THE UNIVERSITY OF HULL

APPLICATION OF MULTI-SCALE ASSESSMENT AND MODELLING OF LANDFILL LEACHATE MIGRATION: IMPLICATIONS FOR RISK-BASED CONTAMINATED LAND ASSESSMENT, LANDFILL REMEDIATION, AND GROUNDWATER PROTECTION

Being a Thesis submitted for the Degree of

PhD

In the University of Hull

By

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February 2004

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ABSTRACT

There are a large number of unlined and historical landfill sites across Britain, contaminating groundwater and soil resources as well as posing a threat to human health and local communities. There is an essential requirement for robust methodology when carrying out risk-based site investigations prior to risk assessment and remediation of landfill sites. This research has focused upon the methods used during site investigations for two reasons. Firstly, the site investigation is often conducted using field instruments and methods that do not account for the heterogeneous conditions found at landfill sites. Interpreting geophysical conditions between sampled points is a common practise. Given the complex and heterogeneous conditions at landfill sites, such methodology introduces uncertainty into data sets. Secondly, risk estimation models that simulate groundwater flow and contaminant transport require extensive field information. The data used during model construction will significantly impact contaminant transport simulations. Modelling guidelines also need further development, ensuring that sound modelling practises are adhered to during model construction.

To address these concerns, four research objectives were identified: (1) Two new multi-spatial field assessment methods (remote sensing and ground penetrating radar), previously applied in other fields of science, were tested on landfill sites; (2) Kriging was used as a tool to improve landfill-sampling strategies; (3 & 4) Groundwater flow and contaminant transport models were used to evaluate whether different scales of field data and modelling practises influenced modelling assumptions and simulation.

The utility of novel field- and airborne-based remote sensing methodologies in identifying the location and intensity of vegetation stress caused by leachate migration and inferring pathways of near surface contamination using patterns of vegetation stress was proven. The results from the kriging investigations demonstrated that additional insight into field conditions could be resolved to identify locations of additional sampling points, and provide information about variability in hydrological data sets. The Ground Penetrating Radar investigations provided three types of valuable

near-surface information that could assist in determining landfill risks: buried landfill features, leachate plume locations and local hydrogeological conditions. These combined methods provided detailed synoptic geophysical and contaminant information that would otherwise be difficult to determine. Their application and acceptance as site assessment methods (used under certain landfill conditions) could increase the accuracy of assessing risks posed by landfill leachate.

These applications also demonstrated that the most effective site assessments are achieved when integrated with other field data such as soil, vegetation, and groundwater quantity measurements, contaminant concentrations and aerial photographs, providing comprehensive information needed for risk estimation modelling.

The modelling analyses found that close attention must be paid to site-specific and model-specific characteristics, as well as modelling practises. These factors influenced model results. By using additional data to infer model parameters, it was evident that the amount of data available will influence the way in which risk will be perceived. The more data that was available during model construction, the higher the risk prediction. This was the case for some seventy- percent of the models.

By improving the accuracy of site investigation methodology, and by adhering to robust assessment and modelling practices, a higher level of quality assurance can be achieved in the risk assessment and remediation of contaminating landfill sites. If the improvements and recommendations presented in this research are considered, uncertainties inherent in the site investigation could be reduced, therefore enhancing the accuracy of landfill risk assessment and remedial decisions.

TABLE OF CONTENTS

CHAPTER 3: LANDFILL ASSESSMENT AND MANAGEMENT

CHAPTER 4: LANDFILL SITE ASSESSMENT METHODS

CHAPTER 7: STUDY SITE DESCRIPTIONS

CHAPTER 8: New Site Assessment Methods- Results

CHAPTER 9: MODELLING ANALYSIS - RESULTS

CHAPTER 10: DISCUSSION

CHAPTER 11: RESEARCH SUMMARY

REFERENCES

LIST OF FIGURES

LIST OF TABLES

LIST OF ABBREVIATIONS

ASTM	AMERICAN STANDARDS FOR TESTING MATERIALS
BH	BOREHOLE
BSI	BRITISH STANDARDS INSTITUTION
CAMASE	CONCERTED ACTION FOR THE DEVELOPMENT AND TESTING OF QUANTITATIVE METHODS FOR RESEARCH ON AGRICULTURAL SYSTEMS AND THE ENVIRONMENT
CASI	COMPACT AIRBORNE SPECTROGRAPHIC IMAGER
CIRIA	CONSTRUCTION INDUSTRY RESEARCH AND INFORMATION ASSOCIATION
CIWM	CHARTERED INSTITUTE OF WASTES MANAGEMENT
CLEA	THE CONTAMINATED LAND EXPOSURE ASSESSMENT MODEL
CSA	CANADIAN STANDARDS ASSOCIATION
DEFRA	DEPARTMENT OF ENVIRONMENT, FOOD AND RURAL AFFAIRS
DN	DIGITAL NUMBER
DNAPL	DENSE NON-AQUEOUS PHASE LIQUIDS
GIS	GEOGRAPHIC INFORMATION SYSTEM
GMP	GOOD MODELLING PRACTICES
GPR	GROUND PENETRATING RADAR
GUI	GRAPHICAL USER INTERPHASE
GWFCT	GROUNDWATER FLOW AND CONTAMINANT TRANSPORT
GWL	GROUNDWATER LEVEL
ICRCL	INTERDEPARTMENTAL COMMITTEE FOR THE REDEVELOPMENT OF CONTAMINATED LAND
IEMA	INSTITUTE OF ENVIRONMENTAL MANAGEMENT AND ASSESSMENT
ISO	INTERNATIONAL STANDARDS ORGANISATION
LNAPL	LIGHT NON-AQUEUOUS PHASE LIQUID
MNF	MINIMUM NOISE FRACTION
NAPL	NON-AQUEOUS PHASE LIQUID
NDVI	NORMALISED DIFFERENCE VEGETATION INDEX
NERC	NATIONAL ENVIRONMENTAL RESEARCH COUNCIL
NRMS	NORMALISED ROOT MEAN SQUARE
PAH	POLY AROMATIC HYDROCARBONS
REIP	RED EDGE INFLECTION POSITION
RMSE	ROOT MEAN SQUARE
SC.	SCENARIO
SNIFFER	SCOTLAND AND NORTHERN IRELAND FORUM FOR ENVIRONMENTAL RESEARCH
UK	UNITED KINGDOM
US	UNITED STATES OF AMERICA
VOC	VOLATILE ORGANIC COMPOUNDS

CHAPTER 1: CONTAMINATED LAND RISK ASSESSMENT

1.1 Introduction

This research project looks at the site investigation of landfill sites, as a particular type of contaminated land. The research focused upon (1) the methods used during site investigation and (2) the modelling process, specifically evaluating the implications of data sets, site assumptions, and modelling practices used during construction of risk estimation models, e.g. groundwater flow and contaminant transport models. The overall research aim was to demonstrate that existing and innovative field sampling methods and modelling tools should be used in an integrated manner to improve the accuracy of data collected during the site assessment conducted on contaminating landfill sites. This chapter will introduce the reader to:

- the difficulties associated with landfill site assessments,
- the research aims and objectives,
- the resources that were available to conduct the research,
- how the research was executed and,
- how the thesis is structured.

Several terms are used in this thesis, requiring clarification. The term 'site investigation' is referred to as 'site assessment'. The term 'landfill site' refers to landfill waste that contains municipal waste, light industrial waste, sewage sludge and water treatment sludge. Reference to 'site assessors' refers to individuals conducting an assessment of land conditions at a contaminated site. Reference to 'modeller' refers to the individual constructing a groundwater flow and contaminant transport model. The term 'spatial' refers to 3-dimensional models. In the UK, the Environment Agency is the governing body responsible for environmental management issues in England and Wales while the Scottish Environment Protection Agency (SEPA) is responsible for management of these issues in Scotland. For ease of reference, when the term 'UK' is used in reference to the Environment Agency, the author is actually referring to the Environment Agency in England and Wales.

1.2 Background: The Difficulties of Landfill Site Assessments

Before remediating a contaminating landfill site, an investigation of geophysical characteristics must be conducted. This investigation forms a conceptual model of site conditions, providing information about the type of contaminant present, the rate and extent of the contaminant plume, and pathways of contaminant migration. The site assessment provides information needed to determine risks posed by the site to local receptors such as water resources, soil, ecosystems, and humans. The methods and instruments applied during the site assessment play an important role in providing accurate information about subsurface site characteristics. This information is also used in risk estimation models to determine the rate and direction of contaminant migration. Such models are often used to understand contaminated conditions, predict levels of risk posed to receptors and simulate remedial situations.

The site assessment of landfill sites currently faces two challenges. Firstly, field instruments and methods that are commonly used for assessment provide spatially limited data. For example, boreholes can be used to measure contaminant concentrations in groundwater around a landfill. The method has a long-standing history in water quality analysis, producing reliable results, and is cost effective. Despite these advantages, such an approach produces data sets that do not represent the extent and distribution of a landfill contaminant plume. Instead it represents contaminant concentrations at one location, at one point in time. It is also standard practice to interpolate geophysical and hydro-chemical conditions (without validation) between sampled points. This spatial interpolation between points is an assumption that is often overlooked when data are interpreted (Kjeldsen, 1998(a); Golder Associates, 2000).

A common approach is to sample at several points across a landfill site, interpolating or statistically estimating conditions between sample points. Guidance is available for selecting appropriate assessment methods and for developing sampling strategies at contaminated sites (e.g. ICRCL, 1990; Department of Environment, 1994; CIRIA, 1995; BSI 2001; Environment Agency, 2000; 2001). However there is a need to address landfill sites assessments as a separate type of contaminated land due to their site-specific and heterogeneous nature. Each site varies drastically in waste type, size, structure, hydrogeological conditions, age, leachate chemistry, plume depth, paths of migration, and risks posed to local receptors. Despite the variety of government and industry publications that offer tools and guidelines for determining assessment methods and sample densities, the site assessment is more often than not limited by financial and time constraints. As a result, the funding available for the site assessment will determine the sample density, sampling methods, and field instruments used.

The second challenge that faces the site assessment process is found in risk estimation models such as groundwater flow and contaminant transport models. Application of such computing tools requires (a) extensive field data for model construction, calibration and validation and (b) trained modellers that understand the field data, the model, and the model's capacities and limitations. If detailed and accurate data are not available, and if the modeller does not adhere to sound modelling practices, then the model will not be able to effectively simulate site-specific contaminant conditions or evaluate levels of risk posed by the site.

The modeller plays an instrumental role in determining model assumptions and dimensions, recognising model capabilities, calibrating and simulating the model to site-specific conditions, and communicating model results, assumptions and limitations to stakeholders involved in the risk assessment and remediation procedures. The data collected during the site assessment and used in model construction will also influence both the conceptual understanding of site conditions and potentially overlook conditions that could lead to groundwater and soil contamination.

1.3 Research Aims and Objectives

1.3.1 Research Aims

The aim of the project was to review and test several methods and models that could be applied when conducting landfill site assessments in order to improve the accuracy of the assessment.

The site assessment of contaminated land in the UK (namely older and abandoned landfill sites) is of growing importance since there are an estimated 20,000 unlined older landfill sites across the country (Environment Agency *cited* 2003(a-b); 2004(b)). With increasing demands for brown field development, derelict land and contaminated sites are becoming increasingly popular options for remediation and re-development. In the case of the estimated 20,000 landfill sites, it will be increasingly important for local authorities to determine the level of risk, remediation and applicable uses that these sites can provide. This is a requirement of Part IIA of the Environmental Protection Act (1990) which came into effect in April 2000 (Walker, 2000).

The research examined methods and tools that are available for use during landfill site assessments, focusing upon landfill sites that pose risk to groundwater and soil resources and upon the influence that field sampling methods, field data assumptions and assessor qualifications can have upon assessment results.

The site assessment of contaminated landfill sites is a relatively new field of applied science. As a result field methods for groundwater, soil and waste sampling have evolved from other similar fields of expertise (e.g. water quality sampling for potable water supplies). Computing innovations both in real-time measurements and multi-dimensional modelling have evolved rapidly in the last 15 years. The 'newness' of the environmental assessment industry, along with the increasing need to address landfill related risks, provided an opportunity for conducting scientific research which (a) explored new methods of site assessment and (b) investigated model sensitivities.

1.3.2 Research Objectives

The research had five objectives:

1. To test two relatively new multi-spatial field methods that have been tested in other fields of science or on other types of contaminated sites. The objective was to evaluate where these methods could provide new types of data that could be used to validate and improve the accuracy of site assessing findings.

2. To test whether geostatistical modelling could provide the site assessor with a better understanding of heterogeneous site conditions and whether it could be used to identify locations where to best place sample points at a landfill site.

3. To test the influence of data sets available when constructing a 3-D model simulating groundwater flow and contaminant transport conditions. The objective was to test whether increasing the amount of field data collected by different field assessment methods could influence how a groundwater flow and contaminant transport model was constructed and again, whether it directly influenced modelled simulations.

4. The objective was to test whether field assumptions derived from field data and 'professional interpretations[1]' influenced (a) how site conditions were constructed in the model, (b) how the model simulated groundwater flow and contaminant transport.

5. To test the influence of modelling practises and modelling assumptions when constructing and simulating groundwater flow and contaminant transport models.

Footnote 1: 'interpretation' refers to the professional judgement of the assessor or modeller

The objectives are similar and linked together in that each objective looks at a field sampling method or modelling tool that can be applied during the landfill site assessment. Each objective also identifies the sources of inaccurate data that may be overlooked when using each type of method or modelling tool. These objectives were developed based on the need to improve site assessment methods and modelling tools, when conducting a site assessment of contaminating landfill sites that pose potential or existing risk to water and soil resources, human communities or ecosystems.

1.4 Resources Available

The resources and study sites available to the project were:

- Three contaminated landfill sites with similar geophysical conditions and an abundance of historical field data. The sites are titled: Study Site A, Study Site B and Study Site C
- Support from the landfill management companies responsible for these sites, allowing full access to historical site data and field research on the sites
- Equipment support from the National Environment Research Council (NERC), allowing their ground penetrating radars (GPR) to be used for research purposes at Study Site A and Study Site C in 1999 and 2000
- Flight data collection by the NERC who conducted airborne flights over Study Site A, collecting remote sensing Compact Airborne Spectrographic Imager (CASI) images of the site
- Equipment support from NERC, using their field-based spectroradiometers for research purposes at Study Site A
- Three years of financial support by the Entrust Fund for a PhD research project focused upon 'Contaminant Flux around Landfill Sites'
- Research and academic support from the Geography Department and the Centre for Waste and Pollution Research at the University of Hull

- Groundwater flow and contaminant transport models constructed using Visual MODFLOW and MT3D and ArcView GIS models. These software packages were selected due to their cost effective, popular, and robust reputation and they are used in both academic and industry applications for groundwater flow and contaminant transport modelling.

1.5 Research Investigations

After defining the four research objectives, six investigations were carried out. Kriging models within GIS were used in investigation 1 to identify new sampling locations for determining groundwater levels at two landfill sites. Investigation 2 used GPR to model subsurface hydrogeological characteristics along leaking edges at two landfill sites. Investigation 3 applied field-based and airborne remote sensing instruments to measure the spectral reflectance of stressed vegetation, identifying locations of leachate-stressed vegetation along landfill edges and inferring pathways of leachate migration.

The second part of the research used groundwater flow and contaminant transport models to test the sensitivity of three parameters that were found to impact model results: different field data sets; grid size and hydraulic conductivity values. These three parameters tested (a) the influence of field data available during model construction and (b) the influence of field data assumptions and modelling practices on model results. Investigation 4 focused upon the implications of additional field data. Different data sets were used to construct various scenarios of each landfill, evaluating the influence of field data sets on model results. Investigations 5 and 6 tested the influence of field data on modelling practices and model results by assuming that grid size and hydraulic conductivity are model parameters that are inferred during the model building process from field data collected during the site assessment of a contaminated site. The values assigned to these two parameters can influence model-simulated groundwater and contaminant flow gradients and plume dimensions.

Before conducting the three modelling investigations, a detailed site assessment was carried out at the three landfill sites. This data was used to construct site-specific 3-D groundwater flow and contaminant transport models using Visual MODFLOW and MT3D modelling software.

1.6 Study Sites

The municipal landfill sites used, as study sites in this project were similar in that all:

- had similar geological conditions – the sites were based on strata with sand-clay lenses
- contained unlined buried landfill cells
- were identified to be leaching off site (surface and subsurface leaching) and
- posed risks to local soil and groundwater resources.

Preliminary and detailed site investigations were conducted at all three sites. The largest part of the field and modelling research was conducted at Site A due to its proximity to the University of Hull from where the research was based, and because NERC agreed to collecting airborne remotely sensed CASI data at this location. The other study sites also made important contributions, confirming the findings of each investigation and meeting the overall project objectives. Each of the six investigations used data sets from at least two landfill sites. This was done to provide stronger evidence of the results in each investigation. The only exception was with the remote sensing investigation in which only Site A data was used.

1.7 Thesis Plan

The thesis is structured into eleven chapters. The sections include: the introduction; the background literature review; a review of the methods used; descriptions of each study site; the investigation results, a discussion of research findings and conclusions (e.g. Figure 1.1).

1.7.1 The Introduction Chapters

Chapter 1 introduces the research problem, aim and objectives. Chapters 2, 3, 4 and 5 aim to give the reader a background explanation into the history of contaminated land in the UK, and the risk-based approach that has been adopted to better manage such sites. Chapter 2 provides the reader with background information about contaminated land management in the UK. The site assessment process is also reviewed, focusing landfill site investigations. Chapter 3 provides background information about landfill sites, their structure and design. Subsurface contaminants commonly found at landfill sites are also discussed, reviewing methods of contaminant plume classification and the risks posed by the chemical nature of leachate.

Chapter 4 looks at the methods used to assess geophysical conditions at a contaminated site, focusing methods used to assess soil, geology, and groundwater quality at landfill sites. Chapter 5 reviews groundwater flow and contaminant transport models, introducing the reader to uncertainties that are caused by poor field data and poor modelling practices.

1.7.2 Methodology and Study Site Descriptions

Chapter 6 describes the methods that were used in the research investigations discussing each method's application in context of the landfill site assessment, justifying why the method was applied and the strengths, weaknesses and difficulties in applying each of the methods used. It also provides background information that was used in each of the six investigations. Chapter 7 describes the geophysical and contaminant conditions at each landfill study site.

1.7.3 The Six Investigations and their Results

Chapter 8 presents the findings of the first three research investigations, relating back to: new data collection methods used during the site assessment and geostatistical modelling as a tool for defining field sampling strategies. The remaining investigations

(investigations 4, 5 and 6) are found in Chapter 9. They look at the influence of modelling practices and modelling assumptions derived from field data on modelling results.

Chapter 10 links the investigation results and their significance in relation to the project objectives, discussing the research limitations and areas needing further scientific investigation. It also summarises the research contributions, forming several recommendations in order to improve the site assessment process and to better understand the geophysical conditions at landfill sites. Chapter 11 provides a brief summary of the research findings and conclusions.

1.8 Chapter Summary

This chapter has presented the objectives and structure of this thesis, in which several cross-discipline approaches were used to conduct this research. The following chapters (chapters 2-5) will provide a background explanation to the complexities associated with contaminated landfill sites in the United Kingdom (UK): the history and legal framework of contaminated land management; the problems and risks posed by landfill sites; and the strengths and weaknesses of field techniques and modelling tools that are available for assessing geophysical conditions at contaminating landfill sites.

Figure 1.1 Chapter structure in thesis

CHAPTER 1: Introduction

CHAPTER 2-5: Background and Literature Review			
CHAPTER 2: History of Risk-based Contaminated Land Management	CHAPTER 3: Landfills and Contaminant Classification	CHAPTER 4: Site Assessment Methods	CHAPTER 5: Groundwater Flow and Contaminant Transport Modelling

CHAPTER 6: Research Methods				
Introduction to Methodology	Methods Applied In All the Investigations	Methods Applied in Three of Six Investigations	Methods Applied To Single Investigations	Limitations Faced By the Investigations

CHAPTER 7: Description of each Study Site		
Site A	Site B	Site C

CHAPTER 8: New Site Assessment Methods - Results		
Investigation 1 Results	Investigation 2 Results	Investigation 3 Results

CHAPTER 9: Modelling Analysis - Results		
Investigation 4 Results	Investigation 5 Results	Investigation 6 Results

CHAPTER 10: Discussion, Recommendations and Conclusions

CHAPTER 11: Summary of research findings and conclusions

CHAPTER 2: CONTAMINATED LAND – BACKGROUND

2.1 Introduction

This chapter will aim to:

- introduce the reader to the inherent problem of contaminated land in the UK as well as the framework of contaminated land management
- define risk management and risk assessment processes when evaluating risks posed by contaminated land
- define terms such as uncertainty, risk and hazard in the context of contaminated land
- introduce and review steps of site assessment in context of the risk assessment
- introduce the concept of risk communication and its role in the risk management of contaminated land.

2.2 The Risk-Based Framework for Contaminated Land Management

Contaminated land in most cases causes local or regional scale of contamination. The cause for alarm is in the abundance of such sites. In the UK, the Environment Agency estimates that there are up to 300,000 hectares of land in the UK affected by contamination (Environment Agency *cited* 2004(a)). Contaminated land can be linked to a number of industries. The highest risk industries and land uses causing contaminated land in England and Wales (Environment Agency, *cited* 2004(a)) are:

- The waste disposal industry – uncontrolled or illegal landfill sites
- The extraction industry – old and abandoned mines
- The energy industry – oil refineries, power stations, gas works, petroleum stations
- Chemical works
- Accidental spillage on roads or industrial sites
- Ministry of Defence sites
- The metal production industry
- Non-metal production and their by-products

- The food processing industry
- The paper, pulp and printing industry
- The textile industry
- The rubber industry
- The infrastructure production industry
- Railways

This research focused upon the waste disposal industry, more specifically upon contaminating landfill sites as a specific type of land contamination. There are an estimated 20,000 landfill sites across the UK, of which 8000 have landfilling licences issued by the Environment Agency (Environment Agency *cited* 2003(a-b); 2004(b)).

To deal with the problem of such lands, a risk-based framework has been established for the evaluation and remediation of contaminated land in the UK. Contaminated land is defined at section 78A(2) of the Environmental Protection Act 1990, cited as:

> *'... any land which appears to the local authority in whose area it is situated to be in such a condition, by reason of substance in, on or under the land, that –*
>
> *(a) significant harm is being caused or there is a significant possibility of such harm being caused, or*
> *(b) pollution of controlled waters is being, or is likely to be caused;...'*

Harm in this statutory guidance is cited as:

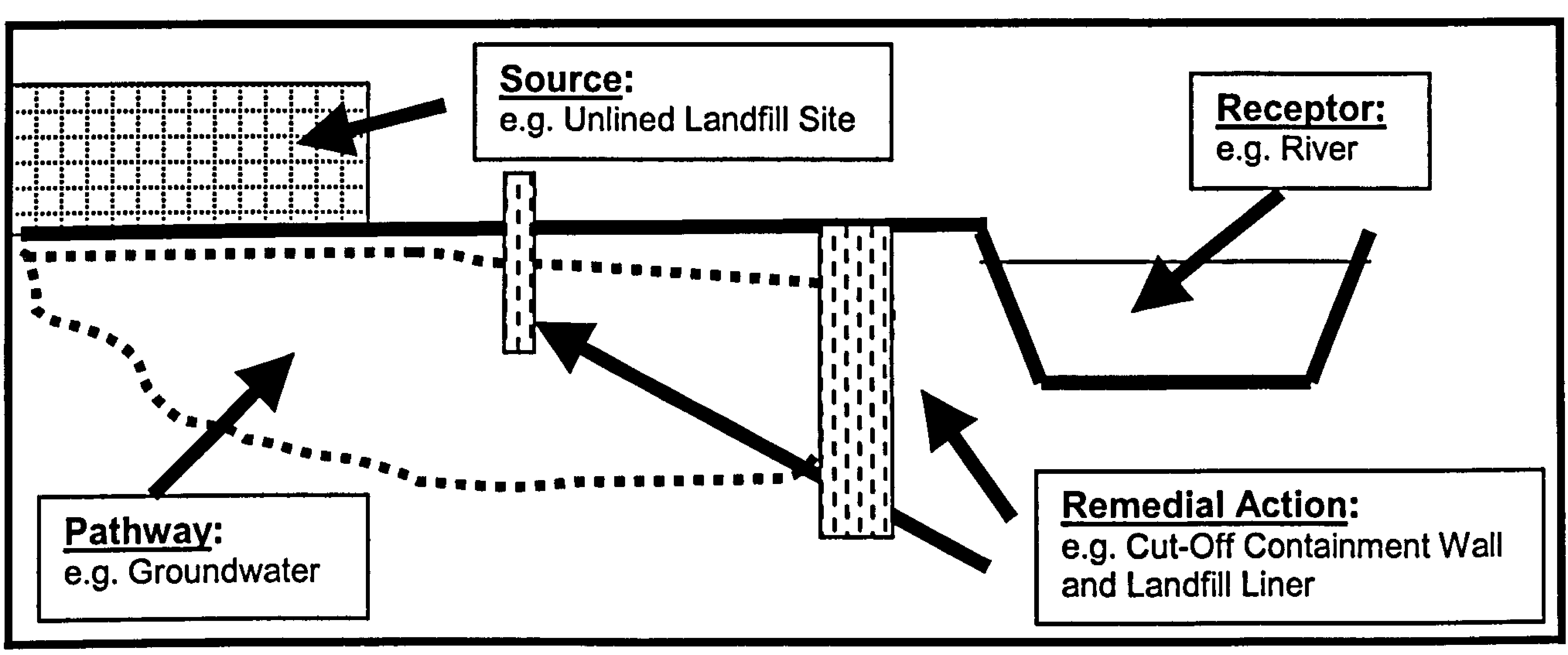
> *'harm to the health of living organisms or other interference with the ecological systems of which they form part and, in the case of man, includes harm to his property'.*

The definition of contaminated land is based on the principles of risk assessment. For the purposes of the guidance, risk is defined and cited in section 78A(2) of the Environmental Protection Act, (1990) as:

> *'(a) the probability, or frequency, of occurrence of a defined hazard (for example, exposure to a property of a substance with the potential to cause harm); and*
> *(b) the magnitude (including the seriousness) of the consequences'.*

The main approach to risk assessment as adopted by statutory guidance is to establish the presence of a source-pathway and receptor (Figure 2.1). The contaminant source is a substance in or under the land with the potential to cause harm or to cause pollution of controlled waters, (e.g. soil and groundwater contaminated with landfill leachate). The pathway is a means or way through which the receptor is being or could be exposed to, or affected by that contaminant (e.g. leachate moving through groundwater into a nearby river). The receptor can be living organisms, groups of living organisms, an ecosystem or some types of property or controlled water (e.g. the endemic ecosystems living in the river, which receives leachate-contaminated groundwater or the local community whose drinking water supply is contaminated from the leachate-contaminated groundwater). There must also be proven risk to the receptor. In order to establish a 'pollutant linkage' all of the three elements must be present and the pathway needs to be a means by which the contaminant is causing or potentially causing significant harm to the receptor (Hooker *et al*, 2000).

Figure 2.1 The Source-Pathway-Receptor framework that underlines the UK's approach to contaminated land management

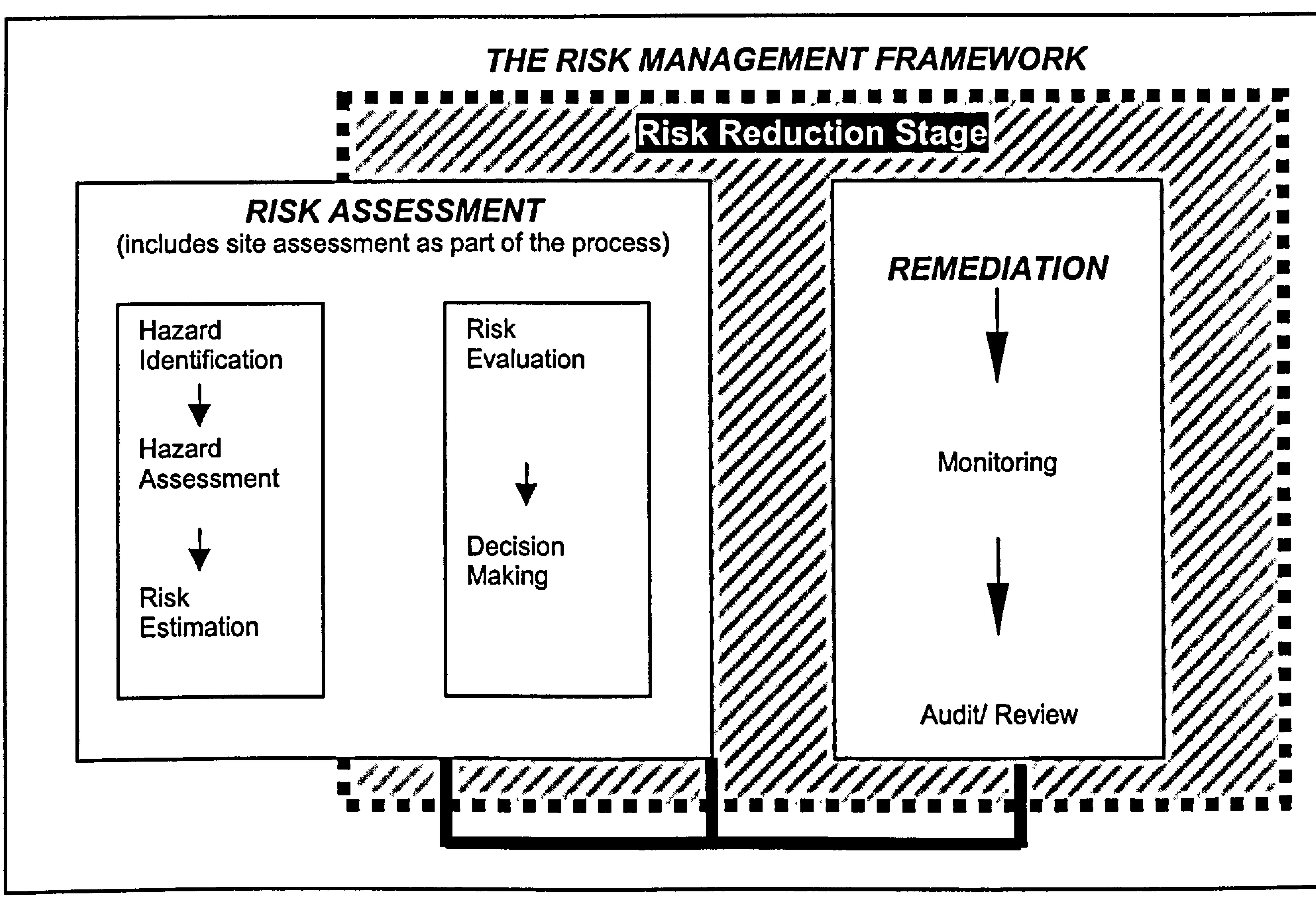


2.3 Risk Management

2.3.1 Risk Management Defined

Risk management in this context, consists of assessing risks (identifying contaminated sites and analysing the existing or potential hazards and harm posed by the site) as well as reducing the risk posed by contaminated sites (remediation of contaminated sites) as shown in Figure 2.2. Risk management incorporates the process of assessing risks, taking decisions based on those risks, and taking actions to reduce the risk as much as possible. It is distinctive from the risk assessment as it deals with the legal, political and administrative aspects of risk. It promotes the application of an objective and systematic assessment, providing a framework for transparent, consistent and defensible discussions for proposed courses of action. It also allows for the assessment of uncertainties that may have been inherited from uncertainties in the site assessment (Petts *et al*, 1998, p.2; 27).

Figure 2.2 The UK risk-based framework for contaminated land management: data collected during the site assessment play an important role in risk assessment results (Adapted from Petts *et al*, 1998, p.3)



The site assessment and the risk estimation are part of one process in which scientific data are collected to provide evidence of site conditions at a contaminated site. The risk estimation is an ongoing process that uses scientific data collected during the site assessment to identify and evaluate whether a source of contamination is present, whether a pathway of contaminant migration exists and whether there is a receptor (e.g. Figures 2.1 and 2.2). If the risk assessment confirms that the pollutant linkage is present and that risk is posed to the receptor, then remediation of site conditions will likely follow.

2.3.2 History of Contaminated Land Management in the UK

The implications of a long industrial history have resulted in a large number of contaminated sites across England and Wales. On April 1, 2000 the 'Contaminated Land Regime' came into effect in England. Prior to this new regime, contaminated land was dealt with and managed under a number of statutory regulations in which UK contaminated land polices that often lacked coherence (Walker, 2000). Before April 2000, contaminated land in the UK was viewed as a problem for town-planners and redevelopers to address. In 1989 the House of Commons Select Committee on Environment presented its first report on the extent of contaminated land in the UK, arguing that there was a lack of knowledge and awareness about the extent of this problem (HMSO 1990). Until then, risk and remediation of such land until then focused upon identifying potential risks for future land users if the land use was to change. The 1990 Environmental Protection Act (EPA) brought increasing attention to contaminated land. In 1991 a draft statutory guidance was added to the 1990 EPA, in which it was estimated that some 100,000 sites have been contaminated as a results of industrial activities of the past (Walker, 2000). The draft, which faced a lot of resistance and watering down in 1992-1994, required local authorities to establish registrars of potentially contaminated land. In 1994, the Department of Environment consultation paper 'Paying for Our Past' which formed the basis for section 57 of the Environment Act 1995, which was inserted as Part IIA of the Environmental

Protection Act 1990. Section 57 made local authorities responsible for land surveying in order to establish locations of contaminated land. Local authorities also became responsible for keeping registers of both contaminated land and land that had been remediated. Part IIA follows the polluter-pays principle and is criticised for being too complex, and for placing too many financial and technical responsibilities on local authorities (Walker, 2000). The risks and awareness of dealing with industrial contaminated land (including derelict waste sites) continued to increase in the late 1990's growing in parallel with the increasing economic opportunities for re-use of such land.

