^2/6$ and $\gamma \simeq 1$ if it is assumed that the local dispersivity α_L is small compared to correlation length λ . In this study γ is considered to be 1 since α_L is taken in the order of centimetres, while λ is in the order of meters.

Similar to the homogeneous aquifer conditions, from a continuous leak such a plume is approximated by repeated small instantaneous plumes at short time intervals to find the mean concentration distribution as the mean groundwater velocity and injection rate are considered to be constant (Vomvoris and Gelhar, 1990). For both instantaneous and continuous leak cases, the mean concentration values will be used to determine the detection probability of a contaminant plume by a given monitoring system.

3.3.3 Probability of detection

Plumes start from a random location (x_0, y_0) where x_0 is fixed and y_0 is between $y_c - L$ and $y_c + L$ where $2L$ is the length of the landfill. Detection of such a plume by a well located at position (x_{mw}, y_{mw}) occurs if the concentration at the monitoring well $C(x_{mw}, y_{mw}, t)$ is greater than or equal to the threshold concentration C_{TH} at some moment in time. By calculating the maximum concentrations on the line $x=x_{mw}$ the maximum width of the plume $2l$, (above a given threshold) at x_{mw} can be found.

The (vertical) width of the plume at time t at a well distance x_{mw} can be found by solving $C(x_{mw},y,t)=C_{TH}$ for y which gives,

$$y^2 = 4\alpha_T v_x t \left[\ln\left(\frac{C_0}{C_{TH}} \frac{1}{4\pi\sqrt{\alpha_L \alpha_T} v_x t} \right) - \frac{\left(x_{mw} - v_x t\right)^2}{4\alpha_L v_x t} \right] \tag{3.6}$$

Define the abbreviation

$$A := \frac{C_0}{C_{TH} 4\pi \sqrt{\alpha_L \alpha_T}} \tag{3.7}$$

This gives

$$y = g\left(t\right) = \sqrt{4\alpha_T v_x t \left(\ln A - \ln v_x t\right) - \frac{\alpha_T}{\alpha_L}\left(x_{mw} - v_x t\right)^2} \tag{3.8}$$

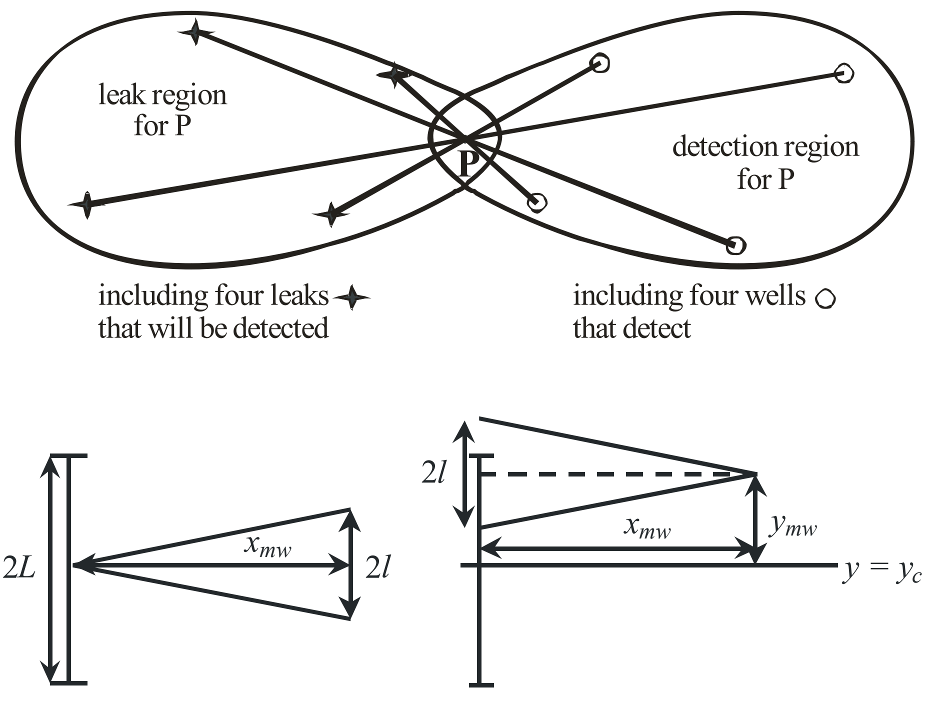


Figure 3.1: Depiction of detect and leak regions.

To find the maximum l ($l = g(t_{\max})$), g is differentiated with respect to t and $g'(t) = 0$ has to be solved. This is not analytically feasible. Note that, for fixed t, the contours $C(x,y,t)=constant$ are ellipses. One would expect the plume has its maximal width at distance x_{mw} when the centre of this ellipse is at x_{mw}, which happens at $t=x_{mw}/v_x$. Using numerical approximations it is found that the width of the plume for this t is very close to the optimal width.

Define the detection region $D(x_0, y_o, C_{TH})$ as the set of the points(x, y) where at some moment in time a plume starting from (x_0, y_0) will be detected at level C_{TH}. Likewise let the leak-region $L(x_{mw}, y_{mw}, C_{TH}.)$ be the set of points (x, y) such that a plume starting from (x, y) will be detected by a well at location (x_{mw}, y_{mw}). In a homogeneous medium the shape of a plume is the same whatever its starting point and the leak region and the detection region for one and the same point (x, y) are each other's image under reflection in the point (x, y) (see Figure 3.1). Suppose that the plume released from (x_0, y_0) has width $2l$ at distance x_{mw} from the source. Any leak on the line $x=x_0$ between $y_{mw} - l$ and $y_{mw} + l$ will be detected; any leaks with other y-values will not. The detection probability is thus simply the fraction of the line segment $x=x_0$, $y_c - L \le y \le y_c + L$ that is covered by $[y_{mw} - l, y_{mw} + l]$. As long as $l < L$ and $[y_{mw} - l, y_{mw} + l]$ falls completely within $[y_c - L, y_c + L]$, which happens if $y_c - L + l \le y_{mw} \le y_c + L - l$, the detection probability is therefore

$$P_{d(mw)} = \frac{2l}{2L} = \frac{l}{L} \tag{3.9}$$

When calculating the detection probability of a well close to the boundaries or when $L \le l \le 2L$ a boundary effect should be taken into account. If $l \le L$ and, say $y_{mw} + l \ge y_c + L,$ the leaks in $[y_c + L, y_{mw} + l]$, which is an interval of length $(y_{mw} + l - y_c - L)$ should not be counted and,

$$P_{d(mw)} = \frac{2l - ((y_{mw} + l) - (y_c + L))}{2L} = \frac{l + L - y_{mw} + y_c}{2L} \tag{3.10}$$

Likewise if $y_w - l \le y_c - L$ then the detection probability equals:

$$P_{d(mw)} = \frac{2l - ((y_c - L) - (y_{mw} - l))}{2L} = \frac{l + L - y_c + y_{mw}}{2L} \tag{3.11}$$

If $L \le l \le 2L$ and if $L - l \le y_w - y_c \le l - L$ the detection probability P_d equals 1. But if $y_w - y_c \le L - l$ then the detection probability equals:

$$P_{d(mw)} = \frac{(y_{mw} + l) - (y_c - L)}{2L} = \frac{L + l + y_{mw} - y_c}{2L} \tag{3.12}$$

Likewise if $l - L \le y_w - y_c$ then,

$$P_{d(mw)} = \frac{(y_c + L) - (y_{mw} - l)}{2L} = \frac{L + l - y_{mw} + y_c}{2L} \tag{3.13}$$

and last of all, if $l > 2L$ then any leak within $[y_c - L, y_c + L]$ will be detected.

3.4 Illustrative example

The model domain is of size L_x= 500 m and L_y=400 m Figure 3.2. The model is discretized with grid cells of 2 m by 2 m in both x- direction and y- direction. The hypothetical landfill is located at $30 \le x \le 50$ m and $180 \le y \le 220$ m in the model domain. The monitoring wells are located in the rectangle $60 \le x \le 450$ m and $180 \le y \le 220$ m. In order to achieve a detailed comparison between the analytical and the simulation model in terms of estimated concentrations and detection probability values the distance between the monitoring wells is set to 10 m (5 grid cells) in the x- direction and 2 m in the y- direction.

The boundary conditions for the groundwater flow are: zero flux at y=0 m (bottom boundary) and y=400 m (top boundary) and constant head along the left and the right boundaries. The head values at x=0 m and x=500 m were chosen to result in a macroscopically constant hydraulic gradient of 0.001. Porosity equals 0.25. The average hydraulic conductivity K is set to 10 m/day and for homogeneous aquifer conditions the location of the leak is the only random input to the model.

For the heterogeneous aquifer, uncertainties due to both the contaminant source location and the subsurface heterogeneity are incorporated in the simulation model. Subsurface heterogeneity is reflected by the spatial variability of the hydraulic conductivity. Hence hydraulic conductivity is treated as a random space function or random field. The logarithm of the isotropic hydraulic conductivity $Y=ln\ (K)$ is modelled as a stationary Gaussian field with a given mean, variance and correlation length (see e.g. Gelhar, 1986).

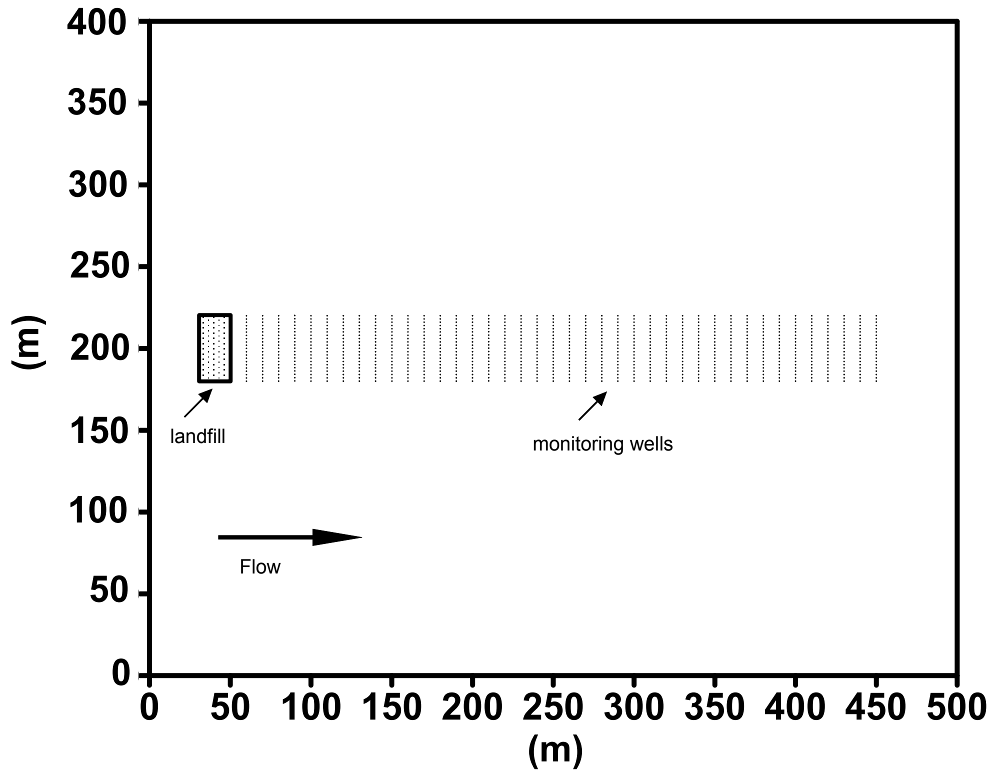


Figure 3.2: Dimensions and components of the example with 840 monitoring well locations.

Random conductivity fields that respect these statistics are generated using the turning bands method (Mantoglou and Wilson, 1982). The value of μ_Y is set to 2.3, whereas the variance of Y, σ_Y^2, is assigned four different values, namely 0.2, 0.4, 1.0 and 1.5, respectively. The value of μ_Y=2.3 corresponds to a geometric mean of the hydraulic conductivity of 10 m/day The isotropic covariance of Y is chosen to be of exponential form with a correlation length=15 m.

For the transport model, a condition of a zero dispersive flux is imposed on the top and bottom boundary, and the initial background concentration in the model domain is set to zero. Since the flow direction is parallel to the x- axis, the only source dimension that is treated as a random variable is its y- coordinate. Potential leak locations occur along the downgradient edge of the landfill. The contaminant leak is assumed to be a point source, as it would result in a plume, which is most difficult to detect, and the source location is drawn from a uniform probability distribution between y-coordinates of $180 \leq y \leq 220$ m for each Monte Carlo run. Calculations are carried out for two types of leak, namely instantaneous and continuous leaks. The initial concentration for the instantaneous leak is assumed to be 1 mg/l whereas for the continuous the leak case injection rate is set to 1 mg/l/day. The threshold concentration (detection limit) at which detection occurs is set at 0.5 % of the initial source concentration. Contaminants are assumed to be conservative and to be completely mixed over the depth of the aquifer. Dispersion is incorporated in the model by introducing microscale longitudinal (α_L) and transversal (α_T) dispersivity. The ratio between α_L and α_T is assumed to be 10, (Bear, 1972). α_L is set to 0.1 m and 0.5 m. Since a two-dimensional model is used in this study it is assumed that the monitoring wells are fully penetrating the aquifer, and that they are located in the centres of the grid cells, having a dimension of one grid cell. It is supposed that sampling is continuous.

3.5 Results and discussion

3.5.1 Assessment of simulations by analytical methods for the homogeneous case

In order to investigate the accuracy of the simulation model and the influence of the parameters on estimated values, the solution of the simulation model is compared to the results obtained by the analytical model.

Instantaneous leak

For plume simulations 500, 1000, 2000, 4000 and 8000 particles are used in order to investigate the influence of the number of particles on the computation of concentration values and to determine the appropriate number of particles to be used throughout the computations.

The simulations are performed for the cases where, α_L=0.1 m, α_T=0.01 m and α_L=0.5 m, α_T=0.05 m respectively. In both cases the plumes originate from an instantaneous leak at the fixed location x=50 m and y=200 m. Figure 3.3 shows the different longitudinal sections of simulated plumes and comparison with the analytical solution for α_L=0.1 m, α_T=0.01 m. As is seen in the figure the differences between the plume simulations with 500, 1000, 2000, 4000 and 8000 particles are minor. Nevertheless, the plume edge (which occurs around y=204 m) is defined the worst by 500 particles and the best by 8000 particles. The same trend is also observed for α_L=0.5 m, α_T=0.05 m. Since simulations of 8000 particles are computationally very expensive, 2000 particles are used in the rest of the analysis, as a compromise value.

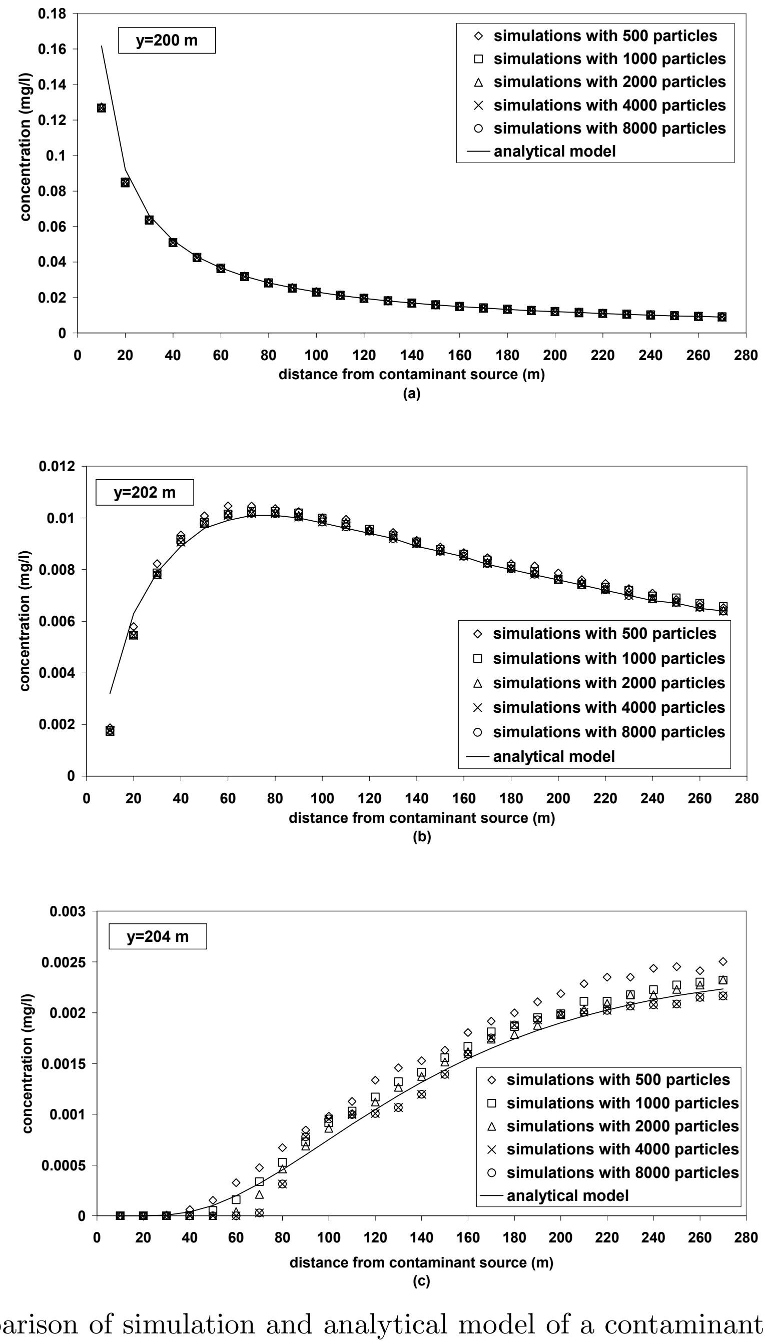
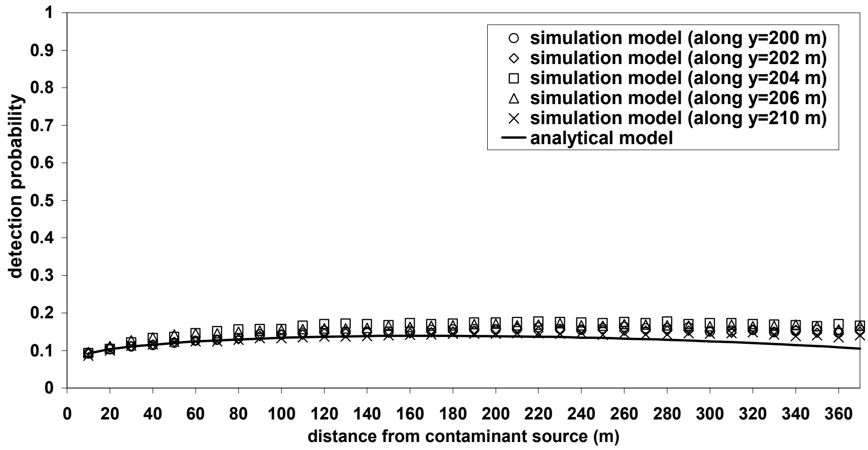


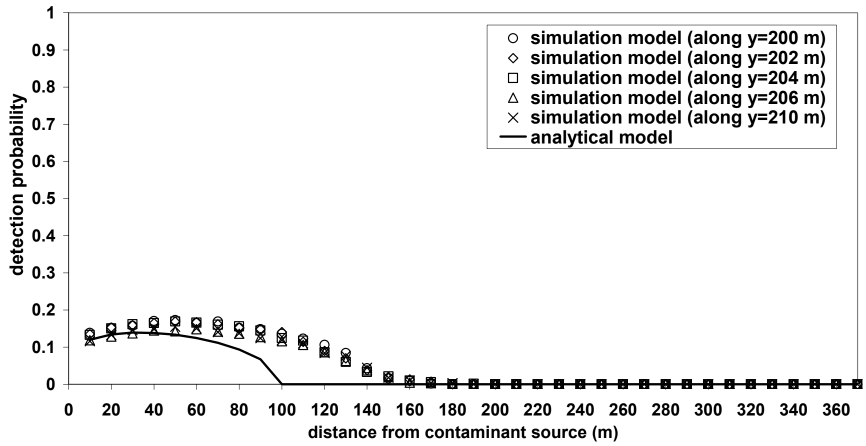
Figure 3.3: Comparison of simulation and analytical model of a contaminant plume originated from an instantaneous leak (y=200 m) in the homogeneous case for α_L=0.1 m, α_T=0.01 m for longitudinal sections along (a) y=200 m, (b) y =202 m and (c) y=204 m.

The concentration values obtained by simulations are quite accurate over most of the plume length. However, near the source there is a slight discrepancy between the simulation and analytical models especially when the dispersivity value is low. The plumes are narrow close to the source and widen as they move away. Therefore close to the source the concentration values determined by the analytical model are more peaked. The averaging of the analytical solution over a grid cell using Simpson's Rule will then overestimate the average concentration. This also leads to higher discrepancy between the two models in the low dispersive medium (α_L=0.1 m, α_T=0.01 m, shown in Figure 3.3) compared to the highly dispersive medium (α_L=0.5 m, α_T=0.05 m, not shown).

Figure 3.4 shows the comparison of the detection probabilities computed by the simulation and the analytical model at the selected well locations for both dispersivity cases. The possible leak locations are now randomly located at x=50 m and over $180 \le y \le 220$ m. The values estimated by the simulation model are compatible with those estimated by the analytical model. The slight discrepancy seen in the graphs is due to the fact that the plume edges are not as sharply defined as in the analytical model.



(a)



(b)

Figure 3.4: Comparison of detection probability values at selected well locations computed by simulation and analytical models for an instantaneous leak in the homogeneous case (a) α_L=0.1 m, α_T=0.01 m and (b) α_L=0.5 m, α_T=0.05 m.

Continuous leak

Plumes originated from a continuous leak located at x=50 m and y=200 m with an injection rate of 1 mg/l/day. As in the instantaneous leak case the simulation model estimated the concentration values correctly over the most of the plume length (see Figure 3.5). As described above, the slight discrepancy between the simulation and analytical model estimations close to the source, particularly in the low dispersive case, is due to the slender nature of the plume when it is close to the source. The results are representative for the case where α_L=0.5 m, α_T=0.05 m as well. Figures 3.6 and 3.7 present the detection probabilities at selected monitoring wells for continuous leak condition in the homogeneous case for α_L=0.1 m, α_T=0.01 m and α_L=0.5 m, α_T=0.05 m, respectively. The possible leak locations are at x=50 m and $180 \leq y \leq 220$ m. As seen from the figures the discrepancy between the analytical and simulation model estimations is much less than in the instantaneous case. This is mainly due to the fact that the convolution procedure described at the end of Section 2.4 yields better approximations of the plume with less particles than in the instantaneous leak case.

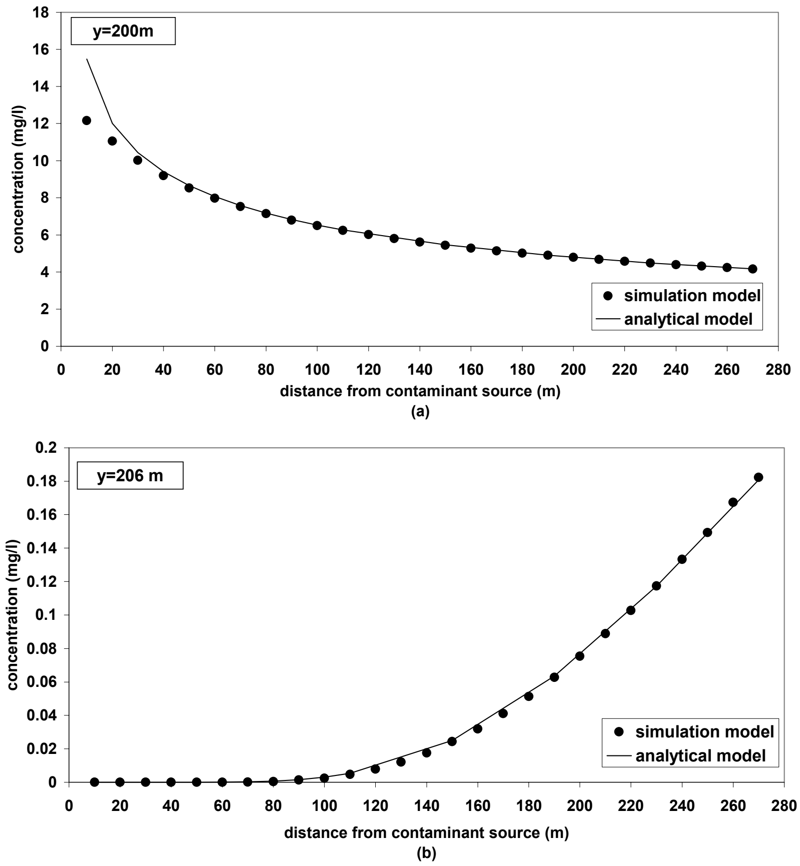


Figure 3.5: Comparison of simulation and analytical model of a contaminant plume originated from a continuous leak in the homogeneous case with α_L=0.1 m, α_T=0.01 m for longitudinal sections along (a) y=200 m and (b) y =206 m.

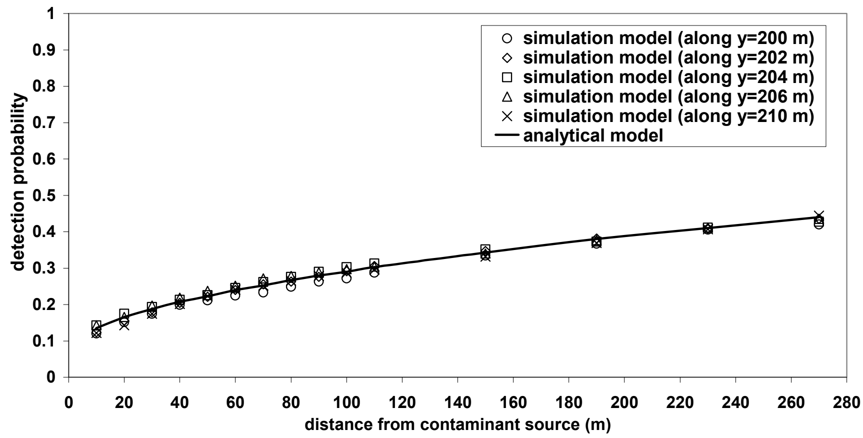


Figure 3.6: Comparison of detection probability values at selected well locations computed by simulation and analytical models for a continuous leak in the homogeneous case (α_L=0.1 m, α_T=0.01 m).

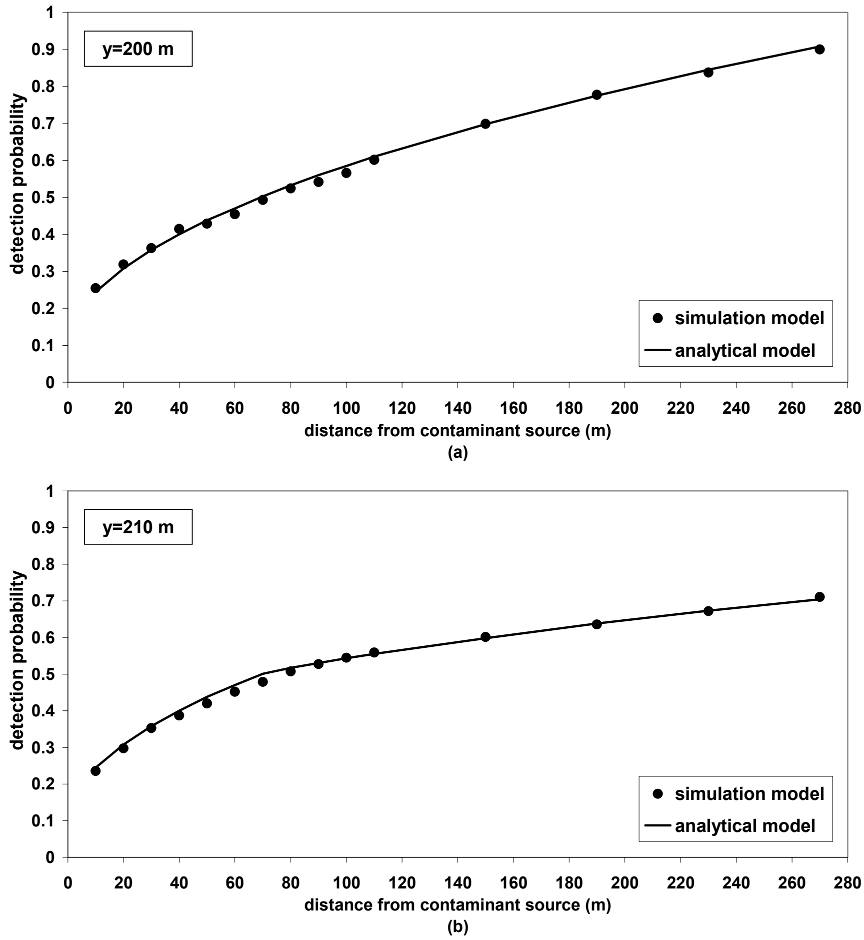


Figure 3.7: Comparison of detection probability values at selected well locations computed by simulation and analytical models for a continuous leak in the homogeneous case (α_L=0.5 m, α_T=0.05 m) (a) along y=200 m and (b) along y=210 m.

3.5.2 Assessment of simulations by analytical methods for the heterogeneous case

The results of the analytical model described in Section 3.3 and the simulation model described in Section 3.2 are expressed in terms of concentration profiles along the specified longitudinal sections and plots of the detection probability as a function of the distance from the contaminant source to determine: (1) how good is the mean concentration as a predictor of the concentration at a given monitoring well location, and (2) how accurate is it to use the mean concentration value in computing the detection probability of a contaminant plume by a given well in a sample realization of the hydraulic conductivity field. The computations are carried out for eight different scenarios. Table 3.1 summarizes the parameters for all cases.

Table 3.1: Parameters used in simulation and analytical models for computations for heterogeneous aquifer conditions

Simulation Model					**Analytical Model**			
Case	**longitudinal dispersivity, α_L (m)**	**transverse dispersivity, α_T (m)**	**mean of Y, μ_Y**	**variance of Y, σ_Y^2**	**correlation length, λ(m)**	**mean velocity, v (m/day)**	**longitudinal macrodispersivity, A_L (m)**	**transverse macrodispersivity, A_T (m)**
Case1a	0.1	0.01	2.3	0.2	15	0.04	3.1	0.01325
Case1b	0.1	0.01	2.3	0.4	15	0.04	6.1	0.0165
Case1c	0.1	0.01	2.3	1.0	15	0.04	15.1	0.02625
Case1d	0.1	0.01	2.3	1.5	15	0.04	22.6	0.034375
Case2a	0.5	0.05	2.3	0.2	15	0.04	3.5	0.06625
Case2b	0.5	0.05	2.3	0.4	15	0.04	6.5	0.0825
Case2c	0.5	0.05	2.3	1.0	15	0.04	15.5	0.13125
Case2d	0.5	0.05	2.3	1.5	15	0.04	23	0.171875

Instantaneous leak

The actual concentration field is observed in a single heterogeneous aquifer and should be viewed as a realization of the stochastic process, whereas the ensemble mean represents the average behaviour of solute plumes in a large number of statistically identical aquifers. The observed concentration distribution does not form a smooth curve, as the mean concentration would, but is quite irregular. Hence the ensemble mean value is not sufficient for the description or prediction of the actual concentration distribution and a successful prediction should be made in a probabilistic context (in terms of predictions accompanied by a quantification of the deviation around the mean values) rather than in the traditional deterministic framework (in terms of mean concentration only). Figure 3.8 presents the concentration values at given monitoring well locations for three different single realizations, the ensemble mean concentration over 700 simulations and their 95% (empirical) confidence interval along with mean concentration values computed by the analytical model for Case 1a and Case 2d. Case 1a represents the lowest while Case 2d represents the highest dispersive and heterogeneous medium among the scenarios considered in this study.

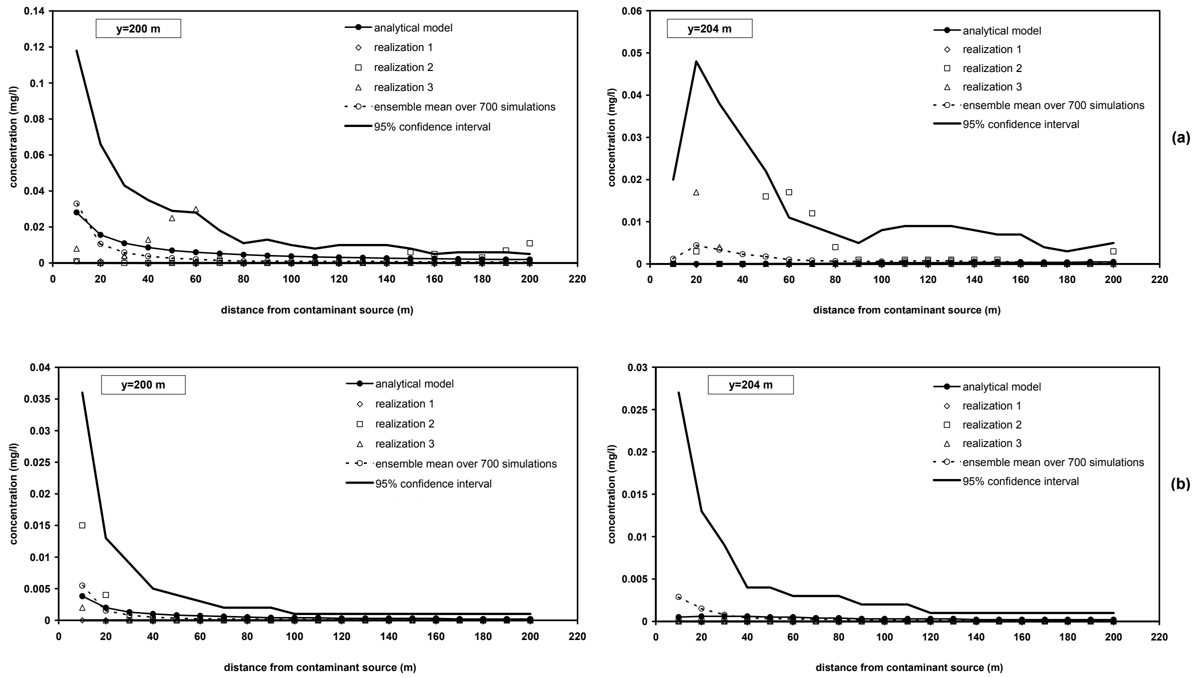


Figure 3.8: Comparison of simulation and analytical model of a contaminant plume originated from an instantaneous leak (y=200 m) in the heterogeneous case for longitudinal sections along y=200 m (left column) and y =204 m (right column) (a) Case 1a (b) Case 2d.

The analysis results corresponding to these cases characterize the others as well. As before the instantaneous leak is located at x=50 m and y=200 m in order to compare concentration profiles while random leaks at x=50 m and along $180 \leq y \leq 220$ m are taken to compare the detection probabilities.