On April 1, 2000, the Contaminated Land Regime came into effect. The regime is based on several activities: identifying the problem, assessing the risks, determining the appropriate remediation requirements, establishing who should pay and implementation and remediation (DETR *cited*, 2004; Environment Agency, *cited* 2004(a)). It requires local authorities to:

(a) Inspect their areas for contaminated land
(b) Determine whether particular sites are contaminated
(c) Act as an enforcing authority for sites that pose risk but are not the responsibility of the Environmental Agency.

The Regime also states that there are two steps in determining whether land is classified as contaminated. Firstly, the local authority must be sure that the three components (e.g. the source, pathway and receptor) have been identified in order to establish a 'pollutant linkage' (e.g. Figure 2.1). Secondly, the local authority must provide evidence that shows that a pollutant linkage exists and that it is:

(a) resulting in significant harm to the receptor
(b) presenting a significant possibility of significant harm being caused to the receptor
(c) resulting or is likely to result in the pollution of controlled waters (Environmental Agency, *cited* 2004(a)).

In most cases it is the local authority that is expected to inspect contaminated areas, establish responsibilities, ensure remediation and keep a public register detailing regulatory action under taken for each contaminated site. However, the Environment Agency takes responsibility for sites that have been formally identified as 'special sites'. These sites include contaminated land where an aquifer is at risk, oil was refined, explosives were manufactured or if the site is occupied for Ministry of Defence purposes (Morgan, Lewis & Bockius LLP, *cited 2004).*

Due to the level of risk posed by contaminated land, a series of guideline documents have been developed to assist in the management of contaminated land. Some examples include the 'CLR' reports such as Department of Environment (1994) and DEFRA and Environment Agency (2002(a-d)). Such documents provide guidance to assist local authorities and other stakeholders as well as professionals dealing with land remediation. The regime contains three statutory parts:

1. The DETR Circular (February 2000) titled 'Contaminated Land' providing a government policy statement that describes the laws and statutory guidance related to this issue.
2. Part IIA of the Environmental Protection Act 1990 that was inserted by section 57 of the Environment Act 1995. This preliminary legislation provides a definition of contaminated land, requiring a risk-based approach to identifying and managing contaminated land (Environmental Act, 1995).
3. The Contaminated Land (England) Regulations 2000 that deals with procedural details including descriptions of special sites, public registers, remediation notices and appeals.

2.4 Risk Assessment Definitions

2.4.1 Risk Assessment

Risk assessment is defined by the US National Research Council (NRC, 1994, p.4) as a systematic approach to organise and analyse scientific knowledge and information for potentially hazardous activities or for substances that might pose risks under given conditions. Risk evaluations may be quantitative, semi-quantitative, or qualitative, aiming to characterise the source, pathway and receptor and evaluate whether the risks posed to the receptor are acceptable or require measures of contaminant control and reduction. The risks posed by contaminated land are related to the potential for damaging soil resources, the ecosystem, water supplies, buildings and infrastructure and public health (Petts *et al*, 1998, p.1 & 29).

2.4.2 Harm, Hazard and Uncertainty

Harm, hazard and uncertainty are commonly used in risk management. Harm has been defined as part of section 78A(2) while hazards in relation to contaminated land are an 'event or situation' which have the potential to cause harm to targets of concern (e.g. human, ecological, physical, financial and psychological). Risk, in this context, combines the frequency and probability of a harm being realised, along with an estimation of the scale and magnitude of its affects (Petts *et al*, 1998, p.29).

Uncertainty, in relation to the management of contaminated land, refers to the amount of estimated or unknown data (often collected during the site assessment) and used in risk assessment. In most cases uncertainty is linked to a lack of knowledge. Common sources of uncertainty, as listed by Columbia-Wharton/Penn Round Table *cited* (2003) are systematic error, subjective judgement, linguistic imprecision, disagreement, approximation, statistical variation, variability and inherent randomness or unpredictability.

These sources of uncertainty can be classified into two groups. The first is a lack of good data due to systematic error, subjective judgement, imprecision, disagreement and approximation. The second is related to variability in data sets, which includes statistical variation, heterogeneity, inherent randomness or unpredictability. These are two distinct characteristics of data collected during the site assessment, having different implications for the risk assessment process and remedial decision-making that follows. It is important that these sources and classes of uncertainty as well as their implications are (a) correctly documented, (b) accurately reported, and (c) explained or communicated effectively to stakeholders during the risk assessment or remedial decision-making process.

2.5 Phases of the Risk Assessment

Risk assessment in the UK (e.g. Figure 2.1) consists of several phases that will be discussed: (a) the hazard identification; (b) the hazard assessment; (c) the risk estimation; (d) the risk evaluation; and (e) the remedial decision-making phases of contaminated land risk assessment.

(a) Hazard Identification and Hazard Assessment

The first phase of risk assessment consists of hazard identification. This phase uses findings from the site assessment (phase I – the preliminary study) to identify the contaminant source, site-specific contaminants of concern, environmental factors that could be affected, potential routes of contaminant migration, potential targets (including their sensitivity and characteristics) and the nature of exposure. It also tries to construct a conceptual model of site-specific conditions (e.g. Table 2.1).

The second step is hazard assessment, which aims to refine the initial conceptual model (e.g. Table 2.1). The aim is to provide a detailed description of existing and potential pathways of contaminant migration, the transport mechanisms and environmental factors that will determine contaminant transport gradients and directions, and a detailed description of the target. This is done using data from the exploratory and detailed investigation of the site assessment. In order to assess hazards, soil and water quality samples are often compared to generic standards and guidelines for water and soil quality. This allows the assessor to judge whether the levels of site-specific contaminant concentrations are excessive when compared to relevant standards or guidelines. An example is to use soil quality guidelines outlining toxicological impacts on human health outlined in DEFRA and Environment Agency, (2002(c)). The hazard assessment usually has one of three outcomes. Firstly, it may conclude that site-specific contaminant levels do not pose a risk to targets meaning further action is not required. Alternatively, further field data and field assessment may be needed. Lastly, it may provide data that are out of standard (e.g. soil quality that can have toxicological impact on human health) indicating the presence of a risk and pointing to the need for remedial action.

(b) Risk Estimation and Risk Evaluation

The risk estimation phase aims to estimate the possibility of an unwanted outcome under given conditions (Harris *et al*, 1995). It uses the findings of the hazard assessment if further investigations are needed or if the need for remedial action is confirmed. Risk estimation can be conducted in two ways: either by estimating the level of exposure (exposure assessment) or by estimating the level of effects (effect assessment). Both methods use data from the detailed investigation (listed in Table 2.1). Exposure assessment can for example estimate the rate of contaminant migration given different (a) pathways of migration, (b) contaminant transport mechanisms and (c) environmental factors (e.g. hydrogeological factors at a landfill site). Groundwater flow and contaminant transport models or similar environmental models integrated with GIS are frequently used tools in exposure assessment of contaminated groundwater and soils, using data collected during the site assessment (Table 2.1) for model construction.

Effect assessment aims to determine the 'dose-response' relationship between the target and the contaminants. It evaluates the impact of contaminant concentrations on different targets by describing the target in detail (e.g. age, gender, health status, species, characteristics, and physical properties) and quantifying the environmental or health effects that the contaminants will have on the target. The different methods and models for estimating threshold contaminant concentrations fall outside the scope of this project. Further details can be found in Suter (1993); McDonald (1996); Petts *et al,* (1998), p.255-259; and DEFRA and EA (2002(a-d)).

The fourth phase of the risk assessment process is risk evaluation. It uses results of the hazard assessment and risk estimation to form conclusions about risks posed by a given contaminated site. It aims to provide a multi-scenario analysis of 'what if' and 'worst-case' situations, identifying the cost-benefits of given remedial actions, and outlining the uncertainties of these evaluations. It also identifies the appropriate standards and guidelines applicable to site-specific conditions.

2.6 The Site Assessment

2.6.1 The Site Assessment Defined

The site assessment an important part of the risk assessment. The accuracy of information collected during the site assessment in context of the Contaminated Land Regime (Environmental Protection Act, 1990 section 78A(2), HMSO, 2000) becomes increasingly important. Identifying geophysical conditions during the site investigation will influence the risk assessment outcome, the remedial decisions made and the ability to protect potential and existing targets (AGS *cited* 2001). When using different risk communication models such as the analytic deliberative model (Stern and Finberg, 1996), the importance of accurately assessing site conditions is also stressed since communication relies upon reputable and rigorous methods of analyses.

In the UK, the site assessment consists of at least five phases: (1) the preliminary study consisting of the desk study and the walk-over study; (2) the exploratory study; (3) the detailed study; (4) the supplementary phase; and (5) the post remedial monitoring and assessment. There are a variety of terms that are used to describe the site assessment, depending on the scope of the risk assessment and depending on the country that assessment is being conducted in. The terms 'phase I, II, III' are commonly used in British, American, Canadian and ISO guidance for conducting land assessments. In contaminated land assessment conducted in Britain, the 'phase I' assessment generally consists of the desk study of historical and documented site information and the walk over study, which is an initial site assessment of conditions. The 'phase II' assessment includes the exploratory study and the detailed study. The former is an initial data collection of background and historical information about site conditions. The latter requires the assessor to conduct a comprehensive review of geophysical conditions of the site. The 'phase III' study includes a supplementary investigation and post-remedial monitoring of a contaminated site. It is important to note that the terms 'phase I, II and III' are popularly used terms in environmental auditing (e.g. CSA, 1999; ASTM, 2000(a); ASTM, 2000(b); CSA, 2000). Canadian and US literature use similar definitions for the three phases however their guidelines are written for any type of environmental site assessment (not exclusively for contaminated land) and they do not extensively focus upon establishing a pollutant linkage (CSA, 1999; ASTM 2000(a-b); CSA, 2000). Ecological assessments in the UK also use the terms 'phase I and II', but they have different meanings (Petts *et al,* 1998, p. 53, 62-64). Alternatively, the British Institute of Environmental Management and Assessment (IEMA) considers the assessment of land conditions to be comprised of phases I (a), I (b) and II which comprise of eight steps: the desk study, the walk over study, the exploratory investigation, the full intrusive investigation, laboratory analysis, modelling, monitoring and verification (IEMA, *cited* 2003). This thesis has focused upon phases I and II (assuming that the phase I includes the preliminary study, and the phase II includes the exploratory and detailed investigations) as these assessments provide geophysical information used to determine risks and remedial actions at a given contaminating landfill site.

2.6.2 Phase I: The Preliminary Study

The assessor conducting a preliminary study has three objectives during the preliminary assessment. The first is to check for contaminants that could affect the suitability of the site in its current form or future use, and to assess the nature and extent of the contamination. The second aim is to determine whether any specific procedures and precautions need to be taken during engineering and other activities on the site. Thirdly, to collect site information, establishing whether further ground investigation is necessary and providing baseline information for the design of an effective ground investigation, should this be necessary. The assessment report needs to reflect the above objectives, describing the site's past uses, identifying the nature and extent of contamination within the site's vicinity, and identifying materials that might need to be removed from the site. The report should formulate recommendations on disposal, and alternative remediation methods. Immediate dangers to public health, safety, and the environment need to be specifically addressed. It should also determine the potential for contaminant migration, earmarking site limitations that might influence the cost and effectiveness of remedial actions (ASTM, 2000(a-b); IEMA *cited* 2003).

2.6.3 Phase II: The Exploratory and Detailed Study

The explanatory study is the interim phase conducted after the preliminary and before the detailed study, aiming to better characterise the findings of the preliminary study, to determine whether a detailed study is needed, and if needed, to determine where to locate the detailed study. It can include initial sampling (e.g. drilling trial pits, water quality samples) to determine study locations for the detailed study.

There are three objectives during the detailed study. The first is to identify the main risks and circumstances based on the preliminary and explanatory study results. The second objective is to integrate the data collected and develop a quantitative simulation model of hydrogeological conditions. The third is to develop an agreement with the regulatory authorities and stakeholders in assigning appropriate protocols for risk evaluation. As listed in Table 2.1, information about soil, geology, contaminant concentrations, and hydrological features needs to be collected and analysed during this phase in order to establish whether the source, pathway, and receptor are linked. The table also lists the objectives of the site assessor during the preliminary, explanatory and detailed study and the type of data that can identify the source, pathway or receptor.

2.6.4 The Site Assessor and Good Assessment Practices

The site assessment (preliminary, explanatory and detailed studies) is often exposed to some level of uncertainty. Site uncertainties that can be linked back to the site assessor include:

(1) Overconfidence in a specific field assessment instrument or method to identify hazards posed by the site during the detailed study

(2) Over-reliance on identifying contaminants present rather than identifying potential pathways of contaminant migration

(3) Fitting generic standards to site-specific conditions in order for a risk assessment to be possible using exploratory and detailed study findings.

The collected data are also open to human error during interpretation which can be influenced by: (a) the methods used to assess site conditions; (b) the spatial and temporal distribution of data collected; (c) the types of data collected; and (d) the amount of money and time available for site assessment (Petts *et al*, 1998, p.72-73; ASTM, 2000 (a-b)).

Good assessment practices when conducting site assessments of contaminated land need to be upheld in order to avoid these uncertainties. A wide variety of documents outlining good assessment practises have been produced in recent years in the UK (Table 2.2). A detailed list summarising the different guidelines is found in 'Assessment of Risks to Human Health from Land Contamination: An Overview of the Development of Soil Guideline Values and Related Research – R&D Publication CLR 7' which was issued in 2002 (DEFRA and Environment Agency, 2002(a)). The individual(s) conducting the site assessment have a direct influence on (a) the assessment findings and (b) in the way that these findings are reported. These two issues require further attention given that the professionals dealing with contaminated land assessments should be highly trained and qualified to conduct such evaluations. One way to overcome uncertainties in the site assessment and reporting is to develop a mechanism that ensures that properly trained professionals conduct site assessments. A new example of this in the UK is the 'SiLC PTP' registration run by the Institute of Environmental Management and Assessment (IEMA) issuing licenses to site assessors (IEMA *cited* 2004). The accreditation verifies the level of understanding needed to conduct land condition assessments and also encourages adherence to standardised assessment procedures and record keeping. A less rigorous alternative, which has also been undertaken in the UK in recent years, is to develop good assessment guidelines that provide guidance on:

(a) conducting a detailed assessment of potential and existing risks posed by hydrogeological and contaminant conditions
(b) risk estimation modelling in order to determine the potential of site contaminants to cause risk and harm to groundwater beneath or off the site
(c) developing a groundwater-chemistry database

(d) analysis of data sets (collected during the site assessment) for trends, and

(e) remedial procedures to prevent contaminant migration from the site during operation

Examples of some guidelines in the UK are DEFRA and Environment Agency, 2002(a-d), and McMahon *et al,* 2001(c). One way to promoting these practises would be to organise training courses through professional industry associations related with contaminated land and landfill assessments (e.g. IEMA, the Chartered Institution of Wastes Management (CIWM) and Association of Geotechnical and Geoenvironmental Specialist (AGS)).

Table 2.1 Data collected during the site assessment of contaminated land, and added into a GIS data base can describe the source, pathway, and receptors (Based on ASTM, 2000(a); ASTM, 2000(b); Hooker *et al*, 2000; AGS, *cited* 2001; IEMA, *cited* 2003)

Data Collected During the Site Assessment:	Phase 1:	Phase 2:			
	Preliminary Study Information	Exploratory and Detailed Study Information	Exploratory and Detailed Study Information	Detailed Study Information	Detailed Study Information
	Information about Site Conditions collected during the Desk Study and Walk-Over Study: • Site location, size and ownership, site uses, past, present, and future • Adjacent land uses within 1 km of the site, and boundaries of the site) • Review of statutory records • Location of services (buried and overhead, live and disused) • Abstraction wells and surface water (presence within 500 m of site) • Water Protection Zones • Access (by assessment, monitoring or remedial equipment, by public etc.) • Proximity to natural areas and wetlands	Description of Soil: Unsaturated zones Thickness Permeability Organic content Temperature Saturation levels pH Clay content Chemical quality	Description of Lithology: Stratigraphy Permeability Structure Thickness	Description of Hydro-geology: Depth to Groundwater Groundwater Chemistry Direction of flow Water table levels Aquifer type Aquifer thickness Hydraulic gradient	Description of Contaminants: Classifying types of species Concentration and extent Solubility and viscosity Physical state Nature of source
Objectives, tasks and evaluation to be conducted by the assessor in order to identify the pollutant linkage	Identify and locate: • indicators of land use change and drainage patterns • geology and hydrological factors at site • threat of contaminate to groundwater and public water supplies • archive information about chemical quality of soil, surface water, groundwater, building fabric to identify contaminants, their mobility and toxicity • contaminated areas adjacent to site	Builds on findings of Phase I by: • verifying and further exploring subsurface features of the site • digging trial pits to sample surface contamination • sampling site conditions to determine optimal sampling locations for the detailed study		Conducts full intrusive investigation of site conditions: • collects soil and water samples and conducts laboratory analysis • geophysical survey of subsurface • biodiversity study of local waterways, flora and fauna to identify ecosystem stress levels • Modelling – GIS and risk estimation modelling to simulate groundwater flow and contaminant transport	
Identifies Source	✓	✓	✓	✓	✓
Identifies Pathway	✓	✓		✓	✓
Identifies Receptor	✓			✓	✓

Legend:
✓ = confirms that data collected at this stage of the site assessment should provide information about the presence of a contaminant source, pathway or receptor

Figure 2.3 Elements of the risk assessment: this research focused upon the site assessment (adapted Fairman and Mead, *cited* 2003)

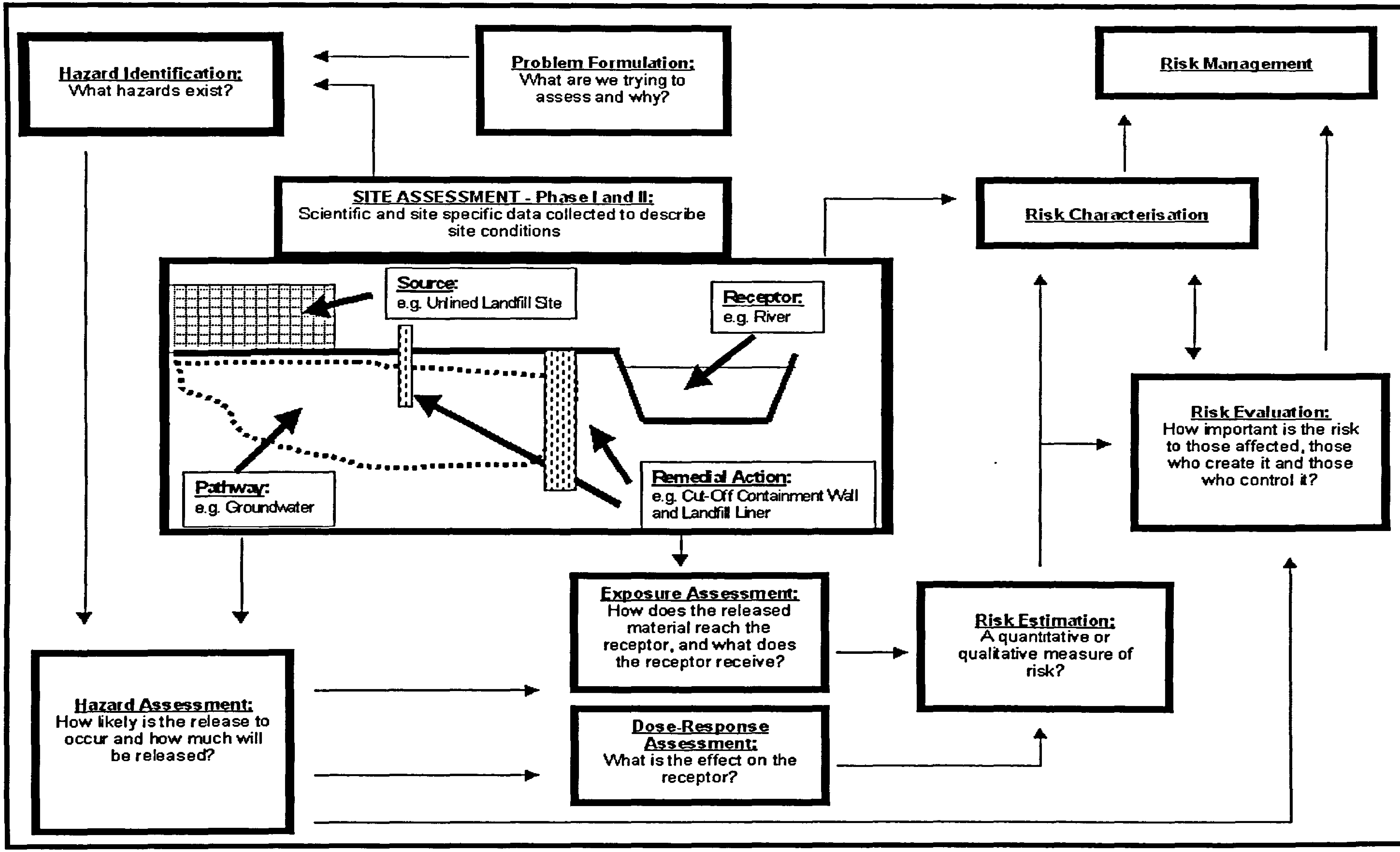


Table 2.2 A list of recent UK 'Contaminated Land Reports' (CLR) outlining good assessment practises that are useful during the site assessment.

CONTAMINATED LAND REPORT (CLR #)	*TITLE.* AUTHOR. PUBLISHER, YEAR OF PUBLICATION.
CLR 1	*A framework for assessing the impact of contaminated land on groundwater and surface water.* Report by Aspinwall & Co. Volumes 1 & 2. Department of Environment, 1994.
CLR 2	*Guidance on preliminary site inspection of contaminated land.* Report by Applied Environmental Research Centre Ltd. Volumes 1 & 2. Department of Environment, 1994.
CLR 3	*Documentary research on industrial sites.* Report by RPS Group plc. Department of Environment, 1994.
CLR 4	*Sampling strategies for contaminated land.* Report by The Centre for Research into the Built Environment, The Nottingham Trent University. Department of Environment, 1994.
CLR 5	*Information systems for land contamination.* Report by Meta Generics Ltd. Department of Environment, 1994.
CLR 6	*Prioritisation and categorisation procedure for sites which may be contaminated.* Report by M J Carter Associates. Department of Environment, 1995.
CLR 7	*Assessment of Risks to Human Health from Land Contamination: An Overview of the Development of Soil Guideline Values and Related Research – R&D Publication CLR 7.* DEFRA and Environment Agency, March 2002.
CLR 8	*Potential Contaminants for the Assessment of Land – R&D Publication CLR 8.* DEFRA and Environment Agency, March 2002.
CLR 9	*Contaminants in Soils: Collation of Toxicological Data and Intake values for Humans – R&D Publication CLR 9.* DEFRA and Environment Agency, March 2002.
CLR 10	*The Contaminated Land Exposure Assessment Model (CLEA): Technical Basis and Algorithms– R&D Publication CLR 10.* DEFRA and Environment Agency, March 2002.
CLR 12	*A quality approach for contaminated land constancy.* Report by the Environmental Industries Commission in association with the Laboratory of the Government Chemist. Department of Environment, 1999.

2.6.5 Site Assessment and Risk Assessment: Integrated

The framework for risk management has been briefly reviewed. The aim was to better explain the importance of site assessment findings and the influence that this scientific data (which provides evidence of site conditions) can have in determining the results of the landfill risk assessment and in determining remedial actions. Figure 2.3 lists the phases of site assessment showing that these data are used through out the risk assessment. If the site assessment findings are inaccurate there will be compounding effects on the risk assessment and on the success of landfill remediation.

2.7 Risk Communication and the Analytic Deliberative Process

Understanding and communicating risks to stakeholders influenced by a given contaminated site is an important element tied to risk management. Communicating landfill-related risks to stakeholders requires specific attention because these sites are often located close to densely populated areas in the UK. They are also often perceived negatively, as contaminating, nuisance causing and health-risk related sites. Stakeholders in many such cases are investors, current landowners, local residents, local authorities, government agencies, non-governmental environmental groups, or academic researchers concerned with the site's effects, risks and remediation. Their interests are varied with different levels of technical understanding, especially when interpreting site conditions and risks. It is important to consider how effectively such risk assessment information is presented and communicated to stakeholders and how these stakeholders can influence the outcome of the risk assessment and remediation.

Risk communication and its implications on the decision-making process has evolved into a new form of scientific research which aims to link the gap between risk conflicts, risk assessment, risk communication and the decision making process. In developed democratic societies, communicating risks to stakeholders is often done through some form of educational process (e.g. presentations to stakeholders, round table discussions, open public forums, and published reports). There are many models that describe risk communication in relation to contaminated land (Contaminated Soils Forum, *cited* 2003). The limitation of many such models has been their one-way communication in that they do not take the stakeholder understanding of scientific issues or social perspectives into account and they do not allow for two-way stakeholder communication and feedback. Some examples of risk communication models that have been applied to contaminated land case studies include the conduit model, the hazard and outrage model and the capacity building model (Contaminated Soils Forum, *cited* 2003). An alternative model, that may overcome the limitations of previous models, is the analytic deliberative process developed by the US Research Council in 1996 (Stern and Finberg, 1996). The model is a tool for dealing with risk, integrating scientific issues and democratic processes such as public deliberation. It considers risk assessment to have two discreet and linked parts. The first part is analysis and the second part is deliberation. Analysis assumes that 'experts' (e.g. land assessors conducting the site assessment of a contaminated site) use reputable and rigorous methods in collecting factual and quantitative information about an issue. Deliberation assumes discussions with stakeholders, reflection, and persuasion, considering stakeholder issues that have been raised and increasing the stakeholders understanding to arrive at substantive decisions. The outcome of this ongoing process is that the analysis brings new information into the risk assessment, while deliberation brings new questions, insights and solutions. The two parts complement and build on each other (Contaminated Soils Forum *cited*, 2003).

There are, however, several disadvantages to this model, similar to other risk communication models. In context of contaminated land assessments, there are a number of uncertainties that stakeholders need to be aware of when understanding site conditions and results from documented reports or interpolating risk estimation models that are used for simulating site conditions or remedial options. The second is that time and money are needed to conduct an effective analysis of field conditions and to allow for deliberation. The lack of support – financial or time - will limit the effectiveness of this model. Guglielmo Kinney and Leschine (2002) state that stakeholders involved in the process need to have adequate technical and scientific knowledge in order to contribute effectively to this process. The public deliberation also needs to be fair and effectively executed. The study was based on a risk assessment conducted at a nuclear reservation which had old plutonium deposits, located near the Columbia River in the US.

Despite these limitations, the analytic deliberative process could be successfully applied within the risk-based framework for contaminated land management since it was initially developed to allow for a better understanding and discussion of uncertainty through stakeholder input as the site assessment is being conducted. The process allows for data and assessment findings to be explored and if necessary, the investigation can be changed during the process.

The Contaminated Land Regime (Environmental Protection Act 1990; Environment Act, 1995; HMSO, 2000, Environment Agency *cited* 2004(a)) states that local authorities are responsible for inspecting their areas, adopting, publishing and implementing formal strategies and time scales for inspection and remediation of such land. The analytic deliberative model provides a feedback platform from which local authorities can communicate these strategies to industry and community representative. An important tool that supports both risk assessment and risk communication with stakeholders is Geographic Information Systems (GIS). It has become a standard tool of contaminated land management in local authorities across the UK, (e.g. Hooker *et al*, 2000) as it is able to map and model contaminated land conditions, monitor changes over time, integrate different data sets and assist in identifying pollutant linkages. Since the public's understanding of complex risks related to contaminated land will rarely be sufficient, GIS provides a valuable tool for improving communication and explaining site conditions, risks, and remedial options to stakeholders. The analytic deliberation model provides a framework for communicating landfill related risks, remedial options and site assessment uncertainties to stakeholders through deliberation (e.g. public discussions, debates, trials). In turn this can educate stakeholders and provide stakeholder-feedback when planning remedial actions.

2.8 Chapter Summary

This chapter has introduced contaminated land management and the legal framework that has been developed in Britain in the last decade. The site assessment process has been reviewed in context of the risk management framework. The following chapter will focus on landfill sites, as a particular type of contaminated land in the UK. The chapter will discuss factors that describe landfill design, geophysical complexity and site-specific nature as well as review the implications of taking a risk-based approach to landfill risk assessment.

CHAPTER 3: LANDFILL ASSESSMENT AND MANAGEMENT

3.1 Introduction

This chapter will focus upon landfill sites as one distinctively defined component of land contamination. The aim of the chapter is to:

- Provide a brief review of landfill management strategies in the UK
- Review facts about landfill leachate creation and how its potential risks can are classified and addressed in UK legislation
- Explain how modern landfills are designed and engineered, aiming to better control and prevent leachate contamination
- Explain the fundamentals of (a) waste degradation and (b) landfill hydrology, as this natural cycle drives leachate migration from a landfill site
- Review two ways in which subsurface contaminant plumes at landfill sites and other types of contaminated sites are characterised, with focus on a risk-based approach to characterising contaminant plumes
- Describe the chemical properties of leachate
- Discuss statutory and best-practise guidance that is needed when taking a risk-based approach to landfill management and the implications of failing to do so.

3.2 Landfill Sites: An Introduction

Waste production is a major issue in the UK. From 2000/2001 through to 2001/2003 there was an increase in 1.3 percent, in which approximately 522 Kg of municipal waste per person per year was collected in 2001/2002 across England (DEFRA, *cited* 2003(a)). In 2001/2002 an estimated 28.8 million tonnes of municipal waste was produced in England. Approximately 77 percent of this waste was disposed of in landfill sites (DEFRA, *cited* 2003(a); 2004(b)). Once waste is deposited in the landfill, water flows through the landfill waste, producing and discharging a toxic liquid called leachate that threatens and contaminates soil and water resources. Landfill sites are usually a small part of very large regional hydrological systems, yet their effects on local and regional groundwater and soil quality are increasingly large-scale and long-term. Risks posed to groundwater and soil quality as a result of leaking landfill sites are a concern for several reasons:

- All types of landfills contain leachate (a mixture of landfill chemicals that react with the local geology and hydrological conditions) that can migrate into local soils and groundwater, deteriorate soil and water quality, damage local ecosystems, and contaminate potable water supplies.
- There are a large number of older and abandoned landfill sites having very little background information available. This makes it difficult to evaluate risks posed by the site.
- Newer landfill sites are engineered to prevent and control leachate migration but are still known to contaminate local and regional groundwater and soil quality.

In order to accurately assess and manage landfill sites, it is important to understand their complexities. This includes knowing and understanding how the landfill was engineered; the waste age and composition; the surrounding hydrogeological factors; and the leachate plume characteristics.

3.3 Landfill Management in the UK

In the UK, landfill management is an important part of the risk-based approach to contaminated land. Solid waste disposal is one of the main sources of groundwater contamination, particularly among old, unlined landfills and those landfills with remedial structures that have weakened. Landfills in the UK are currently regulated by the Environment Agency through either waste management licensing under the Environmental Protection Act (1990) or the Pollution Prevention and Control (PPC) Regulations (2000). Further regulations are being developed, entitled 'The Landfill Regulations', which will implement the additional requirements of the Landfill Directive (1999). These will supplement and amend the PPC Regulations 2000 (Environment Agency, *cited* 2004(b)). Under these regulatory regimes, landfill operators, landfill owners and local authorities have designated roles and responsibilities in preventing, evaluating and remediating contaminated landfill sites. Those who are responsible for the land need to use the best available techniques for preventing contamination, disposing, managing and monitoring leachate, controlling landfill gas emissions, and preventing and controlling other environmental effects (Environment Protection Act, 1990; Landfill Directive, 1999; HMSO, 2000; Pollution Prevention and Control Regulations (2000); Environment Agency *cited* 2004(b)). The Environmental Agency issues site-specific landfill licenses that define conditions for monitoring site conditions, the storage, treatment, recycling, and final disposal of waste (Landfill Directive, 1999; Environment Agency, *cited* 2003 (a, b)). New facilities also have to comply with the Landfill Directive (1999), ensuring their location does not pose environmental, health or other risks. Design, planning and location of any new waste management containment facility must follow statutory and Environmental Agency approval using the best available cost effective techniques (Department of Environment, 1995; Landfill Directive, 1999). The design, construction, operation and preventative assessment of landfill sites is defined in documents such as 'Waste Management Paper 26b Landfill Design, Construction and Operational Practices' (Department of Environment, 1995, currently under revision by the Government) and the Landfill Directive (1999), outlining proactive methods that require landfill managers/operators/owners to focus on preventing and managing site-specific landfill risks. The Environmental Protection Act of 1990 (Section 32) also requires of new sites applying for landfilling licences to conduct a risk assessment of their potential impact on groundwater. The outcome of this assessment is important as it often influences the conditions outlined in the landfill license as well as identifying potential risks posed by the site during operation, prior to closure and after closure.

Leachate quality and its related risks can significantly vary depending upon the waste type and waste age in the landfill. The Environment Agency (Leeson *et al*, 2003) suggests that when analysing leachate, the following site-specific compounds should be carefully reviewed: cadmium, mercury, organic compounds, semi-volatile derivatives, semi-volatiles and volatiles. Leachate compounds have been classified into several lists of typical characteristics that should be considered when evaluating the risks posed to groundwater by landfill sites in the UK. Lists I to III are shown in Table 3.1 (a) – (c). 'List I' (Table 3.1a) comprises of eight groups of substances. If the Environmental Agency determines that a substance produces a low risk, based on the low risk of toxicity, persistence and bioaccumulation, then it can be excluded from 'List I'. In general seventy-nine substances have been identified on this list with further information in the Statutory Guidance on Groundwater Regulations, (2001). List II (Table 3.1(b)) substances are highly toxic, persist and bio-accumulate in the environment and have a harmful effect on groundwater and ecosystems. The Water Framework Directive presents an alternative list of main pollutants that are liable to cause pollution (Table 3.1(c)) which can also be used to evaluate site specific risks at a given landfill. All three lists can be used during the risk assessment of a landfill site to determine the level of risk that site-specific leachate can pose to groundwater, ecosystems, other water resources and human health. It is interesting to note that Tables 3.1(a) and 3.1(c) contain very similar compounds. The main difference is that Table 3.1(c), listing all the possible compounds that could contaminate water resources, contains a much longer list of potential pollutants. The similarity of the two tables again confirms the significant environmental risks posed by landfill leachate.

Table 3.1(a) Typical 'List I' compounds known for causing higher toxicity, persistence and bio-accumulation in groundwater, soil and ecosystems (DETR, 2001, p.37)

• Organohalogen compounds and substances which may form such compounds in the aquatic environment	• Organophosphorus compounds
• Organotin compounds	• Substances with possess carcinogenic, mutagenic or teratogenic properties in or via the aquatic environment
• Mercury and its compounds	• Cadmium and its compounds
• Mineral oils and hydrocarbons	• Cyanides

Table 3.1(b) Typical substances that are recommended for List II (DETR, 2001, p.37-38)

• Metalloids and metals: zinc, tin, copper, barium, nickel, beryllium, chromium, boron, lead, uranium, selenium, vanadium, arsenic, cobalt, antimony, thallium, molybdenum, tellurium, titanium, silver
• Biocides and their derivatives
• Substances that have a deleterious effect on the taste or odour of groundwater, and compounds liable to cause the formation of such substances in such water and to render it unfit for human consumption •
• Toxic or persistent organic compounds of silicon, and substances which may cause the formation of such compounds in water, excluding those which are biologically harmless or are rapidly converted in water into harmless substances
• Inorganic compounds of phosphorus and elemental phosphorus
• Fluorides
• Ammonia and nitrates

Table 3.1(c) The 'indicative list of the main pollutants' liable to cause pollution (Leeson *et al,* 2003, p.59)

• Organohalogen compounds and substances which may form such compounds in the aquatic environment	• Organophosphorus compounds
• Organotin compounds	• Substances and preparations, the breakdown products of such, that have been proved to possess carcinogenic, or mutagenic properties or properties that may affect steroidogenic, thyroid, reproduction or other endocrine-related functions in or via the aquatic environment
• Persistent hydrocarbons and persistent and bioaccumulative organic toxic substances	• Metals and their compounds
• Arsenic and its compounds	• Cyanides
• Biocides and plant-protection products	• Materials in suspension
• Substances that contribute to eutrophication (in particular, nitrates and phosphates)	• Substances that have an unfavourable influence on the oxygen balance (and can be measured using parameters such as BOD, COD and the like).

3.4 Landfill Design and Engineering

3.4.1 Landfill Design and Engineering

Landfill sites today can be found in five forms: as wild dumps; as unlined older dump sites; as unlined existing sites or as poorly engineered existing landfill sites; and as lined and engineered waste disposal areas. Wild dumps contain waste deposited in an uncontrolled manner at an uncontrolled location. Old landfill sites and unlined existing landfill sites are usually derelict or abandoned plots of land that were allocated by local bodies or owners for the deposition of waste. Such sites are found across the world. They are 'unlined,' in that they do not contain synthetic or natural clay liners to contain the landfill leachate from migrating and contaminating local and regional soils and waterways.

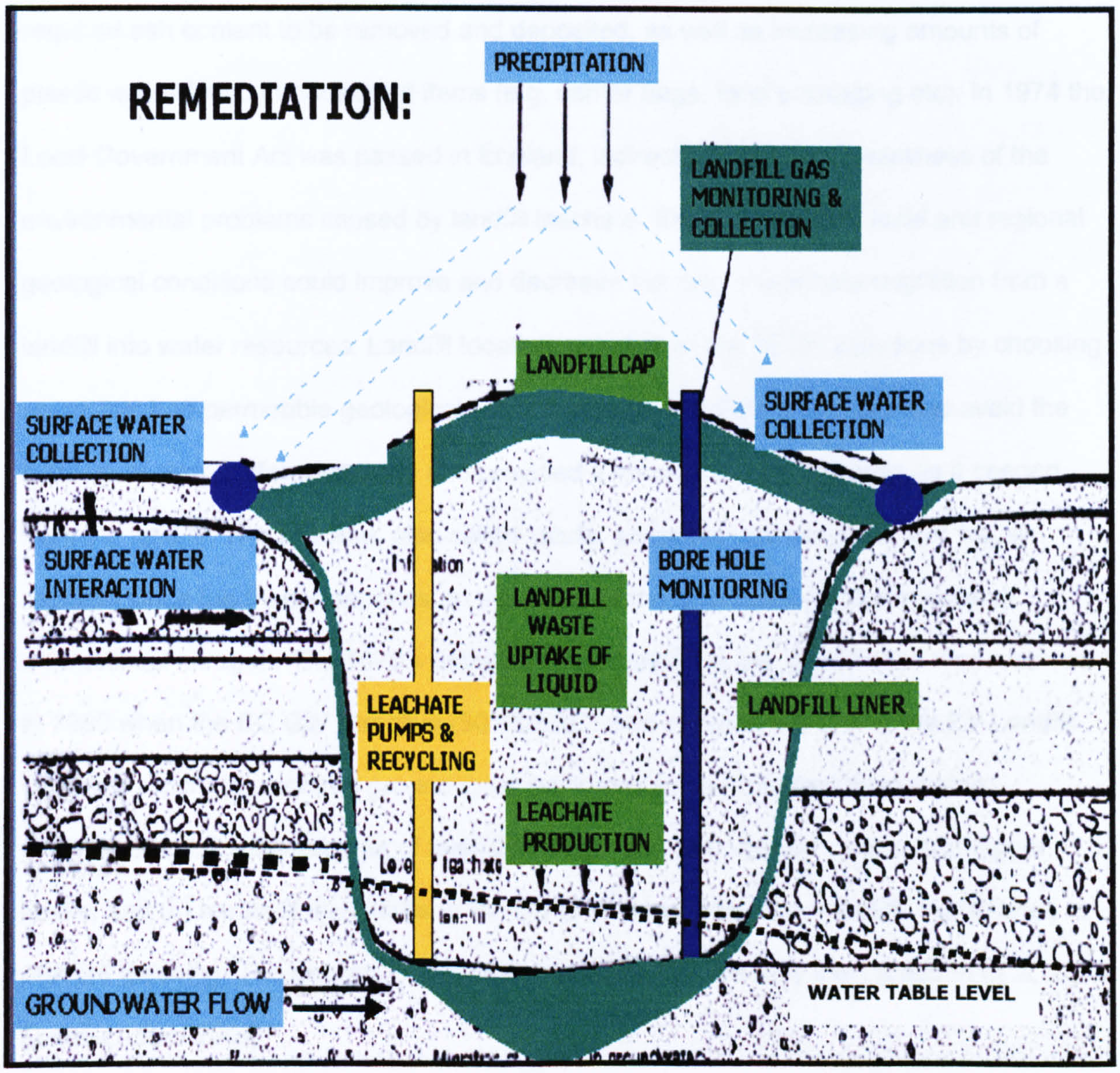
Waste reduction strategies in Britain have intensified in the last decades. Focus in recent years has been on decreasing and managing landfill risks (e.g. contamination of soil, groundwater, surface water from landfill gas and landfill leachate; harm to local ecosystems and water ways), encouraging waste reduction, recycling and reuse and implementing the EU Landfill Directive (Landfill Directive, 1999; CIWM, *cited* 2002; Environment Agency, *cited* 2004(b)). However, old and unlined municipal landfill sites remain major sources of groundwater and air pollution, releasing leachate and landfill gas in the UK (DEFRA, *cited* 2003; Environment Agency, *cited 2004(b)).* The difficulty of classifying and assessing risks posed by such sites is due to the lack of historical information about the age, depth, and type of buried waste. This missing data makes it difficult to accurately: (a) assess landfill conditions; (b) estimate potential risks posed to receptors; and (c) remediate site conditions, without extensive site assessment and monitoring.

Newer landfill sites are less of a threat to groundwater and soil due to their well-engineered structure that controls and prevents leachate from migrating off site. Such sites also monitor and keep detailed records of site changes, waste composition, and site conditions. They are well planned and designed to monitor and decrease environmental impacts (Tanaka, 1997). Engineered landfills in the UK must contain several preventive elements (e.g. Figure 3.1) such as: landfill liners, landfill cells, leachate collection systems, landfill gas collection systems, monitoring wells, leachate re-circulation systems, leachate treatment systems, and energy from waste initiatives (Department of Environment, 1990, 1995). Landfill liners are clay or synthetic liners within specified hydraulic conductivity that line the edge, base and top of the landfill to detain leachate from migrating off site. Landfill cells (also called landfill phases) are part of modern landfill designs that section a landfill into several smaller, self-contained and controlled parts. Leachate collection systems are networks of pumps and collection systems that are constructed into each landfill cell to collect leachate within each cell and allow the leachate to be separated and treated or re-used for secondary purposes. Leachate treatment systems allow the leachate to be re-used for secondary purposes. Leachate re-circulation systems often compliment leachate collection and treatment systems. They re-spray leachate on to landfill areas to stimulate the deterioration and decay of landfill materials in lined cells (Bramryd, 1988; Brumbeloe, 1992). A landfill gas collection system collects landfill gas within each cell. It can be treated and re-used for secondary purposes (e.g. energy for waste initiatives in which landfill gas is used as an energy source for landfill site vehicles). Monitoring wells are installed during the construction of the landfill. Their purpose is to monitor landfill gas and leachate concentration within the landfill cells, around the landfill and off-site of the landfill (e.g. Figure 3.1; Department of Environment, 1990; 1995; 1996).

There have been many examples of engineered landfill sites that have failed. Two examples of published case studies include Cross (1997) and Splajt *et al,* (1999). The former discusses the application of geotechnical engineering techniques for the prevention and control of pollution at landfill sites, also discussing ways in which they can fail. In the case of Splajt *et al,* (1999), the landfill was remediated by constructing a containment cut-off wall. However, the site assessment conducted prior to remediation was poorly conducted and the leachate-groundwater levels in the landfill were incorrectly calculated. The containment cut-off wall was constructed to contain leachate-groundwater levels up to 7m AOD. Leachate re-circulation pumps were added as part of the remediation to control the level of the landfill leachate. The site assessment did not account for periods of heavy rain that caused internal landfill leachate levels to increase above 10 m AOD. This put stress on, and gradually weakened parts of, the containment wall. It also allowed for leachate to seep over and under the containment wall. This case study is only one example of ways in which engineered landfills can fail.

There are several reasons as to why an engineered landfill can fail. Four will be discussed. Firstly, the engineered structures could be build inappropriately, as discussed in Cross (1997). Examples are leaking synthetic landfill liners in which seepage occurs along liner edges if the liner was improperly installed. A second reason for failure could be inappropriate site assessment and risk estimations. An example could be inaccurately calculated landfill cell volumes, which could result in collapsing cell walls or leaking cells during landfill operation. A third example is when the site assessment does not account for regional and local hydrogeological or climatic factors that significantly influence groundwater levels or regional flow velocities (e.g. annual / seasonal precipitation levels). If these factors are not adequately considered during risk estimations and landfill planning phases, then they could be factors that contribute to landfill leaching. A fourth reason for engineered landfill failure is poor maintenance of site conditions. Engineered structures in a landfill need to be monitored and maintained in order to ensure their effectiveness. Examples include torn landfill liners and weak spots in containment cut-off walls. Older engineered landfill sites, which have been closed, also require ongoing maintenance (Department of Environment, 1995; Roche 1996).

Figure 3.1: Engineered elements of a landfill site designed to control and contain leachate with in the landfill (adapted from Department of Environment, 1995)



3.4.2 The Dilute and Disperse Approach to Landfilling

The history of leachate in context of the risk it poses to water, soil, ecosystems and human health, began in the 1960's with technological advances such as central heating which required ash content to be removed and deposited, as well as increasing amounts of plastic waste found in household items (e.g. carrier bags, food packaging etc). In 1974 the Local Government Act was passed in England, indirectly increasing awareness of the environmental problems caused by landfill leachate. It recognised that local and regional geological conditions could improve and decrease the rate of leachate migration from a landfill into water resources. Landfill location selection in the 1970s was done by choosing areas that had permeable geological strata that underlined the site in order to avoid the build up of near-surface leachate and provided a natural filter for leachate as it seeped from the landfill. This practise, also called 'dilute, attenuate and disperse' was highly debated since the theory depends on the ability of the unsaturated zone to remove ammoniacal nitrogen. The 1970's debate about whether to line landfill sites was resolved in 1980 when the EC GW Directive (80/68/EEC) was adopted along with the EU Landfill Directive which: (a) required groundwater resources to be protected by requiring groundwater risk assessments of landfill sites; (b) set groundwater control and trigger levels; and (c) required all landfills in the EU to be lined with impermeable membranes to prevent leaching. The exception is for those landfills that are truly inert (Leeson *et al,* 2003; Enviros, *cited 2004).*

3.5 Waste Degradation & Landfill Hydrology

3.5.1 Waste Degradation

When waste is initially deposited into a landfill site, contaminant concentrations will vary but will generally be low, depending on the regional hydrogeological and climatic conditions as well as the waste type, until the degradation of matter begins. This second stage of degradation can also be described as 'rotting' (a biological process) that initially starts by producing low contaminant concentrations but rapidly starts increasing until the 'acetogenic' stage begins. During the acetogenic stage, aerobic bacterial fermentation occurs in which cells that have died rupture their chemical content producing a sweet smelling odour and leachate. The fermentation process causes the pH to fall, allowing heavy metals to be dissolved in significant concentrations (due to higher acidity levels). Iron is often released from landfill waste in this stage, often leaving a rusty red residue in the path of leachate outbreaks. Leachate generated from recently buried waste has a high level of organic compounds, low acidic pH, unpleasant smells, low dissolved CO_2 and presence of ammonium ions. Acidification occurs at early stages of landfill operations when most soluble organic acids are produced. The acid environment promotes higher concentrations of heavy metals and volatile fatty acids in leachate (McCarthy and Zachara, 1989). When oxygen levels in the landfill deplete, the waste mass becomes anaerobic, allowing for the methanogenic stage to follow the acetogenic stage. Leachate generated from wastes in this stage is several years old. It has lower organic compounds, neutral pH, low BOD and COD ratios and a continued presence of ammonia. Methane generation commences in this stage and can continue for a very long period of time. Methane generation generally rises, until reaching a peak and then tails off. Ammonia nitrogen levels usually remain high throughout the methane generating stage. (McCarthy and Zachara (1989).

Leachate from both acetogenic and methanogenic landfills can produce oxygen deprived conditions in ecosystems and waterways. Ammonia nitrogen varies significantly through all stages of landfill degradation but concentrations of even 100 mg/L of NH_4 can be toxic to fish and other aquatic organisms (Enviros, *cited 2004).*

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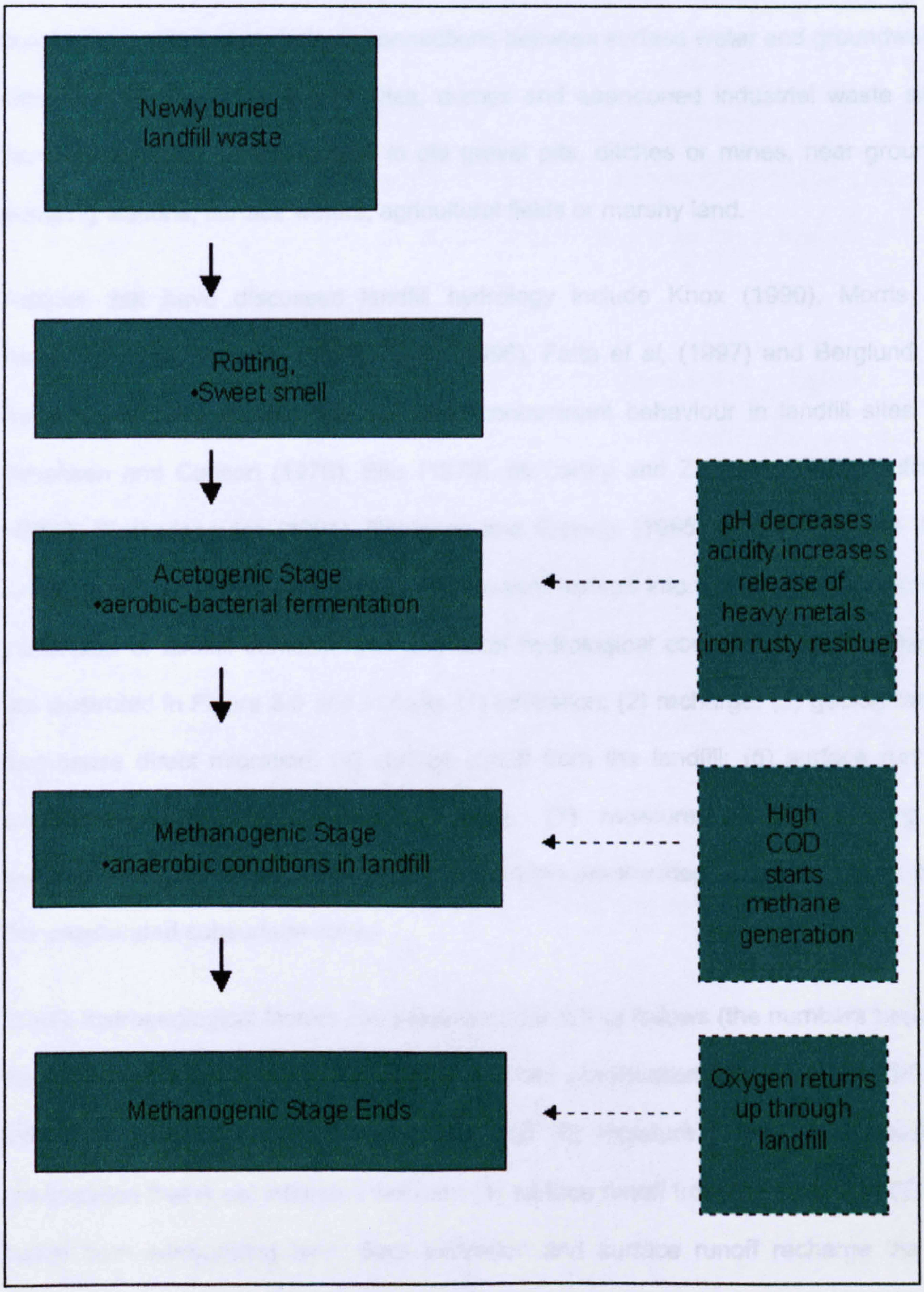
Figure 3.2: Stages of waste degradation: acetogenic & methanogenic



3.5.2 Landfill Hydrology

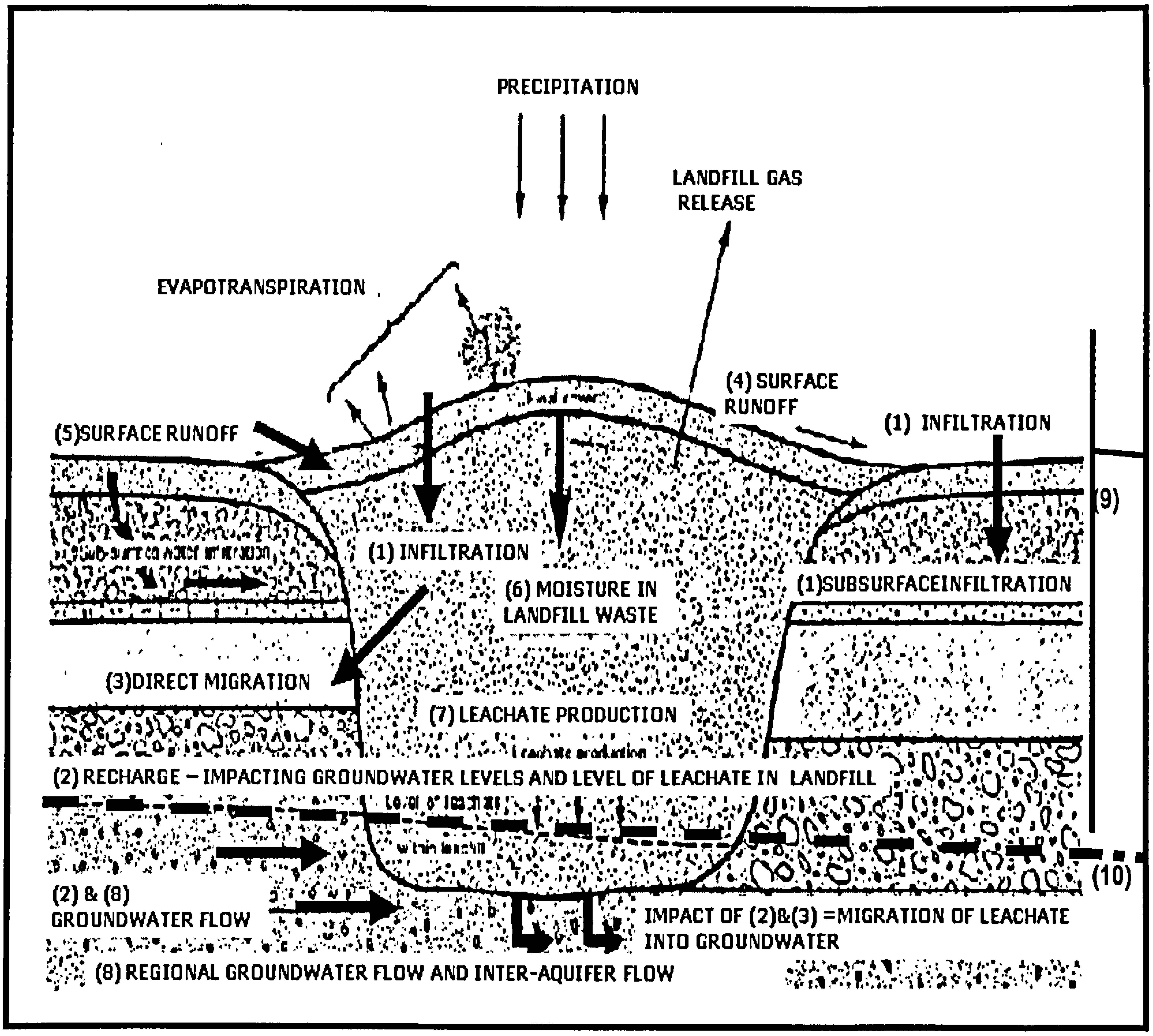
Water reaches an aquifer through recharge zones, which are relatively permeable areas for water to filter into subsurface layers. Streams, lakes, and wetland-recharge zones are commonly marked as hydrologic connections between surface water and groundwater. Unfortunately, many old landfill sites, dumps and abandoned industrial waste sites are found in recharge zones, located in old gravel pits, ditches or mines, near groundwater pumping stations, surface waters, agricultural fields or marshy land.

Authors that have discussed landfill hydrology include Knox (1990), Morris (1994), Radenkova Yaneva *et al* (1995), Chen (1996), Fatta *et al,* (1997) and Berglund (1998). Authors that have studied and classified contaminant behaviour in landfill sites include Johansen and Carlson (1976), Ellis (1979), McCarthy and Zachara (1989), Suflita *et al* (1992), Diamadopoulos (1994), Robinson and Gronow (1995), and Burton and Watson-Craik (1998). The combined studies can be summarised into ten points that describe the interaction of landfill contaminants and local hydrological conditions. These interactions are illustrated in Figure 3.3 and include: (1) infiltration; (2) recharge; (3) geological factors that cause direct migration; (4) surface runoff from the landfill; (5) surface runoff from surrounding land; (6) moisture in waste; (7) moisture produced during waste decomposition; (8) inter-aquifer exchanges; (9) the unsaturated subsurface zone; and (10) the unsaturated subsurface zone.

These hydrogeological factors can influence a landfill as follows (the numbers beside each hydrogeological factor relate to Figure 3.3): when precipitation falls upon a landfill it is (1) infiltrated, impacting (2) recharge levels and (6) moisture levels in the waste. The precipitation that is not infiltrated will form (4) surface runoff from the landfill, or (5) surface runoff from surrounding land. Both infiltration and surface runoff recharge the landfill, affecting (2) leachate levels within the landfill as well as regional groundwater levels. The landfill waste will decompose with time (6 and 7) creating and releasing a liquid cocktail of waste. The landfill's engineered structure (e.g. Figure 3.1) and local or regional (8) hydrogeologic characteristics will determine the landfill's contaminant release modes. These can include point sources of instantaneous, periodic, continuous, and decreasing modes of contaminant migration from the landfill (e.g. Figure 3.6). Three hydrogeologic factors will determine whether the landfill contaminants will remain in near-surface or deeper aquifer layers: (1) infiltration, (2) recharge and regional groundwater gradients, and (8) hydrogeologic features such as aquifer exchange (Radenkova Yaneva *et al*, 1995; Fatta *et al*, 1997).

The subsurface has an unsaturated zone closer to the surface (9, e.g. Figure 3.3) which is followed by an unsaturated zone (10, e.g. Figure 3.3). Many landfill sites are often located in the unsaturated zone (e.g. Ahel *et al,* 1999) making such sites continual sources of contamination due to soil-groundwater processes such as filtration, absorption, and capillary retention. The saturated zone is located below the unsaturated zone, where groundwater mixes with dissolved contaminants in the deeper subsurface layers. Transport to deeper geologic layers is based on groundwater flow gradients and permeability. A landfill site assessment must therefore carefully evaluate the unsaturated and saturated subsurface boundaries, given that leachate transport in groundwater is a function of local and regional geology and hydraulic gradients (Freeze *et al,* 1990; Rafai *et al*, 1999).

Figure 3.3: Ten hydrogeological processes that influence leachate formation in a landfill (adapted from Department of Environment, 1995; Radenkova Yaneva *et al,* 1995; Fatta *et al,* 1997)



Legend:
(1) infiltration
(2) recharge
(3) geological conditions
(4) surface runoff from a landfill
(5) surface runoff from surrounding land
(6) moisture in waste
(7) moisture produced during waste decomposition
(8) inter-aquifer exchanges
(9) the unsaturated zone
(10) the saturated zone

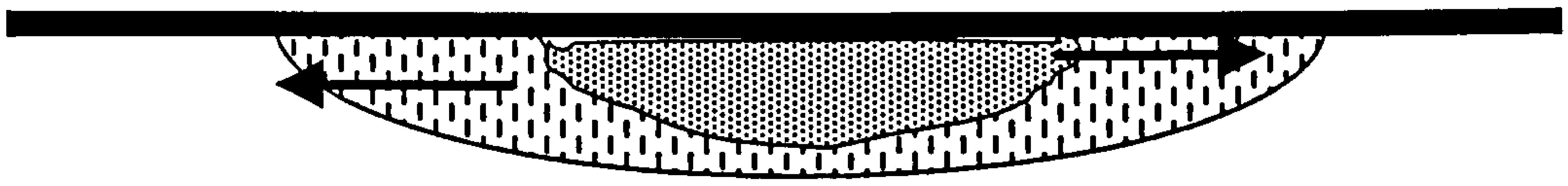
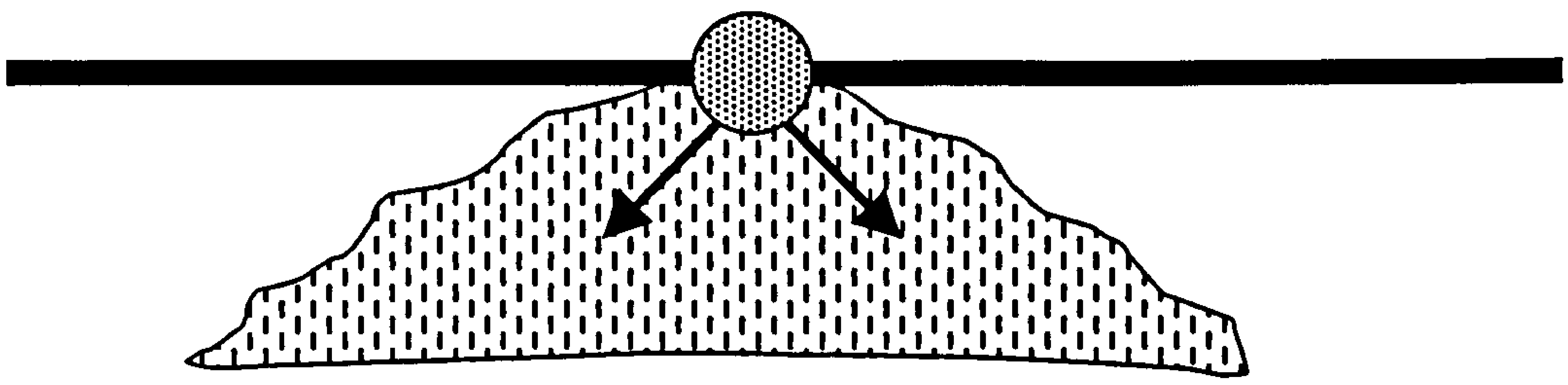
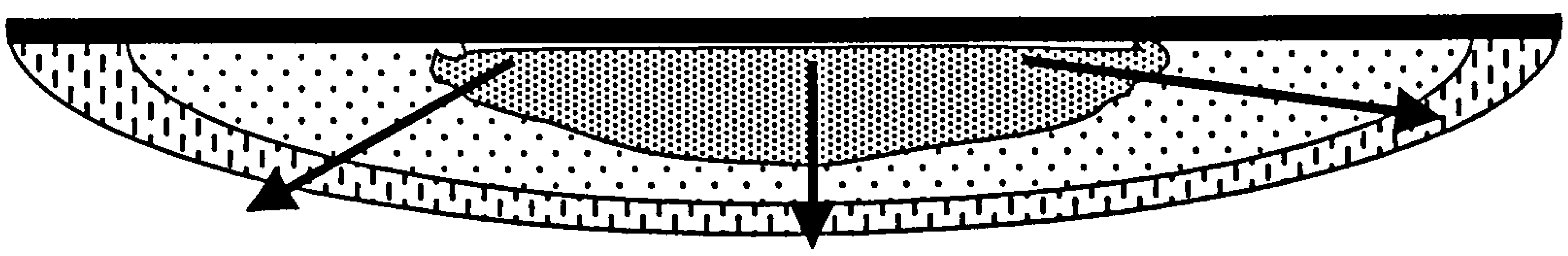
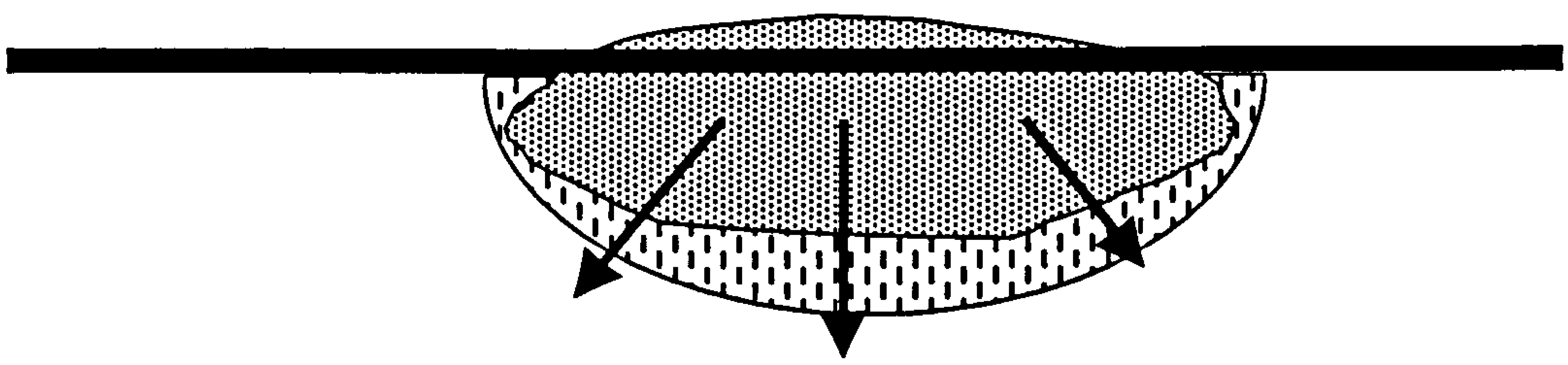
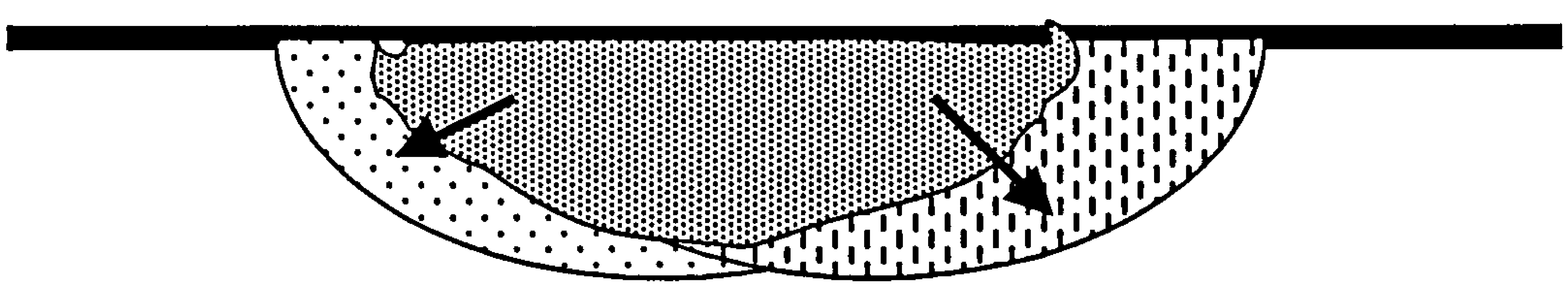
3.6 Classifying Landfill Leachate Plumes

3.6.1 Plume Classification According to Plume Shape

Classification of a contaminant plume can be done in several ways. The primary step is to collect field data about soil, groundwater, and surface water quality. These data are used to classify and conceptualise the size and extent of leachate plume in order to infer the direction and extent of leachate migration. This discussion will focus upon plume characterisation and classification according to the shape of the subsurface plume and according to human activities linked to the contaminated site. It is important to note that this section, along with section 3.6, will use terms such as 'constant' and 'high' concentrations and 'long' periods of time when describing contaminant plume dimensions and leachate composition. These terms are used in a general context, without specifying government or published guidelines. The objective behind taking such a general approach was to have the reader understand that inferring subsurface plume conditions can be quantitative (using government guidelines for example) but it is also influenced the amount of data available to infer leachate plume conditions as well as by qualitative and conceptual perceptions of the site assessor. Information about contaminant concentrations in leachate and in soil and leachate transport in soil can be found in the following literature: Johansen and Carlson (1976), Ellis (1979), McCarthy and Zachara (1989), Suflita *et al,* (1992), Diamadopoulos (1994), Robinson and Gronow (1995), and Burton and Watson-Craik Department of Environment, (1995); Landfill Directive (1999); DEFRA and EA, 2002(c).