The average concentration values computed by the two models are close to each other and present a smooth curve compared to single realizations. Concentration values of the single realizations are relatively scattered as expected, since each realization shows a different plume velocity and a different spreading. The 95% confidence interval is wider close to the source: in all cases uncertainty in concentration prediction decreases with distance from the source. The ensemble standard deviation in the concentration is higher near the source and reduces significantly as plume moves further away. Near the source the plume is narrow and has a large degree of freedom to spread in different forms from one realization to another. However, further away from the source the plume widens and since it covers a larger area the degree of freedom to spread is not that high and uncertainty is less. Near the source the concentration gradient is high and consequently the uncertainty is high (see e.g., Gelhar, 1993). The 95% confidence interval is narrower towards the edge of the plume (y=204 m) for the same reason. The discrepancy between the two models is overall more pronounced in the low dispersive medium.

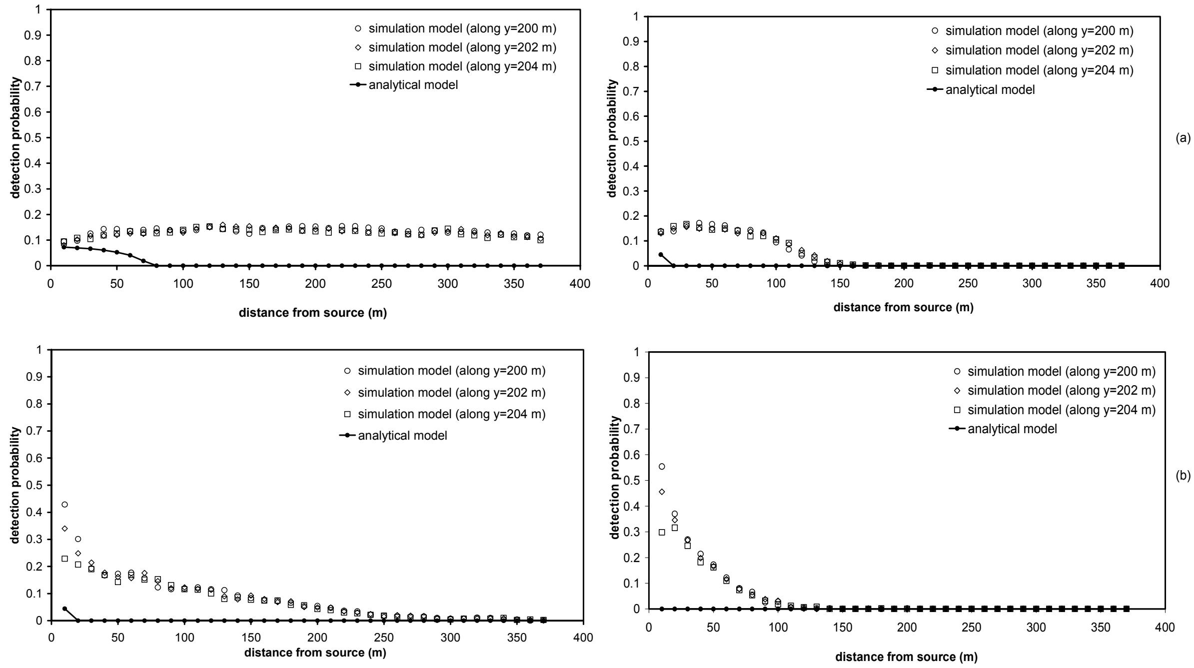


Figure 3.9: Comparison of detection probability values at selected well locations computed by simulation and analytical models for an instantaneous leak in the heterogeneous case (a) Case 1a and Case 2a, (b) Case 1d and Case 2d.

Figure 3.9 shows the comparison of the detection probabilities for four of the eight cases- the cases not shown are similar to Case 1a respectively Case 2a. A discrepancy occurs between the analytical and simulation model, particularly close to the contaminant source. The discrepancy between the detection probability values at given well locations tends to reduce as the distance from the source increases.

The analytical model using effective (macro) dispersivities computes the mean concentration distribution, which corresponds to smoother and relatively wider plumes, consequently a much more diluted plume in the case of an instantaneous leak. This results in lower detection probability values than those obtained by the simulation model. In the simulation model each realization views the possible actual plume observed in a single heterogeneous aquifer, and the detection probability at a given well location is computed accordingly. The influence of homogenization in terms of underestimating the plume size is more pronounced when the values of the dispersivity and/or σ_Y^2 increases. As an increase in both values adds to the macro dispersivities used in the analytical model, the average plume, which embodies the behaviour of the plume in a heterogeneous medium, becomes larger and consequently yields lower concentration values at the wells (see Equations (3.4) and (3.5)).

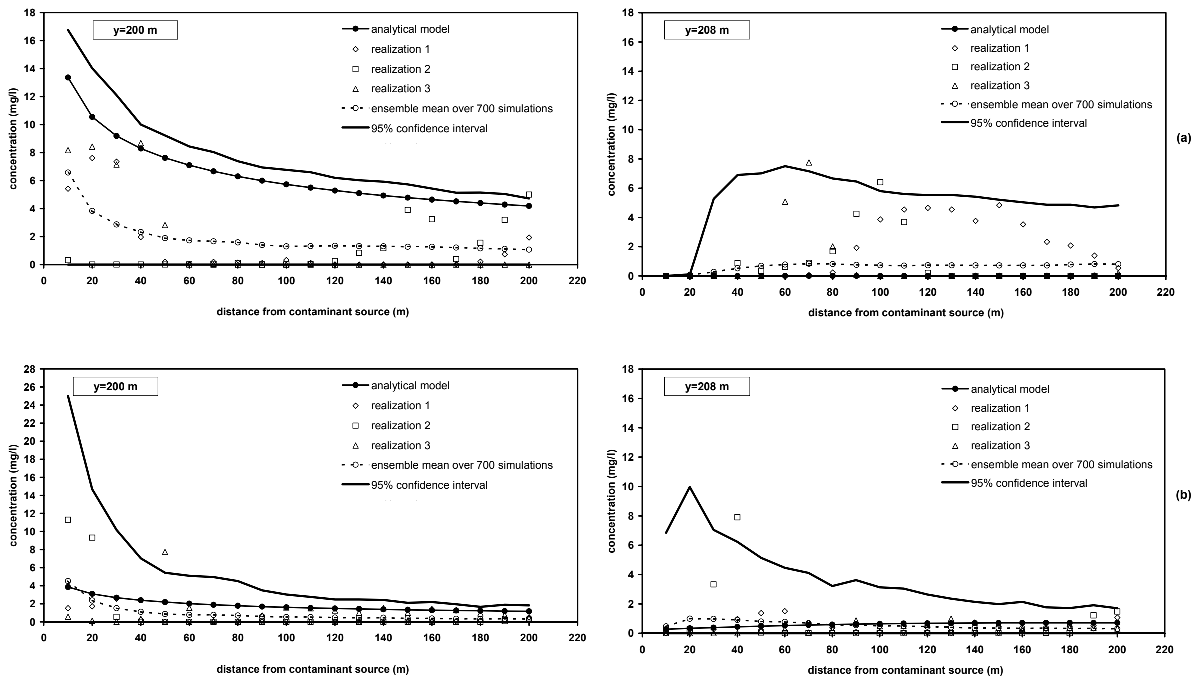


Figure 3.10: Simulation and analytical model comparison of a contaminant plume originated from a continuous leak (y=200 m) in the heterogeneous case for longitudinal sections along y=200 m (left column) and y =204 m (right column) (a) Case 1a, and (b) Case 2d.

Continuous leak

Computations are performed for all cases mentioned in Table 3.1 for the continuous leak case as well, since this type of leaks is mostly considered in monitoring system design at landfill sites unless there are specific data for the type of the leak. Figure 3.10 presents the comparison of concentration profiles computed by the two models in the case of a continuous leak with an injection rate of 1 mg/l/day for Case 1a and Case 2d. The other cases are not shown here as these two cases characterize their behaviour well enough.

The discrepancy between the average concentration values computed by the two models decreases as the dispersivity of the medium increases since the plume gets wider and the concentration gradient is smaller for larger dispersivity values. As described above for the instantaneous leak case the 95% confidence interval is wider close to the source and narrower towards the edge of the plume (y=208 m) in the continuous leak case as well, since the concentration gradient decreases as the distance from the source increases. However, in this case the influence of heterogeneity is more visible compared to the instantaneous leak case: the confidence interval close to the source appears to be wider when σ_Y^2 increases. This is due to the fact that in the instantaneous leak case the Gaussian plumes spread faster when the heterogeneity and dispersivity of the medium increases and accordingly the concentration values and hence concentration gradient is smaller.

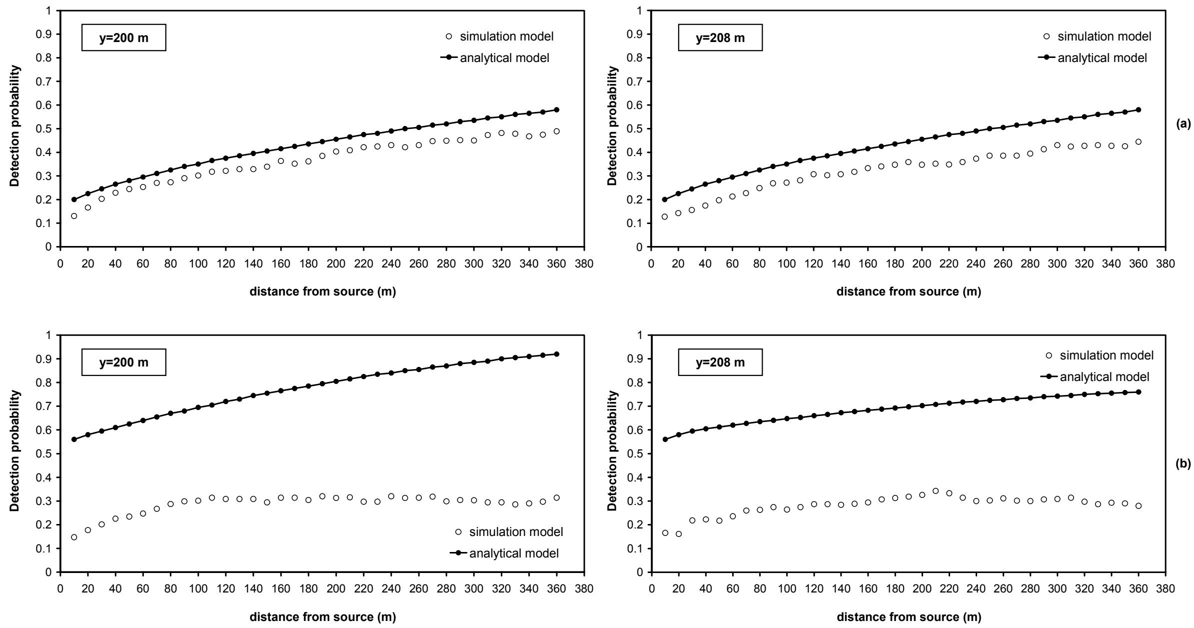


Figure 3.11: Comparison of detection probability values at selected well locations computed by simulation and analytical models for continuous leak in a heterogeneous medium along y=200 m (left column) and y =208 m (right column) (a) Case 1a, and (b) Case 1d.

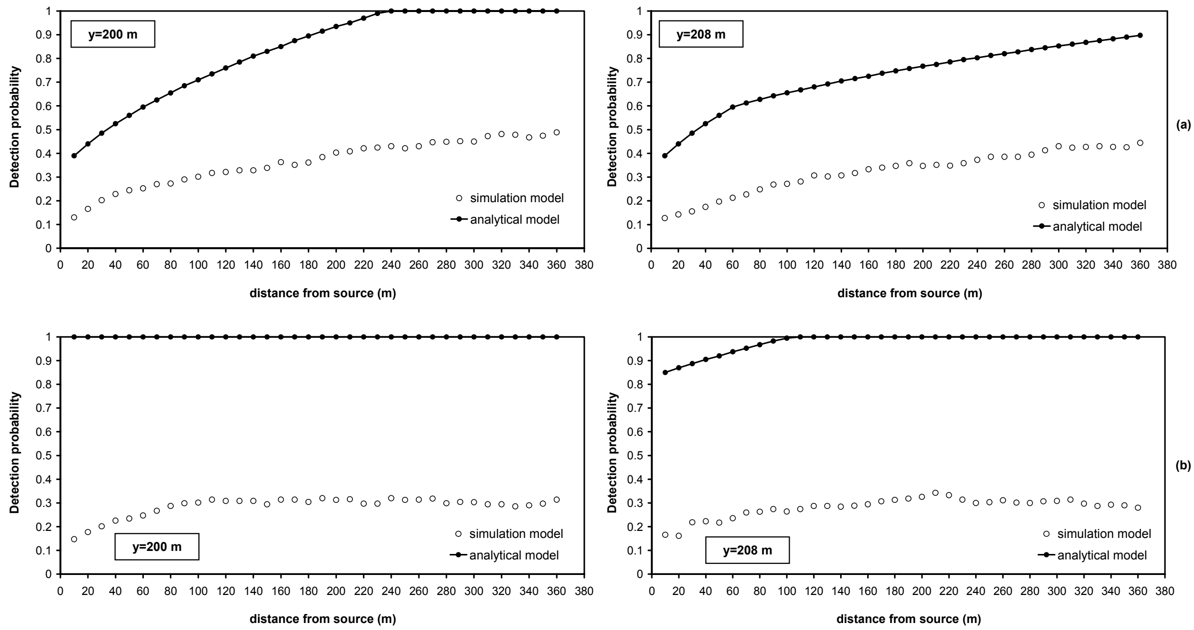


Figure 3.12: Comparison of detection probability values at selected well locations computed by simulation and analytical models for continuous leak in a heterogeneous medium along y=200 m (left column) and y =208 m (right column) (a) Case 2a, and (b) Case 2d.

However in the case of a continuous leak the continuous injection of contaminants results in higher concentration values and therefore a larger concentration gradient, which actually reflects the apparent influence of heterogeneity: the uncertainty in concentration prediction increases as the degree of heterogeneity increases. This also explains why the discrepancy between average concentration values computed by the two models is higher than in the instantaneous case.

The plume described by the analytical model using effective (macro) dispersivities is an average plume or actually an envelope of possible plumes in many single heterogeneous media, therefore it is larger and smoother and overlooks the behaviour of irregular contaminant spreading on a macro scale, particularly when the concentration gradient is high. Furthermore, the large average plume with high concentration gradient leads the analytical model to overestimate the concentration values at given well locations. Eventually the results show that in any case the dispersivity of the medium (both pore scale and macro scale) is the most important parameter, which dominates the spreading of the plume, and hence the uncertainty in predictions of concentration values.

The detection probability of monitoring wells at a given location as a function of the distance from the source is presented in Figure 3.11 and Figure 3.12 for the continuous leak case in a heterogeneous medium. The potential random leaks are assumed to occur along the downgradient edge (x=50 m and $180 \leq y \leq 220$ m) of the landfill as depicted in Figure 3.2. There is a big discrepancy between the detection probability values computed by the two models. The reason for that is as explained above: the overestimation of concentration values computed by the analytical model and hence the overestimation of detection probabilities. Therefore as seen in Figures 3.11 and 3.12 the detection probability of monitoring wells at given locations increases as the heterogeneity increases in contrast to the results of the simulation model. The results of the analysis by the simulation model show that the more heterogeneous the medium is, the less the chance is to detect a contaminant plume at a given monitoring well location. The reason for this is that the plumes become more irregular in shape as the uncertainty in flow paths increases. This result of the simulation model is consistent with other previous studies as well (e.g., Massmann and Freeze , 1987; Meyer et al., 1994; Storck et al., 1997).

3.6 Summary and conclusions

Simulation and analytical models are used to compute concentration distributions and detection probability values at given monitoring well locations. The results of the analysis show that the simulation model estimates the concentration values correctly over most of the plume length for homogeneous aquifer conditions. A slight discrepancy between the two models near the source is due to the fact that the plumes are narrow close to the source and widen as they move away. Therefore close to the source the concentration values determined by the analytical model are more peaked than those determined by the simulation model. An important point is that the accuracy of the estimates by the simulation model is highly dependent on the number of the particles used in the model. In the homogeneous case, particularly for the continuous leak, the comparison of the results in terms of detection probability match quite well.

As an analytical model for the concentration distributions of a contaminant plume for heterogeneous aquifer conditions, effective (macro) dispersion coefficients are used to solve the advective-dispersive transport equation. A discrepancy between the mean concentration values computed by the two models is observed, particularly in the continuous leak case. The mean concentration plume that results from such an approximation is smooth due to loss of the detailed advective heterogeneity. This reflects in overlooking in the determination of the concentration field and consequently in the computation of the detection probability of a contaminant plume by a given monitoring well. The 95% confidence intervals drawn from the simulations show that the uncertainty in concentration predictions decreases with the distance from the source. The ensemble standard deviation of the concentration is higher near the source and reduces as the plume moves further away. Near the source, the plume is narrow and has a large degree of freedom to spread in different forms from one realization to another. However, further away from the source the plume widens and since it covers a larger area the degree of freedom to spread from one realization to another is not that high and uncertainty is less. Near the source, the concentration gradient is high and consequently the uncertainty is high (see e.g., Gelhar, 1993). Furthermore, the uncertainty in concentration predictions increases as heterogeneity and/or dispersivity of the medium increases.

The results show that modelling of contaminant transport using an advection-dispersion equation with effective (macro) dispersivities can be used to describe the average concentration distribution, but this approach is insufficient in monitoring system design when incorporating the subsurface heterogeneity. The discrepancy between the detection probabilities of contaminant plumes at given monitoring well locations computed by the two models is significant, particularly when the dispersivity and heterogeneity of the medium increase. Therefore, despite the computational expenses, the simulation model is more appropriate for monitoring system design under conditions of heterogeneity.

Chapter 4

Reliability Evaluation of Groundwater Monitoring systems

Adapted from Yenigul, N.B., Elfeki, A.M.M., Gehrels, J.C., van den Akker, C., Hensbergen, A.T., and Dekking, F.M., Reliability Assessment of Groundwater Monitoring Networks At Landfill Sites, Journal of Hydrology, 308, 1-17, 2005.

Groundwater monitoring system design using probability tools is more advantageous than the traditional deterministic methods. Through this approach, the sources of uncertainty can be individually comprised, systematically evaluated, and explicitly incorporated in the design process. Thus a reliability concept can be used not only to measure the effectiveness of a given monitoring system but also to compare the efficiency between alternative monitoring systems in detecting contaminant plumes.

This aim of this chapter is to analyze the reliability of groundwater monitoring systems at landfill sites by examining thoroughly the influence of several parameters that play a role in monitoring system design. The next section reviews some previous studies available in relation to reliability assessment to estimate the performance of groundwater monitoring systems at landfill sites, followed by a detailed description of the reliability model and finally its application on a hypothetical example is presented.

4.1 A brief review of previous work

Although several authors have illustrated different aspects of groundwater monitoring design, a few examples of approaches using probability tools are present in the recent literature. Rouhani and Hall (1988) investigated the significance of a sampling program in network design by using a method based on variance reduction analysis, media ranking and risk ranking. Their study showed that risk ranking was the most appropriate criterion to choose the best sequence of sampling points as it combines variance reduction and median ranking. In another study, Haug et al. (1989) presented a geostatistical method to assess the positions and spacing of monitoring wells along the edge of a waste management facility. The reliability of the system was estimated as the probability having at least one monitoring well located in a sand lens of a lithostratigraphically complex geologic profile. Geostatistical tools were used efficiently, but neither groundwater flow nor contaminant transport models were considered in these studies. Thus, their use in prediction of contaminant pathways is limited and consequently they fall short in providing a systematic and consistent approach to design groundwater monitoring systems.

Jang et al. (1994) presented an approach to probabilistic modelling of contaminant transport based on First Order Reliability Method (*FORM*) and Second Order Reliability Method (*SORM*). The models were initially developed for structural reliability analysis to estimate the occurrence of low probability events. Example reliability analysis of one and two-dimensional transport models were used to illustrate the approach and to study the likelihood of exceeding a given contaminant threshold at a specified location. The accuracy of the method was evaluated in comparison with the MC simulation. The results showed that *FORM* increasingly overestimates the probability of exceedance as the spatial variability of the domain increases. *SORM*, on the other hand, gives better results than *FORM* as it accounts for the nonlinearities of the limit state surface. System reliability was also studied using two performance functions where one function was related to the exceedance of a high threshold concentration, and the other was related to the exceedance of a lower threshold concentration for a specified duration. Although the approach is very useful, it is still not well suited to study reliability of groundwater monitoring systems where a performance function is needed for each monitoring well location in addition to random variable and this makes the model computationally too expensive.

Hudak (2001) devised a graphical approach to configure detection wells at the downgradient of a landfill. The reliability method is used to evaluate the effect of a cutoff wall on locations of wells in groundwater monitoring systems. The author considered two five-well monitoring systems for a rectangular landfill oriented oblique to regional groundwater flow. The results showed that downgradient boundary of a landfill might be shortened since upgradient cut off walls induce convergent groundwater flow behind them, and consequently the area over which the detection wells should be located will reduce. Therefore clustering the wells within the critical area may enhance the detection efficiency, or reduced the number of wells required. In his later work, Hudak (2002) presented a deterministic graphical approach to evaluate detection capabilities of perpendicular and equidistant groundwater monitoring networks in aquifers dominated by intergranular porosity. In both studies the author did not consider the uncertainties due to the subsurface heterogeneity and contaminant leak location.

4.2 Reliability model

The simulation model described in Chapter 2 was used to perform an extensive number of numerical experiments in order to investigate the influence of uncertainties due to subsurface heterogeneity and leak location, the dispersivity of the medium, and well spacing well location and the size of the initial contaminant source, on the detection probability of a contaminant plume released from a landfill. Each realization of the *MC* simulation, used in the reliability assessment of groundwater monitoring systems, consists of the following steps:

- Generation of a realization of a random hydraulic conductivity field.
- Solution of the steady state groundwater flow model to determine the velocity field.
- Generation of a random leak location.

- Solution of the random walk transport model to determine the concentration field of the contaminant plume.
- Check whether the concentration value at a given monitoring well location exceeds a given threshold concentration (detection limit), to determine whether a plume is detected or not detected by a given monitoring system.

In this study the probability of failure, P_f of a groundwater monitoring system is defined as the probability of failure of the system to detect a contaminant plume. Hence, the system probability of detection, P_d of a contaminant plume equals $\left(1-P_f\right)$. Since the groundwater monitoring system is composed of a number of individual wells, the system probability of detection depends on the detection probabilities of the individual wells. As described in the previous chapter, detection of a contaminant plume by a monitoring well (mw), is defined as the event where the contaminant concentration at the well location, C_{mw} at some time t is equal to or greater than a given threshold concentration, C_{TH}. For a monitoring system as a whole, failure of the system means failure of all the wells to detect the contaminant plume. Therefore for a monitoring system composed of n wells, failure of the system can be expressed as the intersection of the n individual failure events. The detection probability of the system, P_d is estimated as the ratio of the total number of simulation runs, in which the generated contaminant plumes are detected, over the total number of simulation runs N_{MC},

$$P_d = \frac{1}{N_{MC}} \sum_{i=1}^{N_{MC}} I_d^{(i)} \tag{4.1}$$

Here, $I_d^{(i)}$ is the indicator function of the detection by the monitoring system for realization i, i.e., $I_d^{(i)}$ equals 1 if the simulated contaminant plume i is detected by the given monitoring system, and equals zero otherwise.

4.3 Hypothetical model

4.3.1 Model domain and discretization

A plan view of the hypothetical problem used in the numerical examples is shown in. The overall dimensions of the model domain are 500 m in the x-direction and 300 m in the y- direction. The nodal spacing is equal to 2 m in both directions. A rectangular landfill (L=20 m and W=50 m) is located at the left of the modelled area. Several monitoring systems composed of various numbers of wells placed in a single row at different distances from the down gradient edge of the landfill (see Figure 4.1) are considered in the numerical experiments. The spacing Δs between the wells and the

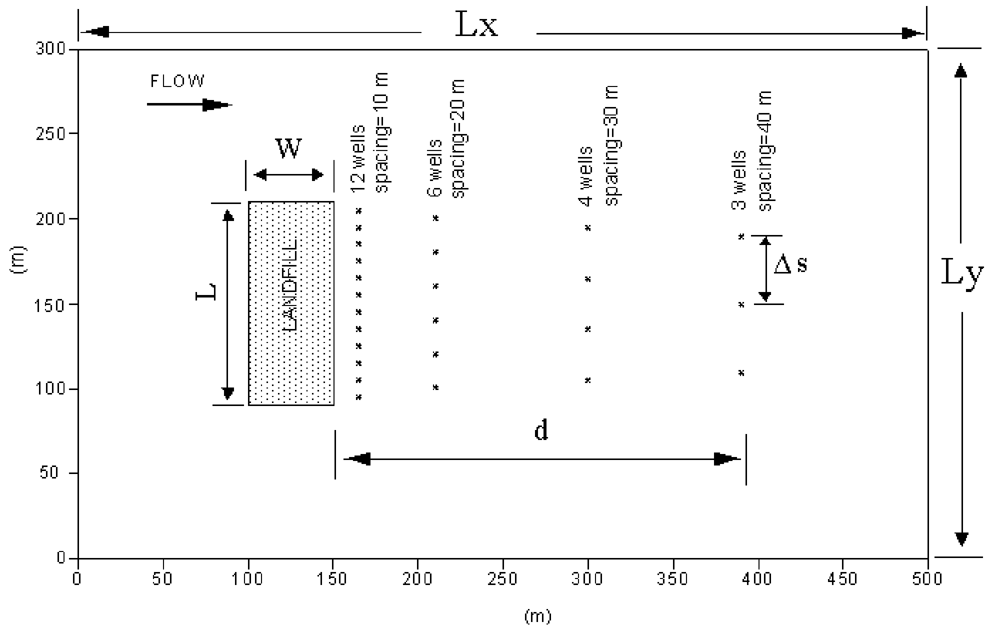


Figure 4.1: Plan view of the hypothetical problem used in numerical experiments.

distance d from the edge of the landfill are normalized with respect to the length of the landfill L perpendicular to the flow, for generalization purposes. The quantities *nws* $(\Delta s/L)$ and *ndfs* (d/L) correspond to the normalized well spacing and the normalized distance from the source, respectively.

4.3.2 Parameter values used in flow model

The boundary conditions for the steady state groundwater flow model are zero flux at y=0 m (bottom boundary) and y=300 m (top boundary) and constant head along the left and the right boundaries. The head values at x=0 m and x=500 m were chosen to result in a macroscopically constant hydraulic gradient of 0.001. Porosity is assumed to be 0.25. The natural logarithm of the isotropic hydraulic conductivity $[Y=ln\ (K)]$ is modelled as a stationary Gaussian random field with a given mean, variance and an isotropic correlation structure. The arithmetic mean value of K is considered to be 10 m/day whereas the variance of Y is assigned several values between σ_Y^2=0 and σ_Y^2=2. The isotropic covariance of Y is chosen to be of exponential form with a correlation length, λ of 20 m. For the numerical experiments 500 random hydraulic conductivity fields are generated.

4.3.3 Parameter values used in random walk particle tracking model

For the transport models a condition of a zero dispersive flux is imposed on the top and bottom boundary, and the initial background concentration in the model domain is set to zero. A local failure that results in an instantaneous leak is assumed to occur at a random location within the area covered by the landfill. The random leak locations are drawn from a uniform probability distribution for each Monte Carlo run. A small instantaneous leak is considered, as it is difficult to detect such leaks. On the other hand, this assumption can reflect the intention of modern design and operation techniques aspiring to minimize both the possibility and the quantity of the contaminant leak in case of a crack or a rupture in the landfill liners.

Furthermore, the numerical experiments are also performed for a one grid cell size source and for a four grid cell size source in order to study the influence of initial source size on the reliability of the monitoring systems. Dispersion is incorporated in the model by introducing micro scale longitudinal α_L and transverse α_T dispersivities. The ratio between α_L and α_T is assumed to be 10, (according to Bear, 1972). α_L is set to different values between 0.01 m and 2 m. The total mass of the 2000 particles used in the simulations is set to 1000 g. Three different contaminant concentration threshold, C_{TH} (detection limit) values of 0.25%, 0.35% and 0.5% of the initial concentration are used to determine whether a plume is detected. Monitoring wells are located in the centre of the grid cell and have a dimension of one grid cell. Furthermore, continuous sampling for the entire monitoring period of 30 years is considered in the numerical experiments.

4.4 Results and discussion

Table 4.1 summarizes the model parameters used in the numerical experiments described below. Parameters for which a single value is given remain constant throughout the numerical experiments. The influence of each parameter on the detection probability is examined by varying the parameter of interest while fixing the others.

4.4.1 Sensitivity to number of Monte Carlo simulations

The Monte Carlo (*MC*) Method is the most commonly used method in simulating stochastic phenomena. One of the major shortcomings of the Monte Carlo approach is that the accuracy of the results highly depends on the number of Monte Carlo realizations, N_{MC}. Furthermore it can be very demanding in terms of computational expenses due to the large number of realizations required to obtain reliable results. A minimum value of N_{MC} for which the estimation of P_d is practically independent of N_{MC} should be identified, while at the same time minimizing the computational expenses. Therefore evaluation of detection probability, P_d as a function of N_{MC} is performed for different monitoring systems located at several distances from the source. Two different dispersivity and σ_Y^2 values are used.

Figure 4.2 shows that as N_{MC} increases the fluctuations of the estimated P_d decrease, showing an asymptotic behaviour between 400 and 2000 *MC* runs. In all of the calculations performed in the numerical experiments 500 *MC* runs are used as it gives an acceptable convergence, while enabling the simulations to be computationally feasible as well.

Table 4.1: Model parameter values used in the numerical experiments.

Model Parameters	**Value(s) assumed in the model**
Length of model domain in x-direction, L_x	500 m
Length of model domain in y-direction, L_y	300 m
Dimension of one grid cell (Δx=Δy)	2 m
Length of landfill (L)	120 m
Width of landfill (W)	50 m
Mean of Y=lnK, μ_Y	2.3
Variance of Y,σ_Y^2	0.0, 0.5,0.75, 1.0, 1.5, 2.0
Correlation length in x- and y- directions (λ_x=λ_y)	20 m
Hydraulic gradient	0.001
Number of particles	2000
Longitudinal dispersivity, α_L	0.01, 0.20, 0.50, 1.0, 2.0 m
Transverse dispersivity, α_T	0.001, 0.02, 0.05, 0.10, 0.20 m
Porosity, ε	0.25
Total simulation time	30 years
Time step, Δt	1 day
Initial contaminant source size	Point source, 1 grid-cell size (2x2 m), 4 grid-cell size (4x4 m)
Normalized well spacing, *nws*	0.08, 0.17, 0.25, 0.33
Normalized distance from the edge of landfill, *ndfs*	0.125, 0.25, 0.5, 1, 1.25, 2, 2.5
Number of Monte Carlo runs, N_{MC}	500
Contaminant threshold concentration (C_{TH}) (as a percentage of the initial contaminant concentration)	0.25%, 0.35%, 0.5%

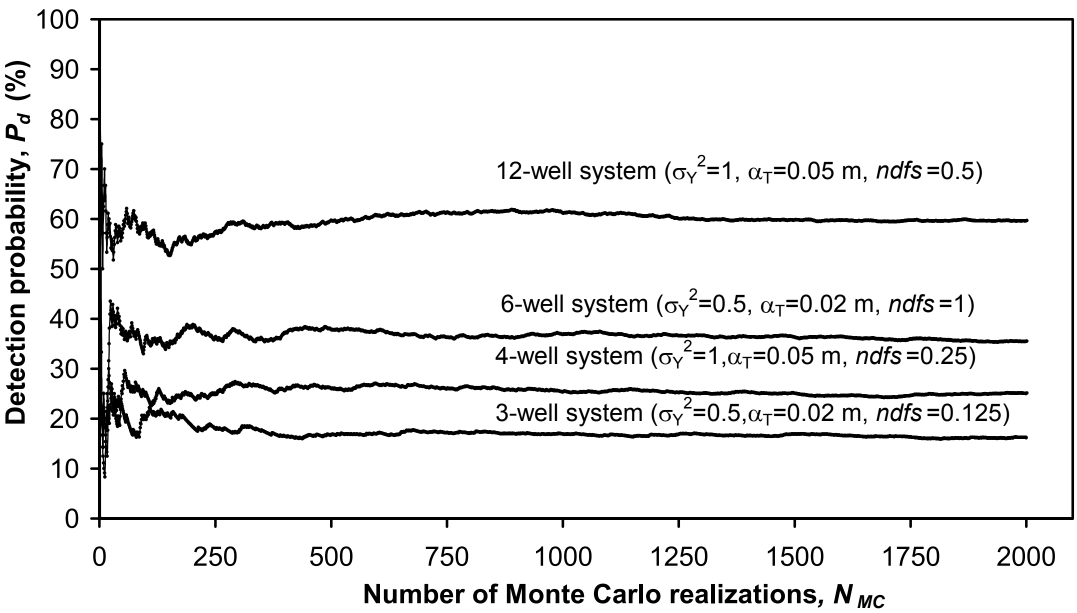


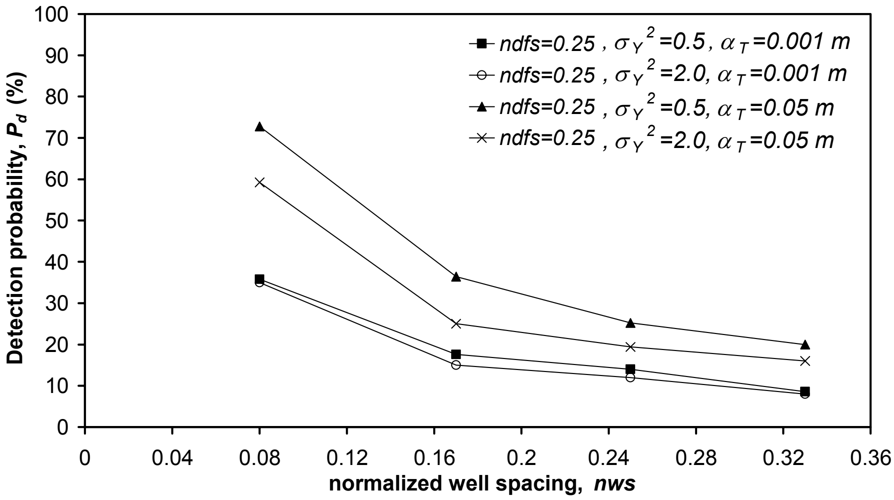
Figure 4.2: Detection probability, P_d as a function of number of Monte Carlo realizations, N_{MC} for different monitoring systems.

4.4.2 Influence of well spacing and location of a single row monitoring system

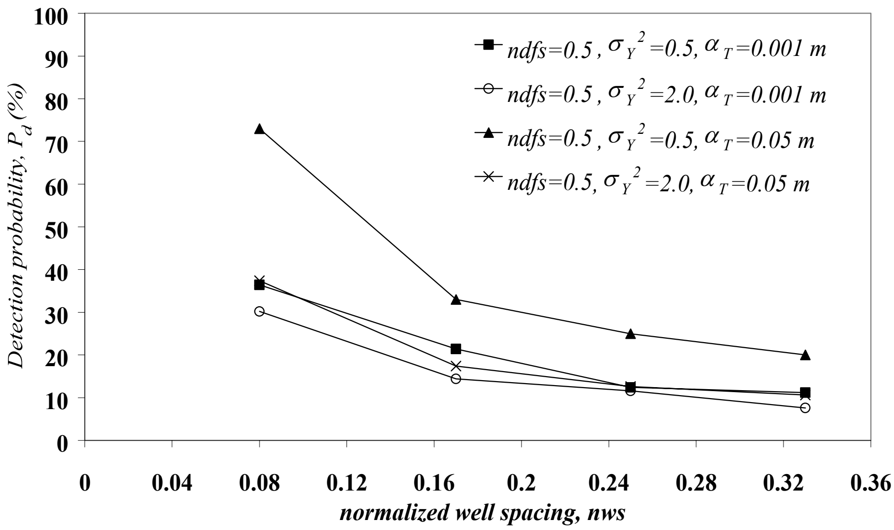
The reliability of monitoring systems is studied by examining the influence of number of wells on the performance of single row systems and the effect of the location of these single row systems. Monitoring systems composed of 3, 4, 6 and 12 wells (*nws*=0.33, 0.25, 0.17 and 0.08, respectively) at seven different distances are evaluated for different heterogeneity and dispersivity conditions (see Table 4.1). For a single row monitoring system, the most efficient design pattern is to locate the monitoring wells evenly spaced (Δs). However one must be ware of the fact that wells located exactly at the top and the bottom boundary of the landfill will lead to less efficient monitoring systems. The problem is that in terms of detecting the contaminant plume, the efficiency of the wells located at the boundaries will be limited to plumes originating from the leaks at the boundaries or at distances that are very close to the boundaries. To prevent this boundary effect and to increase the efficiency of the single row monitoring system, the configuration of the wells should not only be evenly spaced (at distance Δs) but they should be also located at a distance of $\Delta s/2$ from the top and the bottom boundaries of the landfill (see Figure 4.1). Figure 4.3 present P_d as a function of *nws* for σ_Y^2=0.5 and 2.0, α_T=0.001 m and 0.05 m, *ndfs*=0.25 and *ndfs*=0.5, respectively.

For a given distance from the landfill, the detection probability of the contaminant plumes increases with the number of wells, as expected. Moreover, in all numerical experiments, the detection probability of a 3-well system, which is the one required by legislation, is quite low. It has been found that even under the most favourable circumstances with the largest plume widths, for example in a medium specified to be homogenous and highly dispersive (α_L=2 m, α_T=2 m) medium, for a low concentration threshold value (0.25% of the initial contaminant concentration), the P_d of a 3-well system does not exceed 26.4%, whereas under the same conditions a 6-well and a 12-well system can achieve a detection probability 50% and 94% (Figure 4.4). Unlike the transverse dispersivity and initial contaminant source size, C_{TH} has no impact on the actual transport of contaminants but it determines whether or not a simulated contaminant plume is detected by a given monitoring system. As the value of C_{TH} decreases the P_d value increases, due to the enhanced ability to monitor the contaminant, or in other words, the effective detectable plume size increases. The influence of C_{TH} is smallest for low dispersivity values. The reason for this is when advection is the dominant process the plume is narrow and the plume edge is rather sharp. In practice, the threshold concentration is likely to be more precisely defined than many of other physical parameters.

However, in practical applications, it may be unfeasible for a numerical transport model to achieve accuracy at the level of the threshold concentration that represents reality, due to the limitations related to model parameters and computational expenses. For this reason, it is important to understand the consequences of using a threshold concentration in the model that is higher or lower than the one applied in the field. Note that a lower value of C_{TH} than the one applied in the field may lead to more conservative monitoring system designs than actually required or vice versa. Therefore, based on the available circumstances including model parameters, field



(a)



(b)

Figure 4.3: Detection probability, P_d as function of normalized well spacing, *nws* for medium with different degree of heterogeneity and dispersivity for normalized distance from the source, (a) *ndfs*=0.25 and (b) *ndfs*=0.50.

conditions and knowledge, it may be practical to use an intermediate value of C_{TH} in the range between the expected and/or required possible minimum and maximum C_{TH} values so that one can obtain reasonable results enabling the design of appropriate monitoring systems. In this study, threshold concentration value is represented by a percentage of initial concentration so that the model can allow one to design the monitoring system for monitoring a certain contaminant with regard to maximal allowable content of that certain contaminant or as in this study the percentage can represent general amount of contaminant to be monitored regardless of the contaminant type, by simply changing the plume cut off.

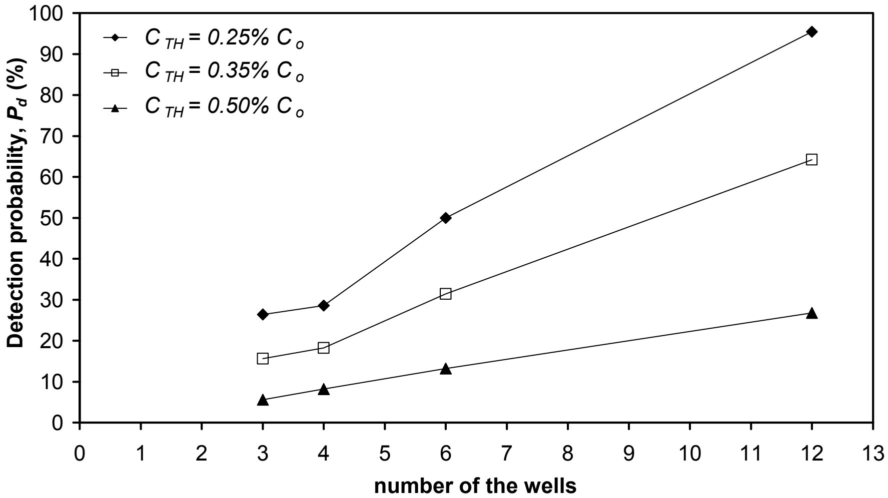


Figure 4.4: Detection probability of monitoring networks in a homogenous and highly dispersive (α_L=2 m, α_T=2 m) medium, for three different concentration threshold values.

In this study, 4000 mg/l was the initial concentration of a point source. A threshold value of 0.35 % of the initial contaminant source corresponds to 14 mg/l. If the contaminant is nitrate, which is one of the wide spread groundwater contamination source, then 0.35 % is representative since in the brochure prepared for the remediation of ground and groundwater in The Netherlands (de Circulaire Streefwaarden en Interwaarden Bodem Sanering, 2000) 15 mg/l is the level that indicates the presence of a nitrate contamination. Furthermore, maximum levels for particular contaminants such as cyclohexanon (used in pesticide formulation or present in fuel) or diethyleneglycol (used in painting stuff) are given in the same document as 15 mg/l and 13 mg/l, respectively. On the other hand, 14 mg/l of a threshold value corresponds to 28 particles, which is a sufficient number for determination of concentration in one grid cell or namely in this study the concentration in a monitoring well (Kinzelbach, 1986). Therefore, mainly the results for a threshold value of 0.35% of initial contamination source are presented in the rest of the paper.

Nevertheless, one can conclude that the most widely applied current practice that fulfils the minimum requirement 3 downgradient monitoring wells is totally inadequate from the point of view of detection of plumes and prevention of groundwater contamination since the subsurface conditions in reality are even much more complicated than those considered in any model.

Table 4.2 summarizes the influence of the location of a single row of wells on P_d for a range of values of transverse dispersivity and hydraulic conductivity variance. Table 4.2 gives the maximum value of P_d and *ndfs* (on average) for given single monitoring systems. A single monitoring system can at most provide a detection probability, which is given in Table 4.2 if it is located at a distance that results in a *ndfs* value equal to or smaller than the one given in the table. For example, for σ_Y^2=0.5 and α_T=0.02 m the P_d of a monitoring system with *nws*=0.17 will be less than 38% if it is located at a distance such that the *ndfs* is greater than 1.00.

Table 4.2: Maximum detection probability, P_d(max) and normalized distance from the source, *ndfs* values for 4 different monitoring network systems corresponding to different degrees of heterogeneity and dispersivity value.

Transverse dispersivity, α_T (m)	***nws***	**Variance of hydraulic conductivity (σ_Y^2)**											
		0.0		0.5		0.75		1.0		1.5		2.0	
		nfds (max)	P_d(%) (max)	nfds (max)	P_d (%) (max)	nfds (max)	P_d(%) (max)	nfds (max)	P_d (%) (max)	nfds (max)	P_d(%) (max)	nfds (max)	P_d (%) (max)
0.001	0.33	2.50	11	1.25	12	1.00	12	0.50	9	0.50	8	0.25	8
	0.25	2.50	17	1.25	16	1.00	16	0.50	12	0.50	12	0.25	12
	0.17	2.50	27	1.25	22	1.00	19	0.50	22	0.50	16	0.25	15
	0.08	2.50	50	1.25	45	1.00	45	0.50	38	0.50	37	0.25	35
0.02	0.33	2.00	19	1.00	17	0.50	16	0.50	15	0.50	15	0.25	15
	0.25	2.00	27	1.00	26	0.50	21	0.50	21	0.50	21	0.25	21
	0.17	2.00	43	1.00	38	0.50	38	0.50	34	0.50	33	0.25	28
	0.08	2.00	78	1.00	70	0.50	66	0.50	65	0.50	60	0.25	55
0.05	0.33	0.50	15	0.50	20	0.50	17	0.25	17	0.25	17	0.25	18
	0.25	0.50	25	0.50	25	0.50	34	0.25	26	0.25	24	0.25	25
	0.17	0.50	43	0.50	33	0.50	37	0.25	37	0.25	35	0.25	36
	0.08	0.50	76	0.50	73	0.50	74	0.25	72	0.25	68	0.25	64
0.10	0.33	0.25	20	0.125	18	0.125	18	0.125	18	0.125	18	0.125	16
	0.25	0.25	27	0.125	26	0.125	25	0.125	24	0.125	21	0.125	21
	0.17	0.25	40	0.125	38	0.125	35	0.125	33	0.125	33	0.125	25
	0.08	0.25	79	0.125	76	0.125	71	0.125	68	0.125	61	0.125	53
0.20	0.33	0.125	15	0.125	11	0.125	11	0.125	11	0.125	11	0.125	10
	0.25	0.125	18	0.125	17	0.125	18	0.125	18	0.125	15	0.125	13
	0.17	0.125	31	0.125	29	0.125	26	0.125	25	0.125	23	0.125	19
	0.08	0.125	64	0.125	54	0.125	50	0.125	48	0.125	40	0.125	37

In the homogenous case (σ_Y^2=0) for low dispersivity values the P_d of the system increases as *ndfs* increases, and a maximum value of P_d is observed at a *ndfs*=2.5. However, when the dispersivity of the medium increases the highest detection probability is obtained for the monitoring systems closer to the landfill. Furthermore, regardless of the degree of subsurface heterogeneity, in a highly dispersive medium (α_L=2 m, α_T=0.2 m), the P_d value of a monitoring system with a *nws*=0.08 (12 well-system) is less than 1% for all three threshold concentration values if *ndfs* >0.5. This can be explained by the width and the dilution of the plume. The plume gets wider as it travels away from the source, hence the larger it gets the lower the concentration is (Figure 4.5). Therefore, P_d decreases with increasing *ndfs* for the medium with higher dispersivity due to dilution of the plume to below C_{TH}, despite the larger plume size. A similar effect is observed when the subsurface heterogeneity increases, since increasing heterogeneity leads to irregular plume shapes due to the so-called fingering effect (Figure 4.6). Hence, the significant decrease in P_d for the monitoring systems much further away from the source is obvious for the more heterogeneous medium with high dispersivity. For instance, for $\sigma_Y^2 \geq 1.5$, the P_d of a 12-well monitoring system does not exceed 15% for all three threshold concentration values used in the numerical experiments when *ndfs*>1.25.

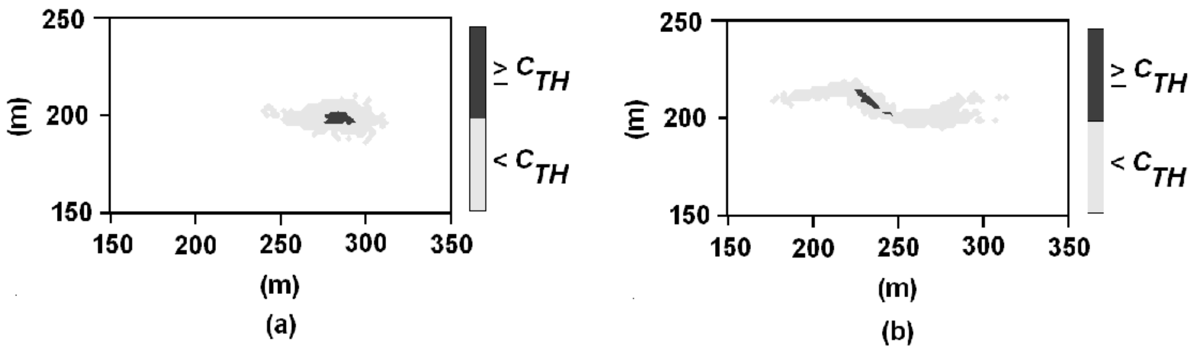


Figure 4.5: Single realization of a plume in homogenous medium a) transverse dispersivity, α_T=0.2 m b) transverse dispersivity, α_T=0.1 m.

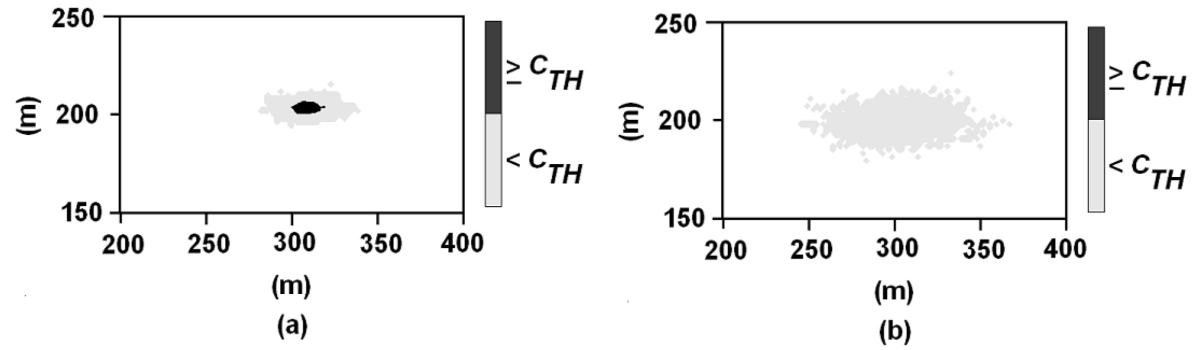


Figure 4.6: Single realization of a plume in a medium with transverse dispersivity, α_T=0.02 m where a) variance of Y, σ_Y^2=0.5 b) variance of Y, σ_Y^2=2.0.

4.4.3 Influence of dispersivity of medium

It is found that the major parameter controlling the spreading of the plume is the dispersivity of the medium, which is in accordance with studies by Meyer et al. (1994) and Storck et al. (1994). The longitudinal dispersivity controls the elongation of the plume with time and distance from the contaminant source in the direction of flow, whereas transverse dispersivity dominates the spreading of the plume (width of the plume) in the direction perpendicular to the flow direction. For single row systems, the main consideration in terms of the well spacing is the plume width. As mentioned earlier, the ratio of longitudinal to transverse dispersivity is taken constant at a value of 10. Therefore, evaluation of P_d as a function of α_T is performed for two different monitoring systems in the homogenous case. The general tendency as shown in Figure 4.7 is that P_d increases as the values of α_T increase up to a certain distance from the landfill. P_d starts to decrease after a certain *ndfs* for higher dispersivity values due to dilution of the wider plumes. Especially for α_T=0.2 m (the highest value used in the numerical experiments) this effect is observed even at distances very close to the source ($ndfs \leq 0.25$).

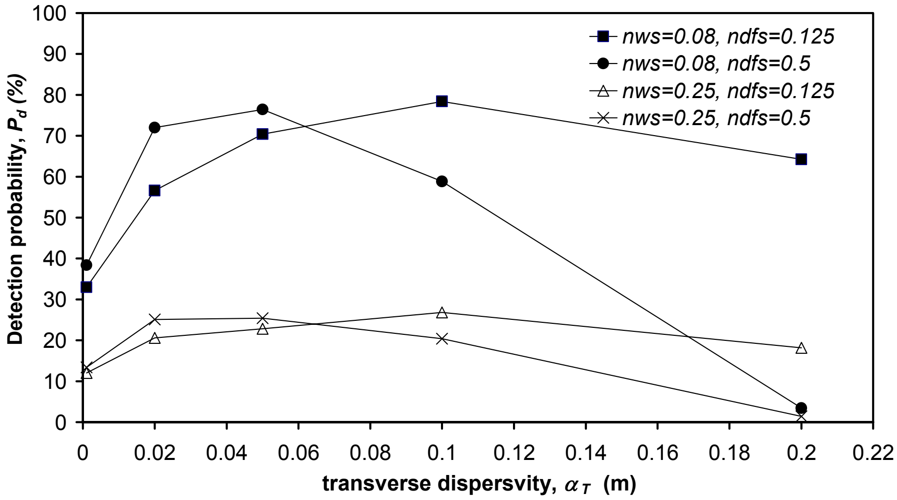
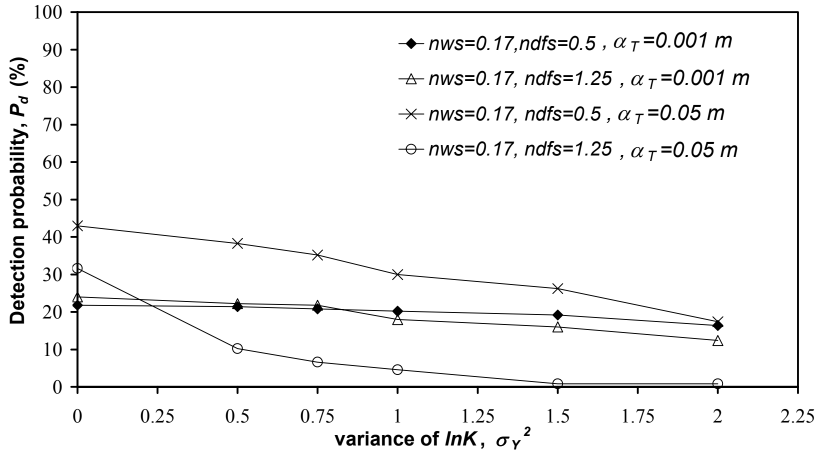


Figure 4.7: Detection probability, P_d as a function of transverse dispersivity, α_T in a homogenous media, for monitoring systems with normalized well spacing, *nws*=0.08 and *nws*=0.25.

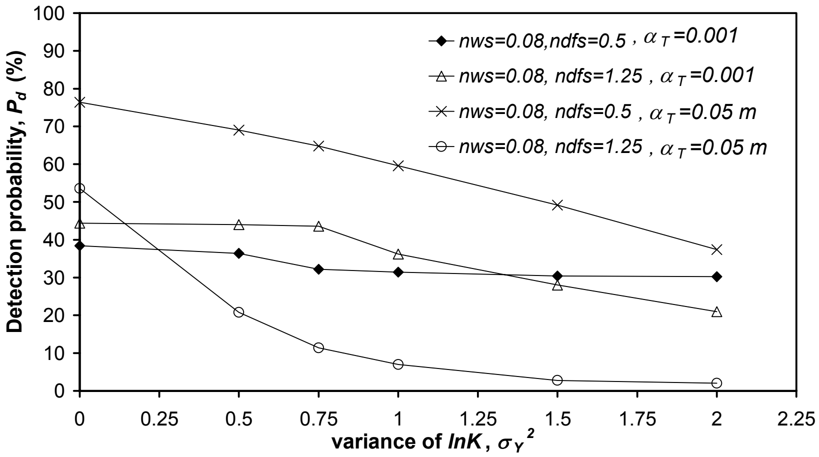
4.4.4 Influence of subsurface heterogeneity

Subsurface heterogeneity, represented here by the spatial variability of the hydraulic conductivity, is one of the important factors controlling the migration of contaminants in porous media. The hydraulic conductivity, K is homogeneous on the scale of discretization (grid cell) but heterogeneous at larger scales. The variance of Y is the parameter that characterizes the degree of heterogeneity of the subsurface. A high variance will produce a highly heterogeneous field with hydraulic conductivity values spanning a wide range, while a low variance will produce a more homogeneous-like field. Figure 4.8 show P_d of a 6 well and 12-well system as a function of σ_Y^2, for *ndfs* equals to 0.5 and 1.25 for α_T equals to 0.001 m and 0.05 m, respectively. The detection probability of the monitoring system decreases as the variance of hydraulic conductivity increases. In a more heterogeneous subsurface it is more difficult to detect a contaminant plume. As explained earlier, this effect is due to the irregular shape of the plume.

On the other hand, the results of analysis show that the influence of heterogeneity is reduced when the monitoring system is located near to the source ($ndfs < 0.25$) for α_T<0.1 m. In fact, this behaviour can be related most likely to two reasons. First, a large number of the simulated plumes may not have a chance to travel more than one correlation length, which occurs on a scale of 40 m in this case, since the correlation length is 20 m. In other words as the plume travels more correlation lengths the influence of heterogeneity is more dominating and this explains why the influence of σ_Y^2 on P_d is rather noticeable for the monitoring systems located further away from the source. Secondly, plumes are still relatively narrow since the influence of low to intermediate values of dispersivity on the spreading of the plume is more dominant at further distances (i.e. *ndfs* >0.25). Nevertheless, analysis also showed that for $\alpha_T \geq 0.1\,\mathrm{m}$ P_d decreases as the heterogeneity increases even for values of the *ndfs*<0.25. This is due to the fact that in a highly dispersive medium, spreading of



(a)



(b)

Figure 4.8: Detection probability, P_d as a function of variance of Y, σ_Y^2 for a monitoring well system, with a normalized well spacing, *nws* of (a) 0.17 and (b) 0.08, for normalized distance from the source, *ndfs*=0.50 and *ndfs*=1.25, where transverse dispersivity, α_T is equal to 0.001 m and 0.05 m, respectively.

the plume is likely dominated more by dispersivity, and therefore in such cases the coupled effect of dispersivity and subsurface heterogeneity on the evolution of the plume can be noticeable even at very close distances from the contaminant source.

4.4.5 Influence of the initial contaminant source size

In the numerical experiments carried out in this study, it has been observed that the efficiency of the monitoring systems is highly dependent on the parameters controlling the average width of the simulated plumes. The initial size of the contaminant source is expected to be another important parameter directly influencing the width of the plume. In the numerical experiments discussed earlier, the initial contaminant source is assumed to be a point source chosen at random within the landfill area. In this section, the influence of initial contaminant source (leak) size is examined by increasing the size of the contaminant source. Hence calculations are performed for initial contaminant source sizes of one grid cell size (2×2 m), and four grid cell sizes

(4×4 m). Realizations with the same velocity field as the point source problem are used. Figure 4.9 shows that the detection probability increases as the initial size of the contaminant source increases for a given monitoring system. This is due to the fact that a larger contaminant source size results in a wider plume. However, a decrease in the value of P_d after certain *ndfs* values for a highly heterogeneous and/or highly dispersive medium, despite the wider plumes, is still valid for larger contaminant source sizes due to the same dilution effect mentioned earlier.

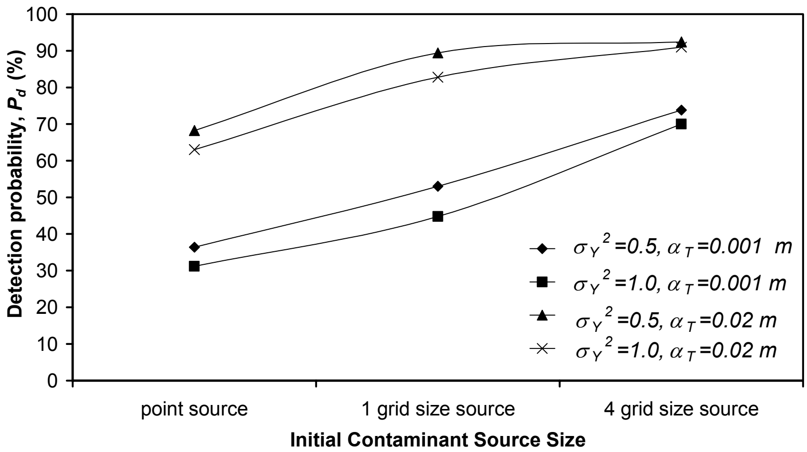


Figure 4.9: Influence of the initial contaminant source size on detection probability, P_d of a 12-well system, for *ndfs*=0.50.

4.5 Summary and conclusions

In this study, a reliability assessment was carried out to estimate the performance of groundwater monitoring systems at landfill sites. Results obtained from extensive numerical experiments show the dependence of the reliability of monitoring systems on several parameters such as dispersivity of the medium, heterogeneity of the medium, size of the initial contaminant leak, detection threshold, and number and location of the wells. The analysis showed that the lateral dispersivity of the medium has one of the most significant influences on the efficiency of the systems, since it is the primary parameter controlling the size of the plume. The detection probability of a monitoring system increases as the initial contaminant size and dispersivity of the medium increases. For transverse dispersivity values greater than 0.02 m the maximum detection probability is obtained when the monitoring systems are closer to the landfill. This is due to the fact that, although the plume gets wider as it travels away from the source, it is diluted to concentrations below the threshold limit. Regardless of the degree of subsurface heterogeneity in a highly dispersive medium (α_L=2 m, α_T =0.2 m), detection probability of a monitoring system with a normalized well spacing of 0.08 (12 well-system) is less than 1% when the normalized distance from the contaminant source is greater than 0.5.

Subsurface heterogeneity is another important factor that affects the reliability of the monitoring systems, since it controls the movement of the contaminant and the shape of the plume. The detection probability of the monitoring well system decreases as the variance of hydraulic conductivity increases. This is caused by the fingering effect due to subsurface heterogeneity. The more heterogeneous the field is, the more irregular the plume shape and the lower the detection probability is. However, the influence of heterogeneity is less when the monitoring system is located close to the source in a medium with transverse dispersivities less than 0.1 m.

Analyses showed that the size of the initial contaminant source is another factor that has influence on the width of the plume. A larger initial contaminant source (leak) size initiates wider plumes and therefore the detection probability of a given row system located at a given distance increases as the size of the initial contaminant source increases.

In homogeneous and low dispersive media the detection probability of the system increases as the normalized distance from the source increases, but in media with higher dispersivities the maximum detection probability is obtained when the monitoring system is located closer to the contaminant source. This is due to the dilution of the plume despite the growth in its size as the plume moves away from the contaminant source. This effect is particularly obvious in a highly dispersive medium regardless of the degree of subsurface heterogeneity. Even a 12-well monitoring system can detect less than 1% of the simulated contaminant plumes when the normalized distance from the contaminant source is greater than 0.5. A similar effect is observed when the variance of hydraulic conductivity increases, because the irregularity in the shape of the plume due to subsurface heterogeneity is more noticeable when the plume moves further away. The detection probability of a 12-well monitoring system does not exceed 15% in the case of a variance of hydraulic conductivity greater than or equal to 1.5, for the normalized distance from the contaminant source greater than 1.25. The analyses showed that the detection probability increases as the normalized well spacing decreases. A striking conclusion from the numerical experiments is that the detection probability of a 3-well system reaches at most 26.4 % even under the most favourable conditions for *all* other parameters. Therefore one can firmly conclude that the widely applied common practice that fulfils the minimum regulatory requirement of regulations, namely 3 downgradient wells to monitor a possible contaminant plume released from a landfill, is definitely inadequate from the point of view of the detection of the contaminant plume and the prevention of groundwater contamination.

Chapter 5

Optimal Systems for Groundwater Detection Monitoring

Adapted from Yenigul, N.B., Elfeki, A.M.M., van den Akker, C., and Dekking, F.M., A decision analysis approach for optimal groundwater monitoring system design under uncertainty, submitted to Hydrological Earth Systems and Sciences and published in HESSD, 3, 27-68, 2006.

The main scope of this chapter is to solve the actual monitoring system design problems by using the simulation and reliability model described in the previous chapters. The objective of the monitoring systems is to detect the contaminant plumes before reaching the regulatory compliance boundary in order to prevent the severe risk to both society and groundwater quality, and also to enable cost-effective counter measures in case of a failure. The detection monitoring problem typically has a multi-objective nature. A multi-objective decision model, which links a classic decision analysis approach with the stochastic simulation model that has been addressed in the previously, is applied to determine the optimal groundwater monitoring system given uncertainties due to the hydrogeological conditions and contaminant source characteristics. The first section includes the description of design objectives and a brief literature review on detection monitoring to provide an overview of the problem while revealing the presented approach in design of detection monitoring systems at landfill sites. The next two sections include a detail description of the methodology and its application to a hypothetical example. Thereafter, analysis results are discussed and general conclusions with respect to the location of the monitoring systems are given.

5.1 Introduction

The growing awareness of environmental issues more often reflects a reactionary response of public throughout the last years. For instance, the fear of groundwater contamination from a leaky landfill has been a major reason for the difficulty in locating landfills, due to such public reactions to the local governments. When faced with sceptical, and even reactionary public, the ability of technical experts to present designs that minimize risk becomes very important. In case of a landfill, concern often centres on the risk of exposure to contaminated groundwater. Regulatory agencies require groundwater monitoring programs at solid waste landfills, hazardous waste sites, and other sites, where potential release of chemicals to the subsurface is a concern, so that the risk of exposure can be further reduced by monitoring the quality of the groundwater.

The object of detection monitoring is to detect the plume early enough that appropriate action can be taken to prevent exposure. As mentioned before in Chapter 1, according to USEPA (1986) detection monitoring wells (at least three downgradient wells) should be placed at locations where they will intersect all possible pathways of contamination. The document also suggests placing the wells as close as possible to the source so that the contaminants are detected as soon as a release occurs. However early detection of contaminants implies that small contaminant plumes must be detected, which can be difficult with a limited number of wells such as proposed by the regulations (namely at least three downgradient monitoring wells). The likelihood of detection increases when a large number of monitoring wells are located, however, the monitoring and construction cost also increases. Hence a trade off exists between the likelihood of detecting contaminant plumes, the plume size, and the associated cost of construction, operation and maintenance of the monitoring systems. Consequently, the design of a groundwater detection monitoring system can be formulated with three conflicting objectives: (1) Maximize the probability of detection of contaminant plumes, (2) minimize the contaminated area, and (3) minimize the total cost of the monitoring system (i.e. construction, maintenance, and remediation cost, if necessary). Therefore the design of an efficient monitoring system is quite complicated and becomes even more difficult when uncertainty, which is characteristic of groundwater problems, is incorporated in the problem.

Several studies are cited in the literature addressing the different aspects of the detection-monitoring problem. Loaiciga et al. (1992) provided a comprehensive review of groundwater monitoring network design. They classified the existing approaches into diffent categories. These approaches are in general based on geostatistical methods (e.g., Rouhani, 1985; Rouhani and Hall, 1988; Haug et al., 1989; Cawlfield and Wu, 1993; McLaughin and Graham, 1986), methods based on simulations (e.g., Meyer and Brill, 1988). Based on graphical methods (e.g., Hudak, 2001; Hudak, 2002; Hudak, 2005) and optimization methods (e.g., Hudak and Loaiciga, 1993; Meyer et al., 1994; Storck et al.; 1997; Mahar and Datta, 1997; Montas et. al., 2000).

Optimization involves the determination of optimal values for a set of decision variables in an engineering system and optimality is defined with respect to a specified objective function and is subject to a set of constraints (Freeze et al., 1990). Identifying the global optima over a nearly continuous decision space is the main advantage of the optimization. The main disadvantages lie in the difficulties introduced by considering capital costs for complex non-linear problems, and the solutions do not typically span multiple technologies (Freeze and Gorelick, 1999). An optimal solution for a multi-objective problem, such as a detection monitoring design, cannot be determined based solely on the values of the objective function since the solution of a multi-objective optimization problem yields an infinite number of optimal solutions referred to as Pareto optimal solutions that are equally good optimal solutions. This solution set is usually large and gives rise to two main problems. First, identifying one solution for implementation can be quite tricky and secondly, because the objectives are usually non-commensurate, finding a preferred point as a compromise or satisfying the solution with a rational procedure can be quite a challenging task. Moreover solution of a multi-objective groundwater detection monitoring problem within a stochastic framework to include uncertainties due to the hydrogeological characteristics may be computationally very expensive and less feasible in practice for most common engineering projects. For example in the work of Meyer et al. (1994), a multi objective stochastic optimization approach is used to determine the 2D location of monitoring wells. The method incorporates uncertainty in hydraulic conductivity and source location through Monte Carlo simulations, and the contaminant leak is assumed to be continuous. Storck et al. (1997), extended this model to three dimensions incorporating local dispersion. They concluded that the influence of local dispersion can typically be ignored and that the method is rather elaborate in terms of computational expenses. The main drawback of both studies is that a huge computational effort is required to perform such a search technique in order to find optimal sampling geometries.

On the other hand, decision analysis provides a rational step by step approach to support the decision making process and most often compares alternatives on a common basis called utility function. In most of the applications the utility function is represented by money and preference is based on a specified objective function, including the risks, costs, and benefits of alternatives. A utility function can be defined as a mathematical function that associates a utility (numerical value) with each alternative solution so that all alternative solutions may be ordered. When more than one performance criteria are used like in a detection monitoring design problem, the utility function is called a multi-attribute utility function. In multi-objective (multi-criteria) decision making problems one usually considers a set of alternatives, which are valued by a family of decision objectives (criteria). Assessment of such a set of overall preferences of an individual decision maker leads to aggregation of all objectives into a unique objective, called a multi-attribute utility function. Since the process of engineering design is often described as a sequence of decisions between alternatives under conditions of uncertainty and hydrogeologists and engineers are often asked to address alternatives in the most traditional engineering practice, decision analysis is well-suited to the risk-based philosophy of engineering design (Gorelick et al., 1993). Moreover, when one incorporates the uncertainties, less computational effort is required to solve the multi-objective decision problem compared to solution of a multi-objective optimization problem in a stochastic framework. However, besides the ease of an incorporation of capital cost and the ability to examine alternative designs that span multiple technologies, the discrete decision space and the need for a full enumeration of the design alternatives are the limitations of the decision analysis. For further reading on the advantages and shortcomings of decision analysis versus optimization methods the reader is referred to the work of Freeze and Gorelick (1999).

However, a limited number of authors have used decision analysis models in solution of problems related to groundwater contamination. For example, Massman and Freeze (1987 a, b) presented a risk-cost-benefit analysis for waste management facilities from the perspective of the owner/operator to make design decisions for the facility plus the role of a regulatory agency as a counterpart to the owner/operator. They covered a wide range of factors relevant to the design of a landfill liner in their analysis. However, they do not detail the evaluation of the reliability of the monitoring system.

Massmann et al. (1991) presented two applications of the framework presented by Freeze et al. (1990). The first example considers a preliminary design of a groundwater extraction system whereas the second one involves the design of a facility to treat contaminated soil. Homogeneous aquifer conditions were considered for both examples. Likewise Massman and Freeze (1987 a, b) the first example considered advective contaminant transport. Freeze et al. (1992) treated site characterization in relation to a landfill liner design. They used search theory, Bayesian updating and prior analyses in a simplified hypothetical example, where advective contaminant transport was considered. All these works do not directly refer to groundwater monitoring system design. On the other hand, Jardine et al. (1996) illustrated a decision model for evaluating monitoring network designs at waste management facilities overlying fractured rocks while considering an instantaneous pulse of contaminants. They did not include uncertainty in the leak location in their analysis.