Classifying a contaminant plume according to the shape of the subsurface plume is done by identifying the chemical transport nature of the plume (using groundwater quality samples) and by spatially deriving the plume shape (e.g. Figure 3.3, Miller, 1980). This approach can be used when time and funding for site assessment are limited (Lefebvre, 2000). There are however several limitations to this classification. Firstly, plume shapes (the plume depth and extent) are difficult to measure. Extensive field samples are therefore needed for accurate plume estimations. Secondly, the field samples collected are assumed to truly represent site conditions, which may not be true if there are a limited the number of samples taken and if the sampling depth does not represent the plume dimensions. Thirdly, in context of the risk-based approach to establishing a pollutant linkage (source-path-receptor), the method can only infer information about the pathway of migration. However, it an adequate number of field samples are available to accurately infer plume dimensions then directions of contaminant migration and existing or potential receptors can be identified.

Figure 3.4 Classifying contaminants according to the shape of the contaminant plume (Adapted from Miller, 1980, p. 80; Lefebvre *et al*, 2000).

(1) Floating Pollution	Floating pollution 
(2) Sinking Pollution	Sinking pollution 
(3) Dispersing Pollution	Dispersion pollution 
(4) Solution from Solid Waste	
(5) Pollution Generated with in the Landfill	Leachate generation and migration from a buried site 

Legend:

- Wider plumes = found at unlined landfill sites (e.g. pictures 1, 3 and 5)
- Narrower plumes = found at leaking engineered landfill site (e.g. pictures 2 and 4)

3.6.2 Risk-Based Approach to Landfill Leachate Classification

A risk-based method for classifying contaminated land and identifying the levels of harm or threat posed by such conditions is illustrated in Figure 3.5. The approach uses a combination of data collected during the site assessment. Assessment and classification is done by identifying the human activities that caused the contamination (contaminant source); defining the plume shape depending on the type of contaminant source; defining the contaminant release mode, depending on the contaminant source; and defining the contaminant group present. This combined information (collected during the preliminary and detailed studies) will assist in defining the contaminant plume, estimating directions and pathways of contaminant transport and in identifying potential risks and receptors posed by the contaminant plume.

Common human activities that often cause land contamination are waste deposited in municipal landfill sites, farming and mining activities, and waste produced as by products from industrial processes. Once the contaminant source is known, then the plume shape and contaminant behaviour can be estimated and further investigated. In general, there are three types of contaminant sources: a point source, a diffuse source or a linear source of pollution (Lefebvre, 2000). Each of these will be explained further:

- Point sources of contamination release high contaminant concentrations in small volumes. Contaminant release is controlled by hydrogeologic factors such as the hydraulic gradient, the hydraulic conductivity, porosity, dispersion, and attenuation processes. Point sources form well-defined plumes that require intensive local characterisation of geophysical conditions.
- Diffuse contaminants include pesticides, nitrates and bacteria. Sources of diffuse contamination are acid rain, radioactive fallout, forest pollution, and farming activities where the effects are near surface - affecting the unsaturated zone, shallow parts of aquifers and unconfined aquifers. Diffuse sources are the largest global factor in

diminishing groundwater quality, affecting wide areas. Delineation of diffuse plumes is difficult due to their small plume sizes combined with infiltration processes that dilute concentrations. Regional studies of soil infiltration and groundwater quality are needed, characterising geophysical conditions and using measurement instruments over large areas.

- Linear contaminants produce plumes with a combination of point and diffuse characteristics in that sources result from the presence of linear infrastructure such as roads (de-icing salt, sand, heavy metals from cars), pipeline and sewer line leakage. They are common in urban settings in which roads, sewers, ditches, and pipes collect surface runoff containing a mixture of contaminants from a variety of sources. Various modes of intervention are needed since controlling the linear source does not control the point or diffuse source from which the contaminants originate. A detailed assessment of geophysical conditions is needed to infer such plumes (Lefebvre, 2000).

Landfilling, mining and industrial activities can produce all of the above, depending upon the type of site-specific operations while farming activities have been known to cause both diffuse and linear sources of contamination.

From a risk assessment perspective, point and diffuse contaminant sources pose the greatest concern. Point sources are easier to control by regulatory standards but usually have a smaller area of impact. Diffuse sources have a much larger area of impact however; introducing contaminants into the environment through indirect sources which make it difficult to control their migration (Petts *et al*, 1998, p.6). Engineered landfill sites are most frequently classified as point sources of contamination while unlined landfill sites often form either point or diffuse sources. However, scientific research has shown that site-specific hydrogeological and landfill conditions most often define the shape of the leachate plume (e.g. Johansen and Carlson, 1976; Ellis, 1979; McCarthy and Zachara, 1989;

Suflita *et al*, 1992; Diamadopoulos, 1994; Robinson and Gronow, 1995; Fatta *et al*, 1997; and Burton and Watson-Craik, 1998).

Figure 3.5: Data collected during the site assessment (shown in Figure 2.3) are integrated in the risk management of contaminating landfill sites (adapted from Lefebvre, 2000; DEFRA and EA, 2002(a))

TYPES OF INFORMATION COLLECTED DURING THE SITE ASSESSMENT OF CONTAMINATING LANDFILL SITES

HUMAN ACTIVITIES CAUSING CONTAMINATION e.g. landfill sites	Phase I Site Assessment
⇩	
PLUME SHAPE DEFINED BY CONTAMINANT SOURCE e.g. point source, linear source, diffuse source	Phase I & II Site Assessments
⇩	
CONTAMINANT RELEASE MODE e.g. instantaneous, continuous, periodic release	Phase II Site Assessment
⇩	
CONTAMINANT GROUP e.g. inorganic contaminants, heavy metals, nutrients	The phase I & II Site Assessment
⇩	

SITE ASSESSMENT DATA
THAT GIVES INFORMATION ABOUT
CONTAMINANT PLUME BEHAVIOUR AND
TYPES OF HEALTH OR ENVIRONMENTAL RISKS
THAT MAY BE CAUSED AS A RESULT OF THE CONTAMINANTS PRESENT

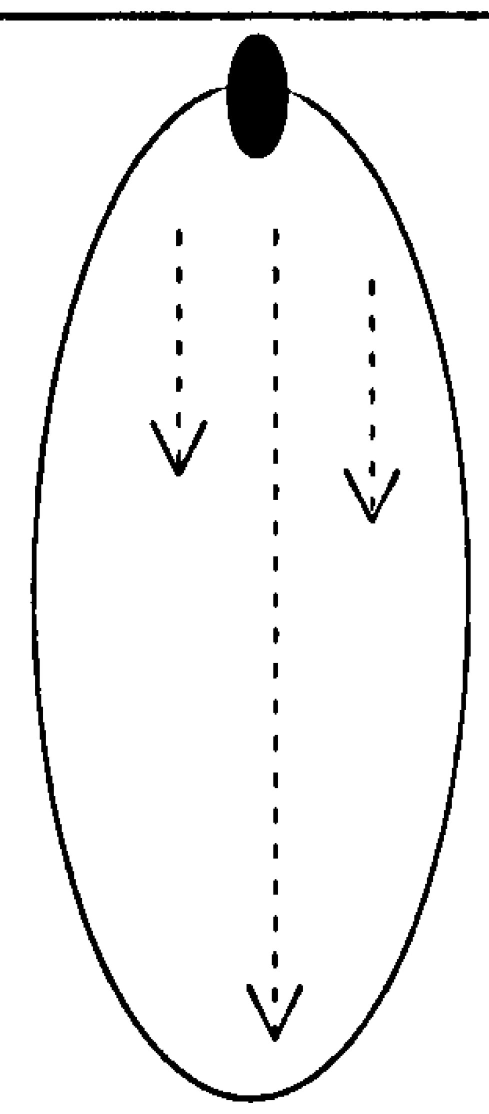
Figure 3.6: Human activities that often cause contamination are classified into three contaminant sources (based on Lefebvre, 2000)

POINT SOURCE

Examples:

- Landfill sites
- Mining
- Industrial sources.

Illustration explained:
The small circle is the point source of contamination (e.g. landfill, mine, industrially contaminated land). The large circle is the shape of the contaminant plume.



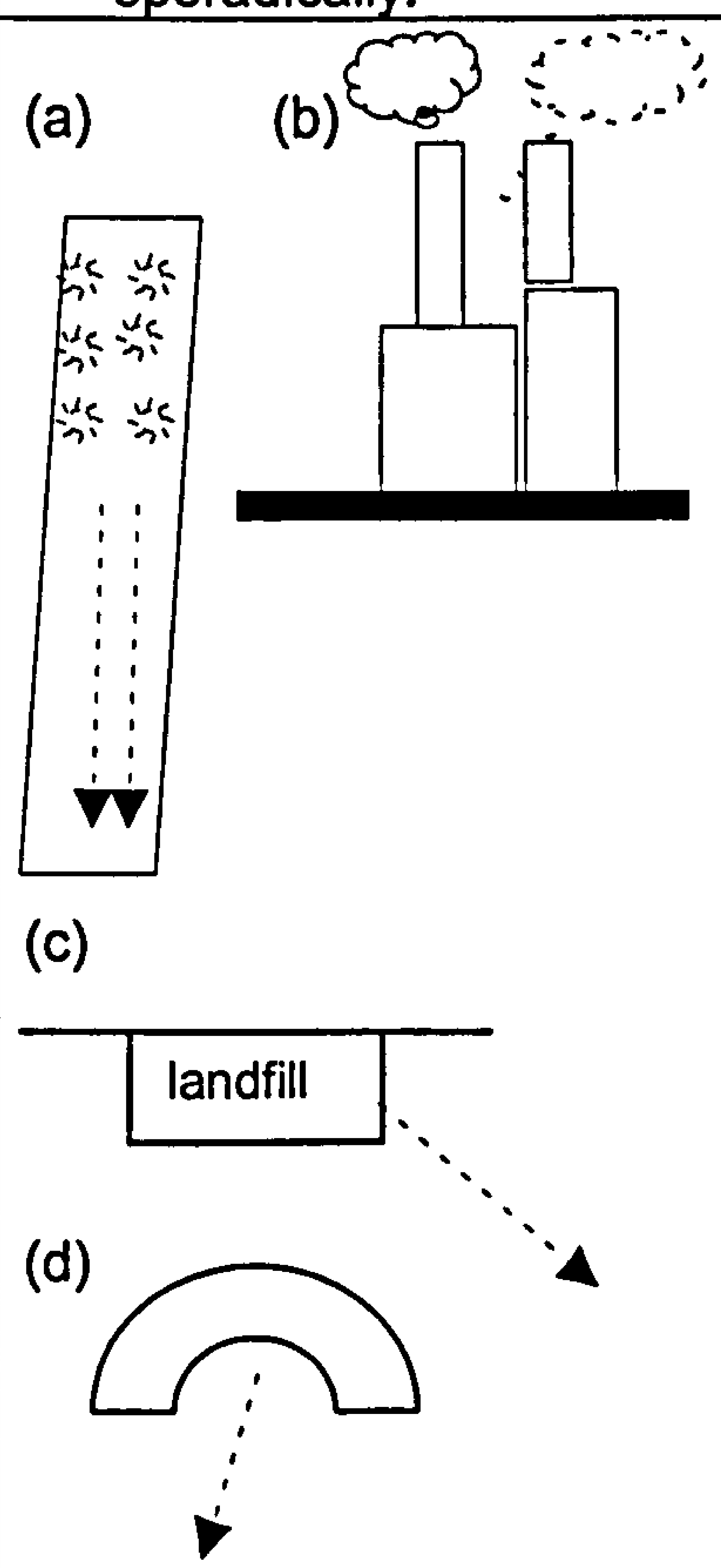
DIFFUSE SOURCE

Examples:

- Engineered landfills
- Farming activities
- Mining waste waters
- Industrial air and water pollution.

Illustration explained:

(a) An agricultural field with the arrows showing diffuse runoff.
(b) Industry producing air pollution.
(c) Landfill site-releasing contaminants.
(d) Mining wastewater releasing contaminants sporadically.



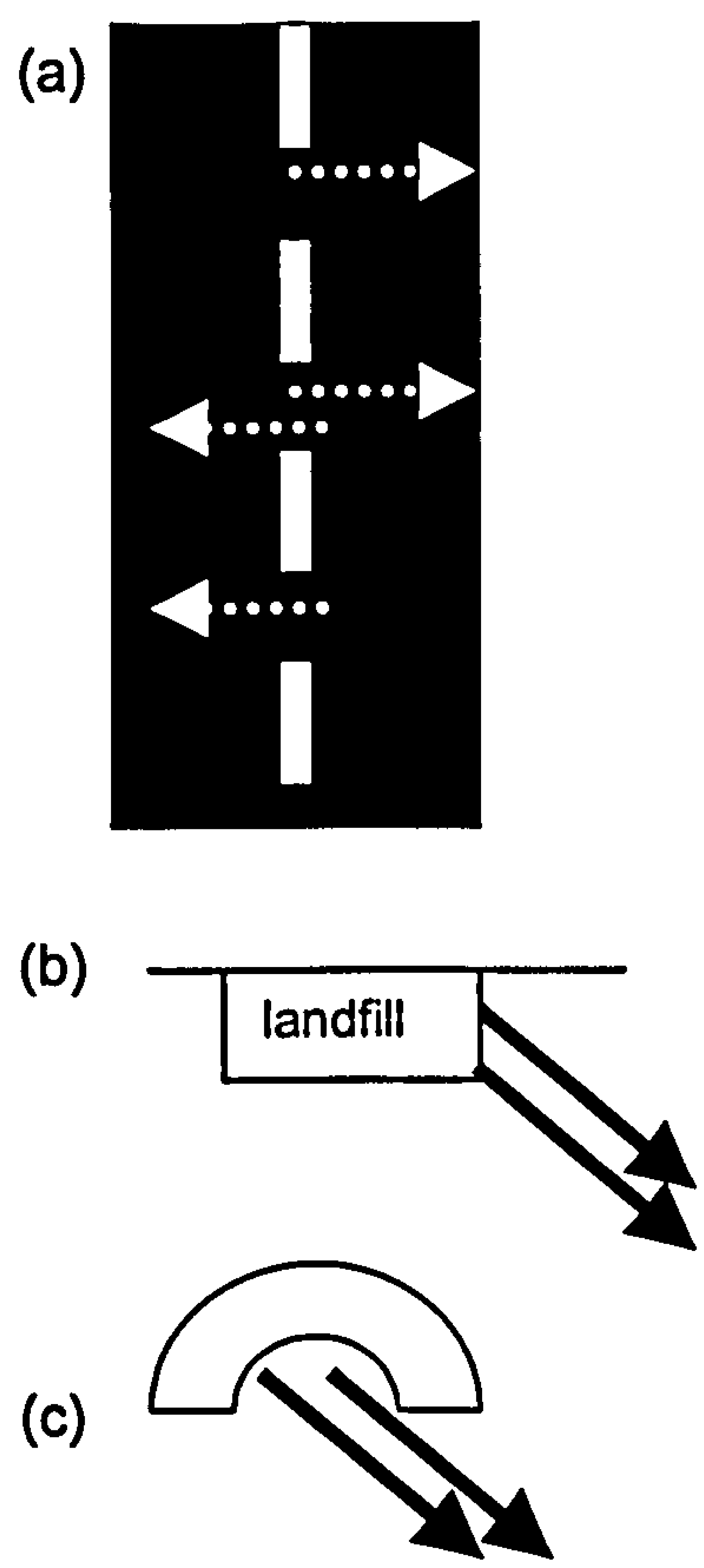
LINEAR SOURCE

Examples:

- Landfill leachate
- Farming activities
- Mining and industrial wastewater
- Roadside runoff.

Illustration explained:

(a) Road side runoff
(b) Linear runoff from a landfill
(c) Linear runoff from a mining site.



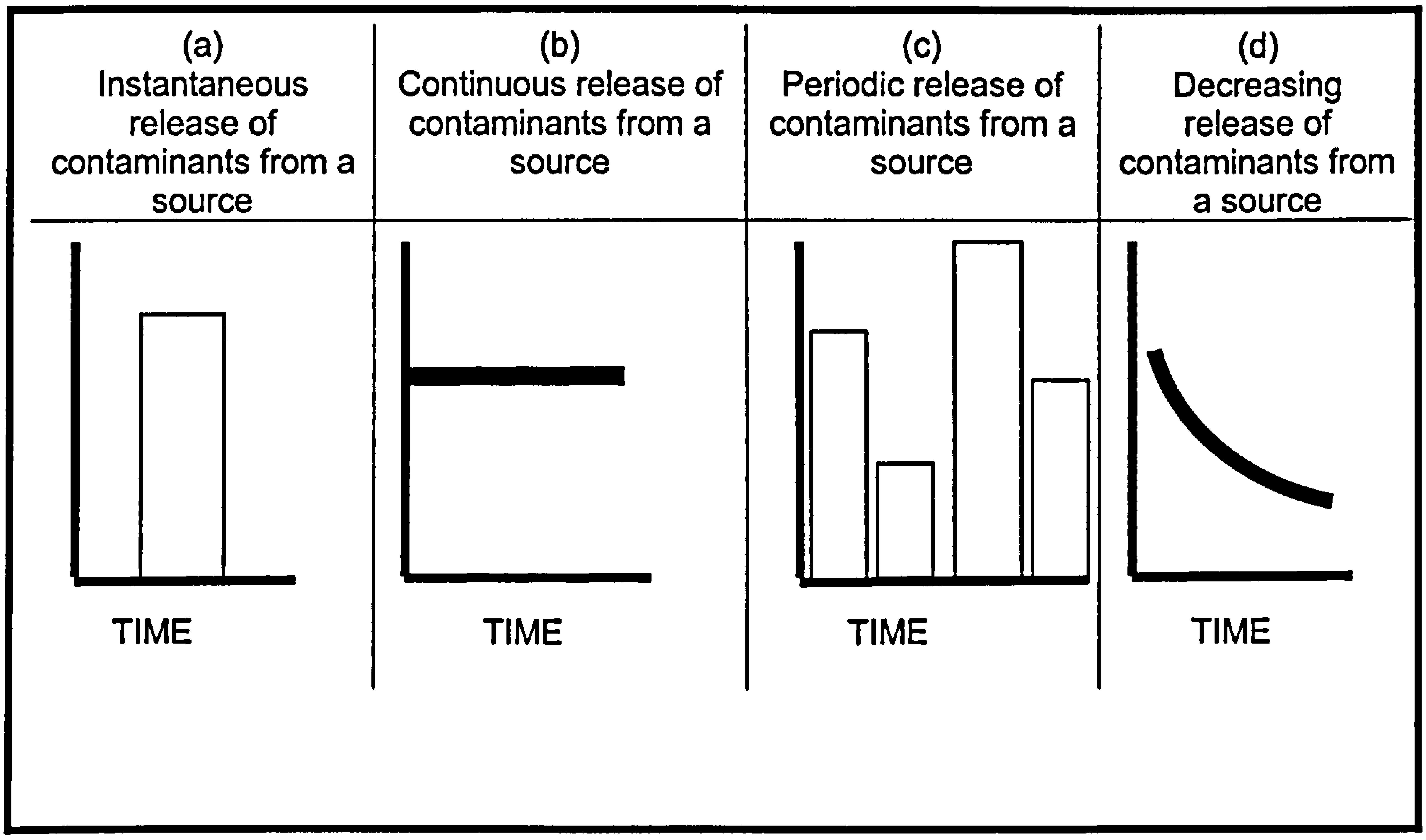
Determining the period of contaminant release, it is also necessary when determining whether contaminants are released from the source instantly, continuously, periodically, or in a degrading over time (e.g. Figure 3.7):

- Instantaneous contaminant releases have high concentrations that quickly disperse as a result of processes such as dilution, adsorption, and reactions with geological materials. Sources include spills and accidents that occur during chemical storage or transport.
- Continuous sources of contaminants release constant concentrations over long periods. Examples are landfill sites, contaminated lagoons, and wild dumps in which the contaminants migrate over large distances. Assessment and remediation of continuous contaminant sources is a long-term process requiring detailed characterisation of local and regional geophysical conditions with instrumentation spread over a large geographical area. Remediation should focus on removing or controlling contaminant migration.
- Periodic release of contaminants is the variable release of contaminants in which migration is cyclical. Sources include landfills and mine leachate, effluent and industrial waste lagoons, mercury in hydroelectric reservoirs, and road de-icing salts. Control requires assessment and sometimes long-term monitoring to identify the periodic cycles and the source.
- Degrading sources of contaminant release are those sources where contaminant concentrations decrease over time. Sources include radioactive products, pesticides, bacteria, and landfill sites.

In the case of unlined landfill sites, they are most often point or diffuse sources of contamination having continuous, periodic and degrading contaminant release modes. Classification of landfill contaminant plumes requires specific attention when evaluating the migration patterns and risks posed since the chemical nature is site-specific, highly toxic.

Integrating information collected during the preliminary and detailed study and classifying these data according to human activities, plume source categories, and contaminant release modes (illustrated in Figures 3.5 - 3.7) will assist in accurately identifying the contaminant group present.

Figure 3.7: Four modes of contaminant release from a contaminated source over time (based on Lefebvre, 2000)



__Legend:__
(a) instantaneous release = e.g. chemical spills, landfills, waste dumps
(b) continuous release = e.g. landfills, waste dumps, leaking pipes
(c) periodic release = e.g. landfills, waste dumps and lagoons, leachate from mines
(d) declining concentration = e.g. radioactive products, landfills, pesticides, waste dumps, bacteria

3.7 Characterising Landfill Leachate

3.7.1 Landfill Leachate

Chemical compounds commonly found in leachate are listed in Table 3.2, which also lists the mean contaminant concentrations found at landfill sites used in this research. There are several groups of contaminants that are commonly found in landfill leachate including inorganic and organic compounds and radioactive contaminants. Each will be discussed further.

Table 3.2 Leachate compounds and their chemical concentrations measured in leachate samples from the three study sites used in this research, landfill sites in the UK (Cross, 1997) and landfill sites in the US State of Wisconsin (Fetter, 1999, p.22).

LEACHATE PARAMETERS	**Site A (Mean)**	**Site B (Mean)**	**Site C (Mean)**	**UK Landfill in Lancaster England (Mean) Cross (1997)**	**US Landfill in Wisconsin (Range of Site Medians) Fetter (1999)**
Total Organic Carbon (mg/l C)	467	496	21.55	9.9	427-5890
Carbon Oxygen Demand (mg/l O)	106	1400	33.73	144	1120-50450
Lead (mg/l Pb)	0.03	.11	0.06	0.1	1.11
Cadmium (mg/l Cd)	0.01	.05	0.01	N/A	180-2651
Chromium (mg/l Cr)	0.02	.12	0.02	N/A	1
Nickel (mg/l Ni)	0.02	.23	0.025	N/A	1.65
Zinc (mg/l Zn)	0.01	1.47	0.01	0.4	0-54
Copper (mg/l Cu)	0.01	0.19	0.02	N/A	0-.32
Ammonium (mg/l N)	1.95	650	8.37	4	0-85
Cyanide (mg/l CN)	0.1	0.1	0.1	0.05	0-.25
Nitrites (mg/l N)	7.3	6.5	0.1	0.2	
pH	7.12	7.56	7.57	7.4	5.4-7.2
Nitrates (mg/l N)	0.1	6.34	0.505	0.2	0-1.4
Phosphates (mg/l P)	1.05	.86	0.3	N/A	
Magnesium (mg/l Mg)	1.13	1.03	17.56	N/A	0.03-25.9
Iron (mg/l Fe)	2.66	13.1	0.03	N/A	2.1-1400
Chloride (mg/l Cl)	2229	1181	136.76	1.9	180-2651
Sulphates (mg/l SO^4)	0.3	53.4	113.6	67	8.4-5000
NUMBER OF SAMPLES USED TO DETERMINE MEAN OR RANGE	20	12	20	Not Available	Not available

Legend:

- Leachate concentrations for Sites A, B and C were measured by the landfill operators from 1997-1999

3.7.2 Inorganic Contaminants

Inorganic contaminants fall into three major groups: (a) trace and heavy metals; (b) nutrients; and (c) other inorganic materials such as non-toxic salts.

(a) Trace and Heavy Metals

Trace metals and heavy metals come from mine effluent, industrial waters, solid waste, agricultural wastes, fertilisers, and fossil fuels. They commonly include lead, chromium, zinc, arsenic, copper and cadmium. Their solubility in water depends on the pH and it becoming soluble under acidic and reducing conditions. 'Trace' metals are metals that are present in the environment or in the human body, in very low concentrations. Examples include copper, iron and zinc. Discarded products containing metals are buried in dumps along with metal-containing ash from coal and trash burning (Harte *et al,* 1991, p.103-105). As metal in landfills is oxidised, it increases heavy metal concentrations in landfill leachate, which is then transported off site, often effecting metal concentrations in local groundwater and soils (e.g. Radenkova Yaneva *et al,* 1995; Ahel *et al,* 1999). In landfill sites, trace and heavy metal levels are monitored measuring chromium, iron, chloride, cyanide, sulphate, sodium, potassium, copper, magnesium, calcium, zinc, cadmium, lead, nickel, mercury and manganese as listed in Table 3.1(a) and 3.2. Metals have a stronger absorption to clay and organic materials, affecting their groundwater transport, allowing heavy concentrations to build up in clay and soil, and transferring higher metal concentrations through to the food chain. Once found in the natural environment, metals find their way into human bodies and animals through drinking water, food and air. Biological accumulation in fatty tissue and can cause problems, particularly in higher levels of the food chain. Small doses for humans are hazardous (values for dose-response in humans can be found in DEFRA and Environment Agency, 2002(c)). In ecosystems, the effects are toxic and potentially poisoning due to the bio-accumulating nature in the food chain and natural environments (e.g. DEFRA and Environment Agency, 2002(b)). Metals can influence human health in several ways:

- By disrupting chemical reactions or blocking the absorption of nutrients of with in body cells or specific body organs;
- Causing acute poisoning that can cause vomiting and diarrhoea, skin reactions, damaging lungs, causing brain damage or death to young children (e.g. through acute concentrations of lead poisoning);
- Causing chronic poisoning from long-term exposure to low concentrations of metals (e.g. cadmium can build up in the kidney and cause kidney disease; lead, methyl mercury and organic tin compounds can cause brain deterioration; arsenic can damage the peripheral nervous system, metal dust can damage lung and skin tissue as well as cause damage to liver and kidneys);
- Causing lung cancer as a result of arsenic, beryllium, cadmium, chromium and nickel dust;
- Causing skin cancer as a result of beryllium and mutations causing cancer as a result of lead, cadmium, chromium, selenium, nickel, and arsenic absorption in human bodies;
- Damaging the nervous system structure or resulting in gross deformities, blindness, language development and low IQ in developing embryos and new born children as a result of exposure to methyl mercury or lead and lead (Harte *et al,* 1991, p. 104-105).

(b) Nutrients and Microbiological Contaminants

Nutrients are potential forms of contaminants. They contain nitrogen and phosphor, as with fertilisers used in farming activities (e.g. arazine, simizine, and dieldrine), irrigation effluent, sewage and landfill leachate. Nitrates and ammonium (NH_4) are the most common nitrogen contaminants. They are highly soluble with low adsorption rates, degrading under reducing conditions. Nitrates and nitrites are salts, commonly added to cured foods, water supplies, inorganic fertilisers and in explosives and glassmaking. Nitrogen can also come from nitrogen fertilisers and wastes from farm animals and feedlots. In the case of landfills, nutrients come from a variety of waste types. Their soluble nature allows for their easy transport from the landfill through to local and regional groundwater supplies. Many nutrient contaminants have been found to have cancer-causing components. Small children are very sensitive to nitrate concentrations and ammonia is hazardous to health (Burton and Watson-Craik, 1998). In general two health effects are (a) methemoglobinemia caused by nitrites and (b) cancer caused by nitrosamines. The former is as a result of high nitrite intake, which interacts with normal haemoglobin in the blood, causing oxygen to be ineffectively transported. Groups at risk are infants (potentially causing fatality), pregnant women, people with ulcers and cancer patients. In most cases, well water containing high nitrate levels or infant food containing high nitrate levels. Although nitrates are not particularly harmful, they are converted into nitrites in the body. The latter effect is cancer in which strong statistical evidence points to a link between nitrites, nitrosamines and stomach, oesophagus and nasal passage cancer (Harte, *et al,* 1991; p.363). Phosphates do not dissolve as easily as nitrates but have higher adsorption rates. Nutrients in landfill sites are measured using phosphorus, nitrogen (NO_2 and NO_3), and ammonia (NH_3 and NH_4) compounds (Robinson and Gronow, 1995; Burton and Watson-Craik, 1998). Values for dose-response in humans and for different receptors can be found in DEFRA and Environment Agency, (2002(b and c)).

Microbiological contaminants are an alternative contaminant that can be detected through measuring various levels of nitrogen ions (nitrate, nitrite and ammonium) as well as through measuring the total count of micro organisms present in a water sample and through conducting further microbiological tests to identify whether pathogenic organisms are present in the sampled water. Coliform as well as yeast and moulds are common microbiological contaminants that can be found in well water and groundwater resources. Some common sources of such contamination can come from accumulation of faeces in soils, poor hygiene conditions around pumping wells, natural fertilisers, microbiological waste produced from laboratory testing, decomposing organic compounds (e.g. animal or human bodies) and various types of food and beverage processing. Pathogenic bacteria, viruses and protozoa can in the least cause diarrhoea, vomiting, and weaken the human immune system (Harte *et al*, 1991, p.57). The implications on human health are a stale taste in food and water, poisoning and acute illnesses and illness caused by pathogenic organisms (Suflita, 1992).

(c) Other Inorganic Contaminants

Other inorganic contaminants that are not based on carbons are ions and various metals naturally found in groundwater. Higher concentrations can be caused by mining or industrial activities, landfill leachate and wastewater sewage due to their highly soluble nature. Their accumulation increases the salinity of groundwater supplies with excessive amounts potentially causing hypertension and other health problems (Suflita *et al*, 1992; Burton and Watson-Craik, 1998; DEFRA and Environment Agency, 2002(b and c)).

3.7.3 Organic Compounds

Organic contaminants are carbon-based, complex mixtures of numerous chemicals, of which most are synthetic chemicals from commercial chemical products that were disposed of or have spilled into soil and groundwater. Synthetic organic compounds are a modern contaminant group that can be classified according to their behaviour. Organic contaminants containing carbon compounds include volatile organic compounds and polyaromatic hydrocarbons. Organic contaminants that can also form distinct phases separate from water are non-aqueous phase liquids (also known as 'NAPLs'). When the organic phase is lighter than water, they are called light non-aqueous phase liquids (also known as 'LNAPLs'). Examples include petroleum products such as gasoline, diesel, motor oil and transmission fluid. When they are denser than water, they are called dense non-aqueous phase liquids (also known as 'DNAPLs', Lefebvre, 2000). Examples include chlorinated solvents such as tar, coal, polychlorinated biphenyl, polychloro ethane, trichloroethene, and trichloroethane. Assessment and control of organic contaminant plumes is a growing issue. It is a subsection of contaminant hydrogeology as such plumes are very difficult to measure and remediate. Assessment requires groundwater flow conditions, soil adsorption behaviour under different geochemical conditions, unsaturated flow in multiple dimensions and oil flow theories to be integrated when evaluating current and potential plume dimensions. The time, cost, and risks of remediation are often very high. Most landfill sites contain smaller concentrations of light non-aqueous phase liquids such as petroleum (gasoline, diesel, motor oil and transmission fluid). Other forms of organic contaminants may be present as municipal waste that is a heterogeneous collection of organic chemical compounds. Organic contaminants have multiple and long term implications for human and ecosystem health causing long-term, local and regional contamination of groundwater drinking supplies, degrading soil quality, destroying ecosystems and killing off animals along the path of migration. Concentrations in drinking water are poisonous to human health are listed in (ASTM D6235, 1998; EA, 2000; Hooker *et al*, 2000; Lefebvre, 2000; CL:AIRE, 2001).

3.7.4 Radioactive Contaminants

Sources of radioactive contaminants include waste and by-products from the nuclear industry, mining sites, municipal waste, industrial waste, and uranium enrichment for fuel generation. The type of radioactive contaminant released depends on the source from which it originates and how it is uses. The largest sources of radioactive contaminants and waste are the manufacture of nuclear weapons and commercial nuclear power plants. In the case of municipal landfill sites, radioactivity can come from a variety of household items such as batteries and fluorescent watch faces. A significant problem with all sources of radioactive contaminants is that they are considered to be a burden for future generations as they can remain a hazard for a very long time, depending on the waste half-life that can vary from 100 to several hundred years (Harte *et al,* 1991, p.163). Exposure, in the least can cause cancer, birth defects, skin deformations and death (ENDS Report, 1996; DEFRA *et al, cited* 2003).

3.8 Taking a Risk-Based Approach to Landfill Leachate Management

3.8.1 Implications of Failing to Take a Risk-Based Approach

If a site assessment fails to adopt a risk-based approach, it will likely lack in one or more of the following elements: a systematic approach, objective assessment methods, consistent and factual information and proper professional care and diligence. The implications of failing to adopt a risk-based approach at landfill sites therefore results in (a) inaccurate risk assessments and (b) ineffective courses of remedial action causing contamination and risk or harm to receptors. There are also human health and business risks and liabilities to consider. The risks to human health are being closely studied to evaluate whether there is a correlation proximity to landfill sites and increasing birth defects and cancer in people living close to landfills in the UK (DEFRA, *cited* 2003). Secondly, the site owner will be responsible for remedial costs if the site produces environmental pollution. Thirdly, the site assessors conducting the assessment may be held liable, facing legal action if their investigation results are proven to be misleading. Lastly, regional and local groundwater and soil resources will be harmed or threatened. Therefore the advantages of a risk-based approach at landfill sites far outweigh the shortcomings of failing to adopt this framework.