In this study detection monitoring design is approached from a decision analysis perspective. Different from the above-mentioned studies, a decision analysis approach particularly for groundwater detection monitoring system design at landfill sites is proposed for heterogeneous sedimentary aquifers while considering advective-dispersive contaminant transport. The simulation model presented in the previous chapter for reliability assessment of monitoring systems at landfill sites is linked with a classic decision analysis approach to formulate a decision analysis model called *MONIDAM.* It incorporates the three conflicting objectives presented previously in a systematic way, as well as the uncertainties due to subsurface heterogeneity and leak location to determine the optimal groundwater monitoring system.

5.2 Methodology

Monte Carlo simulation, an economical analysis of the objective function for monitoring systems, and the selection of the optimal monitoring system are the principal steps of the decision model *MONIDAM.* It allows for the comparison of monitoring systems alternatives (Systems 1 through n). The following paragraphs provide an overview of each step of the decision analysis framework of *MONIDAM* illustrated in Figure 5.1.

5.2.1 Monte Carlo simulation

The simulation model described in Chapter 2 is used to simulate contaminant plumes. The outcome of each Monte Carlo realization consists of two items. The first one is a binary variable representing detection or no detection of the contaminant plume, which is already defined as system reliability model in the previous chapter. The second output variable represents the area of contamination A_d associated with the size of the plume at the time of detection, or the area of contamination A_f when the plume remains undetected at the end of the monitoring period. When a given system j detects a plume, an associated contamination area is obtained.

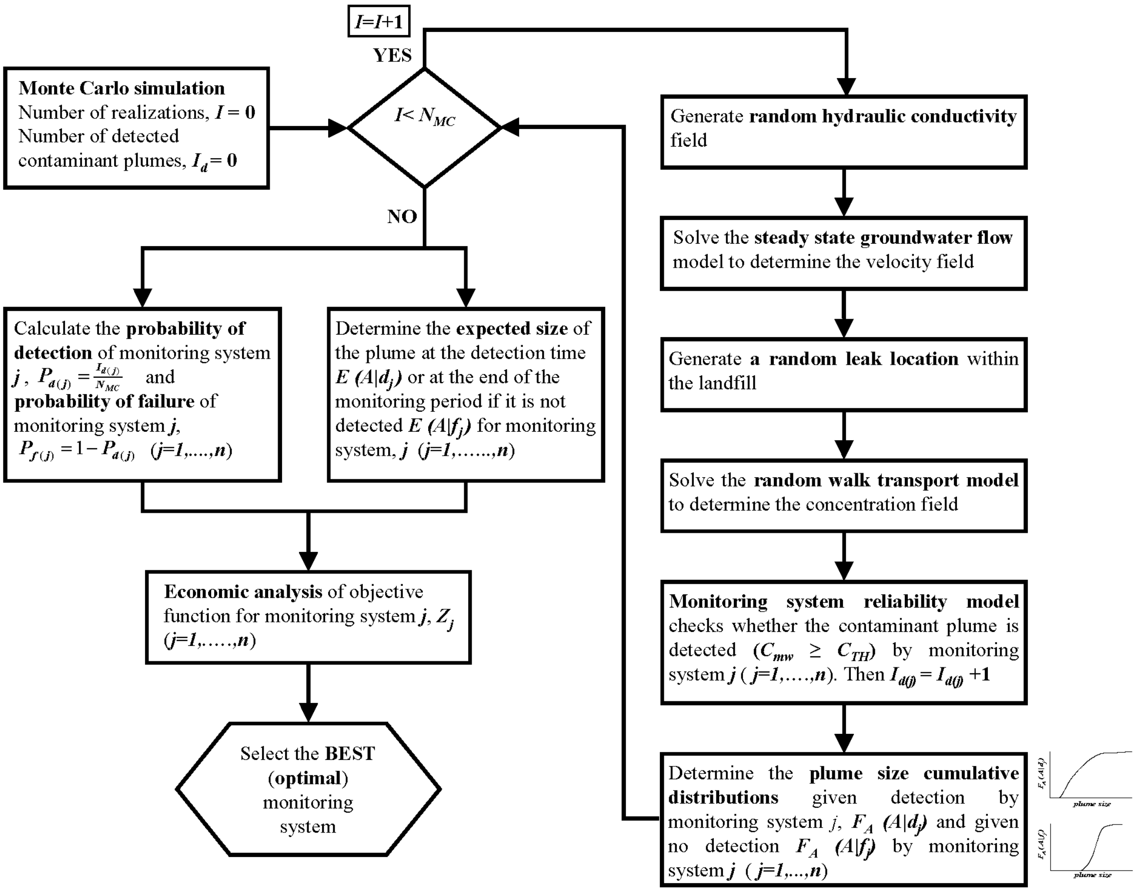


Figure 5.1: Flow chart illustrating the structure of the decision model *MONIDAM*

However, the area of contamination at detection varies from one simulation run to another due to the variability in leak location and subsurface heterogeneity. Therefore, rather than one single contaminant plume, a range of plume sizes at detection is obtained coming from every individual simulation. Using the individual plume sizes at detection, cumulative distributions $F(A|d_j)$ of plume size A can be determined for monitoring system j. Likewise, the cumulative distribution $F(A|f_j)$ of plume size given no detection can be determined by using the individual plume sizes that remain undetected at the end of the monitoring period. These probability distributions are used in the economical analysis of the objective function.

5.2.2 Economic analysis

The economic objective of design must be to meet the technical objective in such a way so as to maximize the profit (or minimize the loss) to the owner-operator (Freeze et al., 1990). From this perspective, an objective function defined as the net present value of the expected stream of benefits, cost, and risks taken over an engineering time horizon and discounted at the market interest rate, (Gorelick et al, 1993):

$$Z_j = \sum_{t=0}^{T} \left[\frac{1}{(1+i)^t} \right] \left[B_j(t) - C_j(t) - R_j(t) \right] \quad (j=1,.....,n) \tag{5.1}$$

where Z_j= objective function for alternative j [$],

T= planning horizon [years],

i= annual discount rate [decimal fraction],

$B_j(t)$= benefits of alternative j in year t [$],

$C_j(t)$= costs of alternative j in year t [$], and

$R_j(t)$= risks of alternative j in year t [$].

The risks, $R_j(t)$, associated with alternative j in year t are defined as:

$$R_j(t) = P_{f(j)}(t) C_{f(j)}(t) \gamma\left(C_{f(j)}\right) \quad (j=1,\ldots..,n) \tag{5.2}$$

where $P_{f(j)}(t)$= probability of failure of alternative j in year t [decimal fraction],

$C_{f(j)}(t)$= cost associated with a failure of alternative j in year t [$]

$\gamma(.)$ = normalized utility function [decimal fraction, ≥1]

The utility function $\gamma(.)$ in Equation (5.2) allows one to take into account the possible risk-averse tendencies of some decision makers. A risk-adverse decision maker will set the utility function to larger than one. Small owner-operators who do not have large net worth are the most likely to use a risk-adverse utility function. Larger companies are more likely to take a risk-neutral approach (Gorelick et al, 1993).

In this study only the construction of a detection monitoring system within the property boundaries of the landfill is considered. There is no analysis of the trade-off between the facility design and monitoring. It is assumed that the revenues generated by the landfill would be the same regardless of the monitoring strategy adopted. Thus it is possible to neglect the benefit terms. Since the trade-off between facility and monitoring is not considered, the capital costs of constructing and operating the landfill are also the same regardless of the monitoring strategy chosen. Hence, these costs are neglected and only those costs directly associated with the construction and operation of the detection monitoring system are taken into account. On the other hand the time dependencies will not be important due to the three assumptions made in this study: (1) the leak in the landfill will occur at the beginning of the monitoring period, and the contaminant plume will be either detected at any time during the monitoring period, or failure will occur if it is not detected at the end of the monitoring period, (2) installation cost of the monitoring wells is the only cost occurs in year zero, the year before the landfill and monitoring system begin operation, and (3) the annual unit costs for monitoring and remediation cost per unit volume of contamination are constant for the whole monitoring period. With all these assumptions, the objective function in Equation (5.1) can be simply written as the summation of the time independent costs and risk:

$$Z_j = -\left[C_j + R_j\right] \quad (j=1,\ldots..,n) \tag{5.3}$$

The minus sign can be removed from Equation (5.3) without loss of generality. The construction and operation cost C_j of monitoring system j can be estimated as:

$$C_j = C_{mw} n_{mw} + C_{sn} n_{mw} s_f \tag{5.4}$$

where C_{mw}= unit installation cost of a monitoring well($/per well),

C_s= sampling cost ($/sample/per well)

n_{mw} = number of the wells in monitoring system j

n_{smw} = number of the sampled wells in monitoring system j.

s_f = number of the total sampling for the total monitoring period.

For a site with no monitoring system, the risk term is equal to the expected costs associated with failure. However, for the sites with a detection monitoring, the risk term is expanded to allow for possibility of the plume being detected and remediated before failure occurs. Therefore, the risk term in this study represents the expected cost associated with both detection of the contaminant plume and failure if it remains undetected at the end of the monitoring period. From a risk-neutral approach perspective the risk term (R_j) associated with monitoring system j can accordingly be defined as:

$$R_j = P_{d(j)} C_{dr(j)} + P_{f(j)} C_{fr(j)} \quad (j=1,\ldots\ldots,n) \tag{5.5}$$

where $P_{d(j)}$= the probability of detection of monitoring system j,

$C_{dr(j)}$= remediation cost associated with detection of contaminant plume by monitoring system j [$],

$P_{f(j)}$= probability of failure of monitoring system j and,

$C_{fr(j)}$= remediation cost when monitoring system j fails to detect the contaminant plume [$].

The clean up cost associated with detection of contaminant plume by monitoring system j can be obtained as:

$$C_{dr(j)} = C_r V_{d(j)} \tag{5.6}$$

where C_r is the remediation cost per unit volume [$/m^3], $V_{d(j)}$ is volume of contamination given detection by monitoring system j [m^3]. Similarly, the cost associated with failure, $C_{fr(j)}$ is C_r times volume of contamination given no detection by monitoring system j, $V_{f(j)}$ [m^3].

As mentioned above in Section 5.2.1 the plume areas associated either with detection or no detection follow a probability distribution and the volume of contamination is defined by the expected plume area times the aquifer thickness, $B=B(A)$ [m]. Let $f(A/d_j)=F'(A/d_j)$ be the probability density of the plume size. Then the risk term for monitoring system j can be found by:

$$R_j = P_{d(j)} \int_0^{+\infty} C_r(A) B(A) A f\left(A \mid d_j\right) dA + P_{f(j)} \int_0^{+\infty} C_r(A) B(A) A f\left(A \mid f_j\right) dA \tag{5.7}$$

Since the unit cost of remediation and the aquifer thickness are assumed to be constant, Equation (5.7) can be simplified to:

$$R_j = P_{d(j)} C_r B E(A_{d(j)}) + P_{f(j)} C_r B E(A_{f(j)}) \tag{5.8}$$

where $E(A_{d(j)})$ is the expected contaminated area given detection by monitoring system j and $E(A_{f(j)})$ contaminated area given no detection by monitoring system j .The final form of the objective function or the so-called multi-attribute utility function is a general form of *MONIDAM*. It represents the expected total cost for each monitoring system $j=1,\ldots,n$ and is given by:

$$\underbrace{Z_j}_{\substack{\text{expected}\\ \text{total}\\ \text{cost}}} = \underbrace{C_j}_{\substack{\text{construction}\\ \text{and operation}\\ \text{cost}}} + P_{d(j)} \underbrace{C_r B E(A_{d(j)})}_{\substack{\text{expected cost}\\ \text{given detection,}\\ E(C|d_j)}} + P_{f(j)} \underbrace{C_r B E(A_{f(j)})}_{\substack{\text{expected cost}\\ \text{given no detection,}\\ E(C|f_j)}} \tag{5.9}$$

5.2.3 Selection of the optimal monitoring system

The 'best' (optimal) system is the system that enables the maximally possible detection probability, while minimizing the expected contaminated area by using the least number of wells. In other words best alternative is the minimization of the cost associated with monitoring and with remediation measures, if necessary. Once Equation (5.9) determines the expected total cost for each monitoring system, the alternative with the minimum expected total cost (Z_{min}) is the '*optimal'* groundwater detection monitoring system. The optimal system does not necessarily enable 100% reliability or does not require a certain pre-specified reliability level. It is defined by the cost associated with maximally possible detection probability and minimally possible contaminated area by using the least number of the wells.

5.3 Example problem

A series of numerical experiments was carried out to study the use of *MONIDAM* in optimal groundwater monitoring system design. The sensitivity of the objective function values of the model is illustrated by varying the model parameters.

5.3.1 Model domain and discretization

The numerical experiments are carried out using a model of a generic landfill facility and groundwater system. Dimensions of the model domain and the parameters that have been used are chosen to reflect conditions of typical solid waste landfills. The model domain is defined by $0 \le x \le 500$ m and $0 \le y \le 400$ m (Figure 5.2). The model is discretized with grid cells of 2 m by 2 m in both x- and y-directions. A rectangular

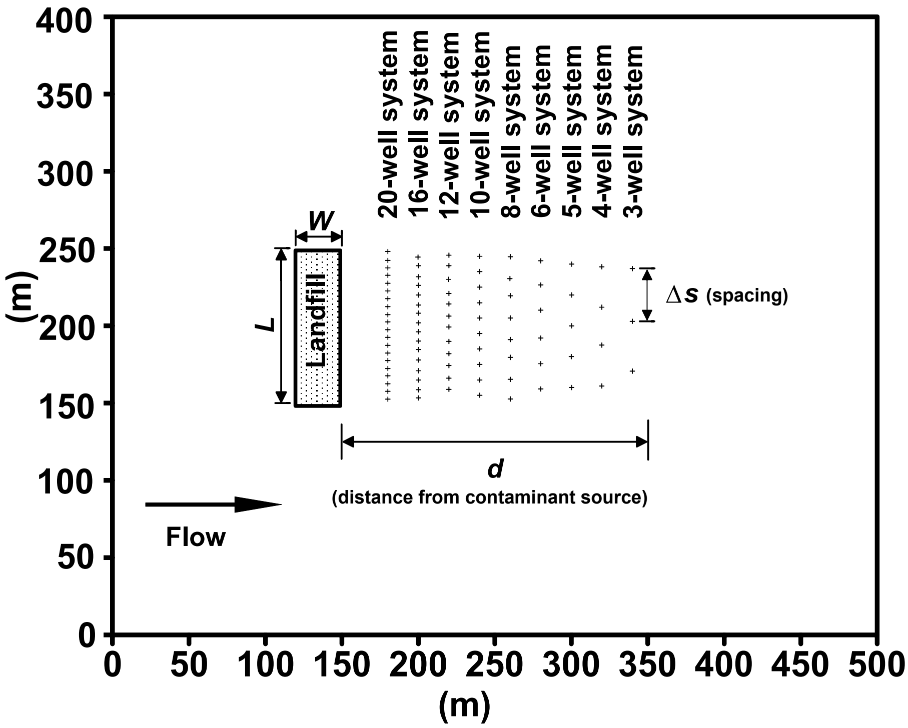


Figure 5.2: Dimensions and components of example problem used in numerical experiments. (Note: the spacing between the first (or last) well and the y=150 m (respectively y=250 m) line equals to $\Delta s/2$ and each monitoring system (i.e., with a fixed number of monitoring wells) is located at a different distance from the landfill which result in 171 potential monitoring system alternatives).

landfill is located at $100 \leq x \leq 150$ m and $150 \leq y \leq 250$ m in the model domain. The monitoring systems considered are composed of a line of wells parallel to the y- axis, covering the length of the landfill. It may be noted that a longer line of wells that extends beyond the length of the landfill could also be considered. However, determination of the appropriate longer length would involve major assumptions on the dispersivity of the medium. This simple pattern adjusts to the shape of the landfill used and the groundwater flow characteristics in the example. For actual case studies with a less ideal shape than in the example, monitoring wells placed along the downstream perimeter of the facility or along a curve parallel to it are the configurations equivalent to the row configurations presented here.

Monitoring systems composed of 3, 4, 5, 6, 8, 10, 12, 16, and 20 monitoring wells are evaluated to study the effect of number of wells on the reliability of single row monitoring systems. Since the number of monitoring wells in the system must cover the length of the landfill, the spacing of wells in the single row monitoring systems decreases as the number of the number of monitoring wells increases. The spacing Δs between the wells and the distance d from the edge of the landfill are normalized with respect to the length of the landfill L perpendicular to the flow, to get rid of the specific landfill size. For instance, a monitoring system composed of 3 monitoring wells has normalized well spacing, nws ($\Delta s/L$) of 0.33, which means that one third of the landfill is monitored by each monitoring well in the system. Each monitoring system (i.e., with a fixed number of monitoring wells) is located from 10 m to 190 m downgradient of the landfill with 10 m distance apart in order to determine the influence of the location of single row monitoring systems. The distances from the source are also normalized with respect to the length of the landfill for generalization purposes, as well as for consistency with *nws*. As the area of potential leak locations is assumed to be the downgradient edge of the landfill the location of the source of contaminant is at x equal to 150 m, the distance from the contaminant source, d can be obtained as the difference between the x- coordinate of the single row monitoring systems and location of the source of contamination (Figure 5.2). Table 5.1 presents spacing, Δs, normalized well spacing, *nws* for different number of monitoring wells in monitoring systems with different x- coordinates, distance from the contaminant source, d, and normalized distance from the source, *ndfs*.

Table 5.1:Spacing, Δs, normalized well spacing, *nws* for different number of monitoring wells in monitoring systems with different x- coordinates, distance from the contaminant source, d, and normalized distance from the source, *ndfs* (L=100 m)

Number of wells	Spacing, Δs (m)	$nws=\Delta s/L$	x-coordinate of the system	Distance from the source, (m)	$nfds=d/L$
3	33.33	0.33	160	10	0.1
4	25	0.25	170	20	0.2
5	20	0.2	180	30	0.3
6	16.66	0.16	190	40	0.4
8	12.5	0.12	•	•	•
10	10	0.1	•	•	•
12	8.33	0.08	•	•	•
16	6.25	0.06	•	•	•
20	5	0.05	340	190	1.9

5.3.2 Parameter values used in flow model

The aquifer is assumed to be confined, with given hydraulic head at left and right boundaries, resulting in a macroscopically constant hydraulic gradient of 0.001. The porosity of the medium equals 0.30. Uncertainties due to contaminant source location and subsurface heterogeneity are incorporated in the model using 1000 Monte Carlo simulations. As mentioned earlier, subsurface heterogeneity is reflected by the spatial variability of the hydraulic conductivity in this study. The natural logarithm of the isotropic hydraulic conductivity [Y=ln(K)] is modelled as a stationary Gaussian field with a geometric mean value of 2.23 m/day , a variance set at σ_Y^2=0.4 and the isotropic covariance of Y is chosen to be of exponential form with correlation length=15 m.

5.3.3 Parameter values used in random walk particle tracking model

For the transport model a condition of a zero dispersive flux is imposed on the top and bottom boundary, and the initial background concentration in the model domain is set to zero. Since the flow direction is aligned with x-axis, the only source dimension that is treated as a random variable is the position along the y-axis. The area of potential leak locations is the downgradient edge of the landfill (Figure 5.2). The contaminant leak is assumed to be a point source (i.e. a single cell represents the location(s) of leakage), as it would result in a plume, which is most difficult to detect and the source location is drawn from a uniform probability distribution between y-coordinates of $150 \leq y \leq 250$ m for each Monte Carlo run. Dispersion is incorporated in the model by introducing micro scale longitudinal (α_L) and transverse (α_T) dispersivities. The ratio between α_L and α_T is assumed to be 10 (Bear, 1972) and. α_L is set to 0.5 m. The simulation procedure assumes that the source is continuous and provides a constant mass rate of 1 mg/l/day. The threshold concentration (detection limit) at which detection occurs is set at 0.5 % of the initial source concentration. This level represents the EPA public health risk level for drinking water for the most common contaminant types released to groundwater mainly via leaks from landfills such as benzene, carbon tetrachloride and 1,1,2-Tricloroethane. And 0.05 mg/l of a threshold value corresponds to 40 particles, which is a sufficient number for determination of concentration in one grid cell (Kinzelbach, 1986). As in the example presented in the previous chapter also here monitoring wells are located in the centre of the grid cell and have a dimension of one grid cell. Contaminants are assumed to be conservative and to be completely mixed over the depth of the aquifer, which is presumed to be 50 m in the example problem. A monitoring period of 30 years is considered while monitoring is assumed to be carried out each quarter of a year.

5.4 Discussion of results

The decision model *MONIDAM* evaluated 171 potential monitoring system alternatives to determine the best monitoring system, which enables the highest detection probability while minimizing the cost of monitoring and remediation with use of the least number of monitoring wells under conditions of pertinent uncertainties. The analysis results of the three principal steps of *MONIDAM*, namely Monte Carlo simulation, economical analysis of the objective function for monitoring systems, the selection of the optimal monitoring system, plus sensitivity to model parameters are discussed in the following subsections.

5.4.1 Monte Carlo simulation results

Reliability evaluation

The reliability of a groundwater monitoring system is measured by the probability of detection P_d. Figure 5.3 presents the reliability of monitoring systems at different distances from the contaminant source. Each data point represents one monitoring system evaluated. The reliability of monitoring systems increases with distance from the contaminant source. Since plumes begin with a small size and spread out as they migrate away from the source, detection of a plume is difficult at close distances and becomes easier as the plume expands with time and distance from the contaminant source. Hence there is a greater chance of detecting plumes for the systems composed

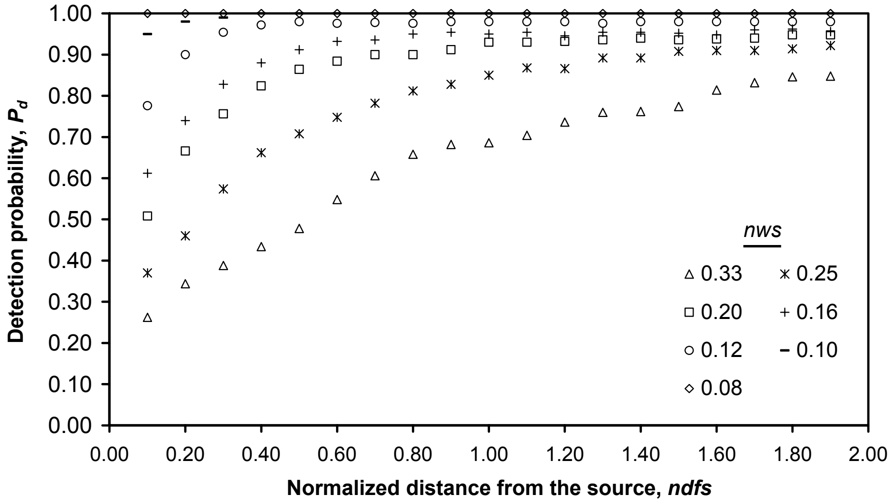


Figure 5.3: Detection probability, P_d as a function of normalized distance from the source *ndfs* for single row monitoring systems with different normalized well spacing *nws.*

of few wells, when they are placed away from the source (i.e. *nws*>0.25 and *ndfs*<0.5). Moreover, for a given *ndfs*, the probability of detection increases when *nws* decreases. However the reliability of the monitoring system is 100 % regardless of the *ndfs* for the systems with *nws* less than 0.08. Additional wells would not be cost effec tive for improving system reliability in such cases. The analysis also showed that for a monitoring system with *nws* greater than 0.08 there is a *ndfs* at which 100% reliability is achieved. For instance for a monitoring system with *nws* equal to 0.10 this point occurs at *ndfs* equal to 0.30. However, for the site dimensions and distance from the source analyzed the common practice of 3-well monitoring system does never reach 100% reliability.

Area of the contaminant plume

As mentioned in Section 5.2.1, the area of the contaminant plume is treated as a random variable and modelled using mean values $E(A_d)$ or $E(A_f)$ and standard deviations $\sigma(A_d)$ or $\sigma(A_f)$ determined from the Monte Carlo simulations. The expected area of the contaminant plume may generally indicate the extent of contamination The contaminated area values estimated during the groundwater transport simulation were normalized by dividing the actual areas (as measured in square meters) by 10000 m^2 simply to render all (normalized) areas less than one. This is just a matter of convenience in presenting the plots.

Figure 5.4 shows a plot of the expected area of contamination given detection $E(A_d)$ as a function of *ndfs* for different *nws* values. For a given *nws* value, $E(A_d)$ increases significantly as the distance from the contaminant source increases. Hence, to minimize the contaminated area, the most effective system is the one located very close to contaminant source. On the other hand, $E(A_d)$ does not increase appreciably and stays more or less constant as the *nws* increase, since *nws* is not a parameter that has effect on spreading and consequently the size of the plume but *ndfs* is a parameter that has influence on the contaminated area in terms of the distance where the

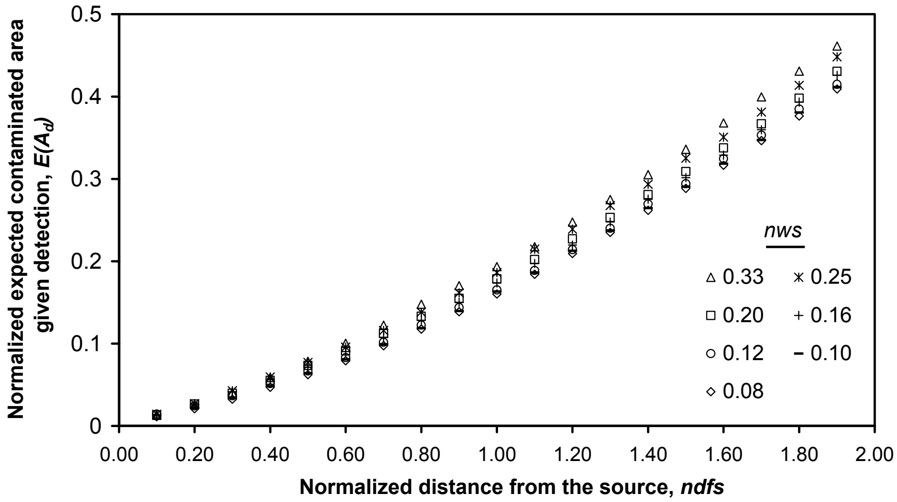


Figure 5.4: Normalized expected contaminated area given detection *$E(A_d)$* as a function of normalized distance from the source *ndfs* for single row monitoring systems with different normalized well spacing *nws.*

plume is detected. Given that A_f is defined as the contaminated area estimated at the end of the monitoring period (in this case 30 years) neither *ndfs* nor *nws* has influence on the plume size that is not detected by the given monitoring system. Hence *$E(A_f)$* remains almost constant with respect to well spacing and distance from the contaminant source.

The distance between the monitoring systems and the contaminant source has influence on the variability of *$E(A_d)$*. Figure 5.5 shows the coefficient of variation of expected contaminated area given detection $CV_{A(d)} = \sigma(A_d)/E(A_d)$ as a function of *ndfs* for monitoring systems with different *nws* values. For the systems close to the contaminant source $CV_{A(d)}$ is very high mainly because *$E(A_d)$* is small at the start and standard deviation relative to the mean plume size is high mainly due to the variation in release location.. As *ndfs* increases, the variability in A_d decreases since the detected plumes becomes more uniform as it moves further away from the source. The effect of leak location is not as significant relative to systems placed close to the contaminant source, since at far distances the plume size is so large that detection is basically unaffected by the initial leak location. This trend is the same for systems composed of any number of wells, with $CV_{A(d)}$ levelling off at large distances from the contaminant source.

5.4.2 Results of economic analysis

The economic analysis of the monitoring system alternatives has been performed considering a unit installation and sampling cost of \$20000 per well a unit remediation cost of 5 \$/m^3 (unit cost estimates are based on James and Gorelick, 1994). A unit installation and sampling cost is considered since it is assumed that each monitoring well in a monitoring system is sampled 4 times per year for whole monitoring period of 30 years. Figure 5.6 presents cost values as a function of *ndfs* for a single row

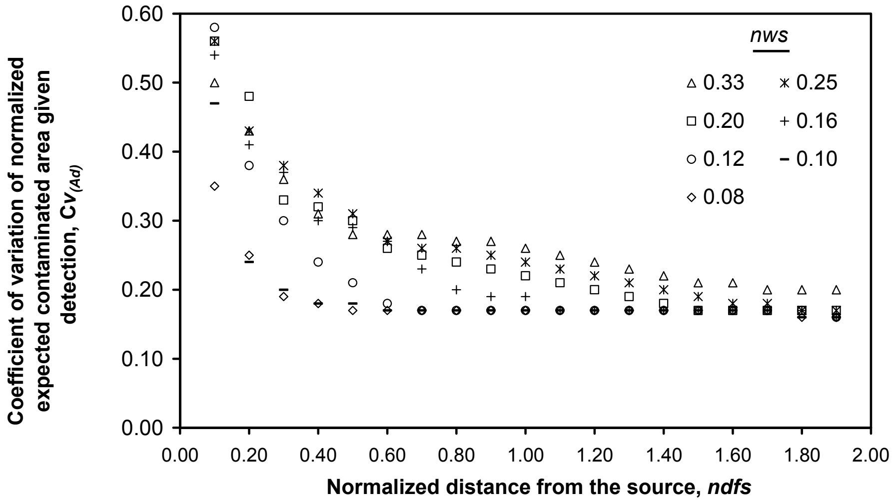


Figure 5.5: Coefficient of variation of expected contaminated area given detection $CV_{A(d)}$ as a function of normalized distance from the source *ndfs* for single row monitoring systems with different normalized well spacing *nws.*

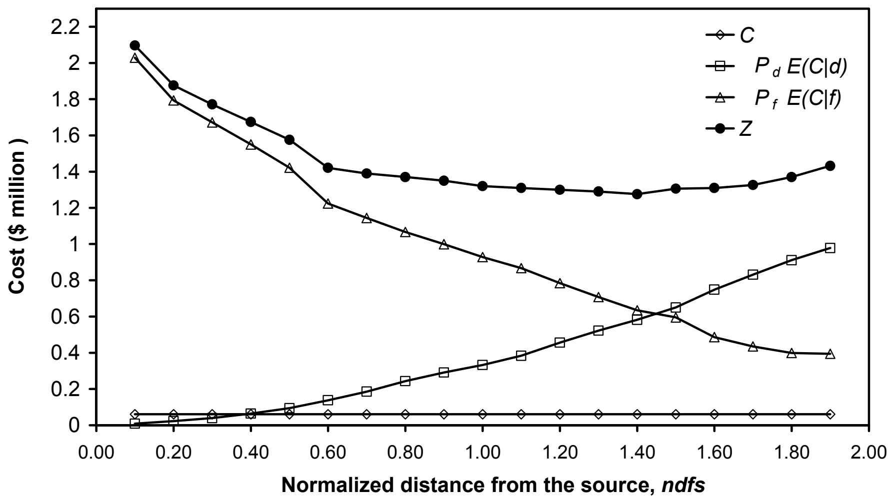


Figure 5.6: Cost values as a function of normalized distance from the source *ndfs* for a single row monitoring system with 3 wells (*nws*=0.33).

system with a *nws* of 0.33. The curves characterized by $P_d\,\mathrm{E}(C|d)$ and $P_f\,E\,(C|f)$ represent the associated costs given detection by the monitoring system and cost associated with the failure of the monitoring system, respectively (cf. Equation (5.9))

The associated cost given detection by the monitoring system increases with distance due to the increase in A_d as the plume moves away from the contaminant source. On the other hand cost associated with the failure of the monitoring system shows a decreasing trend, due to the fact that A_f stays almost constant. The line characterized with C represents the cost of construction and operation of monitoring system. Since only a 3 well monitoring system is considered in this plot, C is constant. Ultimately, the curve symbolized by Z represents the expected total cost for the 3-well monitoring system as a function of *ndfs* and is simply the sum of the three previous curves. It shows that there is a point where the cost is minimized. In other words, there is a trade off between the objectives as a function of *ndfs*.

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5.4.4 Selection of the optimal monitoring system

Figure 5.7 presents the expected total cost, Z as a function of *ndfs* for all single row systems considered in this study. The trough shaped curves for systems with *nws* between 0.33 and 0.12 indicate a trade-off between the contaminated area, detection probability, and cost of the monitoring system. Thus, for systems with *nws*=0.33, it is better to locate wells further away from the contaminant source (*ndfs*=1.4) where a large enough detection probability exists and the contaminated area is somewhat limited .The optimal location, which maximizes the detection probability while minimizing the contaminated area, is at an intermediate distance for the monitoring systems with *nws* between 0.25 and 0.12. As also seen from Figure 5.8, the monitoring systems with *nws* less than 0.12 that are located very close to the contaminant source (*ndfs*<0.3) lead to the lowest expected total cost. This is actually because a high detection probability is reached very close to the contaminant source with use of a multitude of wells for monitoring.

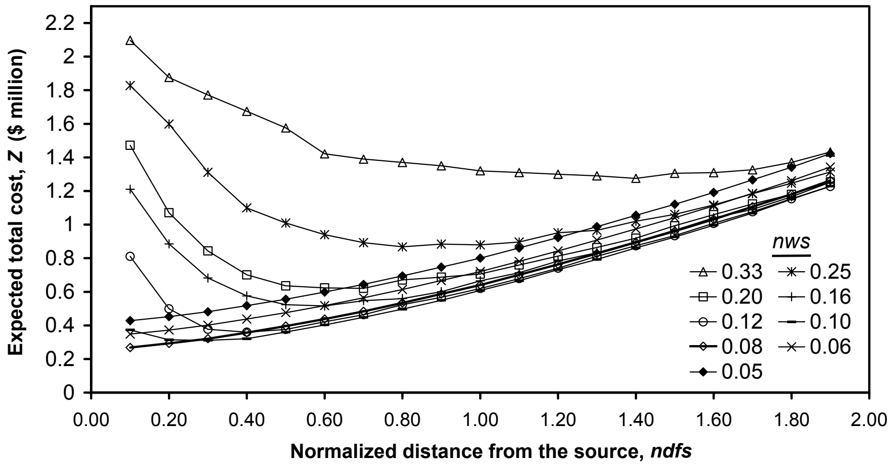

Figure 5.7: Expected total cost, Z as a function of normalized distance from the source *ndfs* for single row monitoring systems with different normalized well spacing *nws.*

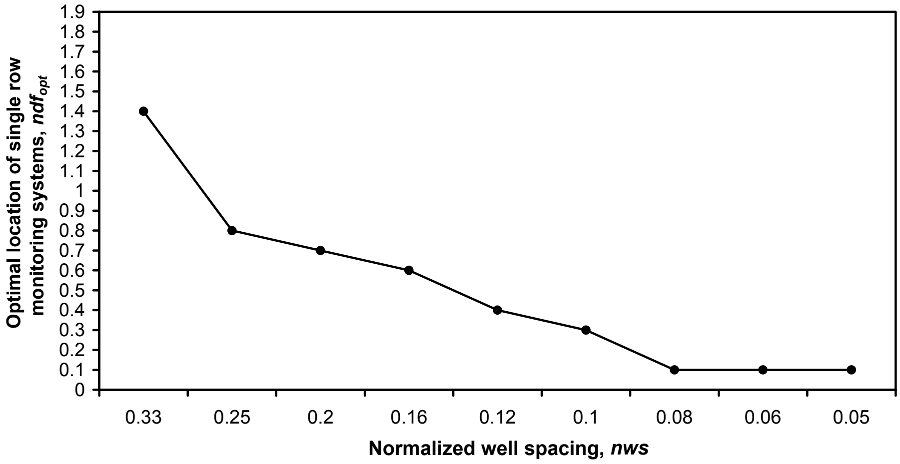

Figure 5.8: Optimal location of single row monitoring systems $ndfs_{opt}$, as a function of normalized well spacing *nws.*

Figure 5.9 shows the minimum expected total cost, Z_{min} as a function of *nws*. For large *nws* values, the minimum expected cost is large, but as the number of monitoring wells increases the expected minimum cost first reaches a minimum, and then increases. This shows that the additional monitoring wells are not cost effective when $nws<0.08$ since they do not contribute to increase the reliability (which is already 100%), and do not reduce the expected area of contamination.

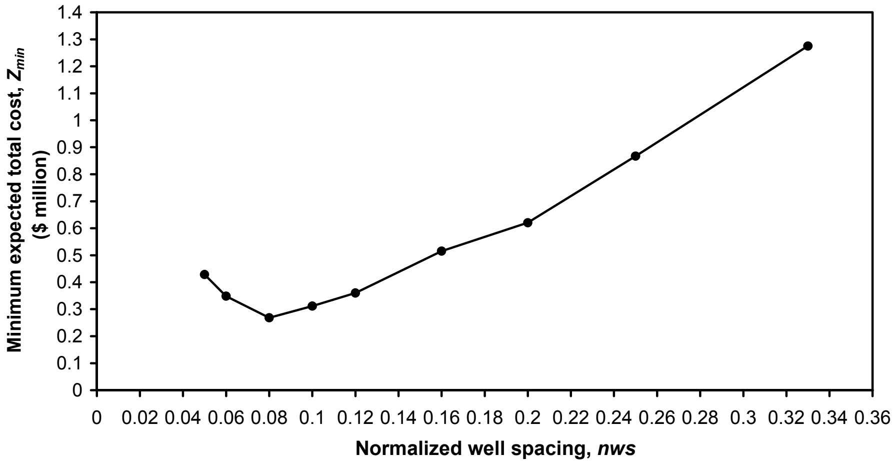

Figure 5.9: Minimum expected total cost as a function of normalized well spacing *nws*.

5.4.5 Sensitivity to model parameters

The objective function values and optimal locations determined by *MONIDAM* are functions of the parameters used in flow and random walk particle tracking model. The results of numerical experiments performed are particularly sensitive to those parameters that have greatest influence on the size and shape of the contaminant plumes. Dispersivity, subsurface heterogeneity, contaminant source size, threshold concentration, sampling frequency, unit installation and sampling cost unit cost and unit remediation cost are generally considered to be the most significant parameters. The influence of these parameters on the detection probability of monitoring systems was studied in the previous Chapter, but for the case of an instantaneous leak. Based on the results of the previous example and the characteristics of the example problem presented here, not only the sensitivity analysis for unit cost and sampling frequency are performed but also the influences of dispersivity and the subsurface heterogeneity on the results are discussed once more in the following sections.

Sensitivity to dispersivity of medium

Dispersivity of the medium is the parameter controlling the spreading of the plume. The longitudinal dispersivity controls the elongation of the plume with time and distance from the contaminant source in the direction of flow, whereas transverse dispersivity dominates the spreading of the plume (width of the plume) in the direction perpendicular to the flow direction. In order to examine the influence of dispersivity α_L is set to values between 0.1 m and 1.2 m. As mentioned earlier, the ratio of α_L to α_T is taken constant at a value of 10. For single row systems, the main consideration in terms of the well spacing is the plume width hence evaluation of P_d and $E(A_d)$ as a function of α_T is presented in Figure 5.10 and Figure 5.11, respectively. As the dispersivity of the medium increases the detection probability of a given monitoring system increases since dispersivity is the parameter that controls the spreading of the plume (see e.g., Meyer et al., 1994; Storck et al., 1997; Yenigul et al., 2005). The higher the

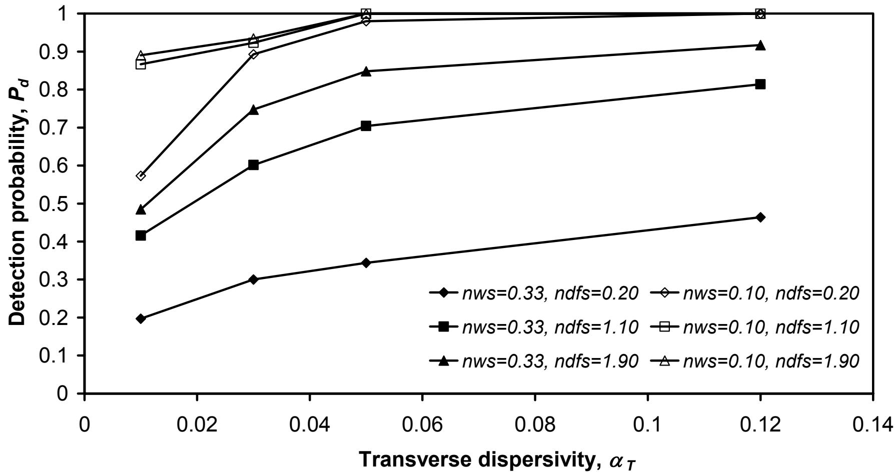

Figure 5.10: Detection probability P_d as a function of transverse dispersivity α_T, for selected single row monitoring systems.

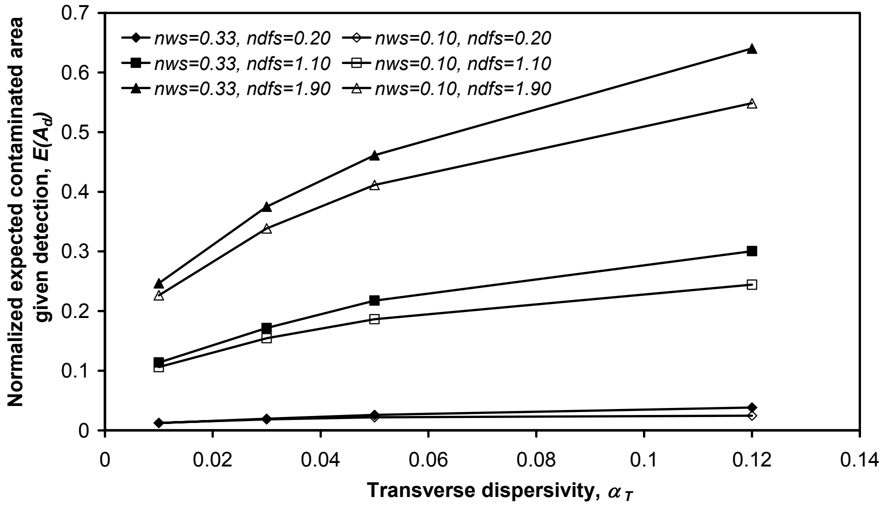

Figure 5.11: Normalized expected contaminated area given detection $E(A_d)$as a function of transverse dispersivity α_T, for selected single row monitoring systems.

dispersivity of the medium the wider the plume gets as it moves further away from the source. Therefore, likewise $E(A_d)$ increases significantly as the dispersivity of medium increases since the plume becomes wider as it moves away from the source.

The results are representative for other monitoring systems evaluated in this study. Furthermore the influence of threshold concentration (detection limit) and initial source size will be similar to the effect of dispersivity on both P_d and $E(A_d)$ as both parameters are directly in relation with the width of the plume. The effective width of the plume increases when the threshold concentration (detection limit) is reduced. Likewise the larger the initial leak size is, the wider the plume will be. Consequently P_d and $E(A_d)$ will increase in both cases (See Yenigul et al., 2005).

Table 5.2 presents the minimum expected total cost Z_{min} and the optimal location of a single row monitoring system $ndfs_{opt}$, as a function of α_T. For a given monitoring system $ndfs_{opt}$ decreases as α_T increases due to the fact that the plumes are wider in highly dispersive media and hence the number of the wells in the optimal monitoring system also decreases. For instance the optimal *nws* of 0.05 for α_T.=.0.01 m goes up to 0.1 when α_T.=.0.12 m. On the other hand the expected total cost for a given system increases as α_T increases because $E(A_d)$ increases. The total expected cost for the optimal monitoring system in the least dispersive medium is 1.5 times higher than that of in the most dispersive medium considered in this study.

Table 5.2: The minimum expected total cost Z_{min} *(million Euro)*, and the optimal location of a single row monitoring systems $ndfs_{opt}$, as a function of transverse dispersivity α_T.

Number of wells	**Transverse dispersivity, α_T**							
	0.01		0.03		0.05		0.12	
	Z_{min}	$nfds_{opt}$	Z_{min}	$nfds_{opt}$	Z_{min}	$nfds_{opt}$	Z_{min}	$nfds_{opt}$
3	1.283	1.1	1.169	1.1	1.275	1.4	1.680	1.4
4	1.242	1.1	0.931	0.8	0.867	0.8	1.606	0.8
5	0.980	1.1	0.755	0.8	0.621	0.7	0.810	0.5
6	0.988	1.1	0.618	0.7	0.515	0.6	0.631	0.2
8	0.770	1.1	0.454	0.6	0.360	0.4	0.448	0.1
10	0.603	0.5	0.479	0.3	0.312	0.3	**0.278**	**0.1**
12	0.591	0.5	**0.337**	**0.3**	**0.268**	**0.1**	0.313	0.1
16	0.492	0.5	0.343	0.1	0.348	0.1	0.393	0.1
20	**0.429**	**0.2**	0.424	0.1	0.428	0.1	0.473	0.1

Sensitivity to subsurface heterogeneity

Subsurface heterogeneity, represented here by the spatial variability of the hydraulic conductivity, is one of the important factors controlling the migration of contaminants in porous media. The variance σ_Y^2 of Y is the parameter that characterizes the degree of heterogeneity of the subsurface. A high variance will produce a highly heterogeneous field with hydraulic conductivity values spanning a wide range, while a low variance will produce a more homogeneous-like field. To investigate the influence of heterogeneity σ_Y^2 is set to 0, 0.4, 0.6, 1.0 and 1.2. Figure 5.12 and Figure 5.13 show P_d and $E(A_d)$ as a function of σ_Y^2, respectively. P_d of the monitoring system decreases as σ_Y^2 increases. This is due to the fact that the contaminant plumes are more likely to become irregularly shaped in heterogeneous media and may go undetected easier due to variability in the flow field, while in a homogeneous medium the plumes have much more uniform shapes and tend to travel in a direction parallel to the average gradient. The maximum difference in detection probability over the range of σ_Y^2 considered is about 20%. Figure 5.13 shows that $E(A_d)$ tends to increase as σ_Y^2 increases. The difference between the homogenous and the heterogeneous case is more pronounceable when the monitoring systems are located further away from the contaminant source, because when the plumes are detected close to the source the plumes do not encounter much of the heterogeneous structure of the hydraulic conductivity as they have less chance to move a distance larger than a few correlation lengths. The results are representative for other monitoring systems evaluated in this study.

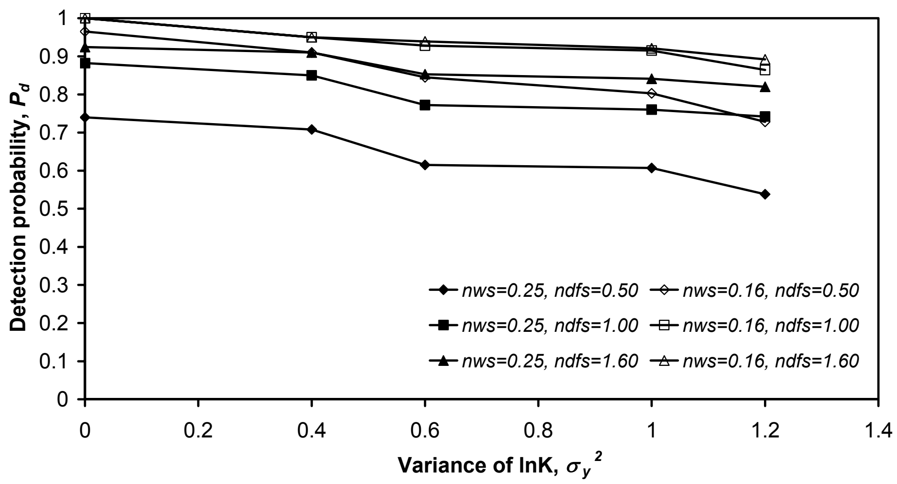

Figure 5.12: Detection probability, P_d as a function of variance σ_Y^2 of Y for selected single row monitoring systems.

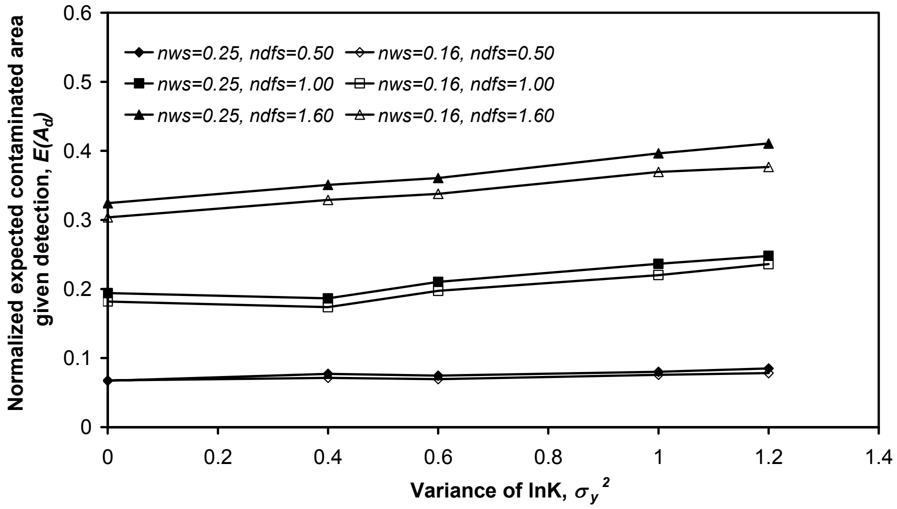

Figure 5.13: Normalized expected contaminated area given detection, $E(A_d)$ as a function of variance σ_Y^2 of Y for selected single row monitoring systems.

Table 5.3 presents the minimum expected total cost Z_{min}, and optimal location of single row monitoring systems $ndfs_{opt}$, as a function of σ_Y^2. For a given monitoring system $ndfs_{opt}$ increases σ_Y^2 due to the fact that P_d of the monitoring system decreases as σ_Y^2 increases. The same effect is also observed in the relation between expected total cost and subsurface heterogeneity, namely the higher the heterogeneity of the medium is, the higher the expected total cost of the given monitoring system. The number of the wells in the optimal monitoring system also increases. For instance, the optimal nws is 0.08 when σ_Y^2 is greater than 0.4.

Table 5.3: The minimum expected total cost Z_{min} *(million Euro)*, and the optimal location of single row monitoring systems $ndfs_{opt}$, as a function of variance σ_Y^2 of *lnK*.

Number of wells	The variance of *lnK*, σ_Y^2									
	0.0		0.4		0.6		1.0		1.2	
	Z_{min}	$ndfs_{opt}$	Z_{min}	$ndfs_{opt}$	Z_{min}	$ndfs_{opt}$	Z_{min}	$ndfs_{opt}$	Z_{min}	$ndfs_{opt}$
3	1.21	1.4	1.275	1.4	1.782	1.4	1.790	1.4	1.815	1.4
4	0.839	0.7	0.867	0.8	1.336	1.4	1.596	1.4	1.671	1.4
5	0.609	0.6	0.621	0.7	0.951	0.8	1.140	0.8	1.184	1.2
6	0.387	0.5	0.515	0.6	0.760	0.8	0.932	0.8	1.178	1.0
8	0.346	0.3	0.360	0.4	0.521	0.5	0.766	0.5	0.865	0.8
10	0.299	0.2	0.312	0.3	0.442	0.4	0.581	0.4	0.655	0.5
12	**0.231**	**0.1**	**0.268**	**0.1**	0.381	0.2	0.446	0.2	0.551	0.2
16	0.238	0.1	0.348	0.1	**0.377**	**0.1**	**0.380**	**0.1**	**0.395**	**0.2**
20	0.319	0.1	0.428	0.1	0.457	0.1	0.460	0.1	0.471	0.1

Sensitivity to unit costs

For the assumed unit installation and sampling cost of \$20000 per well and a unit remediation cost of 5 \$/m^3, the single row monitoring system with *nws*=0.08 gives the minimum expected total cost, and optimal location for such a system occurs at $ndfs_{opt}$=0.1 (See Figure 5.8 and Figure 5.9). Table 5.4 presents the sensitivity of the optimal well spacing and optimal location with respect to changes in the assumed unit installation and sampling cost plus unit remediation cost. The bold phase figures in the table indicate the optimal well spacing and optimal location for the initial assumption of unit cost values.

When unit installation and sampling cost is quite high (i.e. 50000 \$/well) for a unit remediation cost of 5 \$/m^3 the optimal well spacing shifts to 0.12. Fewer monitoring wells should be used to limit the expected total cost and the optimal location of the monitoring system is further from the source, which means that the objective of minimizing the contaminated area can be rather loose (moving to the right of the bold phase figures). On the other hand for lower values of unit installation and sampling cost than assumed neither the optimal *nws* nor the $ndfs_{opt}$ do change. However, for very cheap remediation cost (e.g., 1 \$/m^3) the optimal nws increases as the unit installation and sampling cost increases up to 30000 \$/well and remains the same for 40000 \$/well and 50000 \$/well. Likewise the monitoring wells can be located further away from the contaminant source.

If contamination can be cleaned up easier than assumed in other words if the unit remediation cost is less than the assumed value (i.e. 1 \$/m^3 and 2 \$/m^3) the *nws* shifts to 0.12 and 0.16 respectively, while the $ndfs_{opt}$ values correspond to 0.4 and 0.6. The increase in *nws* can be explained as follows. Cheap remediation allows larger plumes, which allows later detection, which leads to less wells. At the same time if the remediation of the contaminated area is much difficult than assumed due to the nature and concentration of the contaminants (moving down from the bold face figures in Table 5.4) the optimal *nws* remains 0.08 and $ndfs_{opt}$ is kept at 0.1. This means that limiting the extent of contamination to control the costs associated with remediation is still very important.

Table 5.4: Optimal well spacing and optimal location as function of unit remediation cost and unit installation and sampling cost.

Unit remediation cost ($/m^3)	Unit installation and sampling cost ($/well)											
	5000		10000		20000		30000		40000		50000	
	nws	$ndfs_{opt}$	nws	$ndfs_{opt}$	nws	$ndfs_{opt}$	nws	$ndfs_{opt}$	nws	$ndfs_{opt}$	nws	$ndfs_{opt}$
1	0.08	0.1	0.12	0.4	0.16	0.6	0.20	0.7	0.20	0.7	0.20	0.7
2	0.08	0.1	0.08	0.1	0.12	0.4	0.12	0.4	0.16	0.6	0.16	0.6
3	0.08	0.1	0.08	0.1	0.08	0.1	0.12	0.4	0.12	0.4	0.12	0.4
4	0.08	0.1	0.08	0.1	0.08	0.1	0.08	0.1	0.12	0.4	0.12	0.4
5	0.08	0.1	0.08	0.1	**0.08**	**0.1**	0.08	0.1	0.08	0.1	0.12	0.4
6	0.08	0.1	0.08	0.1	0.08	0.1	0.08	0.1	0.08	0.1	0.10	0.3
7	0.08	0.1	0.08	0.1	0.08	0.1	0.08	0.1	0.08	0.1	0.08	0.1
8	0.08	0.1	0.08	0.1	0.08	0.1	0.08	0.1	0.08	0.1	0.08	0.1
9	0.08	0.1	0.08	0.1	0.08	0.1	0.08	0.1	0.08	0.1	0.08	0.1
10	0.08	0.1	0.08	0.1	0.08	0.1	0.08	0.1	0.08	0.1	0.08	0.1

Sensitivity to sampling frequency

Current monitoring program suggested by regulatory agencies (i.e. USEPA, 1986 and ECC, 1999) requires the monitoring of groundwater quarterly, biannually or annually depending on the type of waste, size and design of landfill and aquifer material for 30 years of post closure monitoring duration. In most cases a quarterly monitoring is undertaken; annual monitoring is undertaken mostly for small landfills located in remote places far away from any groundwater use. As mentioned above in this study the most common sampling frequency, namely a quarterly monitoring is considered. However, to determine the influence of sampling frequencies numerical experiments were also carried out for biannually, annually, monthly, and three times a year sampling. Since the leak is assumed to be continuous, sampling frequency will not affect the detection probability of a monitoring system. Because a contaminant plume will sooner or later be detected as long as the single row monitoring system is located such that it coincides with the travel path of the contaminant plume. However one may expect that the main concern, with regard to sampling frequency, is early detection, in other words the extent of the contaminated area. The later the contaminant plume is detected by a given monitoring system the larger the expected contaminated area required to be remediated. The results of the analysis confirmed these arguments. The P_d of all monitoring systems remain the same regardless of sampling frequency, while the $E(A_d)$ increases as the sampling frequency decreases (See Figure 5.14). Since the increase is not so prominent in the current example, the expected total costs will be close to each other for all sampling frequencies considered here. However, the difference in the expected total cost of monitoring systems with different sampling frequencies will be more significant when the remediation cost is much cheaper compared to monitoring costs (similar to the effect presented in Table 5.4).

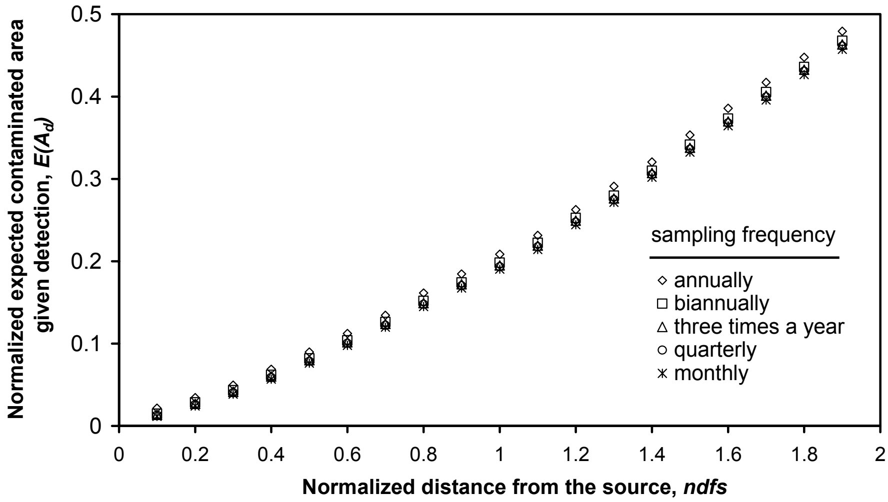

Figure 5.14: Normalized expected contaminated area given detection $E(A_d)$ as a function of normalized distance from the source *ndfs* for a single row monitoring system with 3 wells ($nws = 0.33$) for different sampling frequencies.

5.5 Summary and conclusions

A multi-objective decision analysis model, *MONIDAM* is used to determine the optimal groundwater monitoring system, which maximizes the detection probability while minimizing the contaminated area and, the total cost of the monitoring system under conditions of uncertainties due to subsurface heterogeneity and leak locations. The results of the extensive numerical experiments show that the reliability of monitoring systems increases with distance from the contaminant source. Since plumes begin with a small size and spread out as they migrate away from the source, systems composed of few wells are more likely to detect the contaminant plumes when they are placed away from the contaminant source. For a given distance away from the contaminant source the probability of detection increases as the number of the monitoring wells increase but once 100 % of reliability is achieved by a given monitoring system additional wells would not be cost effective for improving the system reliability. The widely used 3 well monitoring system (minimum regulatory requirement) do not reach 100 % reliability for any of the cases investigated in the presented study.

The contaminated area given detection increases as the distance from the source increases. However the number of the wells used in a monitoring system has no influence on the size of the contaminated area, while the location of the monitoring system is crucial for minimizing the contaminated area. The nearer the monitoring systems to the contaminant source, the smaller the contaminated area will be.

The analysis furthermore demonstrated that the detection probability and the contaminated area increase as the dispersivity of the medium increases since it is the parameter controlling the spreading of the plume. It could give preference to those systems located further away from the contaminant source, as the detection probability is higher at further distances than in a medium with low dispersivity. Also, subsurface heterogeneity is an important parameter that has influence on detection probability and on the extent of the contaminated area. The detection probability decreases while the contaminated area increases with increasing heterogeneity. This is due to the fact that the contaminant plumes are more likely to become irregularly shaped in heterogeneous media, and they may go undetected easier due to variability in the flow field, while in homogeneous medium the plumes have much more uniform shapes and tend to travel in a direction parallel to average gradient.

The results also showed that the optimal location of the monitoring systems would be very close to the contaminant source ($ndfs_{opt} = 0.1$). A large number of wells should be considered (i.e. 12 wells) for optimal monitoring systems except for the cases where the unit installation and monitoring cost are very high and/or the unit remediation cost is very cheap. Furthermore, the widely used 3 well monitoring system (minimum regulatory requirement) is not a good solution for any of the cases presented above. Lastly, sampling frequency has an effect on the extent of contaminated area as well. As the sampling frequency increases the contaminated area decreases. However, the reflection of this effect on the expected total cost is prominent again if the installation and monitoring cost are very high and/or the remediation cost is very low.

Chapter 6

New Strategy to Improve the Efficiency of Monitoring Systems

Adapted from Yenigul, N.B., Elfeki, A.M.M., and van den Akker, C., New approach for groundwater detection monitoring at lined landfills, Groundwater Monitoring and Remediation, 26, 79-86, 2006.

The results of the analysis presented in the previous chapters showed that widely applied 3-well monitoring system (minimum regulatory requirement) is more often inadequate to accomplish groundwater detection monitoring at landfill sites. The limited capture zone of monitoring wells is mostly the key factor. This chapter presents implementation of a new monitoring approach to design a highly efficient cost-effective 3-well system. Following section describes the new approach monitoring approach. Then the application of new proposed monitoring approach (*PMA*) to a hypothetical example is presented. Finally, the results of analysis are evaluated and chapter ends with particular outcome regarding the improvement of efficiency of monitoring systems when the *PMA* is considered.

6.1 Introduction

The purpose of detection monitoring is early detection of a release to groundwater, should one occur, based on comparison of downgradient well data to background data for a limited number of water quality parameters. As mentioned in Chapter 1, for detection monitoring, regulations require at least one background well (hydraulically upgradient from a potential source) and three downgradient wells (USEPA, 1986 and ECC, 1999). The position, number (more than the minimum requirement), and depth of the monitoring wells are proposed by the landfill owners or operators and by local authorities. Conventional monitoring program suggested by regulatory agencies requires the monitoring of groundwater quarterly, biannually or annually depending on the type of waste, size and design of landfill and aquifer material for 30 years of post closure monitoring duration. In most cases a quarterly monitoring is undertaken; annual monitoring is undertaken mostly for small landfills located in remote places far away from any groundwater use source. There is no recognition of uncertainty in regulations conversely to reality.

In previous studies, it has been observed that mostly more than 3-wells are required for an optimal system that will enable satisfactory high detection probability while minimizing the expected contaminated area of the aquifer and/or the total cost of the system (e.g., Meyer et al., 1994; Storck et al., 1997; Hudak, 2001; Yenigul et al., 2005;

Yenigul et al., 2006a). In the analysis presented in the following sections, maximizing the likelihood of detecting contaminants, while minimizing the contaminated area are the conflicting design objectives considered like some of previous studies. However, this study is a pioneer to propose a new groundwater monitoring approach, namely pumping continuously from the monitoring well(s) with a quite low discharge rate, to improve the efficiency of the widely applied common practice of three downgradient wells (the minimum regulatory requirement).

In conventional monitoring approach (*CMA*), by using a bailer or a pump, the water level of each monitoring well is purged by removing four well volumes (internal radius of the well × the height of water column in the well) of water before taking a sample to monitor the groundwater quality. Although it is technically feasible to maximize the likelihood of detecting contaminant plumes with such *CMA*, the cost might be quite high to be practical due to the necessity of a large number of wells to implement the desired high performance particularly at lined landfill sites. Particularly, the initial leakage from a lined landfill will be from point sources such as holes, tears, and imperfections in the liner systems. These point sources will produce groundwater plumes that move as fairly narrow especially in low dispersive medium, where the lateral spreading is relatively limited in the distance between the landfill and the point of compliance for groundwater monitoring. Moreover, in practice the budget constraint allows the use of very limited number of wells in monitoring systems and more often the monitoring systems are composed of 3 downgradient wells, which fulfils the minimum regulatory requirements. Therefore considering the nature of incipient leakage from the lined landfills and the limited capture zone of conventional monitoring wells, which are widely spaced it is very likely that current practice of groundwater monitoring systems at lined landfills instils a false sense of security and is of little use in protecting groundwater resources from pollution by leachates.

However, in the new proposed monitoring approach (*PMA*) the capture zone of monitoring wells broadens due to the pumping from the wells. Therefore the chance of intercepting more contaminated plumes at early times is expected to increase. Thus, the main objective of this study is to evaluate the influence of the new *PMA* on the efficiency of common monitoring systems and to compare the analysis results with those of the conventional monitoring approach in terms of detection probability, average contaminated area and cost, while incorporating uncertainties due to hydrogeological and contaminant source characteristics.

The detection probability P_d, and the expected average contaminated area, $E(A_{av})$, are estimated the simulation model. $E(A_{av})$, is equal to the mean value of expected plume sizes (either detected or when the plume remains undetected at the end of the monitoring period undetected during the monitoring period) estimated from the Monte Carlo simulation data.

6.2 Hypothetical problem

6.2.1 Model domain and discretization

The numerical experiments are carried out using a model of a generic landfill facility and groundwater system however the dimensions of the model domain and the parameters which have been used, are chosen to reflect conditions of typical solid waste landfills. The model domain is defined by $0 \leq x \leq 500$ m and $0 \leq y \leq 400$ m (Figure 6.1). The model is discretized with grid cells of 2 m by 2 m in both x- and y-directions. A hypothetical lined landfill is located at $20 \leq x \leq 50$ m and $140 \leq y \leq 260$ m in the model domain. For *CMA* 48 potential single row monitoring system alternatives consisting of, 3, 4, 5, 6, 8 and 12 wells respectively are located from 20 m to 220 m downgradient of the landfill with 30 m distance apart. For the *PMA*, 8 monitoring systems of 3-wells are located also from 20 m to 220 m downgradient of the landfill with 30 m distance apart. For each alternative, continuous pumping from one well (the well in the middle) is considered. Under the condition of convergence in terms of the number of the Monte Carlo simulations and number of the particles used in the particle tracking model the choice of the well, which pumping occurs, would have negligible even no impact on detection performance of single row monitoring systems as long as the monitoring wells are evenly spaced and located at a distance equals to half of the well spacing from top and bottom boundaries of the landfill in order to prevent the boundary effect on detection probability of monitoring wells. Because the wells located at the boundaries will be limited to plumes originating from the leaks at the boundaries or at distances that are very close to the boundaries and this may result in inaccurate detection probability calculations for these wells.

Four different pumping rates, namely 25 l/day, 50 l/day, 100 l/day, and 125 l/day have been used to investigate the influence of pumping rate on the efficiency and the optimality of the monitoring systems. The monitoring system alternatives considered are composed of a line of wells parallel to the y-axis, extending the length of the source of contamination.

6.2.2 Parameter values used in flow model

The boundary conditions for the groundwater flow are zero flux at $y = 0$ m (bottom boundary) and $y = 400$ m (top boundary) and constant head along the left and the right boundaries. The head values at x = 0 m and x = 500 m were chosen to result in a macroscopically constant hydraulic gradient of 0.001. Porosity equals to 0.25. The natural logarithm of the isotropic hydraulic conductivity [$Y = ln(K)$] is modelled as a stationary Gaussian field. A geometric mean hydraulic conductivity of 2.23 m/day is considered and the variance of Y is set at $\sigma_Y^2 = 0$ for homogeneous and at $\sigma_Y^2 = 0.5$

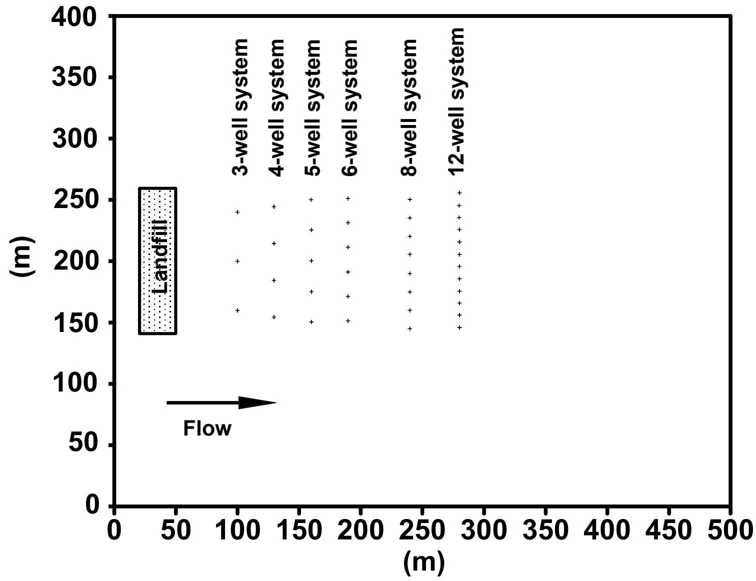

Figure 6.1: Plan view of the hypothetical problem used in numerical experiments.

for heterogeneous medium. The isotropic covariance of Y is chosen to be of exponential form with a correlation length λ=15 m.

6.2.3 Parameter values used in random walk particle tracking model

For the transport model a condition of a zero dispersive flux is imposed on the top and bottom boundary, and the initial background concentration in the model domain is set to zero. Since the flow direction is aligned with x-axis, the only source dimension that is treated, as a random variable is the position long the y-axis. The area of potential leak locations is the downgradient edge of the landfill (Figure 6.1). The contaminant leak is assumed to be a point source, as it would result a plume, which is most difficult to detect and the source location is drawn from a uniform probability distribution between y-coordinates of $140 \leq y \leq 260$ m for each Monte Carlo run. Dispersion is incorporated in the model by introducing micro scale longitudinal (α_L) and transverse (α_T) dispersivities.