3.8.2 Statutory Guidance and Guidelines required for a Risk-Based Approach

The increasing awareness of the environmental impacts from landfill sites is drawing improvement in the way that these sites are managed. The UK's legislative framework for a risk based-approach to landfill management is described through statutory guidance such as Part IIA of the Environment Protection Act (1990), the Landfill Directive, (1999); the HMSO (2000); and Pollution Prevention and Control Regulations, (2000). Part IIA was implemented in England in April 2000. The implementation has been accompanied by a number of technical reports and guidelines for effective landfill management. A handbook of procedures for the management of contaminated land incorporates good technical practises for assessing contaminated land risks are currently being developed (DEFRA *cited* 2004(a)). Best-practise guidelines for waste management licensing and remediation are also being developed (Environment Agency *cited* 2004(a-b)). In reviewing landfill-related problems, there are two areas that would benefit from the development of contaminant management guidelines for problematic and leaching landfill sites. The first area is to developing guidance for communication with stakeholders (who must in many cases pay the cost of the assessment and remediation, and who need to be educated about site-specific risks or decide on remedial actions). The analytic deliberative model for risk communication offers a constructive framework for developing in this area. Documents such as SNIFFER and Environment Agency (1999)-offer advice on how to effectively communicate risk related to contaminated land. However, guidance in dealing with landfill-specific issues would be beneficial for both site assessors and landfill managers.

The second area is to develop a guideline for assessing landfill sites. In other words, recognising the importance of the site assessment – the methods used to conduct the investigation at landfill sites. DEFRA and CIRIA have developed a collection of guidance documents in support of the Part IIA of the Environmental Protection Act. Several publications issued by the British government and industry organisations in the last decade have increased awareness of this issue, (listed in DEFRA and EA, 2002(a); DEFRA *cited* 2003). However further evaluation is needed to identify the most appropriate methods that can be used to evaluate the extent and chemical nature of a subsurface leachate plume and its pathways of migration, under landfill conditions. Such guidelines would be of great assistance to both site assessors and landfill operators, given that landfilling and waste management strategies will be increasingly intensified in coming years through the implementation of the Landfill Directive (Landfill Directive, 1999; DEFRA, *cited* 2003).

3.9 Summary

In summarising the findings of this chapter, landfill sites in Britain can be described as heterogeneous, complex areas in which waste is disposed. The level of threat that they pose depends upon their waste composition, age and quantities, as well as site design, location, and local-regional hydrogeology. They are human-induced contaminant sources that release inorganic, organic, and sometimes radioactive chemical contaminants at continuous, periodic and degrading release modes. The first half of this chapter discussed landfill sites in the UK: the scale of the problem; the statutory and risk-based framework for their management; their design and engineering; the problems that they present and the hydrogeology factors that influence their rate of contaminant plume formation and leachate migration. The second half of the chapter looked the different approaches that can be used to classify contaminant plumes as well as factors that are needed (and are currently either being implemented or already enforced) which support a risk-based approach to landfill site assessments.

The chapter points to two conclusions. Firstly, the site assessment of landfill sites is a difficult task because (a) subsurface characteristics are unknown and (b) background information is often limited. Secondly, landfill site assessments play an important role in determining the level of risk posed by a leachate plume. As a result, sampling strategies and methods of site assessment should be carefully considered given the diverse character of landfill leachate and site-specific influence of hydrogeological factors. Chapter 4 will therefore focus on the methods used to measure and assess geophysical conditions during the site assessment, reviewing some of the more commonly used methods available to characterise soil, geology, and groundwater conditions around landfill sites.

CHAPTER 4: LANDFILL SITE ASSESSMENT METHODS

4.1 Introduction

The site assessment of geophysical conditions at a landfill site is needed to evaluate the level of potential or existing risk posed to receptors and to determine appropriate remedial actions. Such assessments require soil quality, shallow depth geology, and hydrological conditions such as groundwater levels and water quality to be measured, mapped, and monitored. The amount of spatially distributed subsurface information (such as contaminant plume size and hydrogeologic heterogeneity) will determine the accuracy of the site assessment, and consequently influence the precision of the risk estimations and remedial decisions.

The aim of the chapter is to discuss and describe field methods used at landfill sites and similar contaminated sites to assess: (a) soil and contaminant conditions, (b) shallow depth geology, and (c) groundwater and hydrological conditions.

4.2 Assessing Soil and Contaminant Conditions

4.2.1 Introduction

Soil assessment is necessary to identify the contaminant type, concentration, source, migration pathways, and the likelihood of contaminant accumulation in the soil. A soil assessment can be conducted using direct and indirect methods. Direct methods such as soil sampling, trial pits and boreholes provide point information requiring interpretation between sample points. Indirect soil assessment methods have developed rapidly in the last decade. Examples include ground geophysical surveys, as well as the use of aerial photography and remote sensing instruments. There are several new modelling tools to assist in conceptualising site-specific and regional soil conditions, which aid in building a multi-dimensional map of subsurface soil conditions. These include GIS, digital geological models, and an integration of soil quality databases that are linked to modelling interfaces.

4.2.2 Direct Soil and Contaminant Sampling Methods

There are several methods of soil sample collection listed in Table 4.1. These include using hand augers, the shelby tube, the split spoon sampler, and the split barrel sampler. Technological advances in computing have allowed for the development of hand-held soil quality samplers in which soil chemistry can be measured almost immediately on site (Rafai *et al,* 1999).

An alternative and cost effective approach to soil sampling is digging trial pits up to six metres deep. This method allows the initial characterisation of near surface soil and geological layers as well as the preliminary inspection of soil and groundwater for noxious chemicals. Alternatively, in areas where the presence of volatile chemicals is known or suspected, boreholes are a better alternative as samples can be sealed immediately upon collection for analysis in the laboratory and the borehole can be re-filled or sealed after sampling. Light percussion boreholes with depths greater than five metres identify geological layers and can be used in sampling and monitoring soil, groundwater, and gas. Hand-held trawlers, soil punches, and augers are effective for depths under five metres to show and allow sampling of soil strata and near-surface groundwater (British Drilling Association, 1991; Fetter, 1999, p.396-399).

Table 4.1 Common direct and indirect soil assessment methods (Based on Fetter, 1999, p. 396-399; Rafai *et al*, 1999)

	CONTAMINANT TYPE	CONTAMINANT SOURCE	CONTAMINANT CONCENTRATION
Direct Methods			
Hand Augers	✓	X	✓
Shelby Tube	✓	X	✓
Split Spoon Sampler	✓	X	✓
Split Barrel Sampler	✓	X	✓
Geoprobe	✓	X	✓
Large-Bore Soil Sampler	✓	X	✓
Macro-Core Soil Sampler	✓	X	✓
Core Logging	✓	X	✓
Trial Pits with JCB	✓	X	✓
Indirect Methods			
Passive Remote Sensing (GPR)	✓	✓	X
Aerial Photographs	✓	✓	X
Active Remote Sensing (Spectroradiometers)	✓	✓	X

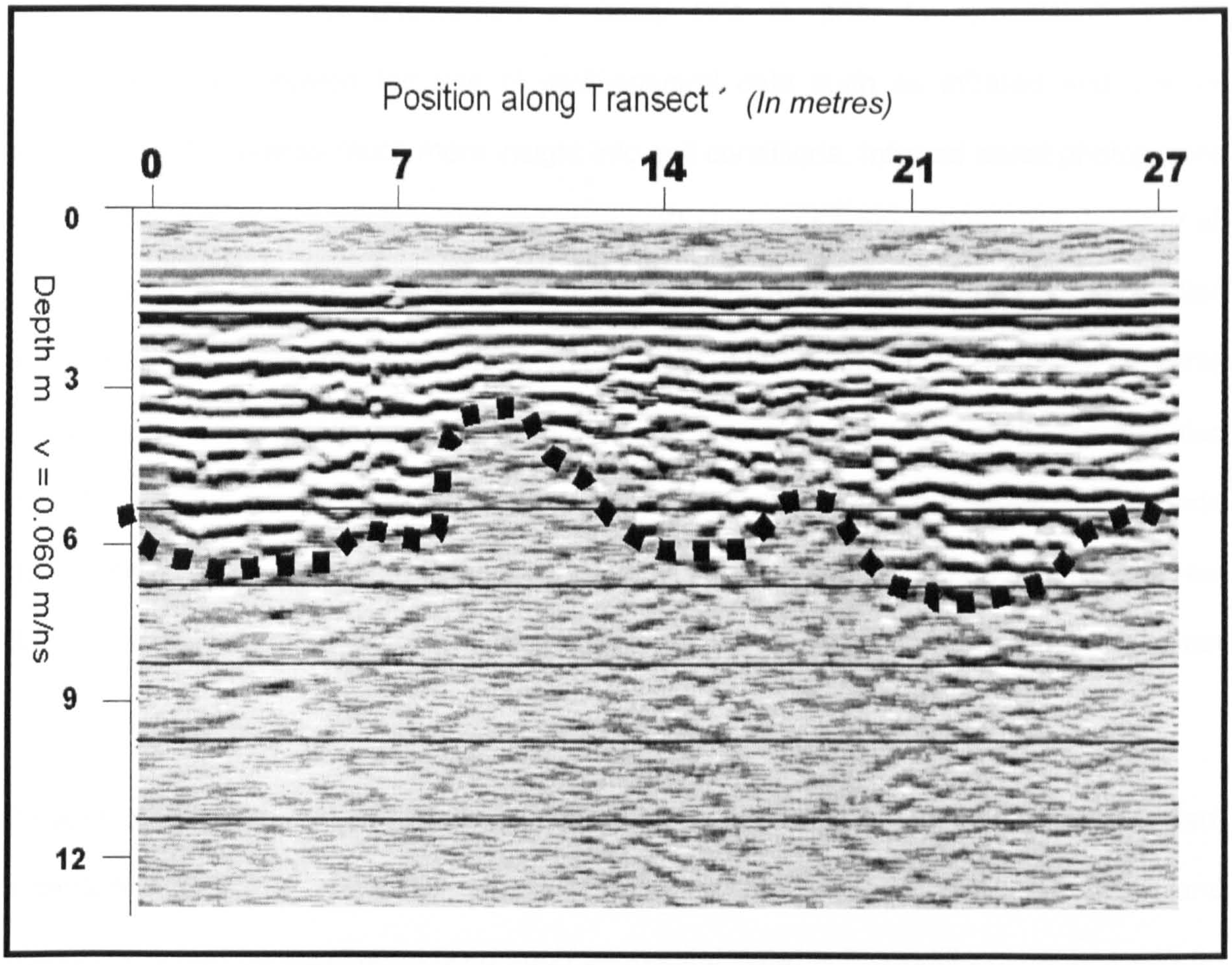
4.2.3 Indirect Soil and Contaminant Assessment Methods

New indirect methods of assessment are non-intrusive and are multi-spatial in nature (Table 4.1). They include geophysical ground surveys, aerial photographs, and remote sensing. During soil assessments, such methods aim to identify the contaminant type, concentration intensities, and sources of contamination. The direct methods listed in Table 4.1 are conventional methods popularly used during the site assessment of landfill sites. They are successful at identifying the contaminant type and concentration but are limited in their capability to represent conditions across a larger area. Indirect methods, however, are successful at identifying contaminant sources, pathways, and spatially distributed conditions across a site. Their disadvantage is that they do not provide quantitative information about contaminant concentrations.

4.2.4 Geophysical Ground Surveys

Geophysical ground surveys have been successful in delineating contaminant plumes and leachate levels. Studies such as Davis and Annan (1989) and Forde (1996) have shown that geophysical methods are effective and non-intrusive soil-assessment tools that can optimise sampling schemes. Reynolds and Taylor (1992) list geophysical techniques that are effective on contaminated land, including electrical resistivity, self potential profiling, electromagnetic conductivity mapping, ground penetrating radar (GPR), magnetic mapping, seismic resistivity, subsurface electrical imaging, transient electromagnetic sounding (TEM), induced polarisation, and spectral induced polarisation. Geophysical ground surveys provide important information needed to understand soil conditions and contaminant attenuation properties. For accurate results, such methods require known targets and familiarity with local and regional hydrogeologic conditions (Forde, 1996). GPR is a relatively new addition to the geophysical methods that can be used to assess landfill sites. Davis and Annan (1989) defined the 'GPR Rules of Thumb', which include knowing the survey objectives, identifying the target and depth, geometry and electrical properties, as well as planning the survey in order to account for topographic obstacles. Contaminant presence in soil alters the electrical properties of the soil material, influencing the hydraulic conductivity of soil materials and allowing for the data analyses to delineate the presence of the contaminant plume. The main disadvantage of GPR is that it is not capable of identifying pollutant concentration. Despite this, studies by Sauck *et al*, (1998) and Splayt *et al*, (2003(b)) show the effectiveness of GPR in mapping soil and contaminant conditions. Figure 4.1 shows an example of what can be identified using GPR on an unlined landfill. The cross-sections drawn using GPR-derived information shows the delineation of leachate levels at older, unlined cells in a landfill.

Figure 4.1 Cross-section drawn using GPR to delineate leachate levels in older, unlined cells of a municipal landfill (Splayt *et al*, 2003(a))

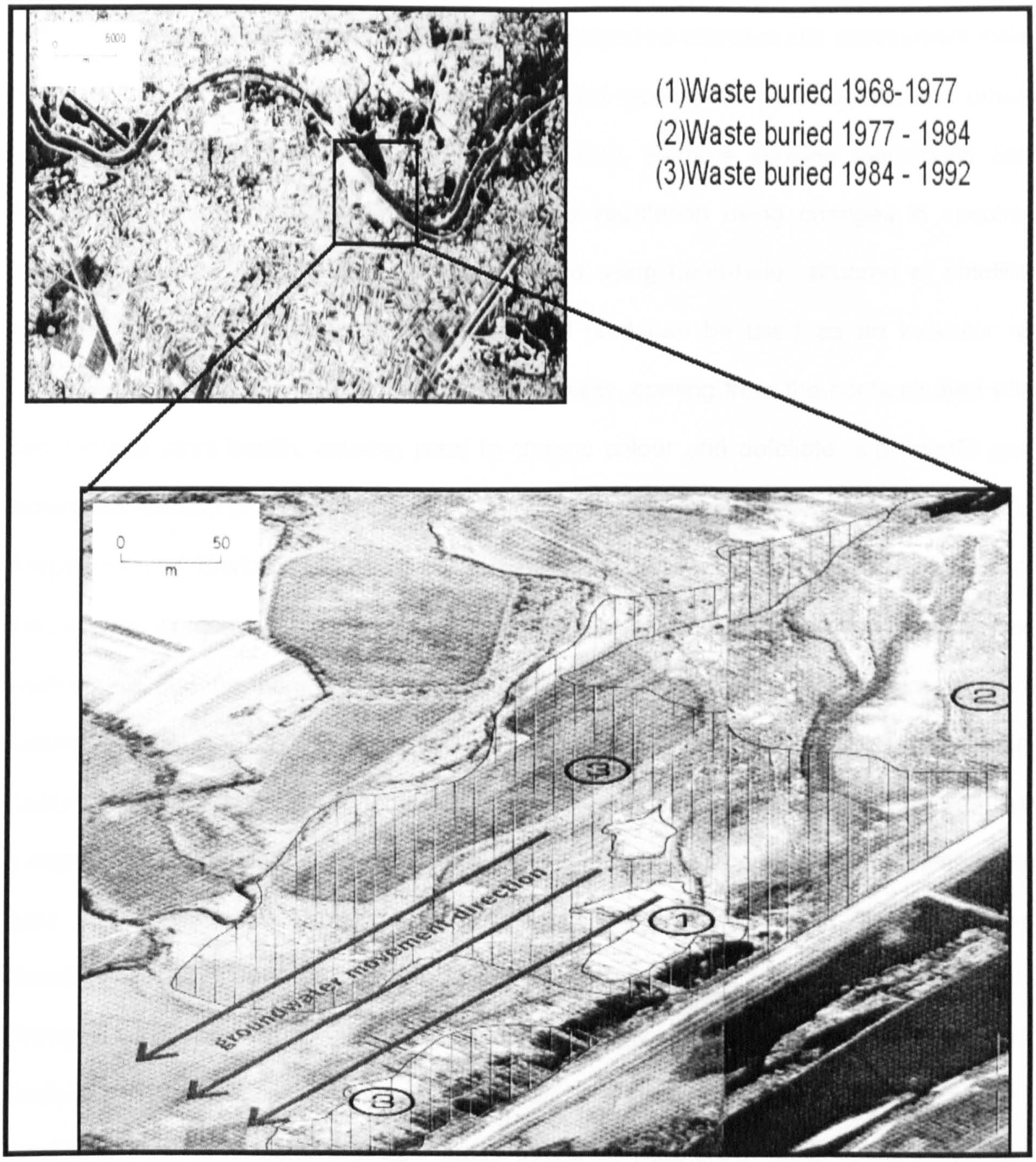


4.2.5 Aerial Photography

Aerial photography is a reliable and cost-effective method of mapping and monitoring changes at intervals across contaminated landfill sites. It can be used to monitor: (a) land use, land cover at local and regional scales; (b) the spatial extent of contaminant plumes; and (c) physical features around the site such as geological shifts, urban development, alteration of drainage patterns, variations in soil colours and vegetation stress. In the case of older and abandoned sites, time series of aerial photos are often the only source of reliable information on spatial and temporal change at the site (Erb *et al,* 1981). Studies of contaminants using aerial photographs date as far back as Erb *et al,* (1981). The method has become a standard method of landfill assessment providing snap shot information about local and regional soil and near-surface conditions. An example is shown in Figure 4.2, showing an aerial photograph of Study Site B, prior to remediation. Digital photography has allowed the use of multi-spectral data such as infrared and thermal imaging which provides much more insight into soil conditions. Infrared aerial photographs provide insights into vegetation and soil stress. Studies by Ferrier (1999) and Splajt *et al,* (2000) have confirmed this, showing that high-resolution aerial photography in combination with other field data provides cost-effective field information. The limitation of aerial photography is that it represents conditions at only a single point in time. The method also provides spatial and visual information about site conditions but does not provide quantitative values of contaminant concentrations. A newer approach has been to use real time airborne video-photography to monitor and assess changes on larger areas at greater frequencies (Folkard, 1999).

If sequential applications of high-resolution aerial photographs or videos can be collected, important information about the impact of contaminants on soil quality, vegetation and ecosystems can be obtained. Using such sequential data, soil condition changes can be identified through (a) barren soil areas; (b) changes to soil reflection; (c) spatial trends and patterns in soil reflection data; (d) accelerated succession from barren soil to weeds and woody plants. Vegetation changes are identifiable through: (a) vegetation discoloration; (b) decreased number of plants; (c) decreased vigour of plants; (d) absence of characteristic plant species; (e) presence of dead trees or shrubs; (f) early 'autumn colours' or discoloration in plants; and (g) presence of plant species adapted to grow under toxic conditions.

Figure 4.2 LANDSAT TM image (top) and aerial photographs of Site B (bottom) provided reliable information about waste quantities buried at the landfill prior to remediation (Olujic, 1995)



4.2.6 Environmental Remote Sensing

Remote sensing is the observation of a target or a process from a distance. Environmental remote sensing applications are increasingly recognised as effective site assessment tools because they can give information about ecological, geological, hydrological, and urban environmental conditions (Green and Chrien, 1999). Remote sensing techniques can measure soil moisture interactions and stressed vegetation using changes in spectral response. Remote sensing data can be acquired using hand-held, airborne or satellite sensors. Vegetation growing near contaminated land can be used as an indicator of contaminant pathways and receptors. Poor air quality, coming from the contaminated site can damage plant health, causing plant to change colour and defoliate, e.g. landfill gas blown downstream of a landfill, can damage crops located several kilometres from the site (Department of Environment, 1990). Alternatively, groundwater can act as a transport mechanism, contaminating soil quality and directly influencing vegetation health. The implications can be negative or positive. If the soil-borne contaminants are high in nutrients, vegetation growth could be induced, producing lush green vegetation. With highly acidic contaminants, vegetation will change colour, defoliate and die off, with the exception of vegetation that can tolerate high soil pH levels, e.g. certain wetland plants will grow well under acidic conditions. In both examples contaminants in the soil will be absorbed by vegetation, damaging the internal plant structure, causing defoliation, colour changes, wilting or internally altering chlorophyll absorption levels. When vegetation shows these symptoms, it is called 'stressed vegetation'. It is important to note that contaminated land is not the only factor that can cause stress to vegetation. Other factors that can damage the internal structure of a plant include poor climatic conditions, land use, erosion, other local sources of air and soil contamination, e.g. other industries in the area, ecosystem factors, e.g. invasion of non-native species in the food chain, and droughts. Landfill sites are known to cause both types of vegetation stress (airborne and soil-based), altering chlorophyll concentrations, damaging the plant, and changing the plant's spectral reflection of the radiation.

The first study to classify contaminant-induced vegetation stress was conducted by Murtha (1976) noted in Griffiths *et al*, (1996). Contaminant-induced vegetation changes were classified into four levels:

- Level 1 - vegetation that is completely defoliated
- Level 2 - vegetation that displays some form of defoliation
- Level 3 - vegetation that has colour changes
- Level 4 - vegetation that does not show visible signs of stress but has a deviation from its normal reflection in the non-visible light spectrum.

Studies that have applied remote sensing data to the assessment of contaminated soil and vegetation are Lyon (1987), Vincent (1994), Griffiths *et al*, (1996), Irvine (1997), Ferrier (1999), Hauff *et al*, (1999), Hauffman *et al*, (1999); Jago *et al*, (1999), Keller and Fischer (1999), McCubbin *et al*, (1999) and Splayt *et al*, (2003(a-b). In order for a remote sensing application to be applicable to contaminated land applications, Lyon (1987) identified five factors that must be present. These include: (1) presence of high concentrations of liquid or gas near-surface contaminants; (2) soil and vegetation under investigation is spatially susceptible to damage by the contaminant source; (3) large areas of homogeneous vegetation cover are present; (4) knowledge of regional soil, vegetation, and hydrogeologic conditions; and (5) field measurements (soil quality, vegetation samples, groundwater quality and levels etc.) to calibrate remote sensing data. The need for homogeneous vegetation cover has until recently been a limiting factor when using such methods over heterogeneous landfill sites. Research presented in section 8.4 - Investigation 3, will show that homogeneous vegetation cover is no longer a prerequisite for remote sensing applications.

Current operational airborne and satellite remote sensing instruments have demonstrated their ability to give robust and cost effective site assessments (e.g. Table 4.2, Folkard, 1999). Irvine (1997) and Keller and Fischer (1999) demonstrated the effectiveness of thermal remote sensing by mapping temperature differences in buried waste. A range of other field, airborne and satellite-based spectroradiometers are under research and developments (NERC-EPFS, cited 2000), focusing upon wavelengths that have been used to measure soil and vegetation stress. Wavelengths showing stress around landfill sites and other contaminated sites are shown in Figure 4.3. Three wavelength regions have been identified as being of use in identifying vegetation stress (Gausman *et al,* 1991):

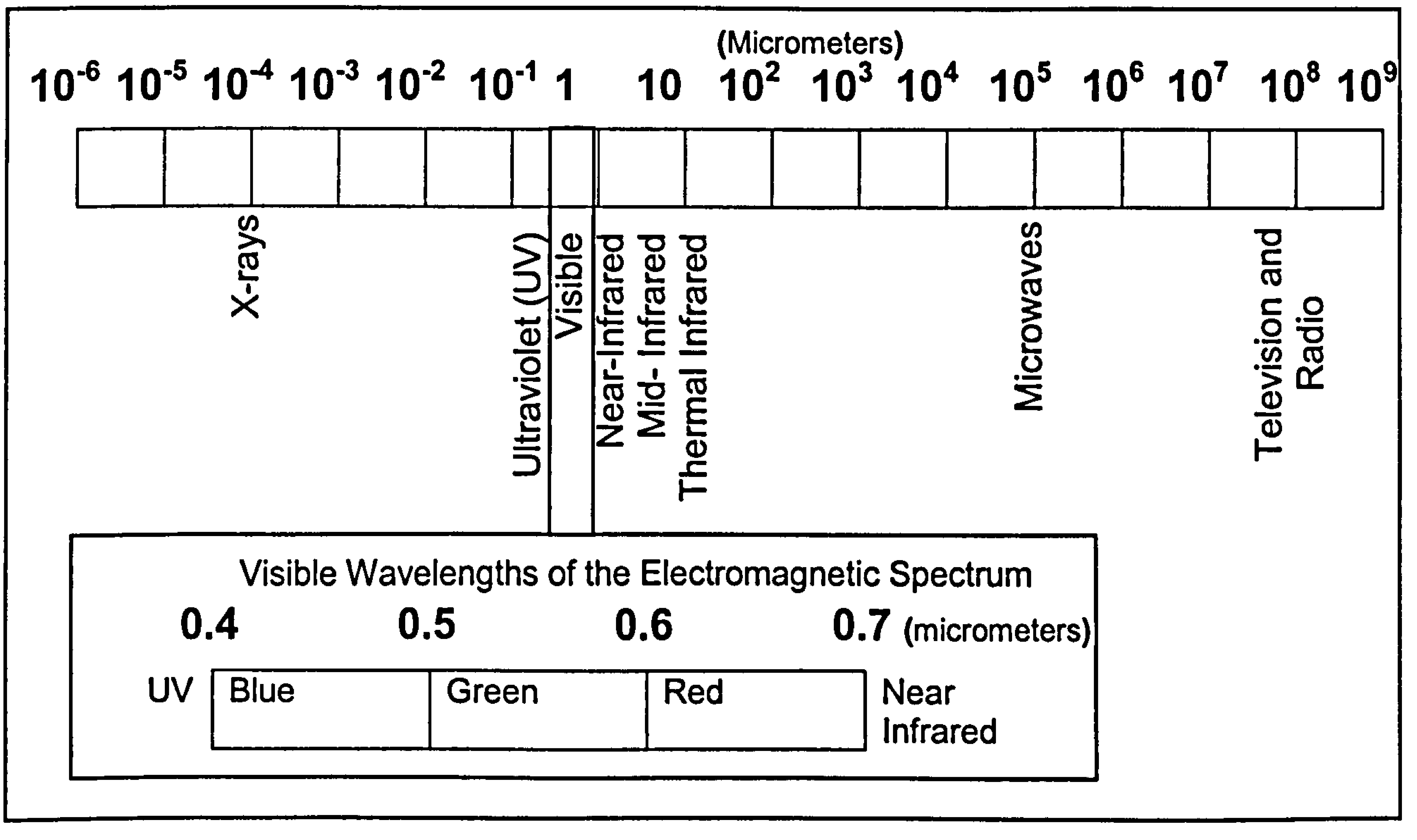
- 500–750nm - visible light region affected by chlorophyll absorption of red light
- 750–1350nm - near infrared region affected by internal leaf structure and dehydration
- 1350–2500nm - light-water absorption region affected by leaf water content.

Table 4.2 Remote sensing instruments use to map soil changes and vegetation stress (Based on Erb, 1981; Lyon, 1987; Irvine, 1997; Folkard, 1999; Hauff *et al*, 1999; Keller and Fischer, 1999, p. 81-88; McCubbin *et al*, 1999; Splayt *et al*, 2003)

	Soil Assessment	Veg. Stress – Chlorophyll Damage	Veg. Stress – Internal Leaf Damage	Veg. Stress – Leaf Water Content
ATM	✓	✓	✓	✓
Aerial Photographs	✓	✓	✓	✓
Airborne Videography	✓	✓	✓	X
SAR	✓	✓	✓	✓
Thermal Long Wave Infrared	✓	X	✓	X
CASI	✓	✓	X	X
Green	X	✓	✓	✓
Red	X	✓	X	✓
Near Infrared	✓	X	✓	✓
Thermal Infrared	✓	✓	X	✓
AVIRIS	✓	✓	✓	✓

Legend:
✓ = Is able to identify the specified type of soil or vegetation changes
X = Is not able to identify the specified type of soil or vegetation changes
Note: SAR = Airborne Synthetic Aperture Radar, CASI = Compact Airborne Spectrographic Imager, LANDSAT ATM = Airborne Thematic Mapper, AVIRIS = Airborne Visible Infrared Imaging Spectroradiometer

Figure 4.3 Wavelength region (micrometers) providing information about soil and vegetation conditions when using remote sensing instruments (Adapted from Barrett and Curtis, 1999, p.3-15)



Wavelengths between 690 and 740nm have shown to be effective indicators of vegetation stress caused by contamination. Research conducted in recent years by Curran *et al*, (1990, 1991), Jago *et al*, (1999), Ferrier (1999) and, Splayt *et al*, (2003(b)) have focused upon the visible to near-infrared (VNIR) wavelengths (500-750nm). This part of the electromagnetic spectrum can be used to identify the Red Edge Inflection Position (REIP) which is defined as the point of maximum slope between 690 and 740nm. It characterises the boundary between the strong absorption of red radiation by chlorophyll and the increased multiple scattering of radiation in near-infrared wavelengths (Curran *et al*, 1990; 1991; Jago *et al*, 1999). Splayt *et al*, (2003(b)) have demonstrated the effectiveness of integrating sequential aerial photographs, and the REIP data within a GIS model. The studies identified and mapped areas of stressed soil and vegetation that were previously unknown. Areas of leachate-induced vegetation stress were identified along landfill edges using field and airborne VNIR spectral data. The anomalous vegetation was analysed showing low chlorophyll concentrations and high levels of contamination. Figure 4.4 shows a field-based spectroradiometer while Figure 4.5 shows reflectance percentages measured for healthy and stressed grass using a field-based spectroradiometer.

The greatest challenge facing soil assessment is the need for spatially distributed information about soil conditions and contaminant plume dimensions. Site assessors require field assessment methods and instruments that will: (a) identify optimal sampling locations and sample numbers as well as; (b) describe spatially distributed information about soil conditions without changing them. The advantages of remote sensing technologies include the spatially distributed nature of information provided, insight into vegetation stress that links the source, pathway, and receptor, and the ability of such data sets to be integrated with other data sets (field data, boreholes, GPR and aerial photographs).