The ratio between α_L and α_T is assumed to be 10 (Bear, 1972). α_L is set to 0.1 m and 0.3 m, respectively to examine the influence of dispersivity. The simulation procedure assumes that the source is continuous and provides a constant mass rate of 1 mg/l/day. The threshold concentration (detection limit) at which detection occurs is set at 0.5% of the initial source concentration. Cadmium, benzene, carbon tetrachloride and 1,1,2-Tricloroethane are the most common contaminant types released to groundwater mainly via leaks from landfills and EPA (USEPA, 1986) defines the maximum contaminant level of 0.005 mg/l for these contaminants in drinking water. Hence considering a 1 mg/l initial source of one of these contaminants, the threshold concentration corresponds to 0.005 mg/l, which is equal to EPA public health risk level for drinking water. Contaminants are assumed to be conservative and to be completely mixed over the depth of the aquifer, which is presumed to be 50 m in the example problem. A monitoring period of 30 years is considered while monitoring is assumed to be carried out each quarter of a year.

6.3 Discussion of results

To find answers to the questions “what is the best monitoring system that optimally meets the desired goals namely, maximizing the likelihood of detecting contaminants while minimizing the contaminated area under pertinent uncertainties?” and “how to improve the efficiency of widely applied common practice that fulfils the minimum regulatory requirement, namely a 3-well system?” and consequently in order to solve the detection monitoring problem, 1000 Monte Carlo simulations are carried out for different subsurface conditions considering both conventional and proposed monitoring approaches. The results of analysis are discussed in the following subsections.

6.3.1 Conventional Monitoring Approach (CMA)

Reliability evaluation

The reliability of a groundwater monitoring system is measured by the probability of detection P_d. Figure 6.2 presents the reliability of monitoring systems at different distances from the source for homogenous and heterogeneous media. Each data point represents one monitoring system evaluated based on the conventional monitoring approach. Both in homogeneous and heterogeneous media, there is an increase in reliability with distance from the source. Since plumes begin with a small size and spread out as they migrate away from the source, detection of a plume is difficult at close distances and becomes easier as the plume expands with time and distance from the source. Hence there is a greater chance of detecting plumes for the systems composed of few wells, when they are placed way from the source. Moreover, as the dispersivity of the medium increases the detection probability of a given monitoring system increases since dispersivity is the parameter that controls the spreading of the plume (see e.g. Meyer et al., 1994; Storck et al.,1997; Yenigul et al., 2005; Yenigul et al, 2006a). The higher the dispersivity of the medium the wider the plume gets as it moves further away from the source. On the other hand, P_d of the monitoring systems are lower in heterogeneous medium since it will be more difficult to detect the contaminant plumes due to their irregular shapes and the uncertainty in the direction that they travel. P_d of a monitoring system at a given distance increases as the number of wells in the system increases. However it is important to note that additional wells would not be cost-effective for improving detection particularly in a case where the reliability of the systems is 100% regardless of the distance away from the sources for a certain number of wells. However, for the site dimensions and distance from the source analyzed the common practice of 3-well does not reach 100% reliability.

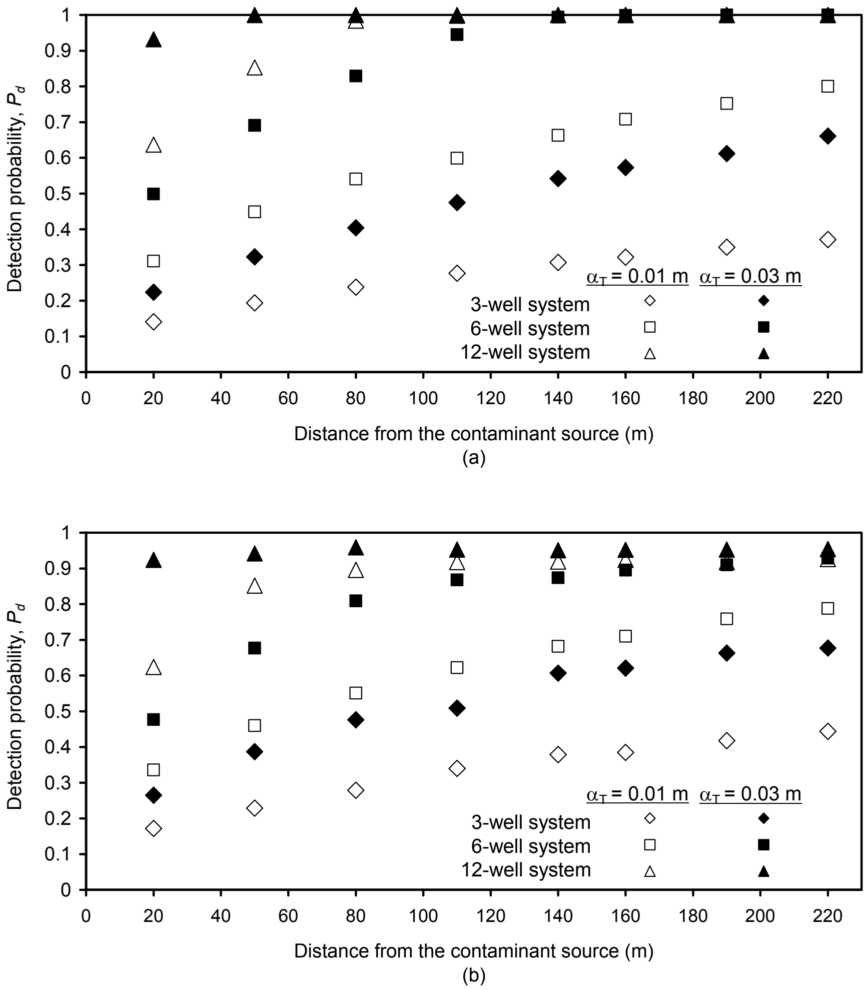

Figure 6.2: System reliability as a function of distance from the source for selected monitoring systems for conventional monitoring approach (*CMA*): (a) homogenous medium, and (b) heterogeneous medium.

Average contaminated area

When a given system detects a contaminant plume, an associated contaminated area is obtained (A_d). When the monitoring system fails to detect the contaminant plume, the contaminated area at the end of the monitoring period (in this case 30 years) is estimated, (A_f). However, the plume size, either detected or not, varies from one realization to another due to the variability in the source location and hydrogeologic characteristics. Therefore rather than producing a single plume size a range of plume sizes determined for each monitoring system and the expected average contaminated area, $E(A_{av})$, is equal to the mean value of expected plume sizes (either detected, A_d or not, A_f) estimated from the Monte Carlo realizations. Both the detection probability of a system and the contaminant area at detection increases as the distance from the source increases whereas the contaminated area associated with no detection remains basically constant with respect to distance from the source.

Therefore, the difference between the size of detected and undetected plumes reduces as the contaminant plumes move away from the source. This will result the curves presented in Figure 6.3. $E(A_{av})$, decreases towards a minimum value up to certain distance and then again increases as distance from the source increases. The distance where the minimum contaminated area occurs is the optimal location where a given monitoring system has to be placed in order to maximize the detection probability while minimizing the contaminated area. $E(A_{av})$ increases as the dispersivity of medium increases. As it was mentioned above this is due to fact that transverse dispersivity is the controlling factor for the spreading and widening of the contaminant plumes. On the other hand, $E(A_{av})$ is larger in heterogeneous medium since it is signified by the plumes those are irregular in shape and are more likely to go undetected due to variability in the flow field. Furthermore, the average contaminated area decreases as the number of wells in the monitoring system increases. This is because the detection probability increases as the number of wells in the system increases and consequently $E(A_{av})$ decreases since the expected area given no detection is larger than the area given detection. However, when the 100% reliability is achieved by a given monitoring system the addition of wells will not affect the expected extent of contamination as well.

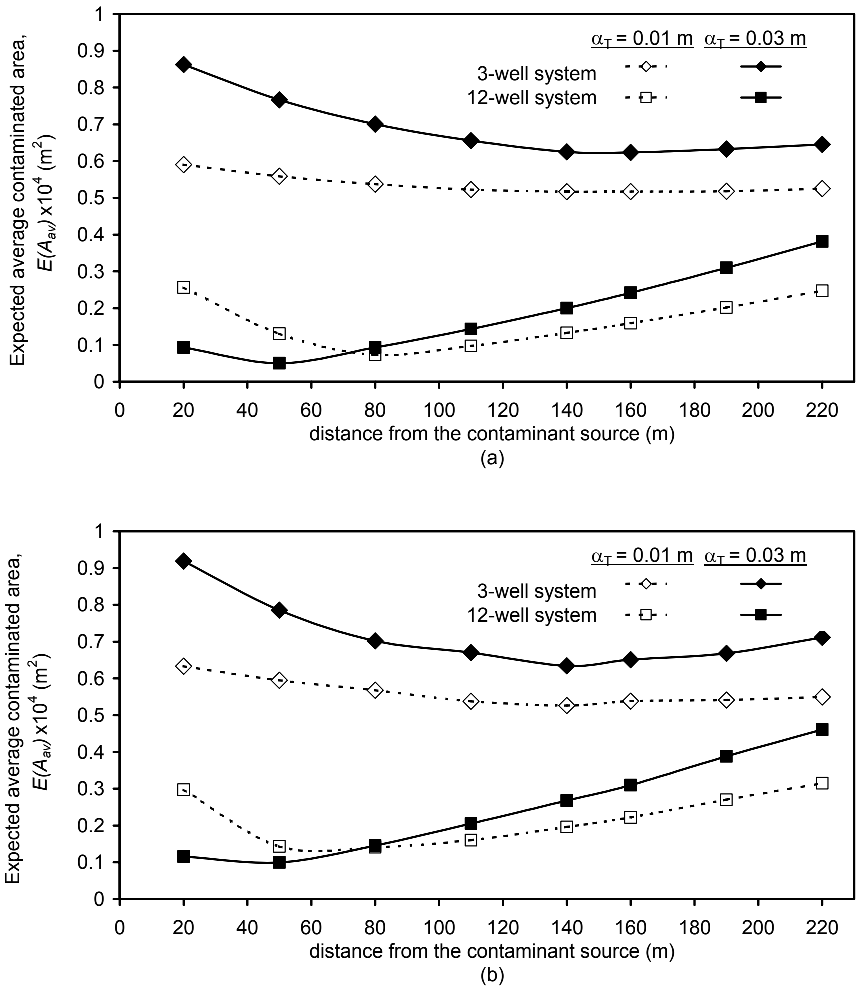

Figure 6.3: Expected average contaminated area $E(A_{av})$ as a function of distance from the source for selected monitoring systems for conventional monitoring approach (CMA):(a) homogenous medium and (b) heterogeneous medium.

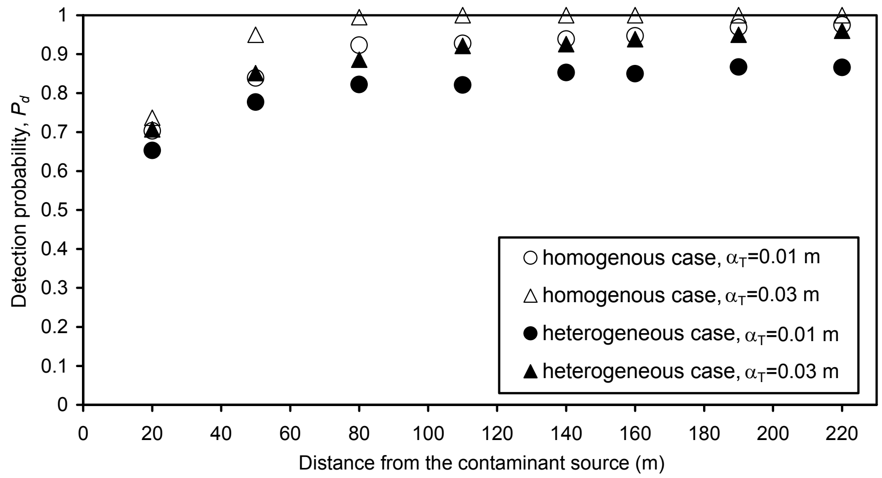

Figure 6.4: System reliability as a function of distance from the source for a 3-well monitoring system for the proposed monitoring approach (*PMA*) (pumping rate is 100 l/day).

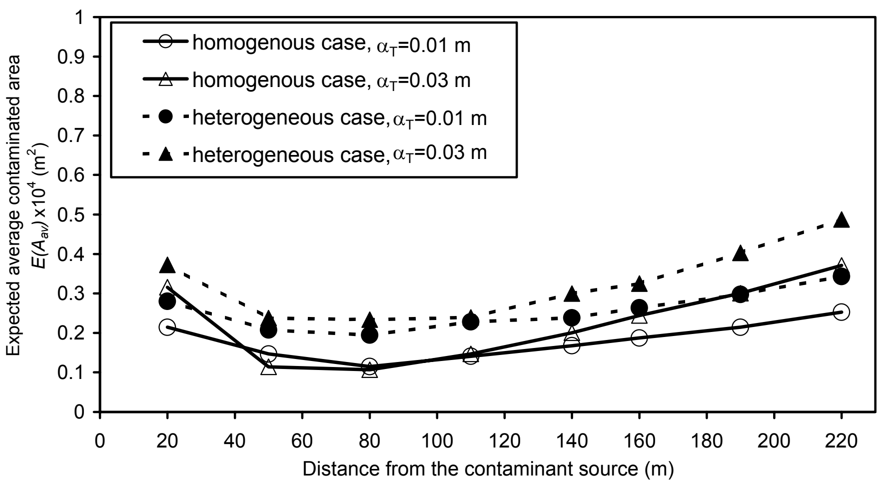

Figure 6.5: Expected average contaminated area $E(A_{av})$ as a function of distance from the source for a 3-well monitoring system for the proposed monitoring approach (*PMA*) (pumping rate is 100 l/day).

6.3.2 Proposed Monitoring Approach (*PMA*)

Reliability evaluation and average contaminated area

The relation between the reliability of monitoring system, P_d, and the distance from the source is the same as it was observed in the results based on the conventional approach (Figure 6.4 and Figure 6.5). Hence, P_d of 3-well monitoring system increases as the distance from the source increases and, $E(A_{av})$ decreases towards a minimum value up to certain distance and then again increases as distance from the source increases. The distance where the minimum contaminated area occurs is the optimal location where a given monitoring system has to be placed in order to maximize P_d

and minimize $E(A_{av})$. The quantitative difference in the results of the analysis based on the two approaches will be presented in detail in the following Section 6.3.3.

Influence of pumping rate

The influence of the pumping rate on the efficiency of a 3-well monitoring system is presented in Figure 6.6. Both in homogenous and heterogeneous media P_d of a 3-well system increases as the pumping rate increases. This is due to the fact that the higher the pumping rate is the broader the capture zone of the monitoring well is. Therefore the chance of intercepting contaminated plumes increases as the pumping rate increases and accordingly $E(A_{av})$ will decrease. However, the increase in the pumping rate will not make any further improvement once 100 % reliability is achieved at a certain pumping rate.

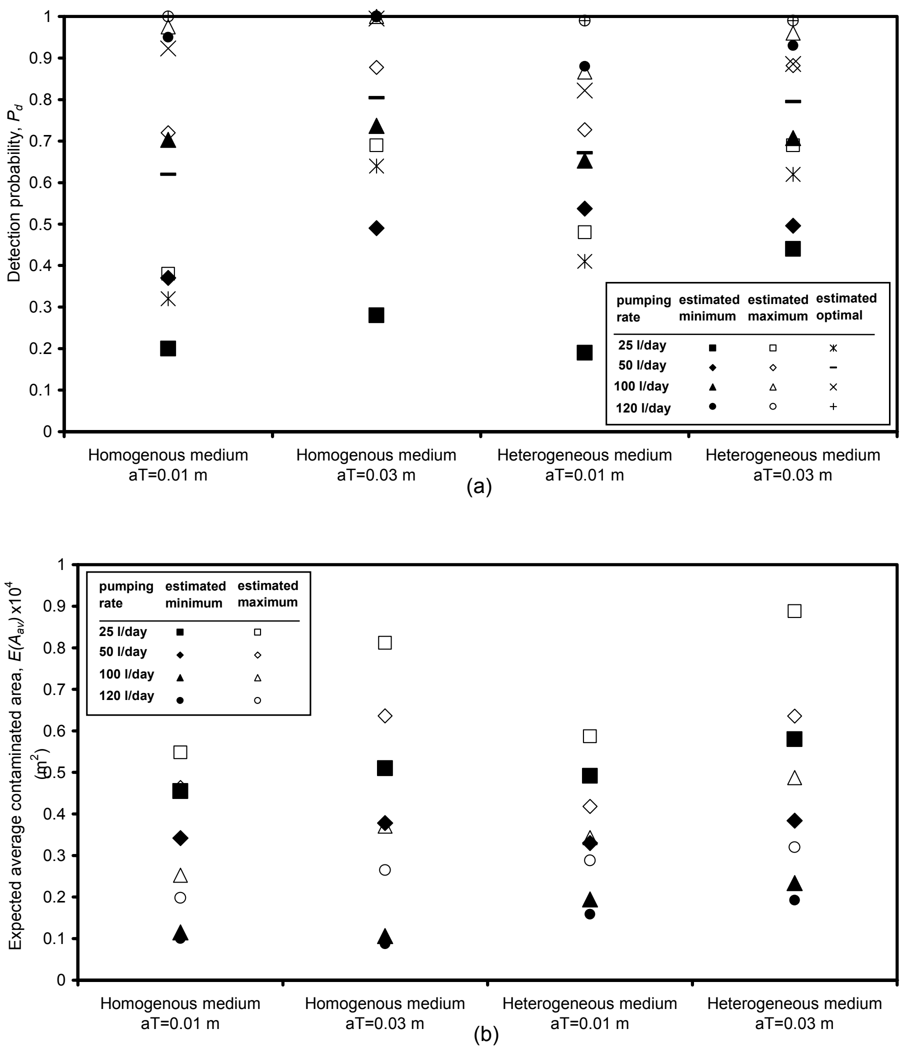

Figure 6.6: Influence of the pumping rate on (a) detection probability P_d of a 3-well system and (b) expected average contaminated area $E(A_{av})$.

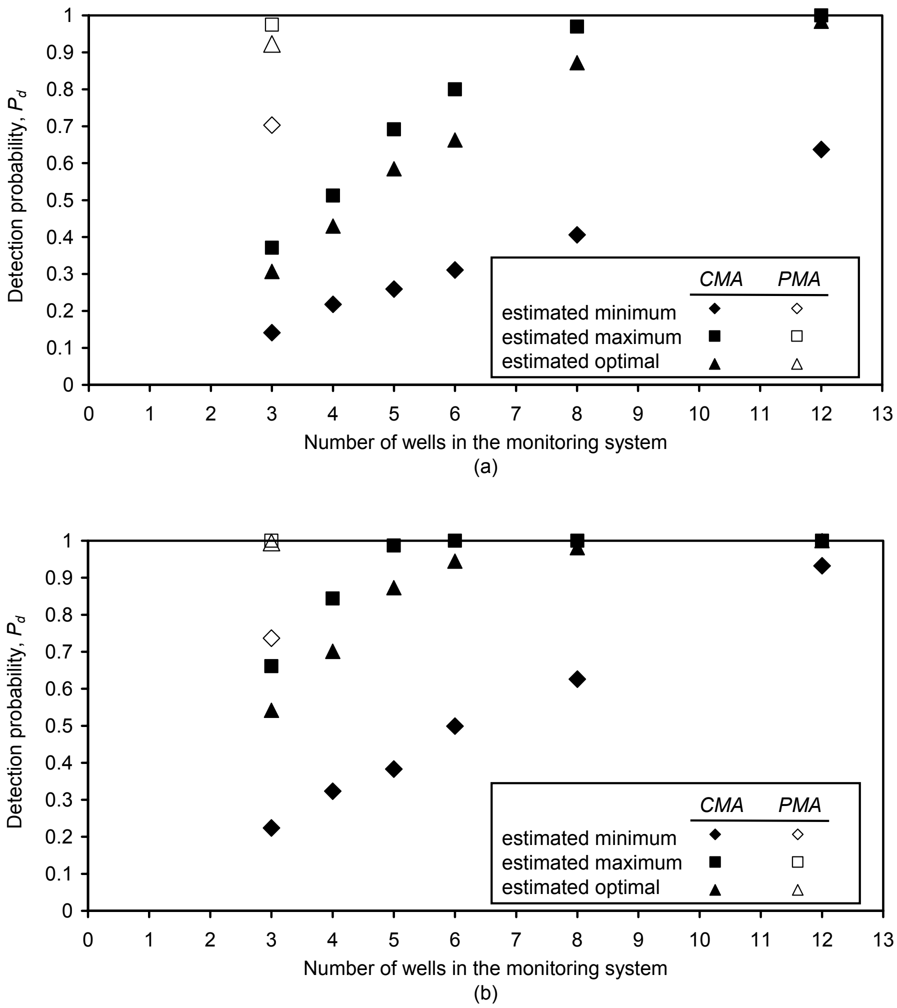

Figure 6.7: Comparison of the conventional monitoring approach (*CMA*) and the proposed monitoring approach (*PMA*) (pumping rate=100 l/day) in terms of reliability "in homogenous medium": (a) transverse dispersivity, α_T=0.01 m, and (b) transverse dispersivity, α_T =0.03 m.

6.3.3 Comparison between the conventional and proposed monitoring approaches

The overall efficiency of a 3-well monitoring system improves enormously by application of the *PMA*. Figure 6.7 and Figure 6.8 present P_d as a function of number of wells in a monitoring system for different subsurface conditions. For a pumping rate of 100 l/day, the *PMA* gives an estimated maximum P_d value of the widely used 3-well monitoring system equals 0.98 in a homogenous medium and 0.87 in heterogeneous medium even for low dispersivity values ($\alpha_L = 0.1$ m, $\alpha_T = 0.01$ m). Under the same circumstances the *CMA* gives a maximum P_d, which does not exceed 0.40. The efficiency of an optimal 3-well system improves with a factor of approximately 2.3 (more than 100%) by simply pumping with a quite small rate (100 l/day). On the other hand, the maximum P_d values estimated based on the *PMA* can be achieved by

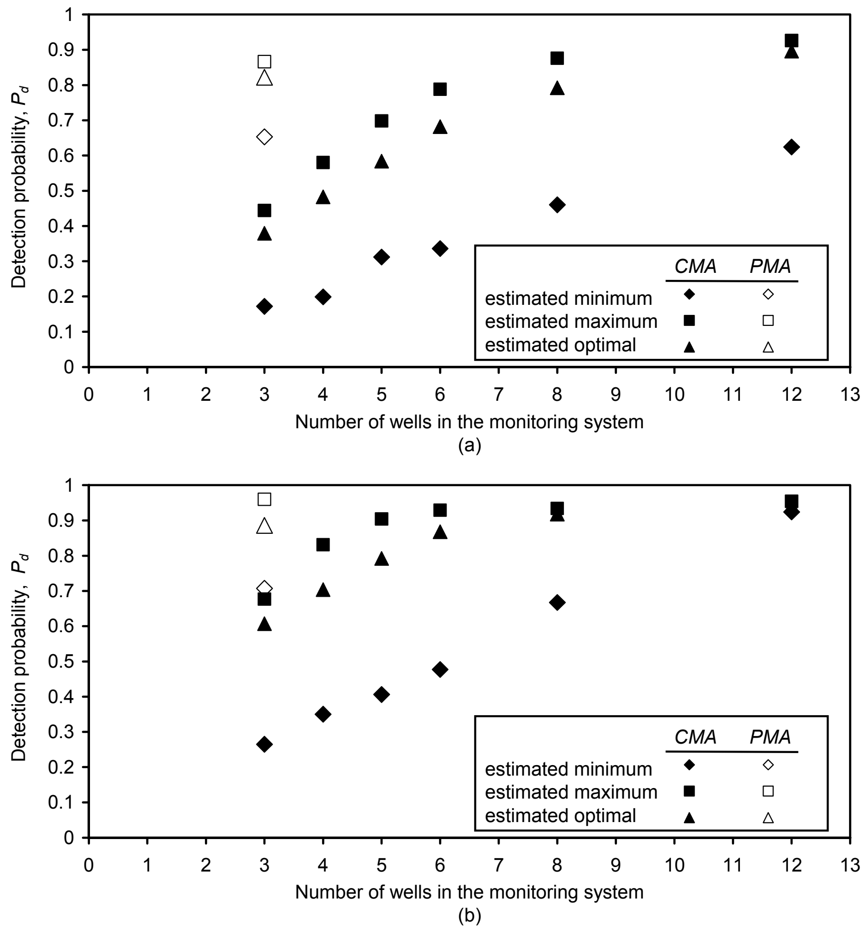

Figure 6.8: Comparison of the conventional monitoring approach (*CMA*) and the proposed monitoring approach (*PMA*) (pumping rate=100 l/day) in terms of reliability "in heterogeneous medium": (a) transverse dispersivity, α_T=0.01 m, (b) transverse dispersivity, α_T=0.03 m.

the *CMA* if 8 or 12 monitoring wells are used, for low and high dispersive medium respectively. The detection probability of an optimal 3-well monitoring system in the least favourable subsurface conditions (heterogeneous low dispersive medium) improves with a factor of 2.2 when the *PMA* is applied. The *CMA* can achieve this level of reliability by 12 well monitoring system. Furthermore the analysis showed that when pumping rate is increased to125 l/day the detection probability of 3-well system reached even up to 100% for all test cases (see Figure 6.6).

The *PMA* leads to a significant improvement in the average contaminated area regardless of heterogeneity and dispersivity of medium as well (see Figures 6.9 and 6.10). For instance the maximum and minimum contaminated area values associated with a 3-well system, for the *CMA* is estimated to be 9190 m^2 and 6340 m^2 in a heterogeneous highly dispersive medium. For the same monitoring system, when the *PMA* is applied, the maximum contaminated area is reduced to 4870 m^2 and the minimum contaminated area is reduced to 2330 m^2. Thus the *PMA* improves the efficiency of a widely used 3-well monitoring system, with a factor of 1.88 and 2.72 in terms of maximum and minimum contaminated area, respectively. The values correspond to those, which can be achieved by the monitoring systems composed of 8 or more wells with regard to the *CMA*

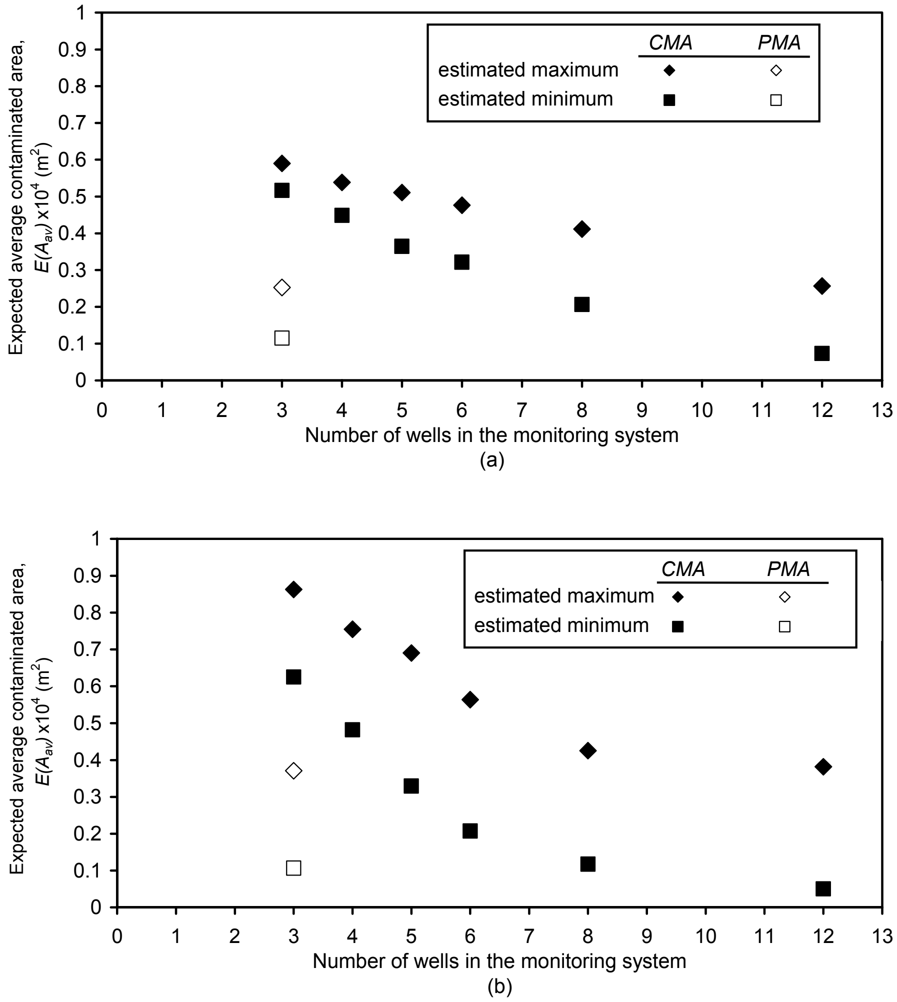

Figure 6.9: Comparison of the conventional monitoring approach (*CMA*) and the proposed monitoring approach (*PMA*) (pumping rate = 100 l/day) in terms of the expected average contaminated area $E(A_{av})$ "in homogenous medium": (a) transverse dispersivity, $\alpha_T = 0.01$ m and (b) transverse dispersivity, $\alpha_T = 0.03$ m.

6.3.4 Cost analysis

Cost analysis has been carried out in order to estimate the improvement achieved by the *PMA* in terms of cost and to investigate its feasibility with respect to the conventional monitoring approach.

Expected total cost, Z, is described as:

$$Z = C_{mw} n_{mw} + C_r V_{av} + C_{pump} Q T_m \tag{6.1}$$

where C_{mw}, is the unit installation and sampling cost of a monitoring well($/per well), n_{mw} is the number of wells in a monitoring system, C_r, is the cost of remediation per unit volume ($/m^3), V_{av}, is the contaminated volume (m^3) to be cleaned up and is defined as E(A_{av}) times the aquifer thickness B, C_{pump} is the unit cost for pumping from a monitoring well ($/m^3/year), Q_p is the pumping rate (m^3/year) and T_m is the total monitoring period (year).

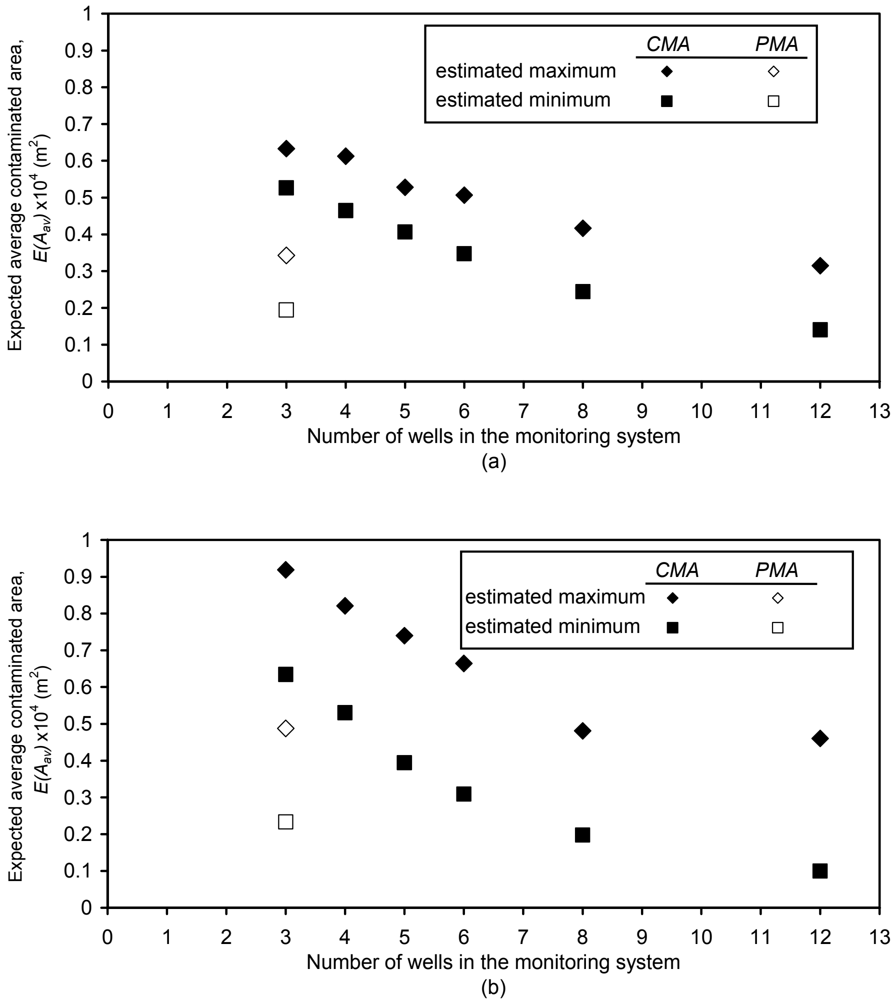

Figure 6.10: Comparison of the conventional monitoring approach (*CMA*) and the proposed monitoring approach (*PMA*) (pumping rate = 100 l/day) in terms of the expected average contaminated area $E(A_{av})$ "in heterogeneous medium": (a) transverse dispersivity, $\alpha_T = 0.01$ m and (b) transverse dispersivity, $\alpha_T = 0.03$ m.

The unit installation and sampling cost is assumed to be fixed and \$20,000 per well. A unit remediation cost of \$5/m^3 and a unit pumping cost of \$2/m^3/day is taken in the cost estimation of monitoring systems (unit cost estimates are based on James and Gorelick, 1994). Figure 6.11 shows the expected cost values of optimal monitoring systems for both *CMA* and *PMA* in homogeneous and heterogeneous medium, where $\alpha_T = 0.03$ m. For *CMA* the expected total cost decreases as the number of monitoring wells increases. However, compared with the *CMA*, the expected total cost of a 3-well system reduces with a factor of 5 in homogeneous and with a factor of 2.5 in heterogeneous medium by pumping continuously from only one well with a pumping rate of 100 l/day. Nevertheless, the unit remediation cost has a great influence on the difference in the expected cost values for both monitoring approaches.