Figure 4.4 A field-based spectroradiometer (GER 3700) used to measure the 'Red Edge' of spectral reflectance on soil and stressed vegetation (photographed by NERC-EPFS, *cited* 2000)

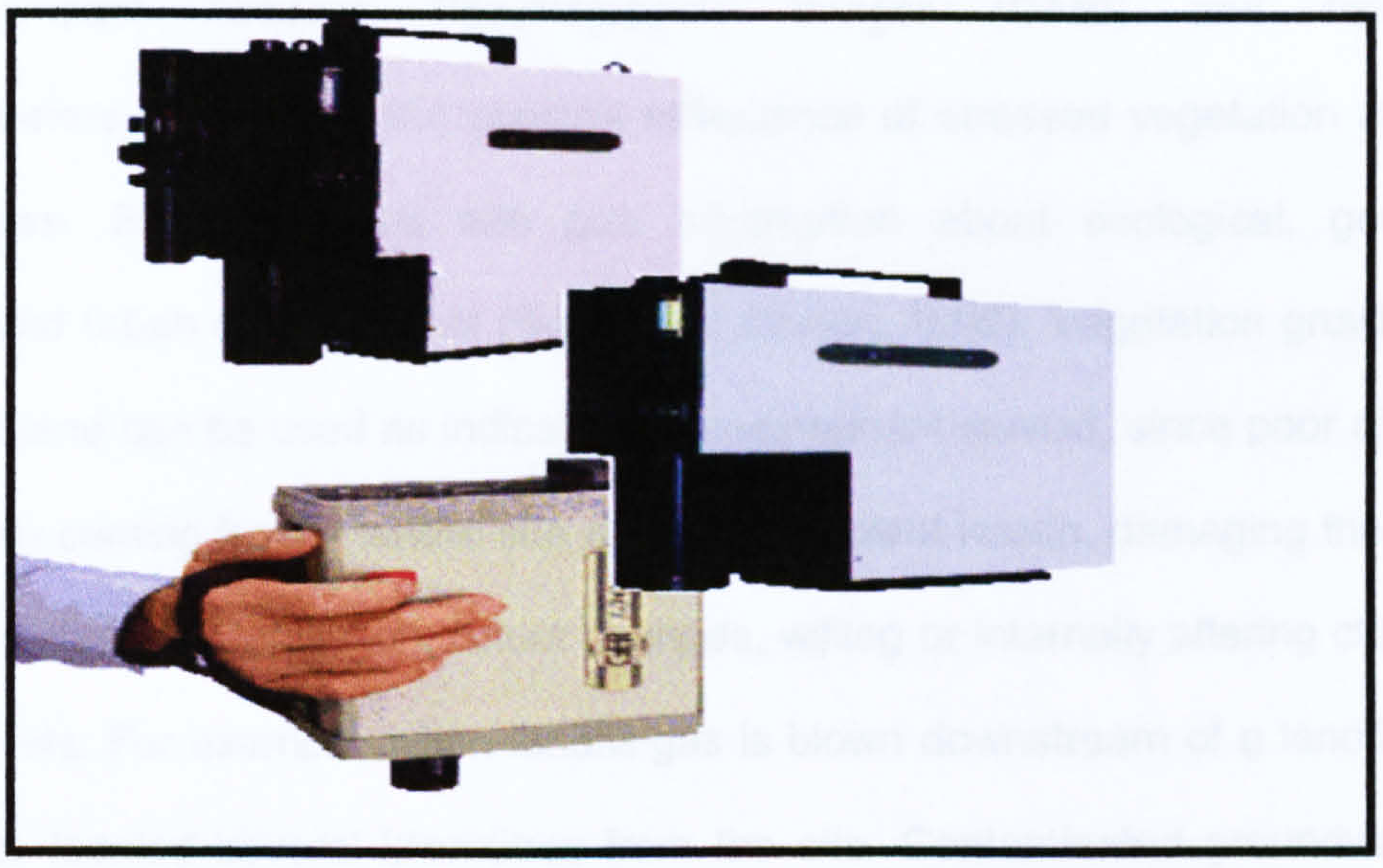
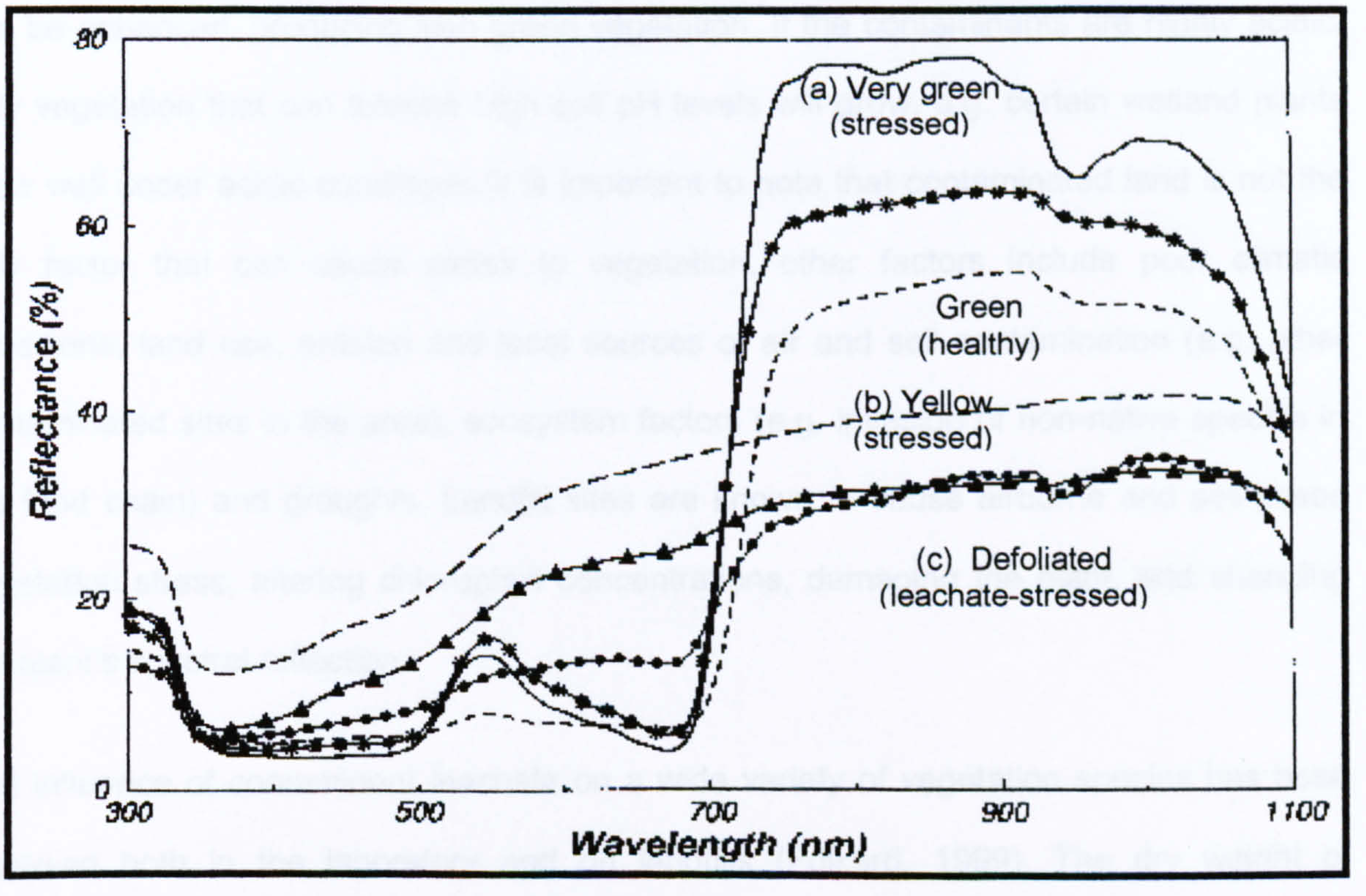


Figure 4.5 Graph showing reflectance data measured using a field-based spectroradiometer over healthy and stressed grass.



Legend:
(a) very green grass; (b) yellow grass; and (c) completely defoliated and discoloured grass

4.2.7 Landfill Remote Sensing: Field and Airborne Instruments

The remote sensing methods that were applied as a part of this PhD research included the use of Compact Airborne Spectrographic Imager (CASI) and field-based spectroradiometers to measure the spectral reflectance of stressed vegetation at one of the study sites. Such scanners can give information about ecological, geological, hydrological and urban environments (Green and Chrien, 1999). Vegetation growing near contaminated land can be used as indicators of contaminant spread, since poor air quality or water quality coming from a landfill site can damage plant health, damaging the internal plant structure, causing defoliation, colour changes, wilting or internally altering chlorophyll absorption levels. For example, when landfill gas is blown downstream of a landfill, it can damage crops located several kilometres from the site. Contaminated groundwater can contaminate soil and directly impact on vegetation health. The implications of contact with contaminants can be negative or positive. If they are high in nutrients, vegetation growth can be enhanced, producing lush green vegetation. If the contaminants are highly acidic, only vegetation that can tolerate high soil pH levels will grow, e.g. certain wetland plants grow well under acidic conditions. It is important to note that contaminated land is not the only factor that can cause stress to vegetation, other factors include poor climatic conditions, land use, erosion and local sources of air and soil contamination (e.g. other contaminated sites in the area), ecosystem factors (e.g. invasion of non-native species in the food chain) and droughts. Landfill sites are known to cause airborne and soil-based vegetation stress, altering chlorophyll concentrations, damaging the plant, and changing the plant's spectral reflection.

The influence of contaminant leachate on a wide variety of vegetation species has been analysed both in the laboratory and on landfills (Folkard, 1999). The dry weight of grassland species decreased significantly when the degree of leachate contamination was increased. Strong correlation exists between the concentration of many biochemicals within vegetation canopies and their reflectance spectra (Curran *et al*, 1991). Derivative analysis of reflectance spectra can identify the point of maximum slope at wavelengths between 690 and 740nm. This point, known as the Red Edge Inflexion Position (REIP), has been widely used as an indicator of foliar chlorophyll concentration (Curran *et al*, 1990; Curran *et al*, *1991;* Jago *et al*, *1999*). Calculation of the REIP depends on the number and spectral resolution of bands within the 650 to 750nm wave range and the smoothing and polynomial approximation algorithms (Ferrier, 1999). Healthy foliage normally has a REIP greater than 0.715 micron whilst foliage experiencing loss of chlorophyll tends to have REIP values below 0.710 micron. If the landfill area is very large and/or the leachate dispersion extends quite far from the landfill site then a ground-based spectroscopy approach becomes prohibitively time-consuming and expensive. Airborne-based spectroscopy offers the potential of overcoming these constraints and providing a cost-effective method for repeated monitoring of large areas.

4.3 Geological Characterisation

4.3.1 Introduction

Geological characterisation is a difficult task due to the heterogeneous nature of landfill sites. The aim of such a survey is to determine the subsurface stratigraphy and its interaction with groundwater flow and contaminant transport (Heron *et al,* 1998). A geological survey can be conducted using direct and indirect methods. The difficulty of geological characterisation is that some level of spatial interpolation between sample points is unavoidable. If direct methods of assessment are used, some level of spatial interpolation must be done to construct conditions between sampled points. If indirect methods are used, (such as ground geophysical surveys, e.g. GPR) then survey data must be calibrated using direct field data (e.g. geological profiles using boreholes), which do not provide spatially distributed information. Again, some level of unvalidated spatial interpolation must be done to construct subsurface conditions between surveyed points.

4.3.2 Characterisation Methods

(a) Direct Conventional Drilling Methods and Direct Push Methods

Conventional drilling is a cost-effective direct method for shallow targets from 33m in depth. Methods include Solid Flight Auger, Hollow Stem Auger, Wet Rotary, Air Rotary, and Sonic Drilling. The methods are not discussed in detail as the focus of this literature review is upon indirect methods of geological assessment. The following authors have reviewed these specific methods, their advantages and disadvantages: British Drilling Association (1991); Petts *et al,* (1998), p.161; Fetter (1999), p.390-396; Rafai *et al,* (1999); and ASTM (2000(a-b).

(b) Direct Push Methods

Direct Push technologies have emerged as efficient tools of geological assessment. They are less costly and enable relatively fast screening of site characteristics and rapid installation of a monitoring well network. Farrar (1998), MacFabe *et al,* (1998) and Rafai *et al,* (1999) discuss direct push methods in detail, explaining that they are generally smaller than conventional drilling equipment and may be mounted on a small vehicle. Examples include the CPT Cone Penetrometer, the Geoprobe and the Hydropunch. Such cost-effective and easily applicable methods are practical solutions for geological surveys, representing physical information at higher densities, decreasing the amount of spatial interpolation needed between sample points. Two landfill-based studies that interpolated geological conditions between borehole points without validating or stating the spatial assumptions between sampled points are Fatta *et al,* (1997); and Kjeldsen (1998(a-b)). Neither of the studies explained the number of boreholes or sample points that were used to interpolate geological conditions in these studies. Such geological assumptions are standard practice during site assessments, since time and funding for field studies is often limited. The cumulative uncertainty of inaccurate geological characterisation carries far-reaching risks. An example would be if a geological assessment at a landfill site failed to identify sand lenses at the base of the landfill that linked the upper unsaturated landfill zone with the middle and deeper subsurface saturated zones. By failing to identify this feature, the scope of risk posed by the site would likely be local and small scale. However, identifying this natural feature brings the potential risk posed by the site to a much higher scale, potentially threatening regional potable water supplies. Direct methods of geological assessment therefore provide invaluable information needed in the risk-based landfill site assessment (e.g. Figure 4.9).

(c) Indirect Geophysical Assessment Methods

Technological advances have given rise to new non-intrusive and multi-spatial geophysical assessment technologies. These include ground geophysical methods that have become widely accepted for geological assessment.

In relation to the site assessment of landfill sites ground geophysics is capable of outlining unsaturated and saturated zones, groundwater levels, and stratigraphy. It is also possible to delineate and locate buried objects such as waste, pipes, cables, and drums (e.g. Ramirez *et al,* 1998; Lemke and Young, 1999; Petersen and Majer, 1999; Powers and Haeni, 1999). Such methods are spatially efficient, providing subsurface information that could not be discovered by any other means. The instruments measure and map reflected or refracted sound, radio, and electromagnetic waves and are often used in combination with direct assessment methods to calibrate field study results (Kearey and Brooks, 1991, p.1-3). Table 4.3 lists different geophysics survey method instruments that are widely used in environmental investigations including seismic, electric and electromagnetic.

Seismic instruments measure travel times of reflected and refracted seismic waves. Electrical resistivity measures differences in the earth's resistivity. Radars using gravity measure spatial variations in the strength of the gravitational field of the earth. All three methods are effective in mapping geological, mineral and groundwater features, as well as subsurface features, contaminant plumes, and pathways of migration at landfills and abandoned industrial sites (Petersen and Majer, 1999; Powers and Haeni, 1999). Electromagnetic instruments measure electrical conductivity and inductance. They have shown to be effective in identifying the direction and extent of plume migration at landfill sites. Their data have also been used in risk estimation models calculating leachate migration in groundwater (Ramirez *et al,* 1998; Lemke and Young, 1999). GPR is a geophysical survey method that will be discussed in detail in section 4.3.4(d). It measures the travel time to reflected radar pulses as discussed in detail by Annan (1992).

Table 4.3 Application of indirect geophysical methods in various environmental engineering applications (Based on Kearey and Brooks, 1991, p.3; Vogelsang, 1995, p.5-54; Splajt *et al,* 2000, Splayt *et al,* 2003(b))

Method	Measured Parameter	Application	Operative Physical Property
Seismic	Travel times of reflected/refracted seismic waves	A, B, C, D, E	Density & Elasticity
Electrical Resistivity	Earth Resistivity	A, B, C, D, E, F, G	Electrical Conductivity
Gravity	Spatial variations in the strength of gravitational field of the earth	A, B, C, D, E, F, G	Density
GPR	Travel times to reflected radar pulses	A, B, C, D, F,G	Dialectic Constant
Electromagnetic	Responses to Electromagnetic radiation	A, B, C, D, E	Electrical Conductivity & Inductance

Legend:
A = Mineral Deposits
B = Engineering & Construction Investigations
C = Underground Water Supply
D = Municipal and Hazardous Landfill Sites
E = Abandoned Industrial Sites
F = Contaminant Plumes and Seepage Pathways
G = Geologic Features

Table 4.4 Expectations and misconceptions when using geophysical instruments and their survey results as part of the site assessment (Based on Vogelsang, 1995, p.3)

	Site Assessor Expectations when using Geophysical Survey Methods	Realistic Characteristics offered by Geophysical Survey Results
1	Results and subsurface images which are accurate without errors, e.g. data derived from survey results are expressed in measured units	Ambiguous results and subsurface images with several possible scenarios, e.g. data derived from survey results are only approximate values
2	No subjective interpretation is necessary when viewing survey results, e.g. clear and correct description of problems is expected since 3-D maps of the subsurface are often provided as survey results.	Interpretation of survey results is subjective, needing trained professionals and field data for validation, e.g. site conditions are presented in 3-D images but even careful image interpretation using highly trained staff can be incorrect or subjective.

The results of geophysical ground surveys are often difficult to interpret due to the heterogeneous conditions near landfill sites. There are two common misconceptions when using such methods. Firstly, assessors often expect a geophysical survey image to provide quantitative information about the subsurface. This includes exact subsurface depths (expressed in measured units) and concise descriptions of the subsurface. This is not possible if detailed field data are not available to calibrate geophysical survey results. Secondly, 3-D images of the subsurface are often produced for reports giving misleading images that can be easily misinterpreted by untrained personnel. This is especially the case for depth interpretations, as accurate depth estimation is not possible. An example is illustrated in Figure 4.1 showing sand lenses at the landfill edge of Study Site A. Images need to be validated by trained professionals using historical data sets and other surveys to overcome these problems (Vogelsang, 1995, p.4).

An additional issue is that geophysical methods require geological and groundwater level field data for validation. Hence, if there are errors in the interpretation of borehole data, this will be inherited in the validation of geophysical ground survey results. This uncertainty is recognised by Folkard (1999) and Hauff *et al,* (1999), who state that insitu sampling will increase in importance in order to validate high-tech multi-spatial measurement data. Such an integrated approach minimises the need for intrusive drilling and optimises the rate of ground cover (e.g. Smith, 1990; Kjeldsen *et al,* 1998(a); Rafai *et al,* (1999); and Splayt *et al,* 2003(a)).

(d) Ground Penetrating Radar

GPR is one form of ground geophysics that can be used to map the depth and location of subsurface features. GPR systems typically operate over the frequency range of 50 to 1000 MHz, with spatial wavelengths in the order of 0.1m to 2m (Davis and Annan, 1989). Research which specifically looks at the field conditions, data collection and instrument parameters that must be carefully assigned include Annan (1992); Annan and Cosway, (1992); Forde (1996); and Reynolds Geo-Sciences Ltd. (1999). GPR has been used in engineering and construction applications for over 15 years and has been used to investigate mineral deposits and map soil moisture variations. Examples of such studies include Davis and Annan (1989); Kilback and Barret (1997); Lanz *et al*, (1998); Smith and Eccles (1998); Lemes *et al,* (1999); and Peretti *et al,* (1999). Splajt *et al,* (2000), and Splayt *et al,* (2003(a)) took these studies as examples and applied the instrument to the mapping of near surface leachate migration from an unlined municipal landfill site, successfully mapping both leachate fluctuations at landfill edges and pathways of leachate seepage off site. GPR has the benefit of providing a visual image of the subsurface which is extremely difficult to sample under heterogeneous landfill conditions. The disadvantage is that the instrument, if not on wheels or in a cart, requires a relatively flat surface. It also needs field data for validation and trained professionals for interpretation. Figure 4.6 illustrates the different image facies that are used in interpretation of GPR cross-sections. Figure 4.7 shows the Pulse EKKO 100 being applied along the edge of a leaking landfill site.

Figure 4.6 Typical cross-section images used to interpret GPR survey data (cited in van Heteren *et al*, 1994, in Smith and Eccles, 1998)

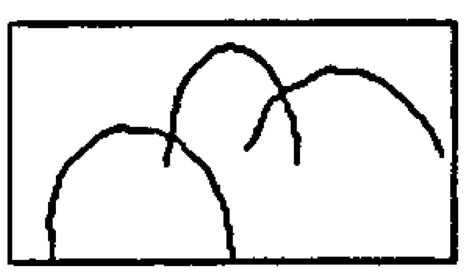
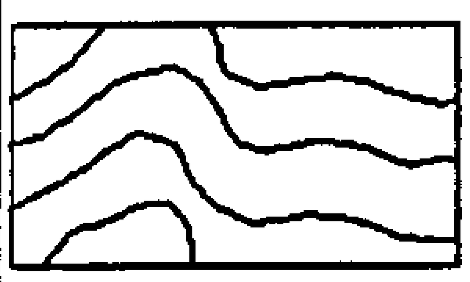
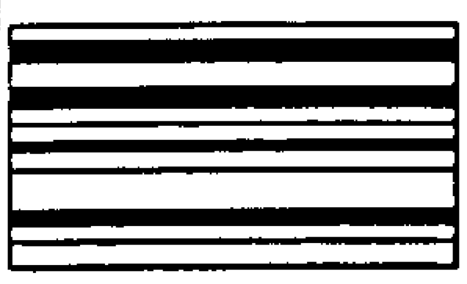
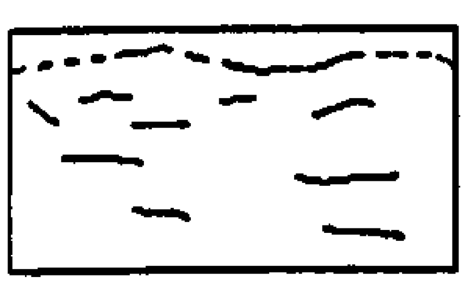
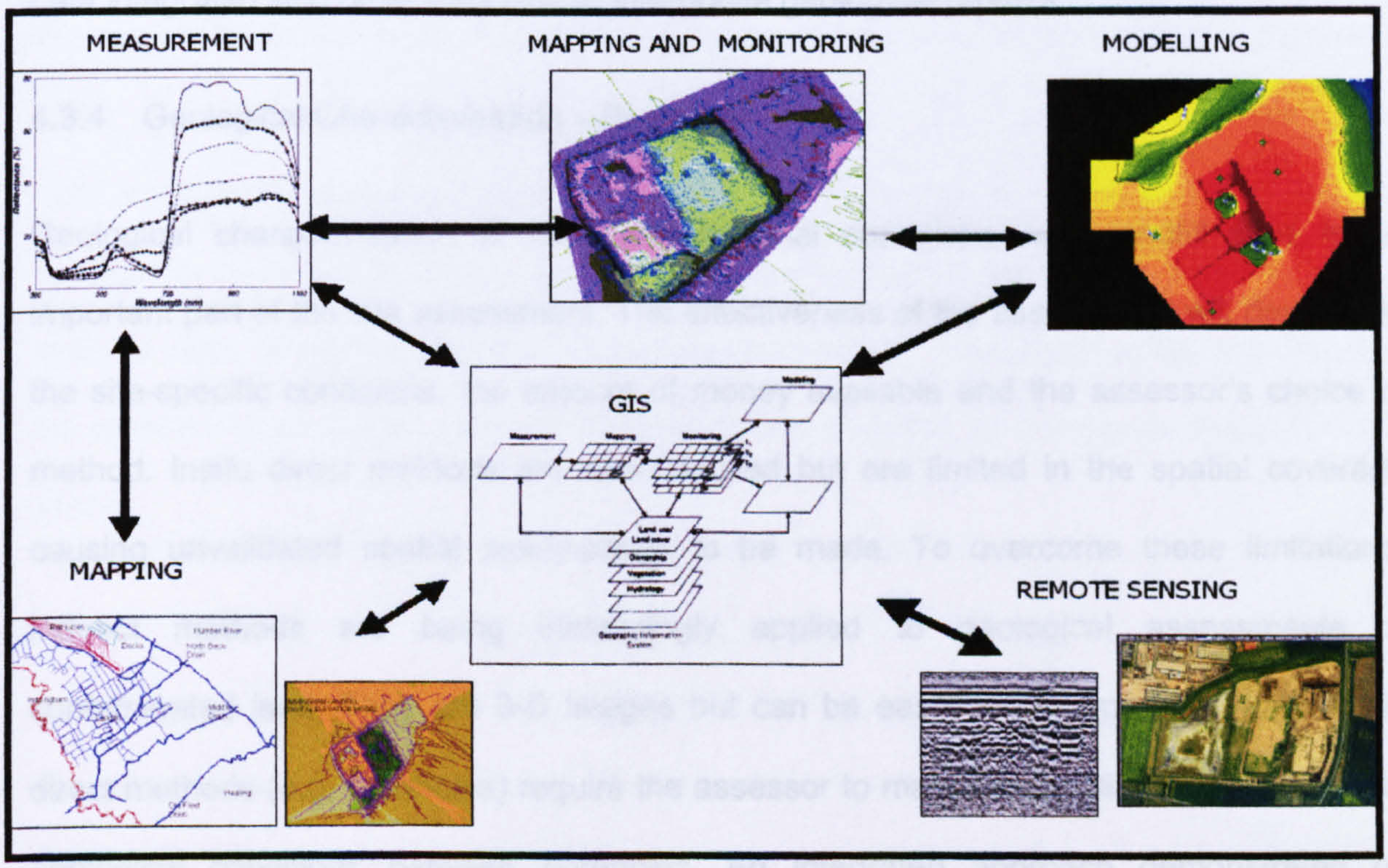
	Hyperbolic configuration with reflection-free areas underneath: Bedrock Pinnacles
	Wavy parallel configuration with uniform deposition rages: Distal glaciomarine clay
	Even parallel configuration with uniform deposition rates: Distal glaciomarine silt and clay with occasional sand layers.
	Non-parallel configuration with horizontal top reflector above irregular bottom reflector. Low energy deposits preferentially filling in low areas, e.g. Back-barrier sediments.
	Chaotic configuration (includes water table: Contoured or distributed material, e.g. Fill.

Figure 4.7 Application of the Pulse EKKO 100 GPR at Site A in August 2000 (photographed by J. Langham, 2000)

Figure 4.8 By integrating remote sensing and other data sets collected during the site assessment, geological, hydrological and contaminant plume conditions can be interpolated and used to validate landfill leachate models.



4.3.3 Environmental Remote Sensing

Another indirect approach to site-specific and regional geological characterisation is the use of remote sensing instruments. Successful examples are discussed in Olujic (1995) and Kaufmann *et al,* (1999). Olujic (1995) was able to quantify the amount of waste added to the Zagreb landfill over 30 years while characterising regional divides. The remote sensing images were validated using regional geological maps, landfilling records and regional borehole records. In a second similar study, Kaufmann *et al,* (1999) used airborne and ground spectroscopic data to obtain spatial information about contaminated landfill sites. Both studies integrated geochemical and hydrogeologic data sets to conduct a risk assessment of site conditions. Although different remote sensing instruments were used by the two studies, both demonstrate the strength of integrating multi-spectral RS technologies with existing data sets. Figure 4.8 shows the integration of remote sensing data integrated with other data sets to interpolate geological profiles.

4.3.4 Geological Characterisation – Summary

Geological characterisation of local and regional conditions near landfill sites is an important part of the site assessment. The effectiveness of the assessment will depend on the site-specific conditions, the amount of money available and the assessor's choice of method. Insitu direct methods are often applied but are limited in the spatial coverage causing unvalidated spatial assumptions to be made. To overcome these limitations, indirect methods are being increasingly applied to geological assessments of contaminated land. They are 3-D images but can be easily misinterpreted. Alternatively direct methods (e.g. boreholes) require the assessor to make assumptions and interpolate geological conditions between boreholes. An integrated approach demonstrated by Kjeldsen *et al,* (1998(a, b)) and Splayt *et al,* (2003(a)) is an effective way of overcoming the uncertainties of both direct and indirect approaches.

4.4 Groundwater Measurement and Assessment

4.4.1 Introduction

Site-specific groundwater contamination is greatly influenced by the fluctuating nature of subsurface flow at local and regional scales, influenced by hydrogeologic factors and by the physical and chemical contaminant properties. The fluctuating nature of water combined with the complex hydrogeologic and contaminant conditions make it difficult to obtain representative water quality samples (Kjeldsen *et al,* 1998(a)).

4.4.2 Groundwater Quality and Quantity

In order to overcome the uncertainties linked to measuring groundwater conditions during a site assessment, it is important to distinguish the difference between measuring (a) groundwater quality, and (b) groundwater quantities. Groundwater quality studies provide information about the contaminant source, contaminant classes and potential behaviour under given hydrogeologic conditions. Groundwater volumes are quantified using instruments that measure groundwater levels, regional recharge and site-specific, as well as regional hydrological fluxes. Such measurement provides information about local and regional rates of hydrological flux, contaminant migration, the nature of the pathway and the potential sinks and sources of contaminant accumulation as it migrates away from the source.

4.4.3 Direct and Indirect Methods of Assessment

Groundwater can be studied using direct or indirect methods. A common feature of direct methods is that they cannot identify contaminant sources or delineate plume extents due to their spatially limited nature. Indirect methods tend to provide spatially distributed information but do not provide quantitative values of contaminant concentrations and groundwater flow velocities. Table 4.5 compares popularly used direct and indirect groundwater assessment and monitoring methods, comparing them with the ability to meet site assessment objectives.

4.4.4 Targeted and Non-Targeted Sampling

When using the methods listed in Table 4.5, a single stage of sampling will not effectively characterise groundwater quality or groundwater quantities. It is common practice to take two to three sets of samples over time to address the variability in space and time. Sampling can be targeted or non-targeted. Targeted sampling for groundwater quality is conducted around known or suspected contaminant sources aiming to confirm the source presence and direction of contaminant migration. It is important to collect both upstream and downstream samples in such cases as well as having an understanding of both background contaminant concentrations and other local sources of contamination. Targeted sampling for groundwater quantities is conducted at known recharge zones, inter-aquifer exchange zones and areas with highly permeable geological conditions. The measured data gives local and regional rates of hydraulic conductivity, permeability and recharge. Non-targeted groundwater quality sampling aims at characterising contaminant-leaking areas or unidentified sources identifying contaminant types, extents, and concentrations (Petts *et al,* 1998, p.96-107).

Table 4.5 Comparing direct and indirect methods for groundwater measurement (Based on Petts *et al*, 1998, p.111-116, 160; Fetter, 1999, p.396 - 404; Rafai *et al*, 1999)

	GROUNDWATER QUALITY ASSESSMENT		GROUNDWATER QUANTITY ASSESSMENT		
GROUNDWATER ASSESSMENT AND MONITORING METHODS	**CONTAMINANT TYPE**	**CONTAMINANT SOURCE**	**CONTAMINANT CONCENTRATION**	**GROUNDWATER LEVELS**	**GROUNDWATER & CONTAMINANT FLOW VELOCITIES**
DIRECT METHODS					
Hand-held Water/ Soil Quality Analysers	✓	-	✓	-	-
Oil/Water interface meters	✓	-	✓	-	-
Samplers	✓	-	✓	-	-
Diffusion samplers	✓	-	-	-	-
Multiplayer samplers	✓	-	✓	-	-
Piezometer	✓	-	✓	✓	-
Borehole monitoring well	✓	✓	✓	✓	✓
Flow meters	-	-	-	✓	✓
Water level meters	-	-	-	✓	✓
INDIRECT METHODS					
Immunoassay sampling	✓		✓		
Invertebrate monitoring	✓		✓		
Soil gas surveys	✓	✓	✓		
Fibre optic sensors	✓		✓		
Natural attenuation	✓		✓		
Aerial photos		✓		✓	
Remote sensing	✓	✓			
GPR		✓		✓	

4.5 Groundwater Quality Assessment

4.5.1 Direct Methods of Groundwater Quality Assessment

Commonly used direct methods for groundwater quality assessments are listed in Table 4.5. These include hand-held water and soil quality analysers, oil and water inter phase measurements, diffusion samplers, multi layer samplers and borehole wells. The uncertainty associated with these methods is related to inappropriate sampling strategies that can produce results that do not represent subsurface plume dimensions. This issue was recognised by Kjeldsen (1998(b)) who used kriging to evaluate the spatial distribution of leachate samples. In order to delineate a contaminant plume it is important to sample off-site, as well as upstream and downstream to delineate contaminant migration pathways and confirm the contaminant source. The details of these methods, their advantages and disadvantages, fall outside the scope of this review. Studies that review direct methods of groundwater quality assessment include Fetter, (1999) p.355-420; and Rafai *et al*, (1999). Borehole wells are most frequently used to sample groundwater quality at landfill sites. Studies that have addressed issues relating to data variability and uncertainty in sampled wells are: Freeze *et al*, (1990) and Kjeldsen *et al*, (1998(a)). Freeze *et al*, (1990) focused upon hydrogeologic uncertainty - hydraulic conductivity values between wells, whilst Kjeldsen *et al*, (1998(a)) considered the heterogeneity and uncertainty in leachate concentrations between sampled points. Both studies used kriging to determine the spatial variation of groundwater heterogeneity. In recent years, several studies have also looked at the effectiveness of direct methods used to derive the level of risks posed by landfill leachate. Goodrich and McCord (1995) argue that sampling of groundwater quality does not take into account groundwater flow and solute transport processes that move the contaminant from the landfill to the receptor. They add that sampling methods often do not account for the heterogeneity in landfill contaminant data. Assumth (1996) and Bernhard *et al*, (1997) had similar findings as well. Assumth (1996) evaluated methods used in Finland to develop risk indices using data from 43 landfill sites. The study concluded that substantial uncertainty is carried in the measurement of site data and in the estimation of sample chemical properties. Bernard *et al*, (1997) tested the effectiveness of ammonia as a risk indicating parameter in leachate. The study found ammonia to be an effective indicator, however, further research is needed as the field data will be pre-determined by the sampling technique used, the frequency of the sampling, and the sample distribution. They also state that local hydrology and geology affect sample results.

All the mentioned studies recognised that three factors significantly impact the quality of data derived from wells. Firstly, the site-specific hydrogeologic conditions, secondly regional hydrological and geological factors and lastly, sampling methods and strategies all influenced the given results.

4.5.2 Indirect Groundwater Quality Assessment

(a) Innovative Methods

Indirect methods are listed in Table 4.5. They include diffusion samplers, multi-layer samplers, immunoassay sampling, invertebrate monitoring, soil gas surveys, fibre optic sensors and natural attenuation. Diffusion samplers are polyethylene bags filled with deionised water. The polyethylene membrane is able to transmit volatile compounds, allowing concentrations in the bag to be equal to that of the well being observed. Multi-layer samplers use diffusion to obtain groundwater samples, using dialysis cells initially filled with distilled water connected to a PVC rod. The cells hold up to 20ml, are able to sample within a few centimetres of the specified depth, and can determine micro-scale gradients. Immunoassay methods measure and detect concentrations in soil and groundwater by using different antibodies that attach to the contaminants. The colour change results from binding, detected using a spectrophotometer or human eye. The advantage of the technique is that it is real time, reproducible, reliable, portable, and easily defines contamination boundaries. Its disadvantages are that under specific conditions, results produce a high number of false positive findings. Also, extraction is difficult in peat and boggy areas and not all contaminants are sensitive to the antibodies. Despite this, the results in many cases are suitable and the method is much less expensive than well monitoring. Invertebrate monitoring comes in two forms - groundwater ecotoxicology and groundwater ecology, studying cause and effect relationships between contaminants, environmental changes, and impacts on organisms. Ecotoxicology studies the organisms within a groundwater system while ecology studies the links between organisms and their interaction with the environment. The methods are useful as vertebrates are good indicators of groundwater contamination and biomonitoring does not require knowledge of past contaminant or environmental status for effective evaluation to occur. The disadvantages are that the methods are still under development and must be site and contaminant specific.

Soil-gas surveys are used to identify VOC contamination such as industrial solvents, cleaning fluids, and petroleum products. Although instrumentation is not yet fully reliable, the method can be a good tracer option. Remote Laser-Induced Fluorescence is a method that uses long, thin plastic or glass flexible fibres. The fibres are coated with sensors to monitor changes in the refractive index that might be linked to contamination in groundwater or soil gas. It is easy to use, portable, has effective data retrieval, is cost-effective and capable of detecting many organic compounds. It works best with known aromatic pollutants, but again the method is still under development. As discussed briefly in section 4.2.3, natural attenuation relies on the groundwater's natural capacity to assimilate contamination, relying on chemical, physical, and biological characteristics of a given aquifer, and developing field protocols to facilitate natural attenuation in an aquifer. The approach is very promising for future site assessments, site monitoring and remedial projects but further research is still needed.

(b) Aerial Photography, Remote Sensing, Ground Geophysics and Groundwater Quality Assessment

Current and future technological developments will enhance the utility of 'indirect' multi-scale assessment methods in groundwater assessment. However, more development is needed to make these indirect technologies competitive in cost with other direct methods of geophysical assessment.

Aerial photography and airborne remote sensing scanners are effective tools for mapping groundwater quality. They provide spatially distributed images of near-surface contaminant plumes and their interaction with regional ecosystems. Data can be collected at various spatial and spectral resolutions, providing images of contaminated drainage and pathways of groundwater migration (Hauff *et al*, 1999). Remote sensing data measures thermal differences and red edge positions providing information about hydrological interactions occurring between the landfill sites and the local environment. Water quality studies that have integrated the use of aerial photography and remote sensing data include Hedge *et al*, (1994); Ferrier (1999); Hauff *et al*, (1999); Splajt *et al*, (2000), and Splayt *et al*, 2003(a-b)). Figure 4.8 illustrates the use of aerial photography, field maps, remote sensing data and other field data sets to construct a models of site conditions which simulate patterns of leachate migration around a landfill site.

Geophysical ground surveys also provide visual images of the subsurface. Examples of such studies include Kilback and Barrett (1997); Lanz *et al*, (1998); and Sauck *et al*, (1998). Geophysical methods discussed in section 4.3.4 and listed in Table 4.3 can also be effective tools for locating and mapping areas that may be experiencing groundwater quality changes. They are effective methods for delineating near-surface groundwater plume boundaries.

The biggest limitation of these indirect methods is the difficulties inherent in quantification of water quality. However, such methods provide valuable information that can be used to confirm subsurface site conditions, validate conditions between existing sample points, and identifying new sampling point locations.

4.6 Groundwater Quantity Assessment

4.6.1 Direct Methods of Groundwater Quantity Measurement

Groundwater quantity assessment encompasses assessing and measuring the local and regional surface and groundwater levels, flow velocities, vertical and horizontal measurements of local and regional hydraulic conductivity, and inter-aquifer flow. These data sets can be collected through boreholes and by taking physical samples of local and site-specific water, sediment and vegetation.

The popularity of using boreholes and piezometers is based in their well-tested reputation for providing quantitative information about: (a) hydrogeologic characteristics; (b) groundwater quality; (c) groundwater levels and groundwater velocities between sampled points (Fatta *et al*, 1997; Kjeldsen *et al*, 1998(a)). Table 4.6 lists borehole applications in hydrological assessments of landfill sites.

Table 4.6 Applications of monitoring wells in groundwater assessment (Based on Fetter, 1999, p.374-420)

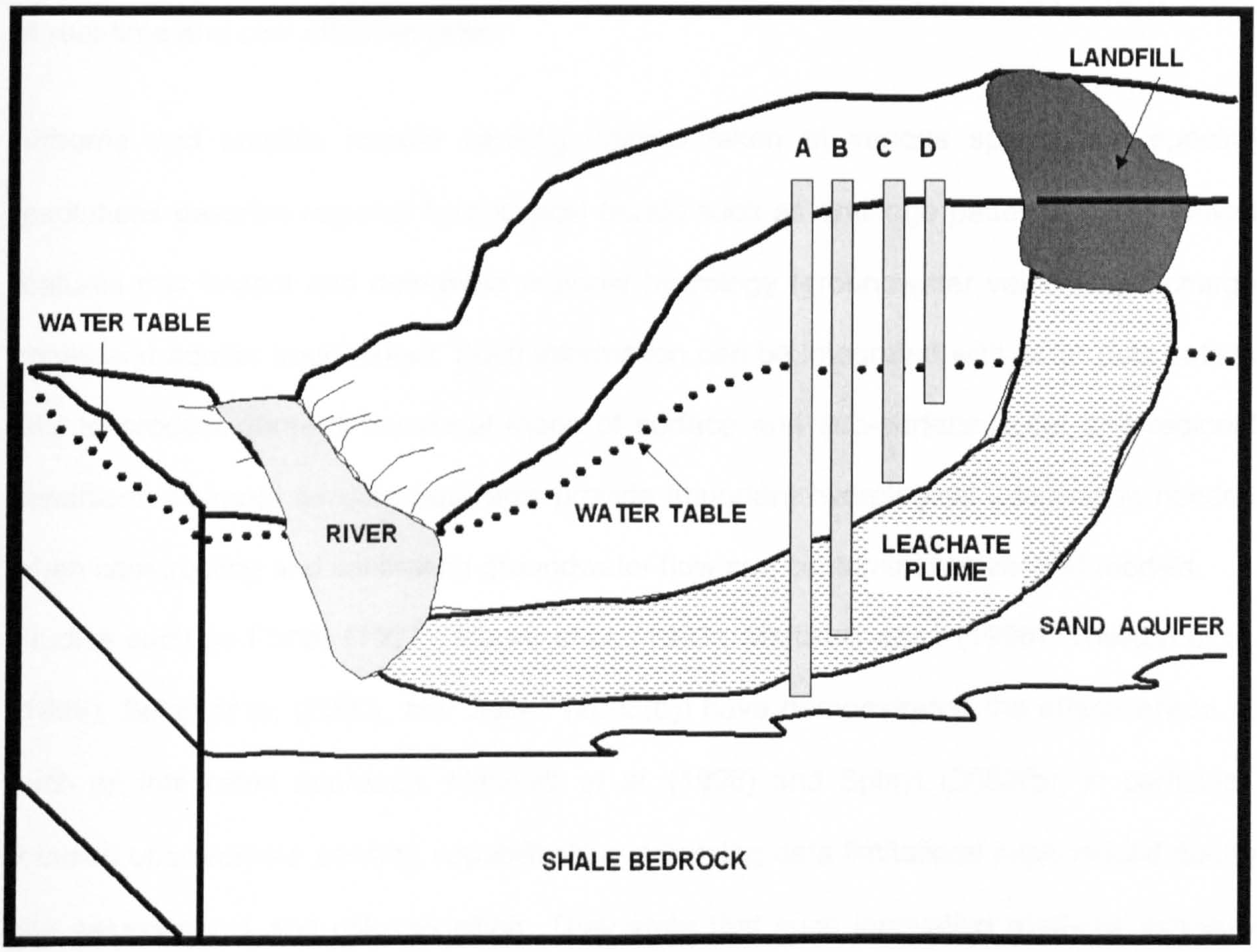
(A) Hydrogeologic Assessment	(B) Water Quality Monitoring and Assessment	(C) Water Quantity Monitoring and Assessment
• Testing the permeability of an aquifer • Providing access for geophysical instruments • Collecting a sample of soil gas.	• Collecting a water sample for chemical analysis • Collecting a sample of a non-aqueous phase liquid that is less dense than water • Collecting a sample of a non-aqueous phase liquid that is denser than water.	• Measuring the elevation of the water table • Measuring the potentiometric water level within an aquifer • Measuring cross-boundary flow between aquifers • Measuring saturated and unsaturated flow in different subsurface materials.

Piezometers are an alternative method that consists of a slotted pipe that is inserted into the subsurface. It has a sensor tip that is either made of porous stone, ceramic, or contains electrical inducers. An electric probe or chalked tape assists in measuring water levels inside the pipe. A discussion of construction and use of piezometers can be found in Fetter (1999), p.385 and Petts *et al,* (1998), p.160. Types of popular piezometers include the single tube open piezometer, the pneumatic-type piezometer, the hydraulic piezometer and the electrical piezometer. The limitation of piezometers is in using an adequate number of piezometers to represent subsurface hydrological and contaminant conditions. This problem is often addressed by placing several piezometers at different depths at the same location, as shown in Figure 4.9.

A few studies that have used piezometers in landfill assessments include Cherry *et al,* (1983); MacFarlane *et al,* (1983); and Kjeldsen *et al,* (1998(b)). Of these, Cherry *et al,* (1983) and MacFarlane e*t al,* (1983) study a landfill sites in Canada. Cherry *et al,* (1983) reviews six types of groundwater monitoring devices while MacFarlane e*t al,* (1983) conducts a site assessment looking at hydrological and contaminant plume properties. Both studies note the importance of recognising small-scale, site-specific heterogeneity at landfill sites. Cherry *et al,* (1983) concludes that piezometers, well location and well type should be chosen with reference to site-specific conditions. Kjeldsen *et al,* (1998(a-b), Hosseini (1993) and Ribiero (1999) also recognise these two factors as important elements of landfill site assessment, in which all three studies evaluated landfill and hydrogeological heterogeneity using kriging and a dense sampling regime.

There are several factors that influence the quality of data derived from piezometers and boreholes. For boreholes these include the quality of well construction, its design and maintenance (well-depth, width, casing and screens). For piezometers it is important to consider the investigation purpose, site conditions (such as depths required) and the number of sampling points needed (Rafai *et al,* 1999). Both methods are designed using a point sampling design, which can be costly if detailed assessment is needed. One way to overcome this problem is to use the 'nested configuration', placing dense piezometer samples in known problem areas. Another approach is to place a few boreholes across the investigated area positioning multi-layer piezometers between the boreholes to measure hydrological conditions at several depths in the subsurface, e.g. Figure 4.9. Alternative approaches are to use multi-level groundwater monitoring devices such as flow meters, oil/water interface meters, samplers and water level meters. Table 4.7 lists other direct groundwater assessment methods that are cost effective and readily used in groundwater. Details of these methods fall outside the scope of this study but they are reviewed in Petts *et al,* (1998), p.111-116 and 160-161; Fetter (1999), p.385-388; and Rafai *et al,* (1999).

Figure 4.9 Evaluating the influence of wells and piezometers used to measure groundwater and contaminant conditions (Adapted from Fetter, 1999, p.387)



<u>**Legend:**</u>

- Borehole A reaches the aquifer and intersects the contaminant plume but the measured concentration will be less then the actual concentration, as water is drawn from both contaminated and uncontaminated parts of the aquifer
- Piezometer B also penetrates the contaminant plume and will have representative leachate samples
- Piezometer C and Borehole D extends through to the saturated zone but do not meet the contaminant plume

4.6.2 Indirect Groundwater Quantity Assessment: Aerial Photography, Remote Sensing and Ground Geophysics

In the last decade, three methods have increasingly been used and integrated to map the hydrological conditions around landfill sites and similar types of contaminated sites. These include using aerial photography, conducting geophysical surveys and using remote sensing scanners to map the spatially distributed hydrological conditions. Studies conducted by Trenholm and Bentley (1998), Splajt *et al,* (2000) and Splayt *et al,* 2003(a)) have shown that using sequential aerial photographs to plan GPR surveys can provide detailed and focused descriptions of groundwater levels, saturation zones, and flux areas at real-time and cost effective rates.

Airborne and satellite remote sensing images taken at various spatial and spectral resolutions describe regional hydrological trends such as drainage patterns, geographical features that impact and determine regional hydrology (groundwater velocities, recharge zones and aquifer boundaries). Such information can be integrated with other data sets in GIS to produce three-dimensional maps of surface and sub-surface, local and regional conditions. Remote sensing data also provide important hydrological information needed when constructing and calibrating groundwater flow and contaminant transport models.
Studies such as Fisher (1993); Moore *et al,* (1993); Mattikalli *et al,* (1996); Theiken *et al,* (1999), Splajt *et al,* (2000), and Splayt (2003(b)) have demonstrated the effectiveness of such an integrated approach. Mattikalli *et al,* (1996) and Splayt (2003(b)) in particular, focused upon remote sensing capability in overcoming data limitations experienced during site assessments and risk modelling. They state that such innovative methods increase the amount of data available from site assessments. The integration of different data sets in GIS has linked site assessments to risk estimation modelling (e.g. Figure 4.8). Table 4.7 lists airborne remote sensing instruments that have demonstrated their effectiveness when measuring and mapping hydrological (water quantity) issues.

Table 4.7: Remote sensing instruments used to map and measure groundwater quantities (Based on Erb, 1981; Lyon, 1987; Njuku *et al,* 1996; Irvine, 1997; Folkard, 1999; Hauff *et al,* 1999; Keller and Fischer, 1999, p. 81-88; McCubbin *et al,* 1999; Splayt *et al,* 2003)

	Mapping Groundwater Levels and Surface Depth Changes	**Mapping Recharge Zones**	**Mapping Thermal Changes to Groundwater Levels**	**Mapping Hydro-geological Features such as Aquifer Boundaries**
ATM	x	✓	x	✓
Aerial Photographs	x	✓	x	x
Aerial Photographs in False Colour	x	✓	x	✓
Airborne Videography	x	✓	x	✓
SAR	x	x	x	✓
Thermal Long Wave Infrared	x	✓	✓	x
CASI	✓	✓	x	x
Blue	✓	✓	x	x
Thermal IR	✓	✓	✓	x
Mid IR	x	✓	x	✓
AVIRIS	✓	✓	x	✓

<u>**Legend:**</u>
✓ = Is able to identify the specified type of soil or vegetation changes
X = Is not able to identify the specified type of soil or vegetation changes
Note: SAR = Airborne Synthetic Aperture Radar, CASI = Compact Airborne Spectrographic Imager, LANDSAT = Airborne Thematic Mapper, AVIRIS = Airborne Visible Infrared Imaging Spectroradiometer

4.7 Summary

This chapter has underlined the importance of field data when assessing landfill site conditions, focusing upon indirect and methods that provide information needed for site assessment in particular, information about the chemical nature of the leachate plume, its sources and mechanisms of migration away from the site. A great deal of uncertainty is based in the methods used to define soil, hydrogeological, contaminant plume and groundwater conditions. In general, uncertainty related to site assessment data can come from several sources including:

1. Ineffective sampling (too few representative samples to represent site specific conditions);

2. Ineffective data collection methods used (e.g. deriving regional geological profiles based on a few borehole records);
3. Inaccurate field or laboratory measurements (caused by human error, instrument inaccuracy, wrong sampling or laboratory procedures, transport conditions that influence the bio-chemical state of the sample as it is transported from the site to the laboratory etc.) that alter the field results from the actual field conditions;
4. Cost of sampling and assessment which often determines and limits the amount and type of field data that can be collected;
5. Up-scaling or down-scaling field data to provide assumptions about site conditions and to be able to construct risk assessment models; and
6. Oversimplifying site-specific conditions (e.g. estimating hydraulic conductivity rates in which all waste across a landfill site which contains waste of different ages, is assigned the same hydraulic conductivity value) in order to understand heterogeneous conditions.

The sources of these uncertainties are often based in assumptions, errors in professional judgement and lack of instructions and methods to tell use everything about the subsurface as required for the site assessment. The most frequent source of uncertainty is the fact that each investigation (site assessment) requires an adequate amount of money available for assessment. Cost-benefit analysis of the type and amount of data that can be acquired by using different sampling and field assessment methods is a common factor in most site assessments. The key to addressing and decreasing sources of uncertainty is in exploring new methods that could provide higher certainty to site assessment data. This may include applying methods and instruments that have been applied in other field of science, or perhaps developing new field sampling methods and instruments. The important point to stress is only limited information about the heterogeneous nature of the subsurface will be available if only selected methods are used to conduct a site assessment.

When assessing soil, geology, groundwater quality or groundwater quantities, the objective is to: characterise contaminant presence; the impact on soil; the influence of geological conditions on contaminant migration; rate of influence of groundwater flow on contaminant migration; and the spatial extent of the sub-surface contaminant plume. The indirect methods used in soil, geological and groundwater assessment for landfill sites can provide insight into the extent of the contaminant plume. Remote sensing methods and GPR are two approaches that have been presented in this chapter, as instruments that could be successfully applied during the site assessment, under certain field conditions. There are four main disadvantages of remote sensing methods that are tied to its application at landfill sites: (a) the high cost of application; (b) the limited number of trained users; (c) the specific geophysical field conditions that are needed for successful application (e.g. clear skies); and (d) the fact that remote sensing data sets do not provide quantitative information, e.g. areas of contaminated soils are identified but soil contaminant concentrations are not measurable. Three of these limitations however are linked to the early stage of development and application. The high quality of information provided by remote sensing technology outweighs its current limitations. Such instruments have a promising future in the site assessment of landfill sites and similar types of contaminated land. However, there is a need to use an integrated approach when selecting instruments used during the site assessment.

The advantage of direct methods is that they provide quantitative information about contaminant concentrations, depth, and rates of groundwater and contaminant transport. Such quantitative information is needed in the risk assessment when estimating the risk and implications of site conditions on the local environment. The data is used to compare whether contaminant concentrations are within legal limits, whether they pose a threat to human health and to ecosystems, or whether they threaten water supplies. The underlying disadvantage of all direct assessment methods is that they are lacking in distributed information that is needed to describe hydrological and contaminant conditions over large areas, which in many cases requires site conditions to be interpolated between sampled points.

Both direct and indirect soil, geological and groundwater assessment methods should be considered, with selection depending upon site-specific landfill conditions. Such an approach is essential for describing, model and assessing the risks posed by leachate. With this in mind, the following chapter will focus upon risk estimation models, modelling groundwater flow and contaminant transport, GIS applications, and good modelling practises.

CHAPTER 5: RISK-BASED MODELLING OF GROUNDWATER AND CONTAMINANTS

5.1 Introduction

Technological advances of the last decade have brought forward new tools that are frequently applied in the site assessment of landfill sites and other types of contaminated land. This chapter will discuss groundwater flow and contaminant transport models, discussing:

- How these models have evolved,
- How modelling guidelines, the modeller, field data and the software used influence modelled results,
- Uncertainties and common sources of predictive model failure linking them to uncertainties and assumptions in field data,
- Good modelling practises,
- GIS and its role in risk estimation modelling of groundwater flow and contaminant transport,
- Geostatistics and its role in the risk assessment and in risk estimation modelling of groundwater and contaminants.

The findings of the literature review found in Chapters 4 and 5 are summarised at the end of the chapter, linking the literature review with the research investigations.

5.2 Mathematical, Environmental and Hydrological Models

Mathematical modelling has developed significantly in recent decades with advances in both scientific theory and computer technology (Stayaert, 1993). Mathematically based environmental models are increasingly being used to simulate, calculate, and better understand physical processes occurring in the natural environment. Advances in computer software and increasing hardware capacities of the last two decades have allowed the development of user-friendly, cost-effective, and multi-dimensional models for assessing risk in different environmental scenarios. As such, there is a high demand for such models as tools for environmental decision-making. Their increased application has changed the way environmental problems are visualised and evaluated, allowing data sets to be integrated in two and three dimensions. Mathematical models that simulate near-surface and sub-surface hydrogeological and contaminant transport processes are often used as risk assessment tools to estimate the concentration and distribution of contaminant migration from the source, along paths and to different receptors. Their strength lies in their ability to simulate past, present and future site conditions. Environmental models can be used in risk assessment in two ways: They can be used to test assumptions about environmental process or conditions, or they can be used to simulate historical and hypothetical risk scenarios.

The second approach uses models as a means of simulating the given environmental problem under different future scenarios. Popov (1968); Beven (1991); Beven (1993); Moore *et al*, (1993); and Stayaert (1993) discuss these two approaches to modelling. Although different hydrological models were tested in each of the studies, they all have similar conclusions; the two approaches contradict each other. The first admits that science does not fully understand the natural environment and its complex processes. The aim of modelling is to continue searching for explanations. The second approach overlooks the scientific uncertainty and searches for visual, quantitative, and cost-effective answers to real-world environmental problems (e.g. Moore *et al*, 1993; Zheng *et al*, 2000). This contradiction has brought environmental modelling to an ethical crossroad. Examples of studies and modelling guidelines that have discussed this contradiction in groundwater flow and contaminant transport models include Anderson (1979); Freeze *et al*, (1990); ASTM (1993(a-b)); CAMASE (1995); Sorooshian and Gupta (1995); ASTM (1996); and Golder Associates (2000). They all called for awareness in adhering to good modelling practises as to avoid diminishing the reputation of risk estimation modelling as effective tools of environmental management.

Water resource management is one area of environmental management that uses hydrological models extensively for risk assessment and management purposes.

Hydrological models have allowed planners and resource managers to quantify and estimate volumetric, spatial, and temporal distributions of past, present and future water supplies (Anderson, 1979; Matanga, 1996; Simmers, 1998; Brezak, 2000). The success of hydrological models in water resource management is tied to their attributes. Firstly, the models impose a level of certainty in risk assessment through the development of different model scenarios and through the calculation of model uncertainty for each given scenario. Secondly, as risk assessment tools, the various modelling software packages are cost effective, applicable to many types of hydrogeological regimes and do not require extensive amounts of background data for simulation. Lastly, the models produce multi-dimensional maps that ease the decision-making and planning process, giving a broader conceptual understanding of conditions, risks, and predictions that are being modelled.

5.3 Groundwater Flow and Contaminant Transport Modelling

Groundwater flow and contaminant transport models are often used as risk estimation models in landfill risk assessments. They are used to estimate the level of risk posed by leachate migration into local regional soils, surface waterways, and groundwater sources. The interest and application of such models have grown exponentially in the last decade for two reasons. Firstly, there is an increasing demand for predictive soil and water management tools. Secondly, there is an increasing scientific need to understand the complex nature of surface and groundwater flow.

In landfill and contaminated land management, they are used to simulate and estimate historical, current and future hydrological conditions (water flow directions and quantities), soil quality, paths of contaminant transport, well pumping capacities, risks posed to receptors, and site-specific remedial engineering options. Examples of such applications can be found in Matanga (1996); Donald and McBean (1997); Rowe and Nadarajah (1997); Wang *et al*, (1998); and Gburek and Floman (1999). Scientific applications of groundwater and contaminant modelling include modelling regional and local groundwater flow and inter aquifer interactions under saturated and unsaturated conditions (Bradley 1996; Soley and Heathcote, 1998), and hydraulic conductivity studies (Sudicky, 1986; Neuman, 1990; Hanor, 1993; Chen, 1996; McDougall *et al,* 1996). They are cost-effective and can be flexibly applied when used in the risk assessment and remedial decision-making process (Cunge and Erlich, 1999; Golder Associate, 2000).

There are many modelling packages used in both industrial and research applications of groundwater flow and contaminant transport studies. Popularly used one-dimensional models such as 'WHI Unsat Suite Plus' are usually used to provide initial insight into site-specific conditions. Examples of two and three-dimensional models are 'SEEP/w' (Geo-slope International, *cited* 2002), 'Flowpath', and 'Fractran' (for 2-D), and 'Visual MODFLOW', 'MODPATH', 'MT3D', 'FEFLOW', 'FRAC3DVS' (Waterloo Hydrogeologic 1999(a-c); *cited* 2003), LandSim (Golder Associates, *cited* 2002), and ConSim 2 (Golder Associates, *cited* 2003). These are popularly applied during the site assessment (to better conceptualise site conditions) and during the risk assessment to estimate the level of risk posed by leachate migration to receptors (Waterloo Hydrogeologic 1999(a); Environment Agency, *cited* 2003(b)). Multi-dimensional models allow the user to visualise model layers in two and three dimensions during model construction and simulation. This allows the modeller to conceptualise site conditions that are included in the model and to better understand how the modelling package works. Studies that have successfully applied 'Visual MODFLOW' for hydrological and landfill leachate simulations include Bradley (1996), Soley and Heathcote (1998), Wang *et al,* (1995), Kladias and Ruskauff (1997) and Garon *et al,* (1998). Effective and highly recommended British risk assessment models for contaminated land and landfill sites are ConSim and LandSim (Environment Agency, *cited* 2003(c)). They were developed by Golder Associates (UK) Ltd. for the Environment Agency and can assess the leakage of contaminants from a contaminated site or leachate from a landfill and the consequent impact on groundwater. Guidance on selecting appropriate groundwater risk assessment models in the UK has be developed by the Environment Agency outlined in McMahon *et al,* (2001a-b); Whittaker *et al,* (2001).

5.4 Groundwater Flow Model Evolution and Uncertainty

5.4.1 Model Evolution and Modelling Standards

Groundwater flow models were initially used in civil engineering, fluid dynamics, and geology (1960s-1980's). They were complex one or two-dimensional single process models that needed highly trained users and large computing capacities. The 1990's saw a revolution in both computing and socio-economic demand for environmental modelling. In reviewing groundwater and contaminant modelling studies conducted from 1979 through to 2003, it becomes apparent that modelling (for risk assessment purposes) has not paid enough attention to model assumption, and model uncertainty. These have been addressed on several occasions in the last 30 years. Examples include Anderson, (1979); Rogers *et al,* (1985); Beven (1991); Bergstrom and Jarvis (1994); Addiscott *et al,* (1995); Diekkrugger *et al,* (1995), Cunge and Erlich (1999); van Clooster *et al,* (2000); and Zheng *et al,* (2000). As a result, reputable government agencies, standard institutes and environmental consultants have begun promoting good modelling practices and publishing groundwater modelling guidelines to assist new and existing modellers in addressing these uncertainties and ensuring quality model results. Such publications include ASTM (1993(a-b)); Beven (1993); CAMASE (1995); Sorooshian and Gupta (1995); ASTM (1996); Golder Associates (2000); and publications such McMahon *et al,* (2001(b)) and Whittaker *et al,* (2001) sponsored by the Environment Agency. A number of approaches could be considered to improve the current state of model uncertainty: licensing modellers that deal with risk-based modelling of environmental issues; certifying modellers through a certification course or certification exam; developing a professional association that deals with the advancement, education and training of risk modellers. Integrating risk modelling into the existing professional certification requirements could be one simple way to improve that good modelling practises are adhered to. In the case of landfill site and contaminated land modelling, the modeller but have an extensive knowledge of land conditions in order to simulate these conditions effectively with in the model domain.

A less stringent approach could be training courses through professional associations (e.g. CIWM). Such approaches would ensure that individuals have adequate understanding of scientific, technical and environmental aspects of modelling as well as good modelling practises. Regardless of the form taken, modelling standards need to be encouraged, promoted and adhered to.

5.4.2 Software, Stakeholders and Gaps

The problems inherent in modelling software are being addressed through distributing or selling upgrade versions (at very reasonable costs). Improvements are available in five areas:

1. Compatibility with GIS and other database programs;
2. Spatial distribution abilities of model domain;
3. Calibration algorithms to encourage modellers to calibrate and test models (e.g. PEST, Doherty, 1999) and;
4. Increasing the availability of training sessions organised with government and academic institutions encouraging good modelling practices.
5. Increasing three-dimensional modelling capacities

However, there is still much to be done. There are no guidelines for stakeholders on how to evaluate the robustness of risk assessment models. Also, as stated above, there are no professional levels of control encouraging modellers to adhere to good modelling practices. Presentation of model results is another area of landfill risk assessment that needs attention. Stakeholders involved in the risk assessment need to be included and educated about site conditions and potential or existing risks. The way in which site conditions, model assumptions or model results are presented or explained may have a significant influence on how stakeholders interpret the potential risks, how they react to the information they have received and how they evaluated the remedial options and risks posed by site conditions. In order to address these areas, the root cause of model uncertainty needs to be documented in the model report and communicated clearly to stakeholders.

5.5 Model Uncertainty

5.5.1 Assessing The Root Cause of Model Uncertainty

In general there are four possibilities of 'model failure' or model uncertainty. These are:

(1) The software code – the groundwater flow and contaminant transport modelling software selected and used to model site-specific conditions.

(2) The availability of field data for model construction.

(3) The modeller and the conceptual model.

(4) The modeller and good modelling practices.

Modellers and software developers are aware of the possibility of predictive failure, however uncertainties in field data and model assumptions are not communicated as clearly to the stakeholder. Although a model will never fully represent field conditions, assumptions made about local and regional hydrogeological conditions, the contaminant source, parameter ranges, and lack of field data should be clearly stated in the model report or when presenting model results. One way to address model assumptions is through calibration. In hydrological models, this a detailed and time consuming process that requires careful modeller attention, however it is non-unique and cannot compensate for a lack of field data or poor modelling practices (Anderson, 1979; Beven, 1991; Doherty, 1999; COST 67, 2000; Golder Associates, 2000; Whittaker *et al,* 2001).

Literature was reviewed to compile three lists that linked common assumptions in field data to common assumptions in groundwater flow models to the above listed possibilities of 'model failure' (e.g. ASTM 1993(a); 1994(b-c); 1995(a-b); 1996; 1998; 2000(a-b); McMahon *et al,* (2001(a-c)); Whittaker *et al,* (2001)). The first, a list of common assumptions found in models was derived from groundwater modelling guidelines (Column 1, Table 5.1). The second list compiled field conditions that are difficult to add to a model in which assumptions need to be made to infer site conditions or site data into the groundwater model (Column 2, Table 5.1). Common reasons for 'model failure' (e.g. software, data, conceptual model, modelling practises) were then cross-referenced to the appropriate model and field assumption (Column 3, Table 5.1). Al eight common assumptions can cause the four types of model failure. However 'assumption B' - assuming isotropic transport in cells in zones is the most common assumption causing all four types of model failure.

This is an interesting finding for two reasons. Firstly, it is common practice for both research and industrial modelling to assume isotropic conditions as a default parameter (e.g. Sykes *et al,* 1982(a-b); Goodrich and McCord, 1995; Zhang and Schwartz, 1995; Nixon *et al,* 1997; Mooder and Mendoza, 1999). Secondly, in modelling applications, small-scale heterogeneity of hydraulic conductivity is often lumped to form large-scale homogenous areas within the model domain. This is a concern since field and geological conditions are seldom isotropic and homogeneous, and the hydraulic conductivity parameter directly determines groundwater and contaminant velocities and quantities in model simulations (Neuman, 1990). The data presented in Table 5.1 shows that the availability of field data for model construction and good modelling practices along with errors in the conceptual model are the three most common causes of model failure. Errors in the conceptual model also have a high frequency of occurrence however they can be corrected through model calibration and validation. The data presented in Table 5.1 underlines the importance of using adequate field data and best practises during model construction. The influence of modelling code errors had a low frequency of occurrence however it is commonly found in groundwater and contaminant modelling codes for two reasons. Firstly, scientific research needs to be strengthened, especially when describing groundwater and contaminant migration under heterogeneous saturated and unsaturated conditions. This was stressed by Zheng *et al,* (2000) who called for further research in this area stating that the current scientific understanding of such processes was validated only under controlled laboratory and field conditions. The second reason is that many proprietary-modelling codes are not capable of representing site-specific landfill conditions. In such circumstances it is the modeller who must (a) alter the modelling code to represent site-specific conditions, (b) find an alternative model which reflects site-specific conditions, or (c) take these code assumptions into account when modelling and when presenting the model results to stakeholders.

Table 5.1 Linking common model assumptions (A-H) with field conditions and with categories of predictive failure (Based on ASTM, 1993(a); 1994(a-b); 1995(a-b); 1996; 1998; 2000(a-b); McMahon *et al,* (2001(a-c); Whittaker *et al,* (2001).

	COLUMN 1: ASSUMPTION	COLUMN 2: FIELD CONDITION	COLUMN 3: CATEGORY
A	Homogeneous layers	Geological layers are often heterogeneous.	2-FIELD DATA 3-CONCEPTUAL MODEL 4-GMP
B	Isotropic transport in cells and zones.	Transport is often anisotropic but is difficult to measure due to its distributed and heterogeneous nature.	1-MODELLING CODE 2-FIELD DATA 3-CONCEPTUAL MODEL 4-GMP
C	No dispersion or diffusion occurs other then what is specified as the source.	There may be other local sources of dispersion & diffusion.	2-FIELD DATA 4-GMP
D	Constant contaminant and hydrogeological flow properties.	These properties vary depending upon climatic and groundwater flow conditions.	2-FIELD DATA 3-CONCEPTUAL MODEL 4-GMP
E	Constant hydraulic conductivity across model layers and zones.	Hydraulic conductivity and other hydraulic parameters are difficult to measure because the subsurface is variable and heterogeneous.	2-FIELD DATA 3-CONCEPTUAL MODEL 4-GMP
F	Sorption approximated by linear isotherm.	Sorption under field conditions does not always follow linear paths.	1-MODELLING CODE 2-FIELD DATA 4-GMP
G	Unknown parameter values estimated from regional averages, past measurements, published values etc.	Hydrogeological and contaminant parameters are difficult to measure because the subsurface is variable and heterogeneous.	2-FIELD DATA 3-CONCEPTUAL MODEL 4-GMP
H	Routine up scaling and down scaling of field data into model domain.	Maps and point samples of field conditions are often used to derive values for spatially distributed model parameters, historical conditions and boundary conditions.	2-FIELD DATA 3-CONCEPTUAL MODEL 4-GMP

Legend:
1=Modelling Code
2=Field Data
3=Conceptual Model
4=Good Modelling Practices (GMP)
A-H = Column 1, Assumptions that are commonly found in models

5.5.2 Modelling Code Selection

Selection of modelling software is an important issue for four reasons. Firstly, not all modelling packages will simulate site-specific conditions. For example, it is important to consider whether groundwater flow is saturated, unsaturated or a combination of saturated-unsaturated flow. This is an important issue because many models are based on saturated groundwater flow equations. In addition, there is a distinct gap in the scientific understanding and ability to model unsaturated flow (Goodrich and McCord, 1995; Fatta *et al*, 1997; Zheng *et al,* 2000) and not all software packages are able to effectively model all types of contaminants (Visual MODFLOW, 1999(a-c)). It is important to consider the type of contaminants present in order to identify the type of transport that is likely to occur (e.g. advection, dispersion, diffusion, retardation, etc). Finally, it is important to understand the scientific uncertainty incorporated into the finite difference or finite element equations used to describe flow. In most cases, the flow theories have been validated under laboratory conditions but have not been fully tested under diverse field scales and conditions (Anderson, 1979; Zheng *et al,* 2000). These code issues are of particular concern for site assessment and risk assessment of landfill sites because modelling code capabilities can have a significant impact on simulating contaminant transport directions and gradients that approach receptors (e.g. Rogers *et al,* 1985; Addiscott *et al*, 1995; Golder Associates, 2000 and Zheng *et al,* 2000).

5.5.3 Field Data

Models contain many parameters requiring a great deal of field information during construction. However, data can be unrepresentative of site conditions or inappropriate for model construction due to several factors:

(a) Site-specific conditions - the site-specific conditions may limit the amount and type of field data that can be measured

(b) Budget - the budget available for the site assessment may be limited (this influences factors (c), (d) and (f) in Table 5.1)

(c) Sampling method - the sampling method used to collect field data will determine the extent, amount and type of site-specific information that is available

(d) Sample distribution - the spatial distribution of collected field data may not represent field conditions

(e) Measurement error - errors in measurement and calculation resulting from field sampling and lab analysis may occur

(f) Parameter estimation - field data may be collected at different scales than model grid dimensions requiring field data to be downscaled or up scaled (spatially lumped) to fit into the model domain

(g) Calibration – the validity of model simulations is based on the amount of field data collected over space and time that is available for model calibration and validation.

5.5.4 The Modeller and The Conceptual Model

The modeller also has a large role to play in the determining the quality of a model and its ability to simulate fluctuations. It is important that the modeller has a good understanding of hydrogeology, site-specific conditions and experience in modelling in order to adequately transfer conceptual assumptions into the model domain.

The model will be influenced by the modeller's expertise depending on the modeller's:

(a) professional qualifications and background,

(b) level of experience in using a particular software package,

(c) understanding of site-specific conditions, and

(d) assumptions during model construction.