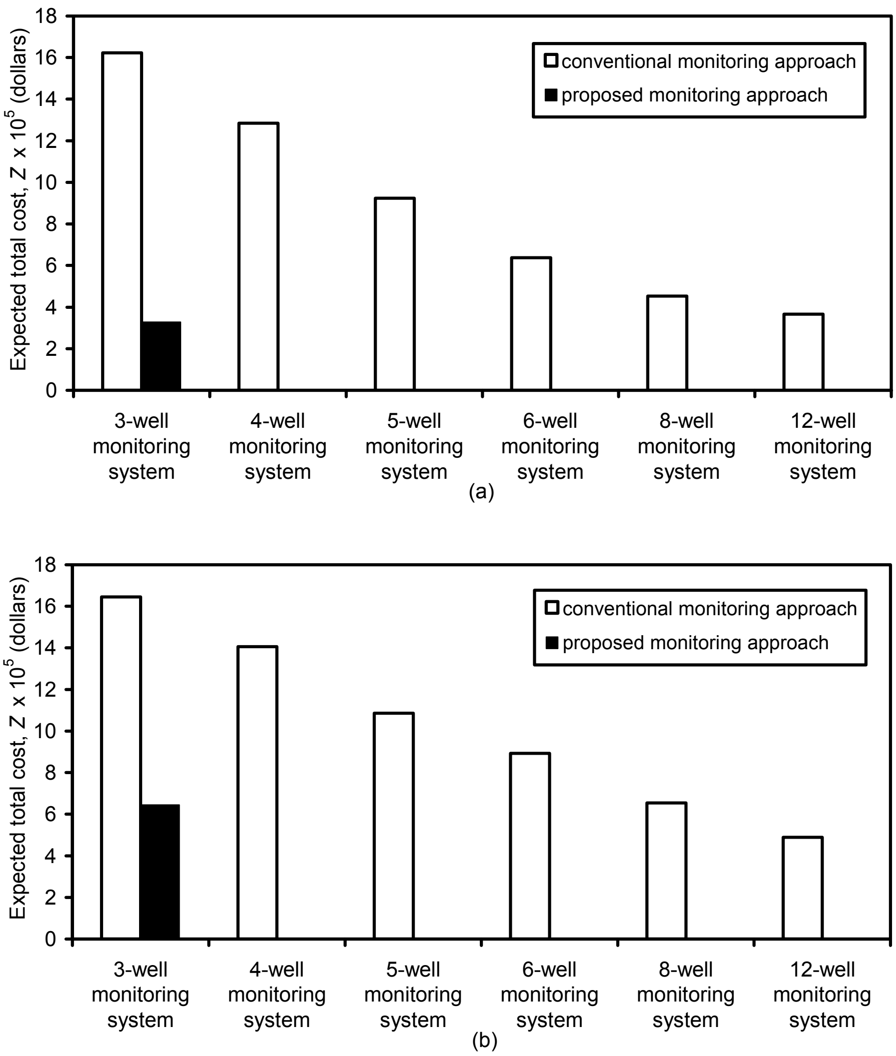

Figure 6.11: Expected cost as a function of number of wells in a monitoring system for transverse dispersivity, $\alpha_T = 0.03$ m: (a) homogenous medium and (b) heterogeneous medium.

If the remediation of the contamination problem is easier than the expected, due to a cheaper unit cost of remediation the reduction may be lesser. Conversely, if the remediation of the contamination problem is more difficult than expected then the reduction in the cost will be even more since the unit cost of remediation will be higher.

6.4 Summary and conclusions

Maximizing the likelihood of detecting contaminants and minimizing the contaminated area have been considered as design objectives. A simulation model is used to determine the optimal monitoring system where uncertainties due to subsurface heterogeneity and leak locations as well as the mentioned objectives are incorporated. A new monitoring approach is proposed to increase the efficiency of the widely used 3-well monitoring system (minimum regulatory requirement) since, with the conventional monitoring approach (*CMA*), such a system more often is not adequate to accomplish the objectives of maximizing the detection probability while minimizing the contaminated area particularly at lined landfills. This is mainly due to the limited capture zone of monitoring wells. Therefore, in the proposed monitoring approach (*PMA*) the main point is to increase the interception of contaminant plumes at early stages by broadening the capture zone of monitoring well(s) simply by continuous pumping from the monitoring well(s) with a small pumping rate.

Results from a detailed study of a hypothetical example showed that new monitoring approach has improved the efficiency of an optimal 3-well monitoring system by more than 2 times even under the least favourable circumstances, namely in a heterogeneous highly dispersive ($\alpha_L = 0.3$ m, $\alpha_T = 0.03$ m) medium. The same level of efficiency can be achieved for the conventional monitoring approach if a monitoring system is composed of more than 8 wells. Dispersivity of the medium, heterogeneity, and the number of the wells are the important parameters that play a role in the efficiency of the monitoring systems. Furthermore, for the proposed new monitoring approach pumping rate is an important factor that has influence on the reliability of the monitoring systems. However, the increase in the pumping rate will not make any further improvement once 100% reliability is achieved. On the other hand, analysis also showed that *PMA* is cost effective compared to *CMA*. However, one should keep in mind that, although the minor changes would not influence expected cost values, still the unit remediation cost has a great influence on the difference between the expected cost values for both monitoring approaches. For instance, the easier the remediation of the contamination problem is, the cheaper the unit cost of remediation will be and vice versa.

Chapter 7

Maarsbergen (Netherlands)

Landfill Site Application

Adapted from Yenigul, N.B., Keijzer, Th.J.S., and van den Akker, C., An application of decision model for groundwater detection monitoring system design for a landfill site in The Netherlands, is submitted to Journal of Contaminant Hydrology, 2006.

This chapter presents the application of simulation decision models, *MONIDAM* and its extension *MONIDAM-P*, to Maarsbergen Landfill site in order to determine the optimal groundwater detection and compare its efficiency to that of the existing monitoring system. The next section gives a general view of the problem to be addressed in this actual site application. The following two sections detail first the extension of the *MONIDAM* described in Chapter 5, and second the Maarsbergen Landfill site. The next section, describing the application of the model to the site, is followed by the analysis of the results.

7.1 Introduction

The groundwater contamination risk cannot be completely eliminated despite the current measures, such as optimal site selection, landfill design, and construction techniques to minimize the chance of leakage. Moreover, most of the old landfills have no top and/or bottom liners or protective measurements installed. The construction and the contents of the landfills, and therefore, the extent and concentration of potential contaminant sources, are mostly unknown. The local soil and groundwater are the first to be threatened by leachates emanating from such landfills. There are 11000 landfills in The Netherlands, and approximately 4000 of them are old landfills (i.e. Maarsbergen Landfill) creating excessive risks for groundwater quality as they are lacking protective measurements.

The decision model developed and described in detail in Chapter 5 and its extension that will be described in the following section is applied to Maarsbergen Landfill Site, in order to (1) determine the optimal groundwater detection monitoring system for the site, (2) evaluate the efficiency of the existing monitoring system, (3) compare the efficiency of the estimated optimal monitoring systems with the efficiency of the existing monitoring system, and (4), if necessary, augment the existing monitoring system. Maximizing the probability of detection of contaminant plumes, minimizing the contaminated area, and minimizing the total cost of the monitoring system (i.e., construction, maintenance, continuous pumping, and remediation cost, if necessary), are the design objectives considered while determining the optimal groundwater monitoring system for the Maarsbergen Landfill site under uncertainties due to hydrogeological conditions and leak location.

7.2 Model description

MONIDAM (see Chapter 5) is extended to *MONIDAM-P* by implementing *PMA* in the model since the results of analysis presented in the previous chapter showed that the efficiency of the monitoring system improves significantly by broadening the capture zone of monitoring well(s) simply by continuous pumping from the monitoring well(s) with a small pumping rate. In this case only, the cost due to the continuous pumping should be added as a cost item to risk-cost based decision model since the influence of *PMA* on the risks term will be included via the variables obtained from simulation model. Similar to the previous chapters these variables represent detection or no detection of the contaminant plume, and the plume sizes associated with detection or failure. Finally, the objective function or the so-called multi-attribute utility function for *MONIDAM-P* represents the expected total cost for each monitoring system $j=1,\ldots n$ and it is can be defined as

$$\underbrace{Z_j}_{\substack{\text{expected}\\ \text{total}\\ \text{cost}}} = C_{mw}n_{mw} + C_s n_{smw} s_f + C_{p(j)} + P_{d(j)} \underbrace{C_r B E_{A(d_j)}}_{\substack{\text{expected cost}\\ \text{given detection,}\\ E(C|d_j)}} + P_{f(j)} \underbrace{C_r B E_{A(f_j)}}_{\substack{\text{expected cost}\\ \text{given no detection,}\\ E(C|f_j)}} \tag{7.1}$$

where C_{mw} =unit installation cost of a monitoring well (Euro/per well)

C_s =sampling cost (Euro /sample/per well)

n_{mw}=number of the wells in monitoring system j

n_{smw}=number of the sampled wells in monitoring system j.

s_f=number of the total sampling for the monitoring period.

$C_{p(j)}$ =total pumping cost for monitoring system j

$P_{d(j)}$ =the probability of detection of monitoring system j,

C_r =the remediation cost per unit volume [Euro/m^3]

B =aquifer thickness [m],

$E_{A(d_j)}$ =the expected contaminated area given detection by monitoring system j [m^2],

$P_{f(j)}$ =the probability of failure of monitoring system j,

$E_{A(d_j)}$ =the expected contaminated area given no detection by monitoring system j [m^2]

The total pumping cost $C_{p(j)}$ of monitoring system j can be estimated as:

$$C_{p(j)} = T_m \sum_{i=1}^{nw_p} Q_i C_{pump(i)} \tag{7.2}$$

where T_m is the total monitoring period (year), nw_p is the number of the monitoring wells at which continuous pumping occurs, $C_{pump(i)}$ is the unit cost for pumping (Euro/m^3/year) from monitoring well i, Q is the pumping rate (m^3/year) of monitoring well i . Note that unit cost for pumping includes the costs for the limited lifetime and breakdown of the pumping system and energy costs.

7.3 Description of the Maarsbergen Landfill site

The Maarsbergen Landfill is located at the Utrechtse Heuvelrug, The Netherlands (Figure 7.1). The landfill is situated in a former sand quarry and the deposited waste varies strongly. The landfill has been operated and filled in two periods, between 1969 and 1979, and 1987 and 1994. The surface area of the landfill is approximately 8 hectare (200 m by 400 m) and currently covered by grassland. As most of the old landfills in the Netherlands no bottom liner is present. The surface level of the landfill is between 45 to 50 m +NAP whereas the base is approximately at 9.2 m +NAP (NAP being the Dutch reference level; Normaal Amsterdams Peil).

7.3.1 Site geology

The regional geology is summarized in Table 7.1 based on Vernes and van Doorn (2005). The site is covered by a medium to coarse sand layer including some gravel so called Drenthe formation to a depth of approximately 80 m –NAP. Underlying Tegelen Unit of Waalre formation is characterized by silty sandy clay and has a thickness of 10 to 20 m. Underneath the Tegelen Unit of Walre Formation the Maassluis Formation is present at a depth greater than 95 m –NAP, and comprises fine to medium sand with clay lenses and clay layers. Groundwater table is at 4—5.5 m +NAP.

Table 7.1: Geological profile

Depth [m+NAP]	Formation	Texture
+45 — -70~-80	Drenthe	Gravelly medium to coarse sand
-70~-80 — -90	Waalre (Tegelen unit)	Silty sand and clay
>-95	Maassluis	Fine and medium sand, with clay lenses and clay inter-layers

7.3.2 Site hydrogeology

The extent of the Tegelen clay is uncertain and therefore the bottom of the aquifer is not exactly known. However, based on the information from borings the aquifer thickness is determined to be around 80 m. The transmissivity of the aquifer T is about 3600 m^2/day, which results in average hydraulic conductivity K value of 45 m/day. The hydraulic resistance c of the top layer is about 1.6 days. As this is a relatively low value the vertical flow will be neglected in the analysis.

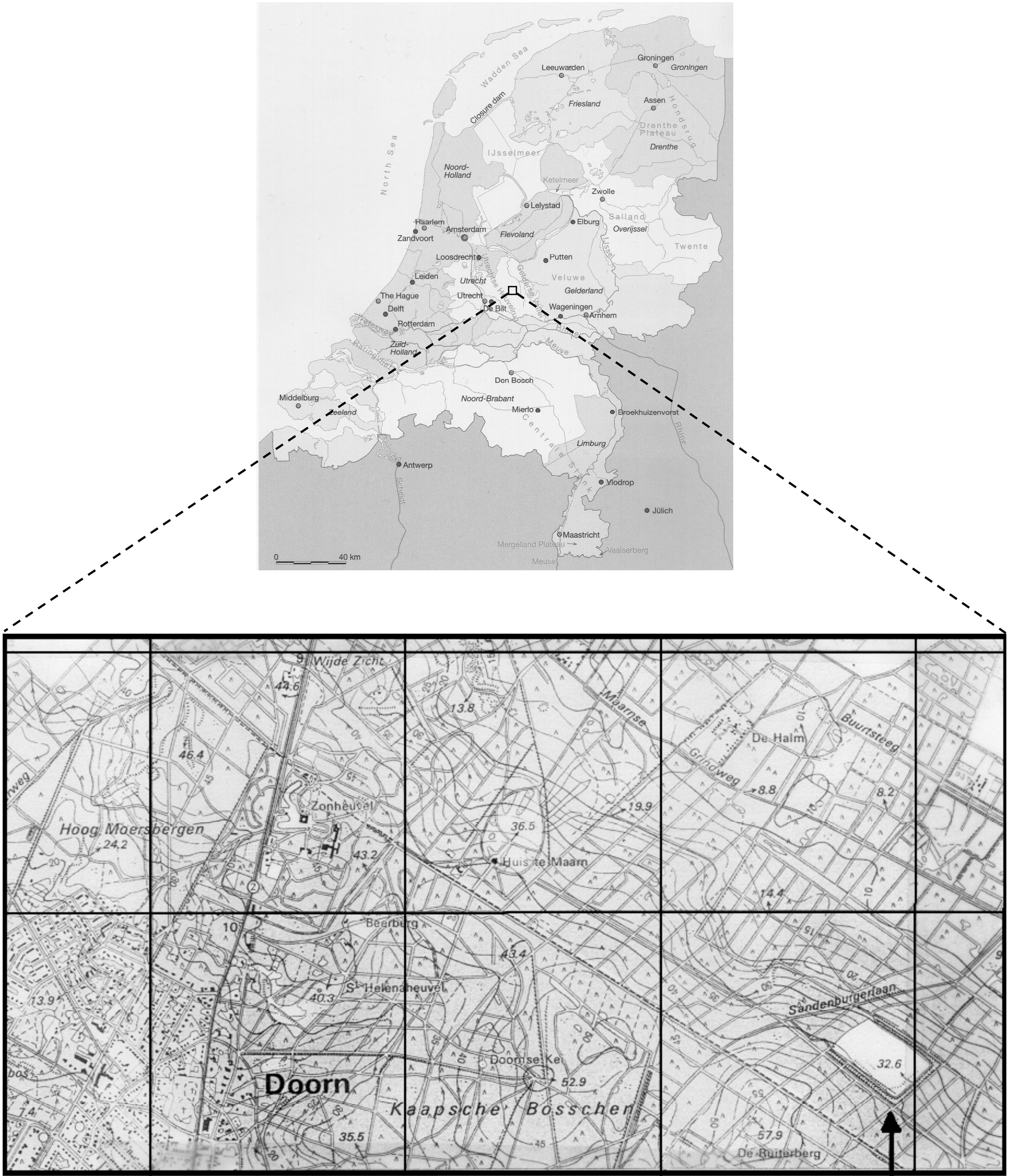

Figure 7.1: Location map of the Maarsbergen Landfill site.

The Utrechtse Heuvelrug lies on the catchment boundary between the Gelderse Vallei and the Utrecht river valley (Figure 7.1). Groundwater flow direction is southwest. The regional hydraulic gradient of 0.0004 m/m in the area results in a flow velocity of 18.8 m/year.

7.3.3 Monitoring well history

Figure 7.2 shows a map of the site illustrating the main disposal area and existing monitoring wells. Before 1997, four wells, B1, B2, B3 and B4 were installed at the landfill site. B1 is situated upstream and represents the original water quality whereas B2 and B3 are placed in the landfill itself in order to characterize the landfill. These two wells have a rather limited value as detection monitoring wells. Therefore only B4, which is placed downstream of the landfill, is included as a monitoring well in the calculations. Early 1997, monitoring wells N1 and N2 were installed in the landfill to determine the depth of the landfill and additionally N3 was installed (in late 1997) downstream of the landfill to monitor the quality of groundwater. Two more monitoring wells, P1 and P2 were installed downstream early 2000 to determine groundwater contamination due to possible contaminant releases from the landfill. Since chlorinated hydrocarbons *CHC* contamination was observed in the samples from wells P1, P2 and N3, in 2003 an extra monitoring well; P3 was installed to estimate the extent of the contaminant plume for remediation purposes. Concluding, in terms of downstream groundwater detection monitoring wells, there are four wells (P1, P2, N3 and B4) to be considered in the evaluation and comparison between the existing monitoring system and the optimal monitoring system estimated by the *MONIDAM* and *MONIDAM-P.* As shown in Figure 7.2 the positions of the wells allow for a large unmonitored area between the east edge of the landfill and the nearest downgradient monitoring well. This large unmonitored area exists even though there are no impediments to well installation in this part of the site.

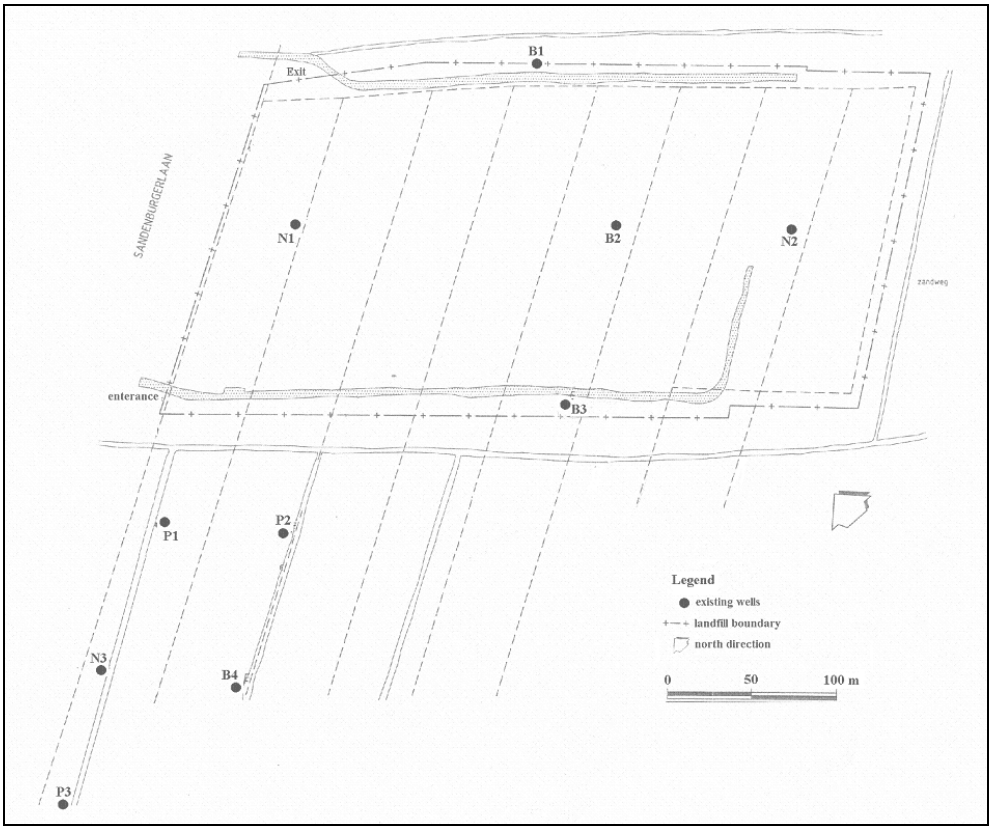

Figure 7.2: Map of the Maarsbergen Landfill site illustrating the main disposal area and existing monitoring wells.

7.4 Application of the model to the site

7.4.1 Model domain and discretization

The investigated model domain is defined by $0 \le x \le 800$ m and $0 \le y \le 500$ m The model is discretized to be 400 grid cells long and 250 grid cells wide, with all cells having dimensions of 2 m by 2 m. The location of the landfill in the model domain is between $60 \le x \le 260$ m and $50 \le y \le 450$ m. Figure 7.3a shows the discretized model domain with the existing detection monitoring wells and the landfill. There are 99 potential single monitoring system alternatives (3-well, 4-well, 5-well, 6-well, 7-well, 8-well, 10-well, 12-well and 14-well monitoring systems) located from 20 m to 220 m downgradient of the landfill with 20 m distance apart to determine the optimal monitoring system for the Maarsbergen Landfill. The considered monitoring system alternatives are composed of a line of wells parallel to the y-axis, extending the entire length of the potential source of contamination, being the downgradient edge of the landfill. Figure 7.3b shows some of the selected monitoring system alternatives in relation to the landfill. In this study monitoring wells are located in the centre of the grid cell and have a dimension of one grid cell, and it is assumed that the monitoring wells are fully penetrating the 80 m thick aquifer. A sampling program of quarterly a year for a period of 30 years is considered. 1000 Monte Carlo realizations are used in all calculations.

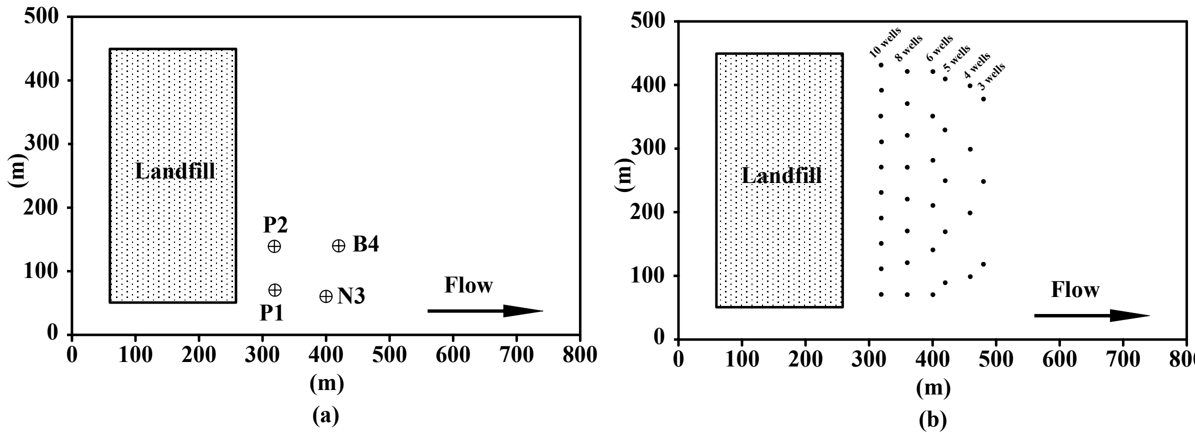

Figure 7.3: Discretized Maarsbergen Landfill model domain (a) with existing detection monitoring system, and (b) with selected monitoring system alternatives in relation to the landfill.

7.4.2 Data used in flow model

The boundary conditions for the groundwater flow are zero flux at $y = 0$ m (bottom boundary) and $y = 500$ m (top boundary) and a constant head along the left and the right boundaries. The head values at $x = 0$ m and $x = 800$ m were chosen to result in a macroscopically constant hydraulic gradient of 0.0004. Porosity is set to be 0.35.

In this study, hydraulic conductivity is treated as a random space function since the uncertainty due to subsurface heterogeneity is reflected by the spatial variability of the hydraulic conductivity. The natural logarithm of the isotropic hydraulic conductivity [Y=*ln (K)*] is modeled as a stationary Gaussian random field. The mean μ_Y of Y is set to 3.8. This value corresponds to a mean hydraulic conductivity K of 45 m/day. Since previous investigations modeled the site as a homogenous aquifer, no values for the variance of hydraulic conductivity were available. However, information about the site geology and borehole descriptions leads to the conclusion that the aquifer is fairly heterogeneous. Therefore, the variance σ_Y^2 of Y was set to 0.3 for the Maarsbergen Landfill application. It should be considered that the subsurface heterogeneity is one of the important factors controlling the migration of contaminants in porous media and hence has an important effect on the performance of the monitoring systems. Previous studies (see e.g., Meyer et al., 1994; Storck et al., 1997; Yenigul et al., 2005; Yenigul et al., 2006a and b) showed that the performance of the monitoring systems decreases as the degree of heterogeneity increases, in other words σ_Y^2 increases, since it will be more difficult to detect the contaminant plumes in more heterogeneous medium due to their irregular shapes and the uncertainty in the direction that they travel.

Data on appropriate correlation lengths was is also non-existent. However, the turning bands algorithm works best with a minimum correlation length in each dimension of twice the grid spacing in that dimension (Tompson et al., 1987). Furthermore, Ababou (1989) suggested that a correlation length four times greater or equal to the domain discretization and smaller or equal to one twenty-fifth of the domain dimension is required for statistically meaningful results from replicates of stationary hydraulic conductivity field. Accordingly, a correlation length, λ, of 20 m is chosen in both directions for the current application.

7.4.3 Data used in random walk particle tracking model

For the transport models a condition of a zero dispersive flux is imposed on both the top and bottom boundary. The initial background concentration in the model domain is set to zero. Since the flow direction is aligned with the x-axis, the only source dimension that is treated as a random variable is the position along the y-axis. The source location is drawn from a uniform probability distribution between y-coordinates of $50 \leq y \leq 450$ m (Figure 7.3b) for each Monte Carlo run.

Although there is no data available on the size of the leak, the most likely scenario suggested in the reports requires the leak to cover the entire extent of the landfill as the widespread migration of the plume is detected by the existing monitoring wells and because the landfill does not have a bottom liner. Obviously this mode of failure would result in an easily detectable massive plume and conflicts with the design philosophy of such facilities. On the other hand from the environment protection point of view regulations demands the optimal site selection, landfill design, and construction techniques aims to minimize the chance of leakage. Therefore the design philosophy of groundwater detection monitoring systems should always consider the worst-case scenario, which mostly dictates a point source or quite a small size mode of failure since it would result in a plume that is difficult to detect. Furthermore, for an old landfill case that entirely leaks, a small size mode of failure may also represent the contaminant source for more troublesome contaminants (such as BTEX), which may result thinner plumes with in the large plume due to the entire landfill leak. Thus, a leak of 1m is considered in the simulations. On the other hand one should keep in mind that the size of the leak has a large effect on the performance of optimal monitoring system since the detection probability of a plume increases as the initial contaminant size increases due to the fact that a larger contaminant source size results in a wider plume (see e.g., Yenigul et al., 2005; Hudak, 2005).

Dispersion is incorporated in the model by introducing micro scale longitudinal (α_L) and transversal (α_T) dispersivities. The ratio between α_L and α_T is assumed to be 10, (Bear, 1972) and α_L is set to 1.0 m as it reflects the previous modeling related to the migration of the plume. On the other hand, the dispersivity of the medium has a significant effect on the performance of the optimal monitoring systems since dispersivity is the parameter that controls the spreading of the plume. As the dispersivity of the medium increases the detection probability of a given monitoring system increases (see e.g., Meyer et al., 1994; Storck et al., 1997; Yenigul et al., 2005; Yenigul et al., 2006a and b). The higher the dispersivity of the medium the wider the plume gets as it moves further away from the source. Therefore, likewise the contaminated area increases significantly as the dispersivity of medium increases since the plume becomes wider as it moves away from the source (see e.g., Meyer et al., 1994; Yenigul et al., 2005; Yenigul et al., 2006a and b). Based on the available data the initial source concentration for *CHC* contaminant has been estimated at 1000 μg/l while the Dutch threshold value of 20 μg/l (de Circulaire Streefwaarden en Interwaarden Bodem Sanering, 2000) is considered a threshold concentration at which detection occurs. These results yield a detection ratio of 2% relative to source concentration. Contaminants are assumed to be conservative (note that since the natural attenuation is not very effective *CHC* might be treated as conservative) and to be completely mixed over the depth of the aquifer.

7.4.4 Unit cost values used in the model application

Table 7.2 gives average unit costs, which are used to perform the economic analysis of the potential and existing monitoring systems and to determine the value of Z_j for *MONIDAM* and *MONIDAM-P.* These values are estimated on average unit costs based on confidential documents of several companies in The Netherlands and discussions with the engineers working in these companies. On the other hand not only are there various price lists available but the before mentioned confidential discussions also showed that the unit costs depends on many factors and may differ markedly. Therefore one should keep in mind that unit costs have great influence on the determination of optimal monitoring system. The sampling cost in Table 2 includes taking the sample, routine laboratory analysis, and reporting the results. The sampling cost reduces 25% if there is continuous pumping from the wells as in the *MONIDAM-P.* Another important outcome of the cost enquiry was that the pumping cost given in Table 7.2 is the constant regardless of the pumping rate unless it exceeds 1m^3 /day.

The figure corresponds closely with the energy consumption and pump operation at its minimum capacity ($1m^3$ /day in this case).

Table 7.2: Average unit costs used in economic analysis of the monitoring systems.

Unit costs	
Installation cost per well (Euro/well)	2500
Sampling and analyses cost (Euro/sample)	300
Pumping cost (Euro/m^3/day/year)	2300
Remediation cost (Euro/m^3)	5

7.5 Results and discussion

7.5.1 Analysis results for application of *MONIDAM*

Maarsbergen Landfill site model results for the 99 potential single monitoring system alternatives

Figure 7.4 shows the reliability system as a function of distance from the contaminant source d for 99 potential single monitoring system alternatives for *MONIDAM*. The P_d of a given monitoring system increases as the contaminant plume travels away from the source as the contaminants plumes gets wider when they travel further away from the source. The P_d also increases as the number of the wells in the system increases. For instance, for d=20 m, a 3 well monitoring system detects 26 % of the simulated contaminant plumes, whereas the P_d of a 14 well system reaches 100%. However, the P_d for a common practice 3 well system does not exceed 39% even at the furthest distance (d=220 m). Therefore, to maximize the likelihood of detecting contaminants at close distances to the contaminant source a large number of wells are required. On the other hand the monitoring systems with few wells would achieve a high reliability when located as far as possible from the source. However, this may result in higher cost. Furthermore, once 100% reliability is reached at a given d increasing the number of wells does not improve the reliability of the monitoring system.

Figure 7.5 presents the $E(A_d)$ as a function of distance from the contaminant source for different monitoring systems for *MONIDAM*. The $E(A_d)$ decreases as the number of the wells in the monitoring wells increases, while it remains more or less constant despite the increase in the number of wells in a monitoring system since the distance from the source has influence on plume size as it controls the spreading of the plume.

On the other hand given that A_f is defined as the contaminated area estimated at the end of the monitoring period (in this case 30 years) neither distance from the contaminant source nor the number of the wells has influence on the plume size that is not detected by the given monitoring system. Hence, $E(A_f)$ remains almost constant with respect to well spacing and distance from the contaminant source. Thus to minimize the contaminated area, the most effective system is the one located as close as possible to contaminant source. Although, unless the monitoring wells are spaced

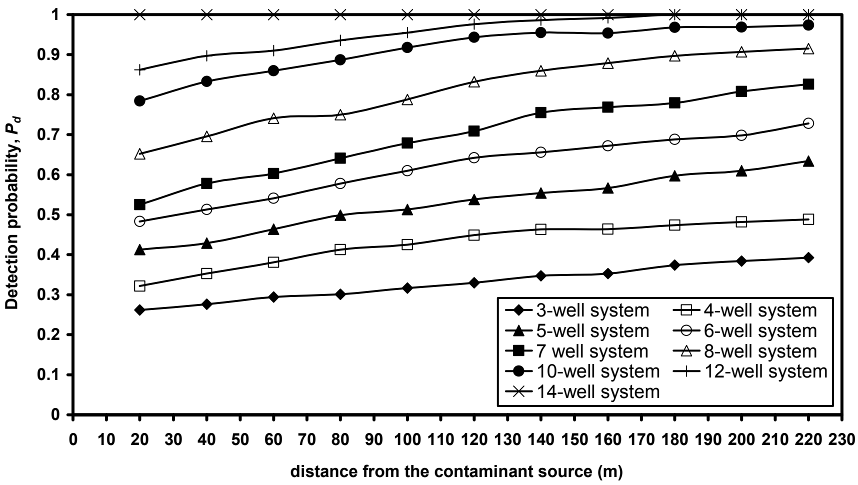

Figure 7.4: System reliability as a function of distance from the contaminant source d for different monitoring systems for the *MONIDAM* application.

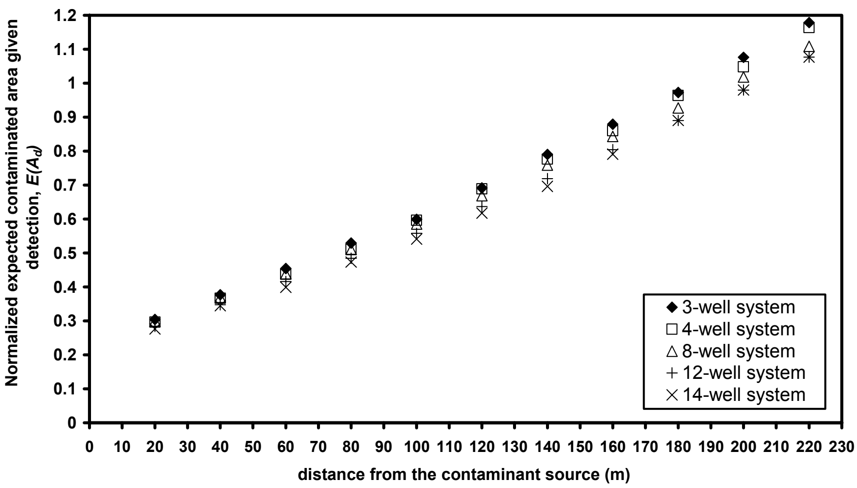

Figure 7.5: Normalized expected contaminated area given detection $E(A_d)$ as a function of distance from the contaminant source d for selected monitoring systems for *MONIDAM* application.

close to each other, the systems located close to the source also have a higher chance to miss the narrow plumes, which will widen away from the source. Yet, once the 100% reliability is achieved by a given monitoring system the addition of wells will not affect the expected size of the plume given detection as well.

Table 7.3 presents the Z values of potential monitoring systems, which were calculated using the unit cost values given in Table 7.2. For a given d the Z decreases significantly due to the fact that the P_d increases as the plume size increases. For a given monitoring system with less than 12 wells the Z decreases up to a certain d and then starts to increase. The d corresponding with the minimum Z value is the optimal location where the maximum P_d can be achieved with respect to the design objective of minimizing the contaminated area for a given monitoring system. The values in the shaded cells correspond to the minimum Z when a given monitoring system is located optimally. Note that the cost is not capitalized since detail cost analysis is not the main goal of this study and Z values is used to represent the one objective function value that defines the multi-objective nature of monitoring problem and intended to be a tool in portraying the solution of monitoring problem. Furthermore, in a thorough cost analysis modern approaches like hedging policy using real option theory should be considered; hence including a discount rate in the calculations performed here is barely significant from the point of optimal system determined by the presented methodology and has little effect on the results from the conceptual point of view.

Table 7.3: Expected total cost Z *(million Euro)* as a function of number of wells and distance from the contaminant source d for 99 potential single-row monitoring systems for *MONIDAM* application.

***d* (m)**	**Number of the wells in the monitoring system**								
	3	4	5	6	7	8	10	12	14
20	7.281	6.846	6.101	5.606	5.309	4.335	3.342	2.723	1.644
40	7.246	6.698	6.099	5.511	5.044	4.173	3.195	2.866	1.918
60	7.228	6.584	5.958	5.451	5.033	4.023	3.196	2.946	2.137
80	7.225	6.460	5.927	5.351	4.910	4.097	3.217	3.160	2.433
100	7.190	6.435	5.850	5.302	4.881	4.148	3.286	3.396	2.704
120	7.232	6.557	5.961	5.322	4.905	4.185	3.429	3.681	3.010
140	7.264	6.570	6.045	5.471	4.915	4.219	3.678	3.990	3.323
160	7.344	6.711	6.155	5.628	5.073	4.404	4.020	4.287	3.704
180	7.367	6.860	6.246	5.764	5.288	4.583	4.290	4.324	4.101
200	7.468	6.975	6.396	5.964	5.425	4.847	4.608	4.384	4.461
220	7.575	7.163	6.541	6.116	5.639	5.154	4.957	4.769	4.846

The application of *MONIDAM* to Maarsbergen Landfill shows that the 14-well monitoring system located at d=20 m is the optimal monitoring system for the site among the 99 potential monitoring systems. Because this monitoring system enables the trade off between the design objectives namely, maximizing the likelihood of detecting contaminant plumes, minimizing the contaminated area, and minimizing the total cost of construction, operation and maintenance of the monitoring system and remediation measures, if necessary.

Evaluation of the existing monitoring system

The analysis shows that the existing 4-well monitoring system detects only 22.6% of the 1000 simulated contaminant plumes (each plume is the product of one Monte Carlo realization). The $E(A_d)$ estimated from the Monte Carlo simulation data is 5560 m^2 and $E(A_f)$ equals to 23200 m^2. Despite the limited number of monitoring wells the existing system presents a poor performance mainly due to its configuration. Firstly, since the placement of monitoring wells allows the monitoring of a small portion (1/5) of the landfill (see Figure 7.2), the plumes originated from the large unmonitored portion of the landfill cannot be detected by the existing systems. This causes low system reliability. Secondly, since the monitoring wells are not located so close (60 m–160 m away from the landfill) to the contaminant source, plume sizes are large even when they are detected. Using the unit cost values given in Table 7.2 the expected total cost Z of the existing monitoring system is calculated as 7.837 million Euros.

7.5.2 Analysis results for application of *MONIDAM-P*

Maarsbergen Landfill site model results for 99 the potential monitoring system alternatives

The study by Yenigul et al. (2006b) showed that continuous pumping from one well improves significantly the efficiency of 3-well system. They also stated that P_d increases as the pumping rate increases, but once 100% reliability is achieved the increase in the pumping rate will not make any further improvement. In this study, calculations are performed using four different pumping rates (100 l/day, 125 l/day, 150 l/day and, 175 l/day) in order to determine how far can the efficiency of the potential monitoring systems be improved and to investigate whether this new monitoring strategy is cost effective. All of the potential monitoring systems (3, 4, 5, 6, 7, 8, 10, 12, and 14-well systems) achieved 100% reliability for a continuous pumping rate of 175 l/day when they are located at d=20 m. Therefore the pumping rate was not increased further. Table 7.4 presents the Z of potential monitoring systems for a pumping rate of 175 l/day. As seen the 3-well system is the optimal monitoring system as it costs the least.

Table 7.4: Expected total cost Z (million Euro) as a function of number of wells for potential single row monitoring systems located at d= 20 m for *MONIDAM-P* application.

Number of the wells in the system	**Expected total cost, *Z* (million Euro)**
3	1.487
4	1.491
5	1.523
6	1.506
7	1.537
8	1.569
10	1.613
12	1.656
14	1.704

Evaluation of the existing monitoring system

The analysis shows that pumping with a rate of 175 l/day from the existing monitoring well P1 raises the P_d of the existing system to 69.6%. The $E(A_d)$and $E(A_f)$ are estimated as 5771 m^2 and 26904 m^2, respectively. The slight increase in these values is due to the fact that, as a result of pumping, the contaminant plumes originated from the points that are quite above or below are forced to move towards the well due to radial flow towards the well. However the average contaminated area that has to be cleaned up is much smaller since more of the possible contaminant plumes (compared to the *CMA*) are detected. The Z calculated according to Equation (7.1) using the unit cost values in Table 7.2 is 5.092 million Euros.

7.5.3 Comparison of the existing monitoring system to the proposed monitoring systems

Figure 7.6 illustrates the comparison of the existing monitoring system to the proposed monitoring systems including, 3, 4, 5, 6, 7, 8, 10, 12 and 14 wells for *MONIDAM* application plus the comparison between the widely used 3-well system and existing system for *MONIDAM-P* application in terms of system reliability. For instance, a 4-well system located to at d=20 m detects 32.2% (minimum detection probability) of the 1000 simulated contaminant plumes, whilst the maximum P_d that a 4-well system can achieve is 48.8% if it is located at d=220 m. And the P_d of an optimal 4-well system, the P_d associated with the minimum contaminated area, is 42.5 %. The 4-well existing monitoring system is not only sub-optimal compared to the P_d of a proposed 4-well system but to the proposed 3-well system regardless of its location since P_d of the existing monitoring system is estimated as 22.6% considering the *CMA*. However, the results of the analysis performed by *MONIDAM-P* show that pumping from one well with a small pumping rate enables 100 % reliability for proposed 3-well and 4-well monitoring systems while enables the existing monitoring system to detect contaminant plumes three times more.

Figure 7.7 presents the comparison between the expected total cost of the proposed optimal monitoring systems and the existing monitoring system. The cost values for the proposed monitoring systems correspond to the minimum cost when they are located at optimal locations that maximizes the P_d, while minimizing the contaminated area. The best monitoring system alternative for Maarsbergen Landfill site application is the 3-well system, which is continuously pumped with a rate of 175 l/day and located at d=20 m. This optimal 3-well monitoring system cost is 9.5% cheaper than the optimal 14-well monitoring system considering the *CMA* while the Z of a conventional 3-well monitoring system is 3.8 times higher. On the other hand, the application of the *PMA* reduces the Z of the existing monitoring system by 35%.

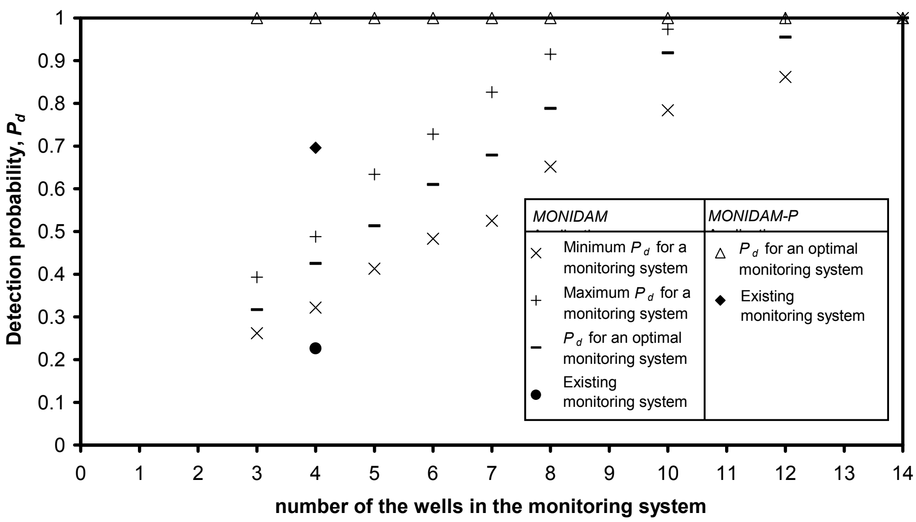

Figure 7.6: Comparison of proposed monitoring systems to existing monitoring system in terms of detection probability, P_d.

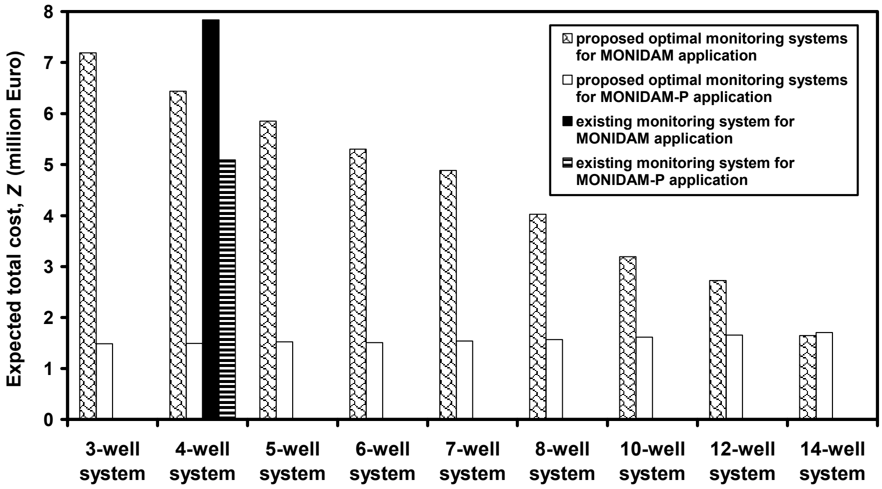

Figure 7.7: Comparison of the proposed monitoring systems to the existing monitoring system in terms of expected total cost, Z.

7.5.4 Augmentation of the existing monitoring system

The existing wells are also considered as potential wells for composing 99 single row potential monitoring system. However, none of the existing wells were part of the optimal monitoring systems defined either by *MONIDAM* or *MONIDAM-P* application. Hence, analyses are performed to determine which potential well(s) should be added to the existing monitoring system to achieve the best performance, namely maximizing the detection probability while minimizing the expected contaminated area and minimizing the cost of the monitoring systems.

Table 7.5 gives the number of the added well(s) to the existing monitoring system, their location (x- and y- coordinates of the added well(s)), P_d, $E(A_d)$ and the Z values of the augmented monitoring systems for *MONIDAM* application. The P_d increases and $E(A_d)$ decreases by adding wells to the existing monitoring system. Still the augmented monitoring systems are suboptimal with respect to those optimal monitoring systems determined among 99 potential monitoring alternatives. For instance, for an optimally located 5-well monitoring system P_d is 34% higher, $E(A_d)$ is 12.5 % lower and Z is 17.8% cheaper compared to the values estimated for an augmented 5-well system. Nevertheless, by adding only one well the reliability of the existing system can be improved by 1.5 times and the Z will reduce by 10%. Furthermore, an augmented 14-well system by addition of ten optimally located wells costs 47.4% more than the optimal 14-well system.

Table 7.5: Detection probability P_d, the expected contaminated area given detection $E(A_d)$ and the expected total cost Z of augmented monitoring system as a function of the number of added well(s) to the existing monitoring system for *MONIDAM* application.

Number of added wells	Coordinates of the added monitoring wells	P_d	$E(A_d)$ (m^2)	Z
1	(260,250)	0.338	6980	7.114
2	(360,280), (360,350)	0.436	6810	6.452
3	(360,260), (360,320), (360,380)	0.554	6686	5.909
4	(360,140), (360,280), (360,350), (360,420)	0.633	5838	5.169
6	(340,190), (340,230), (340,270), (340,310), (340,350), (340,390)	0.732	4733	4.263
8	(300,160), (300,190), (300,220), (300,250), (300,280), (300,310), (300,340), (300,370)	0.758	3348	3.721
10	(300,70), (300,110), (300,150), (300,190), (300,230), (300,270), (300,310), (300,350), (300,390), (300,430)	0.863	3746	3.124

On the other hand if a well (20,250) is added and continuously pumped with a rate of 175 l/day (*MONIDAM-P* application) then reliability of the system increases to %100 and $E(A_d)$ reduces to 3913 m^2, which result in the Z of 1.803 million Euro. Although the Z of this system is 17.5 % more expensive than the optimal 3-well system located at d=20 m, it is still approximately 77% cheaper than the Z of existing monitoring system and 42% cheaper than the augmented monitoring system by adding ten optimally located wells considering the *CMA*.

7.6 Summary and conclusions

The decision model *MONIDAM* is extended to *MONIDAM-P* by implementing a new monitoring approach into the model. The main idea in the new proposed monitoring approach (*PMA*) is to increase the interception of contaminant plumes at early stages by broadening the capture zone of monitoring well(s) by continuous pumping from the monitoring well(s) with a low pumping rate. The two models are applied to a landfill site in The Netherlands to determine the optimal monitoring system for the site considering the *CMA* and *PMA*, and to evaluate the existing monitoring system. The results show that the existing monitoring system is sub-optimal with respect to the design objectives, namely maximizing the detection probability while minimizing the expected contaminated area and minimizing the cost of the monitoring systems. A 14-well monitoring system located 20 m away from the downgradient edge of the landfill is the optimal monitoring system for *MONIDAM* application in other words for the *CMA*.

On the other hand the analyses show that the new monitoring strategy improves significantly the performance of monitoring systems. By continuous pumping with a rate of 175 l/day, a 3-well monitoring system that is located 20 m away from the downgradient detects all of the potential contaminant plumes and cost 35% less than the optimal monitoring system considering the *CMA*.

Furthermore, considering *CMA* addition of 10 optimally located wells to the existing monitoring system, the system reliability can be approximately improved by 74%

with a total cost of 77% less. However, the augmented monitoring system still costs 47% more than the optimal monitoring system estimated. On the other hand if a well (20, 250) is added and pumped continuously with a rate of 175 l/day (*MONIDAM-P* application) then the reliability of the system increases to %100 while the expected total cost is reduced by 77% compared to the cost of existing monitoring system. Furthermore the expected total is also reduced by 42% compared to the augmented monitoring system to which ten optimally located wells added considering the *CMA*. Yet one should consider that no direct comparison is possible between the existing and the determined optimal monitoring systems since actual site problems are always more complicated and usually they are simplified to conform to models.

Chapter 8

Conclusions and Recommendations

The objective of this thesis was to formulate a methodology for the design of optimal groundwater monitoring system design at landfill sites under conditions of uncertainty. A simulation model coupling a Monte-Carlo framework with a two-dimensional finite difference flow model and a random walk particle-tracking model was used to simulate contaminant plumes. Uncertainties due to subsurface heterogeneity and leak location were incorporated in the model. The detection probability of a contaminant plume released from a landfill was investigated by means of both a simulation and an analytical model for both homogeneous and heterogeneous aquifer conditions. The results of the two models are compared for homogeneous aquifer conditions to illustrate the errors that might be encountered with the simulation model. The results of the analytical model using effective (macro) dispersivities were compared with simulation model results to find out how far an analytical model can be used in groundwater monitoring system design while incorporating the effects of various heterogeneities on contaminant transport. Then reliability assessment of monitoring systems at landfill site was performed to evaluate the influence of several parameters (e.g., locations and the number of monitoring wells, dispersivity of medium, threshold concentration, leak size, type of leak and sampling frequency) on the detection probability of contaminant plumes by given monitoring systems.

Afterwards a decision analysis model was developed for optimal design of groundwater monitoring systems under conditions of uncertainty. The methodology accounts for the multi objective nature of detection monitoring problem as well. Maximizing the probability of detecting contaminant plumes, minimizing the contaminated area, and the total cost of the monitoring system (i.e., construction, maintenance, and remediation cost, if necessary) were the conflicting objectives incorporated to find the optimal monitoring system in terms of location and number of the wells. Then a new monitoring approach was proposed and implemented to find out how to improve the efficiency of groundwater monitoring systems, particularly the efficiency of a 3-well system that fulfils the minimum regulatory requirement. To increase the interception of contaminant plumes at early stages by broadening the capture zone of monitoring well(s) simply by continuous pumping from the monitoring well(s) with a small pumping rate is the essence of this approach. Then by implementing this monitoring approach in the model, the former decision model considering the current conventional monitoring approach, was extended. Finally the applications of both simulation-decision models to Maasbergen landfill site (Netherlands) were presented.

8.1 Detection of contaminant plumes from landfills

8.1.1 Homogeneous aquifer conditions

Simulation and analytical models are used to compute concentration distributions and detection probability values at given monitoring well locations for instantaneous and continuous leaks. For plume simulations 500, 1000, 2000, 4000 and 8000 particles are used and the analysis showed that the accuracy of the estimates by the simulation model is highly dependent on the number of the particles used in the model. Obviously plume edges are better defined by 8000 particles than by 500, but the differences are minor. Although the detection probability values computed by simulation and analytical model are compatible for both instantaneous and continuous leak cases the match is better in the continuous leak case. This is mainly due to the fact that the convolution procedure used to mimic a continuous leak yields better approximations of the plume with less particles than in the instantaneous leak case.

8.1.2 Heterogeneous aquifer conditions

Effective (macro) dispersion coefficients are used to solve the advective-dispersive transport equation in order to model contaminant transport in heterogeneous media by analytical methods. The results show that the modelling of contaminant transport using an advection-dispersion equation with effective (macro) dispersivities can be used to describe the average concentration distribution, but this approach is insufficient in monitoring system design when incorporating the subsurface heterogeneity. The mean concentration plume that results from such an approximation is smooth due to loss of the detailed advective heterogeneity. This reflects in an overlook in the determination of the concentration field and consequently in the computation of the detection probability of a contaminant plume by a given monitoring well. The 95% confidence intervals drawn from the simulations show that the uncertainty in concentration predictions decreases with the distance from the source. The concentration gradient is high near the source as the plume is narrow in the beginning and has a large degree of freedom to spread in different forms from one realization to another. On the other hand, the degree of freedom to spread from one realization to another is not that high and uncertainty is less when the contaminant plume moves further away and widens. Another important point is that the uncertainty in concentration predictions based on such analytical models increase as heterogeneity and/or dispersivity of the medium increases.

8.2 Reliability evaluation of monitoring systems at landfill sites

The reliability model is capable of simultaneously evaluating several monitoring systems composed of different numbers and locations of wells. The model can be used to evaluate existing monitoring systems and can be easily applied for new designs in an efficient manner. Results obtained from extensive numerical experiments show the dependence of the reliability of monitoring systems on several parameters such as dispersivity of the medium, heterogeneity of the medium, size of the initial contaminant leak, detection threshold, and number and location of the wells. The analysis showed that:

- Subsurface heterogeneity is an important factor that affects the reliability of the monitoring systems, since it controls the movement of the contaminant and the shape of the plume. The detection probability decreases as the variance of hydraulic conductivity increases. This is due to the fact that the contaminant plumes are more likely to become irregularly shaped in heterogeneous media, and they may go undetected easier due to variability in the flow field, while in homogeneous media the plumes have much more uniform shapes and tend to travel in a direction parallel to the average gradient.
- The lateral dispersivity of the medium has a significant influence on the reliability of the monitoring systems, since it is the primary parameter controlling the size of the plume. The detection probability increases when the dispersivity of medium increases, since the plume gets wider as it travels away from the source.
- The size of the initial contaminant source is another factor that has influence on the width of the plume. The detection probability of a given monitoring system increases as the size of the initial contaminant source increases since a larger leak size initiates wider plumes.
- The reliability of monitoring systems increases with distance from the contaminant source. Since plumes begin with a small size and spread out as they migrate away from the source, systems composed of few wells are more likely to detect the contaminant plumes when they are placed away from the contaminant source.
- For a given distance away from the contaminant source the probability of detection increases as the spacing between the monitoring wells decreases in other words as the number of the monitoring wells increase. However, once 100 % of reliability is achieved by a given monitoring system additional wells would not improve the system reliability.
- The widely used 3-well monitoring system (minimum regulatory requirement) is often too small from the point of view of the detection of the contaminant plume and the prevention of groundwater contamination.
- Although the detection threshold has no impact on the actual transport of contaminants it has influence on reliability of monitoring systems as it defines the plume size to be detected. The effect is similar to that of dispersivity of the medium. The lower the detection threshold the bigger the plume size. Therefore the detection probability increases as the value of threshold concentration decreases.

8.3 Design of an optimal groundwater detection monitoring system

The design of a groundwater detection monitoring system is a multi objective problem. The likelihood of detection increases when a large number of monitoring wells are located, however, the monitoring and construction cost also increases. Hence a trade off exists between the likelihood of detecting contaminant plumes, the contaminated area, and the associated cost of construction, operation and maintenance of the monitoring system. A decision analysis model has been developed *(MONIDAM)* which accounts for these three conflicting design objectives, while incorporating the uncertainties due to the subsurface heterogeneity and leak location. The analysis showed the dispersivity and heterogeneity of medium has influence on the size of the contaminant plume as well as the reliability of the monitoring systems. The contaminated area given detection increases as the distance from the source increases. However the number of the wells used in a monitoring system has no influence on the size of the plume, while the location of the monitoring system is crucial for minimizing the contaminated area. The nearer the monitoring systems to the contaminant source, the smaller the contaminated area will be. Furthermore the contaminated area increases as the dispersivity of the medium increases since the higher the dispersivity of the medium the larger the plume size is. This could give preference to those systems located further away from the contaminant source, as the detection probability is higher at further distances than in a medium with low dispersivity. The degree of heterogeneity is an important parameter that has influence on the extent of the contaminated area. The contaminated area increases with increasing heterogeneity. The analysis also demonstrated that sampling frequency has an effect on the extent of contaminated area as well. As the sampling frequency increases the contaminated area decreases. However, the reflection of this effect on the expected total cost is prominent if the installation and monitoring cost are very high and/or the remediation cost is very cheap.

It was observed that the most efficient design for detection monitoring should consist rather of a large number of wells located close to contaminant source, except for the cases where the unit installation and monitoring cost are very high and/or the unit remediation cost is very cheap. Furthermore, the widely used 3-well monitoring system (minimum regulatory requirement) is not an optimal solution for any of the cases considered in the analysis. Therefore implementation of a new monitoring approach was proposed to design a highly efficient cost-effective 3-well system. In this new approach the main idea was to increase the interception of contaminant plumes at early stages by broadening the capture zone of monitoring well(s) simply by continuous pumping from the monitoring well(s) with a small pumping rate. The efficiency of the 3-well monitoring system was compared for conventional and proposed monitoring approaches. It was observed that the efficiency of the monitoring system improves significantly by the application of the proposed monitoring approach (more than twice). Furthermore, for the proposed new monitoring approach pumping rate is an important factor that has influence on the reliability of the monitoring systems. However, the increase in the pumping rate will not make any further improvement once 100 %

reliability is achieved. Considering these promising results *MONIDAM* is extended to *MONIDAM-P* by implementing the new monitoring approach into *MONIDAM*.

8.4 Application of the Methodology to a Real Landfill Site

MONIDAM and *MONIDAM-P* were applied to an actual site, Maarsbergen Landfill Site (The Netherlands). The results of the site application agree with the trends stated above. Moreover, the results indicated that the existing 4-well monitoring system is sub-optimal with respect to the design objectives, to that obtained by application of *MONIDAM* and *MONIDAM-P.* A 14-well monitoring system located 20 m away from the downgradient edge of the landfill is the optimal monitoring system in the *MONIDAM* application. The reliability of existing system can be approximately improved by 74% with a total cost of 77% less with addition of ten optimally located wells to the existing monitoring system. However, the augmented monitoring system still costs 47% more than the estimated optimal monitoring system.

On the other hand the results of *MONIDAM-P* application showed that all of the potential contaminant plumes can be detected even by a 3-well monitoring system that is located 20 m away from the downgradient and it costs 35% less than the optimal monitoring system determined by the *MONIDAM* application.

Furthermore, when an optimally located monitoring well is added and pumped continuously, the reliability of the system increases to 100% while the expected total cost is reduced by 77% compared to the cost of existing monitoring system. The decision maker can be assured that the monitoring systems designed by the application of the methodology are the best for that site's model parameters However, no direct comparison is possible between the existing and the determined optimal monitoring systems as the solutions are dependent on the complicated site characteristics that are usually simplified to conform to models. Furthermore professional judgment and experience for interpretation of the available data and their modeling is of high importance for the success of the monitoring system design.

8.5 Recommendations

The following general tentative recommendations and further research suggestions on groundwater monitoring system design can be given.

- Monitoring frequently takes place in an environment where vertical flow is important and well screen length is also important in the efficiency of the monitoring system. Therefore extension of the presented methodology to three dimensions is desirable.

- In the groundwater flow model used in this thesis, steady state conditions were assumed. Given the dynamic nature of subsurface flow the extent of contamination may become time dependent. Therefore extension of the present model to incorporate transient conditions would be an attractive future research topic.
- A practical methodology should be able to incorporate information about the system at the time that it is gathered since reducing the uncertainty can provide a better estimate of monitoring system performance. Conditional simulation incorporating hydraulic conductivity, head measurements or concentration values in the Monte Carlo framework, could be used to reduce further the level of uncertainty and to update the design.
- A conservative contaminant transport was considered in the analyses. Extension of the present methodology to incorporate chemical processes, sorption, desorption, retardation, decay and biological transformations would be desirable for a more realistic modelling.
- Although the hydraulic conductivity and leak location were assumed to be the major contributors to transport uncertainty, there are other parameters, such as dispersivity, type and rate of release of contaminants, the magnitude and direction of regional groundwater gradient, sampling frequency, concentration threshold contributing to the uncertainty in concentration of contaminants at a specified location. These parameters were treated deterministically and uncertainty in some of them was investigated via sensitivity analyses in this thesis. Since sensitivity analysis is a simple instrument it would be valuable to have an efficient method to find alternatives that are robust with respect to parameters.
- A Monte Carlo simulation procedure was used to evaluate the reliability of groundwater monitoring systems. Although it is efficient and provides all necessary data for decision analyses, in terms of detection and plume size, it is still computationally heavy. Techniques such as first-order, second moment groundwater uncertainty analysis may demand less computational effort. Given the recent advancements using spreadsheet programs for reliability purposes, development and application of these methods to the situation presented here is highly desirable for practice.
- Dispersion was incorporated as one of the transport mechanisms, assuming constant values for the dispersivities. However, correct values of the longitudinal and transverse dispersivity are very hard to determine given the various factors influencing these parameters. Performing more research to enlighten the topic further and development of criteria to choose appropriate values for accurate modelling would be interesting.
- In the case of scarce data, it is important to develop a criterion that takes into account the data available and other constraints, to determine the levels of uncertainty to be imposed. Such a criterion will act as a guide in an attempt to design robust optimal strategies under different scenarios of data availability.
- An actual site application was used to illustrate the methodology. However gathering the available data was hard due to difficulties obtaining information sources and references. Publication of well-organized case study data would greatly facilitate the analysis and application of such methodologies.

- The analyses results showed that the minimum regulatory requirement of three down gradient well is not a sufficiently large minimum for detecting the contaminant plumes. Furthermore the proposed new monitoring approach, namely increasing the interception of contaminant plumes at early stages by broadening the capture zone of monitoring well(s) simply by continuous pumping from the monitoring well(s) with a small pumping rate, improves the efficiency of such systems significantly. Policy makers should reconsider the regulations with respect to groundwater detection monitoring at landfill sites. As a first step the minimum number of the wells required could be increased. At the same time the proposed monitoring approach could be tested via pilot studies to evaluate the efficiency and applicability of the methodology in reality so that the approach might be considered in the legislations in the future.

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Symbols and abbreviations

UPPERCASE ROMAN SYMBOLS

A_d	plume area at the time of detection
A_f	plume area that remains undetected at the end of the monitoring period
A_L	longitudinal macrodispersivity
A_T	transverse macrodispersivity
B	aquifer thickness
$B_j(t)$	benefits of alternative j in year t
C	concentration of the contaminant
$\overline{C}$	mean concentration
C_0	initial contaminant concentration
C	construction and operation cost of a monitoring system
C_{dr}	remediation cost associated with detection of contaminant plume by a monitoring system
C_{fr}	remediation cost when a monitoring system fails to detect the contaminant plume
$C_j(t)$	costs of alternative j in year
C_{mw}	unit installation cost of a monitoring well
C_p	total pumping cost of a monitoring system
C_{pump}	unit cost for pumping from a monitoring well
C_r	remediation cost per unit volume
C_s	sampling cost
$C_{f(j)}(t)$	cost associated with a failure of alternative j in year t
C_{mw}	contaminant concentration at the well location
C_{TH}	threshold concentration

$CV_{A(d)}$	coefficient of variation of expected contaminated area given detection
D_m	molecular diffusion coefficient
D_{xx}, D_{xy}, D_{yx}, D_{yy}	components of the pore scale hydrodynamic dispersion tensor
$D(x_0,\ y_o,\ C_{TH})$	detection region
$E(A_{av})$	expected average contaminated area
$E(A_d)$	expected contaminated area given detection by a monitoring system
$E(A_f)$	expected contaminated area given no detection by a monitoring system
I_d	indicator function of detection by a monitoring system for a realization
K	hydraulic conductivity
K_{xx}	hydraulic conductivity in the x- direction
K_{yy}	hydraulic conductivity in the y- direction
L	length of the landfill
L_x,	length of model domain
L_y	width of model domain
$L(x_{mw},\ y_{mw,}\ C_{TH}.)$	leak-region
M_o	mass of initial contaminant
N	total number of particles
N_{MC}	total number of Monte Carlo simulation runs
P_d	probability of detection of a groundwater monitoring system
P_d(max)	maximum detection probability
$P_{d(mw)}$	probability of detection of a given plume by a given monitoring well
P_f	probability of failure of a groundwater monitoring system
Q_p	pumping rate of a monitoring well
$R_j(t)$	risks of alternative j in year t
T	transmissivity
T_m	total monitoring period
V_{av}	average contaminated volume

V_d	volume of contamination given detection by a monitoring system
V_f	volume of contamination given no detection by a monitoring system
W	width of the landfill
Y	natural logarithm of hydraulic conductivity
X	random variable
X_A	arithmetic mean
X_G	geometric mean
X_H	harmonic mean
X_p	x- coordinate of a particle
Y_p	y- coordinate of a particle
Z, Z'	two independent random numbers drawn from a normal distribution
Z	expected total cost of a monitoring system
Z_{min}	minimum expected total cost of a monitoring system

LOWERCASE ROMAN SYMBOLS

b	thickness of a grid cell
c	hydraulic resistance
d	distance between the monitoring well and contaminant source
h	hydraulic head
i	annual discount rate
l	half of the maximum plume width
n	number of particles in a grid cell
$ndfs$	normalized distance from the source
$ndfs_{opt}$	optimal location of a single row monitoring systems
n_{mw}	number of the wells in a monitoring system
n_{smw}	number of the sampled wells in a monitoring system
nws	normalized well spacing
nw_p	number of the monitoring wells at which continuous pumping occurs
q_x	Darcy's velocity components in the x- direction

q_y	Darcy's velocity components in the y- direction
s_f	number of the total sampling for the total monitoring period
$\lvert v \rvert$	mean groundwater velocity
v_{max}	maximum velocity in the flow field
v_x	average groundwater flow velocities in the x- direction
v_y	average groundwater flow velocities in the y- direction
x_0, y_0	coordinates of leak location
x_{mw}, y_{mw}	coordinates of a monitoring well

GREEK SYMBOLS

α_L	longitudinal dispersivity
α_T	transverse dispersivity
Δx	grid spacing in x- direction
Δy	grid spacing in y-direction
Δt	time step
δ_{ij}	Kronecker delta
ε	effective porosity
$\gamma(.)$	normalized utility function [decimal fraction, ≥1]
λ_x	correlation length in x- direction
λ_y	correlation length in y- direction
μ_Y	mean of Y
σ_Y	standard deviation of Y
σ_Y^2	variance of Y

ABBREVATIONS

CEC	Council of European Communities
CHC	Chlorinated Hydrocarbons
CMA	Conventional Monitoring Approach
ECC	European Community Council
FORM	First Order Reliability Method
ISWA	International Solid Wastes Association
LCRS	Leachate Collection and Removal System
MC	Monte Carlo
NAP	Normaal Amsterdams Peil (the Dutch reference level)
SORM	Second Order Reliability Method
PMA	Proposed new Monitoring Approach
RCRA	Resource Conservation and Recovery Act
USEPA	U.S. Environmental Protection Agency

LIST OF FIGURES

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PROPOSITIONS

accompanying the thesis:

Groundwater Detection Monitoring System Design under Conditions of Uncertainty

by

N. Buket Yenigül

1. The European regulations, which require a minimum of three downgradient monitoring wells at landfill sites, actually instil a false sense of security rather than protecting the groundwater from contamination (This thesis).

2. A monitoring method based on continuous pumping with a small pumping rate is more efficient and cost-effective as compared to monitoring the groundwater quality using the conventional method. (This thesis).

3. Uncertainties should be incorporated when designing groundwater quality monitoring systems as they have significant influence on the efficiency of the systems. However, it becomes more complicated to design efficient monitoring systems as the level of uncertainty increases. (Related to this thesis)

4. The application of new methodologies and consequently changes in the regulations and standards should not be delayed due to ancient ties to conventional methods or to the prejudice to new developments.

5. Multidisciplinary research and progress in science to solve water resources related problems should be extended to the social sciences, so that the human attitude towards water resources can be improved.

6. Water scarcity will lead to more severe conflicts and wars in the near future than problems related to energy resources. Therefore, a decision maker should rather let the long-term consequences be dominating than his immediate preference when dealing with water resources management problems.

7. The real orphans are not those without parents but those without science and knowledge. (Arabic Saying)

8. The punishment of talking without thinking is to be convicted to think for life after talking.

9. Never justify yourself. Your enemies won't believe you and your friends won't need it.

10. Love and knowledge are so alike. Both demand care and effort. Both increase when shared and diminish when hidden. Together with good intentions both lead to miracles while along with bad intentions they can be disastrous.

These propositions are regarded as defendable, and have been approved as such by the supervisors, Prof. dr. ir. C. van den Akker and Prof. dr. F.M. Dekking.

STELLINGEN

behorende bij het proefschrift:

Het ontwerpen van grondwater monitoring systemen onder onzekerheid
door
N. Buket Yenigül

1– De Europese regelgeving die tot doel heeft het grondwater te beschermen tegen vervuiling, geeft in werkelijkheid een vals gevoel van veiligheid (Dit proefschrift).

2– Een kwaliteitscontrole gebaseerd op continu pompen met een geringe intensiteit is efficiënter en goedkoper in vergelijking met de conventionele methode voor grondwaterkwaliteit controle (Dit proefschrift).

3– Er moet rekening gehouden worden met onzekerheden bij het ontwerp van grondwaterkwaliteit monitoring systemen omdat zij een significante invloed hebben op de efficiëntie van de systemen. Echter, het wordt gecompliceerder en verontrustender om een efficiënt monitoringssysteem te ontwerpen naarmate de onzekerheid toeneemt (Gerelateerd aan dit proefschrift).

4– De toepassing van nieuwe methoden en die daaruit voortvloeiende veranderingen in de regelgeving en standaardisatie zouden niet vertraagd moeten worden door oude banden met conventionele methoden of door vooroordelen tegen nieuwe ontwikkelingen.

5– Multidisciplinair onderzoek en vooruitgang in de wetenschap om probleem gerelateerd aan grondwaterbeheer op te lossen moeten uitgebreid worden naar de sociale wetenschappen, zo dat het begrip van de bevolking t.a.v. grondwaterbeheer verbeterd kan worden.

6– Schaarste aan water zal tot meer conflicten en oorlogen leiden in de nabije toekomst dan energie gerelateerde problemen. Daarom zou een beslisser de consequenties moeten laten overheersen in plaats van zijn directe voorkeur wanneer hij met een probleem zoals grondwaterbeheer wordt geconfronteerd.

7– De werkelijke weeskinderen zijn niet degene zonder ouders maar diegene zonder wetenschap en kennis (Arabisch citaat) .

8– De straf voor spreken zonder te denken is veroordeeld te zijn tot levenslang denken na het spreken.

9– Rechtvaardig jezelf nooit. Je vijanden zullen je niet geloven en voor je vrienden hoeft het niet.

10– Liefde en kennis hebben zoveel met elkaar gemeen. Beide eisen zorg en aandacht. Beide groeien wanneer ze gedeeld worden en nemen af wanneer ze verborgen blijven. Samen met goede bedoelingen leiden beide tot wonderen, samen met slechte bedoelingen kunnen ze leiden tot rampspoed.

Deze stellingen worden als verdedigbaar beschouwd en als zodanig goedgekeurd door de promoter, Prof. dr. ir. C. van den Akker and Prof. dr. F.M. Dekking.