Many authors have raised the need to adhere to guidelines promoting best practices during model construction (e.g. Klemeš 1986; Beven, 1993; Addiscott *et al,* 1995; Golder Associates, 2000). However there is some dispute about the appropriateness of guidelines on the basis that models are site-specific. The view among many groundwater and modelling professionals is that robust calibration techniques, Monte

Carlo analysis and other forms of sensitivity analysis are more important and likely to be more effective than developing professional associations or other forms of formal control (van Clooster *et al,* 2000). However, during model construction the modeller makes critical assumptions about site-specific characteristics that are transferred from the conceptual into the mathematical model. In this transfer, the individual determines the model's structure by selecting the software and defining dimensions (grid size, spatial scale with in the model and time frame), parameter values, parameter distributions, boundary conditions across the domain, and calibration time. As a result, there is a growing need and recognition of the importance of appropriate training and adherence to good modelling practices during construction and presentation. A demonstration of this is the CD-guide published in the UK by the Environment Agency and a leading environmental constancy (Golder Associates, 2000). The CD outlines the steps needed when constructing a risk-based flow model for contaminated land, identifying trigger values and presenting the 'CLEA' model that is used to calculate critical contaminant concentrations in soils across the UK. 'CLEA' (standing for Contaminated Land Exposure Model), is a promoted by British governing bodies as a consistent framework for risk assessment. The model has been used to develop government-supported documents such as soil guideline values and risk assessment fact sheets (DEFRA and EA, 2002(d)). Other examples also include the LandSim and ConSim models, also promoted by the Environment Agency for modelling of groundwater and contaminants at landfill sites and other types of contaminated land (Environment Agency, *cited* 2003(b)).

5.5.5 The Modeller and Good Modelling Practices

In summarising the different publications that promote good modelling practices, the focus is placed upon model construction and calibration methods. Yet, there are two distinct gaps in the modelling guidelines. The first is that there is no clear specification of the number of field samples needed to provide information for modelling. There are a variety of publications that give guidance on sample numbers based on the size of the contaminated site (e.g. Department of Environment, 1994; BSI/ISO, 1995; CIRIA, 1995; BSI 2001), however these documents do not consider the number of samples needed for model construction. It is largely because of heterogeneous site-specific conditions and the mathematical differences from one model to the next. Therefore, the responsibility falls upon the modeller to evaluate the amount of data needed for model construction and to adhere to best practices during construction, validation and simulation.

The second is that no guidance is given about the implications of parameter estimations during model construction. The modelling investigations conducted as part of this research found that the distance (to a lesser extent) and concentration (to a greater extent) of modelled contaminants varied, depending upon the amount of field data available and upon the assumptions made during model construction (e.g. parameter range, model boundaries, and high flux areas of the model etc). Similar studies include Bergstrom and Jarvis (1994); Diekkrugger *et al,* (1995); Nixon *et al,* (1997); and van Clooster *et al,* (2000).

5.6 Good Modelling Practices

5.6.1 Introduction

A lot has been written in the last decade about good modelling practices during model construction (e.g. ASTM, 1993(a); Beven, 1993; CAMASE, 1995; Sorooshian and Gupta, 1995; ASTM, 1996; Golder Associates, 2000; McMahon *et al,* 2001(a-c); Whittaker *et al,* 2001). These best practice guides are written in three forms. The first is a step-by-step approach to model construction, taking the modeller from one phase of model construction to the next (e.g. ASTM, 1993(a); ASTM, 1994(a); CAMASE, 1995; and Golder Associates, 2000). The second approach offers scientific explanations to specific steps or elements of the modelling process that cause uncertainty or alter model results (e.g. Anderson, 1979; Beven, 1993; Sorooshian and Gupta, 1995; and Zheng *et al,* 2000). The third form is often found in modelling software manuals and training course materials, describing modelling steps, and individual parameter capabilities as well as the science that explains the model domain. One weakness of such literature is that modelling code errors or assumptions are not explained in detail (Zheng, 1990; Waterloo Hydrogeologic 1999(a-c)). During investigations 4,5, and 6, all three types of modelling guides were used. The combined approach was most effective during the initial phases while training to use modelling software (Visual MODFLOW and MT3D). Literature that provided step-by-step guidance to model building was most useful during the initial construction stages. Literature offering scientific explanations was effective during the sensitivity analysis, calibration, and verification of models.

5.6.2 A Review of Modelling Steps

The literature listed above applauds model simplicity, modesty, accuracy and testability. The modeller is also responsible for evaluating the quality of field data used in model construction, interpreting its patterns (over space and time) and evaluating whether the model reflects the data and field conditions (e.g. ASTM, 1993(a); ASTM, 1993(b); Hillel, 1986 in Moore *et al*, 1993; CAMASE, 1995; ASTM, 1996; and Golder Associates, 2000). Care must be taken to consider the assumptions in each step of the construction process including: defining study objectives; developing a conceptual model; selecting a computer code; constructing a groundwater flow model; calibrating the model and performing a sensitivity analysis; making predictive simulations; documenting the modelling study; and performing a post audit of model results. In the context of the site assessment, a primary step in the model building process is to identify the objectives by reviewing the findings of the preliminary study. A conceptual model of the site-specific geophysical conditions needs to be formed by reviewing the data collected during the preliminary study. The conceptual model can be considered as a working description of site-specific physical characteristics. It must test assumptions about the site's regional and local geology, hydrology, regional and local hydraulic properties, contaminant source, and contaminant properties (CAMASE, 1995).

The transfer from conceptual to mathematical model is an important step that affects the eventual accuracy of simulations (Popov, 1968; Anderson, 1979; Rogers *et al*, 1985; Stayaert, 1993). The ongoing adjustment of the site-specific conceptual model is an important tool, influenced by two factors; (a) the lack of field data and (b) impact of modeller bias (Freeze *et al*, 1990; ASTM, 1993(a); CAMASE, 1995). Applying different sampling strategies could form a wide variation of conceptual models (e.g. Argyraki *et al*, 1995; Diekkrugger *et al*, 1995). The conceptual model combines field information about site-specific hydrogeological conditions, contaminant concentrations, the waste types, age, and contaminant distributions to better assign applicable model parameters and transport equations suitable for the site-specific contaminant conditions. The ability of the modeller to accurately transfer this data into the model domain is also a factor that will influence model accuracy (McMahon *et al*, 2001(b-c); Whittaker *et al*, 2001). The problem however is that there is very little guidance for modellers on how much field data is needed for a valid site-specific model and how the modeller's decisions and assumptions could influence model dimensions and results. The reason for this is likely related to the site-specific nature of models and to the differences among modelling software. Some publications which have attempted to address how the model will react to parameter changes are ASTM, (1996); McMahon *et al*, (2001(b-c)); and Whittaker *et al*, (2001) in which Table 5.2 lists steps that the modeller can take to achieving successful calibration. The listed suggestions are intended for groundwater flow model calibration and may or may not apply in every situation. The fact that most modelling guidelines were published in the last decade reflects the 'newness' of risk-based groundwater and contaminant transport modelling explaining why gaps might be present.

Despite such gaps, one area that has indirectly addressed modeller decisions and assumptions in some guidelines (e.g. Golder Associates, 2000) is estimation of parameter distribution. As shown in Figure 5.1, different parameter distribution techniques can be used to set a parameter range. These include:

a) Uniform distribution in which a minimum and maximum distribution of parameter values is possible giving other parameter values between this range an equal chance;

Table 5.2 Suggestions for achieving successful calibration and avoiding non uniqueness (Cited ASTM, 1996)

1	As long as the input values are reasonable and the uniqueness problem is eventually addressed, matching historical groundwater levels and flow rates is justified for use of specific aquifer hydraulic properties
2	If recharge values are not changed during calibration, it is best to begin matching heads near the specified or constant head boundary and work toward the flux boundary
3	Hydraulic head depends on the resistance to flow
4	For transient models, begin with steady state scenarios, calibrating the hydraulic conductivity and then calibrating the boundary conditions in later transient scenarios
5	To raise hydraulic head at a point in the model, decrease hydraulic conductivity or transitivity, increase recharge, decrease conductance at nodes per cell near the specified area, and increase flow of groundwater at nodes near the specified area
6	To speed the response of water levels at a point to a change in boundary conditions, increase hydraulic conductivity or transitivity and change boundary conditions in that area or decrease specific storage in that area
7	For near surface water bodies, vary the hydraulic conductivity to raise or lower the slope of the water table and vary the conductance in the nearby boundary conditions to raise or lower water levels nearby, by the same amount
8	When two specified head boundaries with different levels are placed close together with the model domain, expect groundwater flow paths
9	Increasing leakage of a confining layer can cause groundwater levels in adjacent layers to become equal. Decreasing leakage causes levels in upper and lower levels to differ
10	Begin with a simple pattern of distribution in hydraulic properties, and then split them into zones of similarity. Avoid making too many zones
11	If there is undesired spatial correlation between residuals, re-parameterise model inputs and redefine parameter zones
12	If a model is too difficult to calibrate, there may be too many constant head boundaries that constrain the search for a stable model solution
13	The conceptual model identifies constant and flux head boundary regions along with similar parameter zones. Redefining the conceptual model is an ongoing process and an important role of calibration.

b) Triangular distribution in which the minimum, maximum and mode values set the parameter ranges;

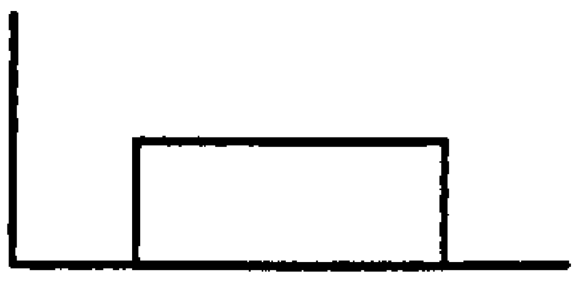
c) Log triangular distribution in which parameters vary by log order of magnitude, in which the triangular distribution of log values avoids skewing the distribution toward upper or lower values;

d) Normal distribution, which represents the statistically normal distribution, that many parameters fall into using a mean value and a standard deviation from the mean; and

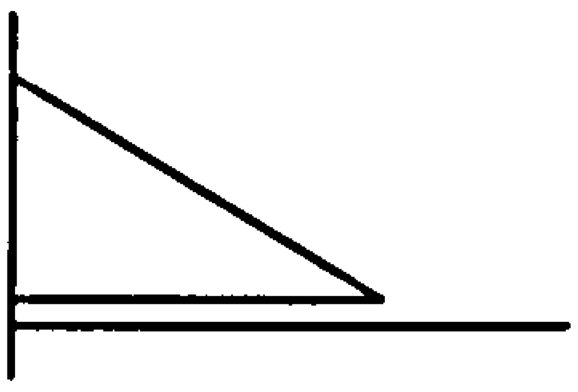
e) Setting guidelines according to US or UK guidelines for health risk in which log normal distributions of parameter values are used.

Of these listed techniques, two are most frequently applied. The first approach estimates the maximum and minimum parameter value by reviewing published values, industry standards, and prior knowledge and then tests the model using either uniform or normal distribution (Visual MODFLOW, 1999; Golder Associates, 2000). The second approach uses national guidelines for risks to human health (e.g. DEFRA and EA, 2002(c and d)). The model parameters are changed one at a time or in sets, testing to see which model conditions are needed to produce worst-case scenario results. The first approach is used due to its easy application. The second approach is used to test worst-case scenarios of human health implications. Since the objective of many models is to estimate the level of risk posed to sensitive site-specific targets such as ecosystems and local communities, reference indicators can be used to test the risk posed to these receptors. Examples of such referenced indicators are: (a) national or local water quality parameters; (b) national human health indicator parameters such as British 'Soil Guideline Values' derived using the CLEA model for residential, allotments, industrial and commercial contaminated sites (DEFRA and EA, 2002(d)); and (c) site-specific guidelines developed based on site assessment findings (Golder Associates, 2000).

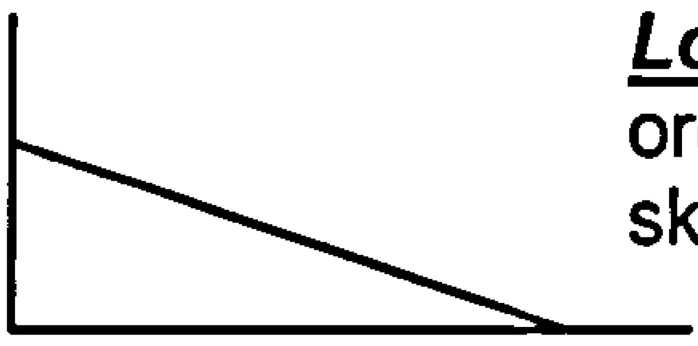
Figure 5.1 Various parameter ranges that can be used to account for data uncertainties in the conceptual model (Golder Associates, 2000)



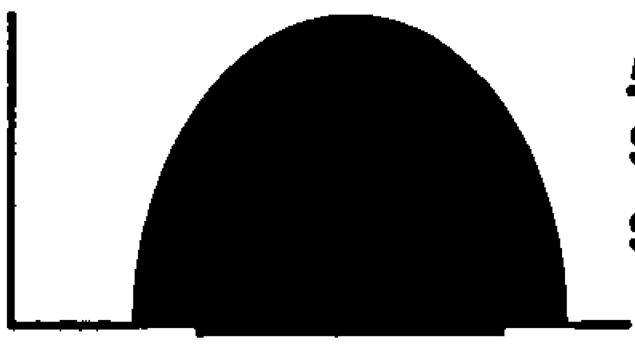
Uniform Distribution: Where a minimum and maximum distribution of parameter values is possible in which there is also an equal chance for all parameter values in between.



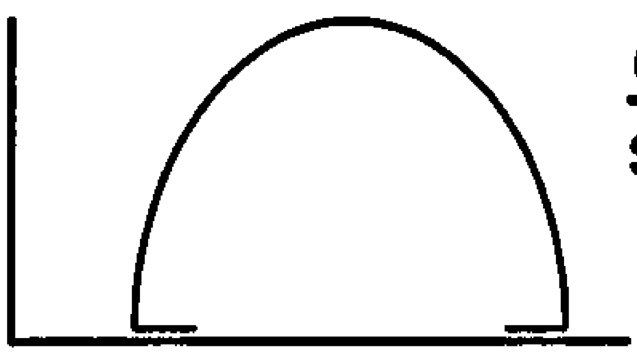
Triangular Distribution: In addition to the minimum and maximum value, it is possible to identify the mode, the value most likely to occur.



Log Triangular Distribution: When parameters vary by log order of magnitude the triangular distribution of log values avoids skewing the distribution toward upper or lower values.



Normal Distribution: Many natural parameters fall into a statistically normal distribution using a mean value and a standard deviation from the mean.



Other distribution types: If data available is limited or very specific, other input distributions will take shape.

5.6.3 Evaluating Gaps in Good Modelling Practices

To summarise there are three main gaps in good modelling practice guidelines:

a) Field data used and its impact on model results

b) The degree to which the modeller can impact model results.

c) Effective methods of communicating model results to stakeholders.

Model results are very sensitive to variability in parameter values selected by the modeller. Tests carried out by Bergstrom and Jarvis (1994) on seven modellers using identical data sets gave seven different results. The conclusions were that laboratory data does not always represent field conditions; parameter estimation on the part of modeller show great sensitivity to model results and that if a model is not carefully calibrated, it should not be considered as a viable management tool. The study conclusions were that modellers need to be trained and some form of professional accreditation is needed to regulate the quality of model construction. A study was conducted by Diekkrugger *et al,* (1995) used 19 different models, and 19 modellers with different data sets derived from the one study. Nineteen different simulations concluded that the complexity (2-D or 3-D) of a software model does not affect the quality of results; the modelling software code impacted model results. This research also concluded that the modeller influenced and is responsible for ensuring good results. They noted that there is a need to develop methods for deriving parameter values such as hydraulic conductivity and longitudinal and transverse dispersion. Such parameters are often inaccurately estimated because they are difficult to measure, heterogeneous and site-specific. The study concluded that the modeller's interpretation of data and their knowledge of the modelling software impact model results. In a third study, van Clooster *et al,* (2000) used a vertical, one-dimensional water, solute, heat and pesticide transport model and one data set. The model and data set were given to 36 experienced modellers. The study results showed that field and laboratory data as well as the software package impacted model results. The modeller influenced parameter values. The study strongly suggests that good modelling practises need to be adhered to and registration of pesticide models is needed for effective risk assessment and decision making.

Bergstrom and Jarvis (1994) note that most contaminant transport models that are present today were initially developed as research tools, calling for caution when applying them to risk assessments. The success and popularity of such models however is unlikely to fade with time. Diekkrugger *et al,* (1995) summarises the impact of the modeller stating that the experience of the scientist applying the model is as important as the differences between various modelling codes. van Clooster *et al,* (2000) points to a crisis in the state of groundwater modelling with an urgent plea to both regulators and industry users calling for the application of good modelling principles when modelling contaminant behaviour in a regulatory context. They state:

'...the strong impact of the modeller on modelling results requires the development of robust techniques, which can be easily applied in an engineering context... (cited in van Clooster *et al,* 2000)'

The three studies show that field data and the type of modelling software used were determining factors influencing model results. In modelling situations where field data was scarce, the modeller's knowledge of contaminant hydrogeology and conceptual understanding of site-specific conditions affected model results. Quantifying and minimising the impact of the modeller on model results is a difficult task as it is problematic to measure the number of assumptions made during model construction. Calibration and validation is one way to over come some assumptions however if field data are not representative, then it is up to the modeller to make assumptions about site conditions.

The one remaining gap that has not been effectively addressed adequately in modelling literature is communication with stakeholders. Some guides are available (e.g. ASTM, 1995(a); SNIFFER and Environment Agency, 1999) however professional ethics are needed to ensure that model assumptions and related risks are effectively communicated to the stakeholder. Model results presented to stakeholders, should also be accompanied by three clear explanations: (a) the model objectives; (b) the model framework; and (c) the system being modelling. The model objectives outline the reasons for model construction and simulation. In the case of landfill sites, many risk estimation models aim to simulate the direction, distance and concentration of leachate migration away from the site. The model framework identifies the parts of the site that will be modelled. For landfill site models these can include leachate migration from a landfill cell, leachate transport through subsurface hydrogeological features and leachate arrival at a receptor (abstraction wells, potable groundwater supplies etc). The system being modelled must also be described. It is at this point that modellers must explain the different conceptual models, general model assumptions, the extent of field data collected and used in model calibration and the degree of uncertainty that is inherent in the model results.

The analytic-deliberative process has been successfully applied in recent years when dealing with communicating and resolving contaminated soil case studies in the US (e.g. Contaminated Soils Forum, *cited* 2000; Guglielmo Kinney and Leschine, 2002). One approach that would strengthen risk communication with stakeholders with regards to modelling would be to build elements of this process into modelling guidelines in which deliberation with stakeholders could be an ongoing process during site-specific model construction and simulation. Such an approach would educate the stakeholder about the amount of data needed for model construction, the assumptions made during model construction and how these factors influence the model simulations and results. Alternatively, stakeholder feedback could ease the modelling process, e.g. providing undocumented site-specific information about the site or its history that may not be documented.

Two tools that have been refined and developed in the last decade, in support of the risk-based approach to contaminated land and landfill site assessment are GIS and geostatistical modelling. These two modelling tools offer practical solutions to over come predictive failures (described in Table 5.1) such as unrepresentative field data, inaccurate conceptual models and inappropriate modelling practices during model construction and results presentation.

5.7 GIS in Support of Good Modelling Practices in the Risk Estimation Modelling of Groundwater Flow and Contaminant Transport

5.7.1 GIS and the Conceptual Model

GIS has brought major geo-computing advances to contaminated land remediation, and related risk-estimation modelling. It allows for different scales of data to be integrated, mapped and spatially analysed for trends or features that otherwise have not been apparent. Field data collected during the site assessment are easily mapped and integrating with other data sets within GIS, producing multi-dimensional spatial and geostatistical models that provide additional information about site conditions. Studies that have demonstrated this application include Harris *et al,* 1993; Townsend and Walsh, 1996; Theiken *et al,* 1999; Hooker *et al,* (2000) and Splajt *et al,* (2000).

5.7.2 GIS and the Groundwater Flow Contaminant Transport Models

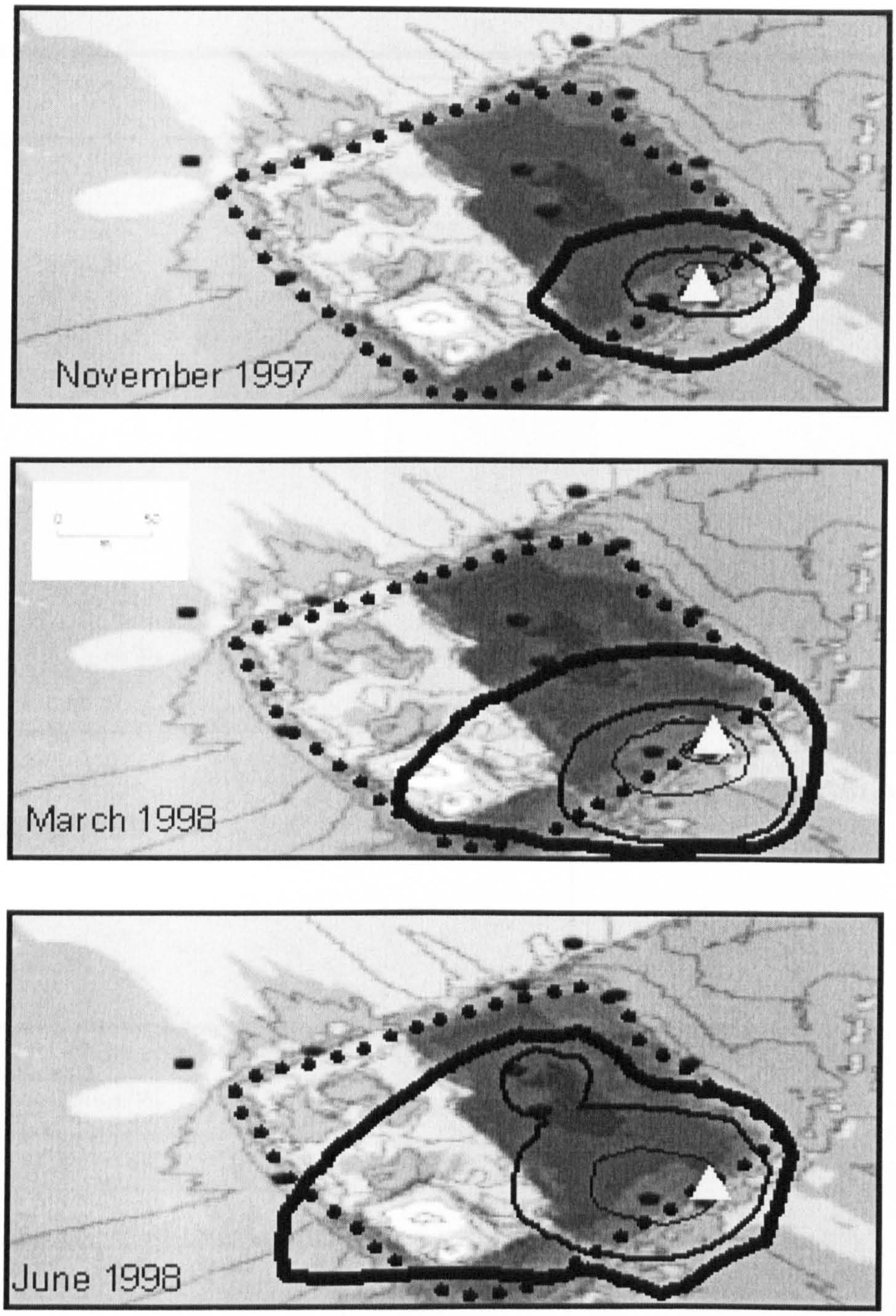
When constructing models there are four ways of using GIS to strengthen the modelling process. These are: (a) using it as a database for model construction, (b) using it as a modelling tool to spatially evaluate patterns in data sets and spatially estimate parameter distributions, (c) building and integrating spatial and hydrological models, and (d) constructing risk maps to validate risk estimation model results. The use of GIS as a hydro-geological and contaminant database allows the integration of different scales of data sets (e.g. Townsend and Walsh, 1996; Ferrier and Wadge, 1997; and Simmers, 1998).

GIS also offers two quantitative solutions during model construction if there is a lack of spatially representative field data. The first approach lumps all the available field data into a contour map of site conditions, but does not account for spatial variations between sample points. The second approach accounts for spatial variations between sample points using geostatistical approaches to model the spatial variability of a parameter (e.g. Simmers, 1998; Theiken, 1998; Goodwin and Hardy, 1999; Hooker *et al,* 2000; and Splajt *et al,* 2000). Both applications supported the calibration and verification process during model construction, as illustrated in Figure 5.2 which shows GIS-based maps of groundwater levels at a landfill site in North Eastern England. Such contour maps provide valuable site information about groundwater and leachate fluctuations between sampling phases.

The third application is to link the GIS and a groundwater flow and contaminant transport model. Integration of the two systems can be done manually or automatically. Model results can be exported into a GIS, calibrating and validating results with other data sets in the GIS (e.g. aerial photographs, remote sensing data and field data).

Figure 5.3 illustrates an integrated application using 'ArcView GIS v.3.2' and 'Visual MODFLOW' in which groundwater flow was simulated for a landfill site in North Eastern England. The groundwater level results were then exported into the GIS to compare modelled and measured groundwater contours. Such 'linked' systems are illustrated in Figure 5.4, where model parameters are calculated using data from various GIS layers. The data are manually or automatically transferred between the GIS and model's parameters. The advantages of such integrated models are shown in river management studies in which the GIS are linked to hydrological and rainfall in-line monitoring instruments. As the GIS databases are updated, so are the maps and risk estimation models used for calibration and validation of river models (Goodwin and Hardy, 1999). In the case of groundwater flow and contaminant transport models, GIS is capable of supporting the modelling process providing a platform for field data inventory, data integration and data spatial analysis that can be used to better define model parameter ranges, spatial boundaries and site conditions. These attributes have made the system a standard tool of contaminated land management in local authorities across the United Kingdom (Hooker *et al*, 2000).

Figure 5.2 Groundwater level contour maps that were created in ArcView GIS v. 2.3 to identify areas of high recharge



Legend:

= reference point representing borehole 13

= >4m AOD

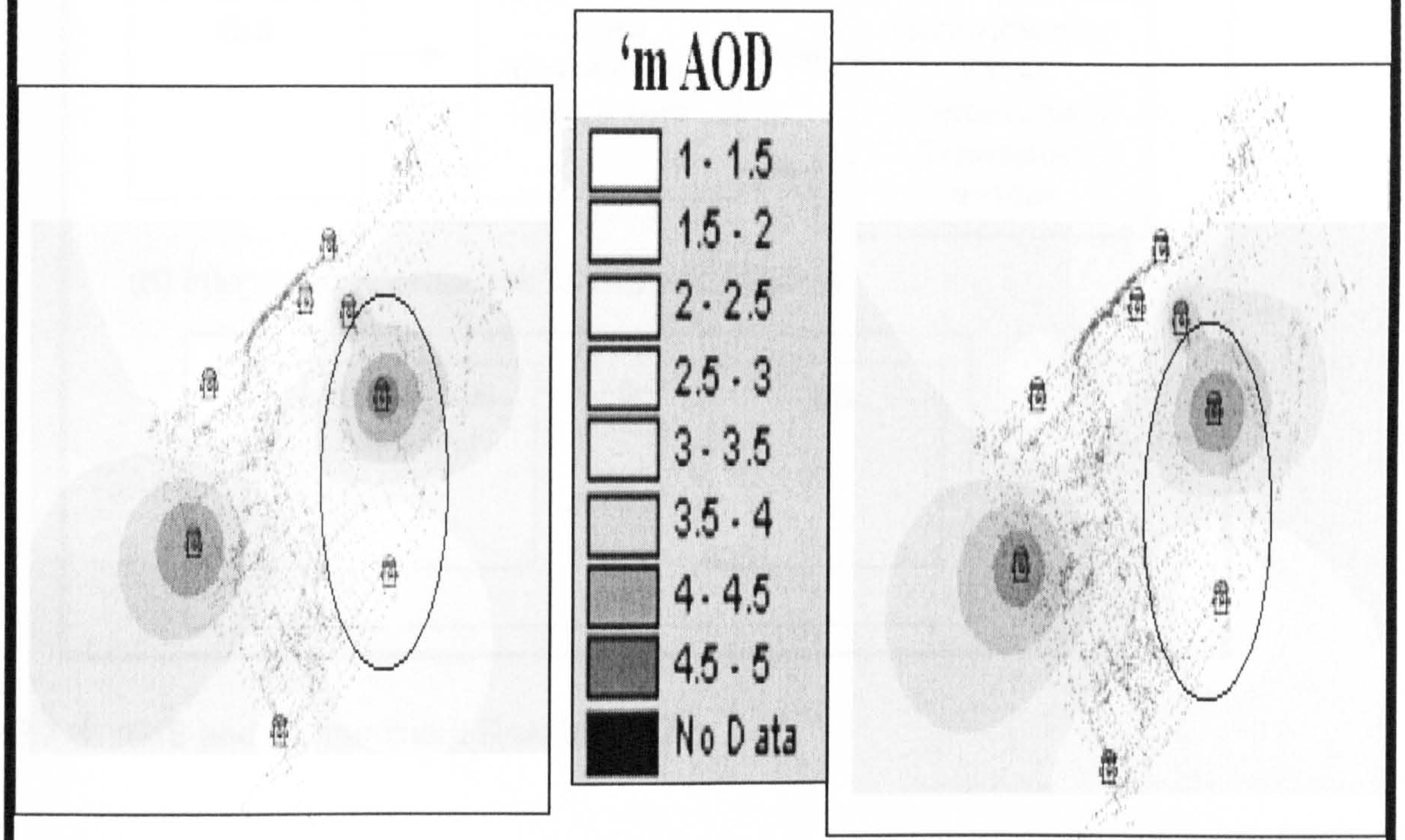
= 3 – 4 m AOD

= 1 – 3 m AOD

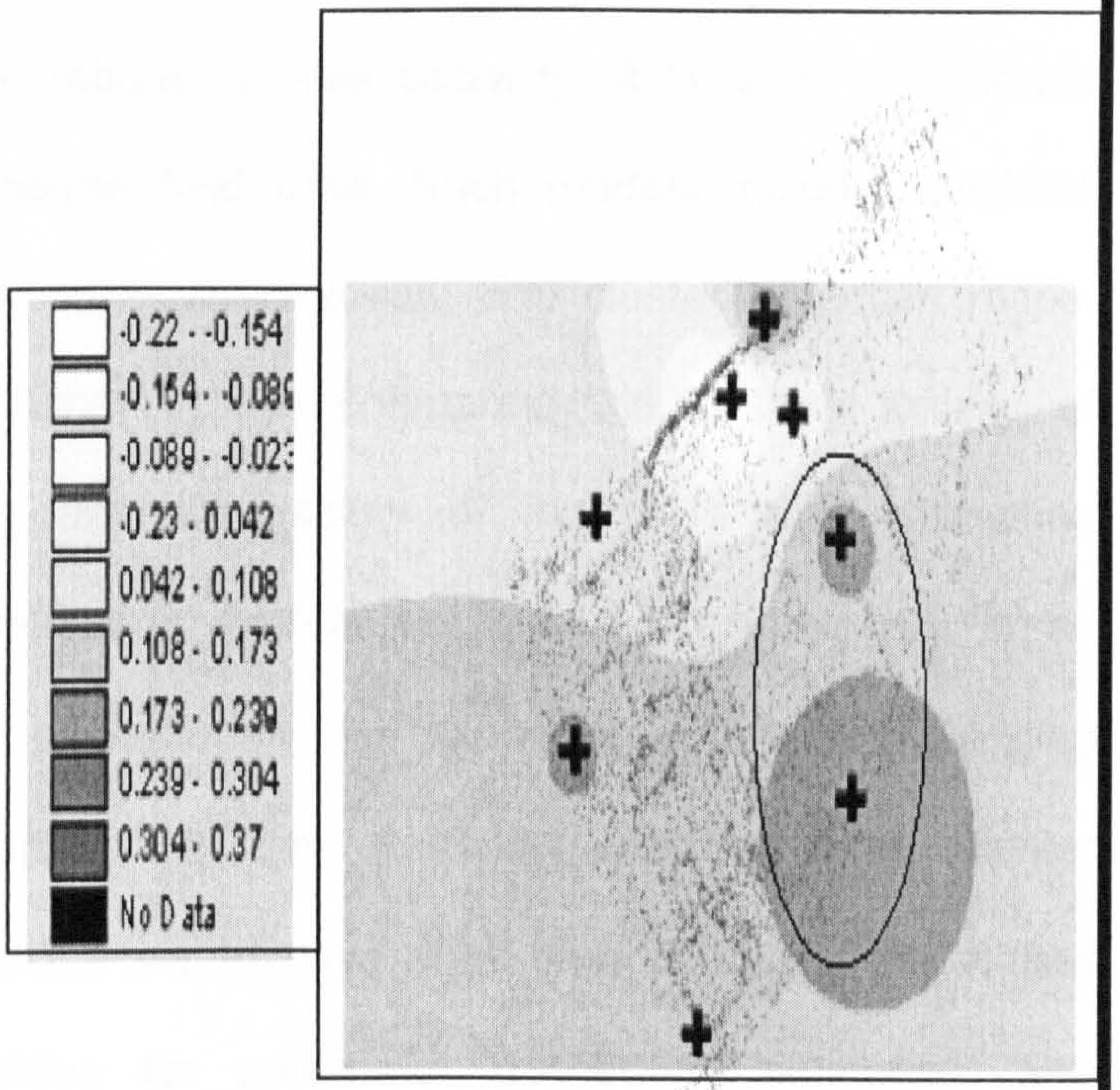
= unlined part of landfill

=landfill edge

Figure 5.3 Modelled groundwater levels were exported into 'ArcView GIS v.3.2' to show areas of greatest difference (circled areas)

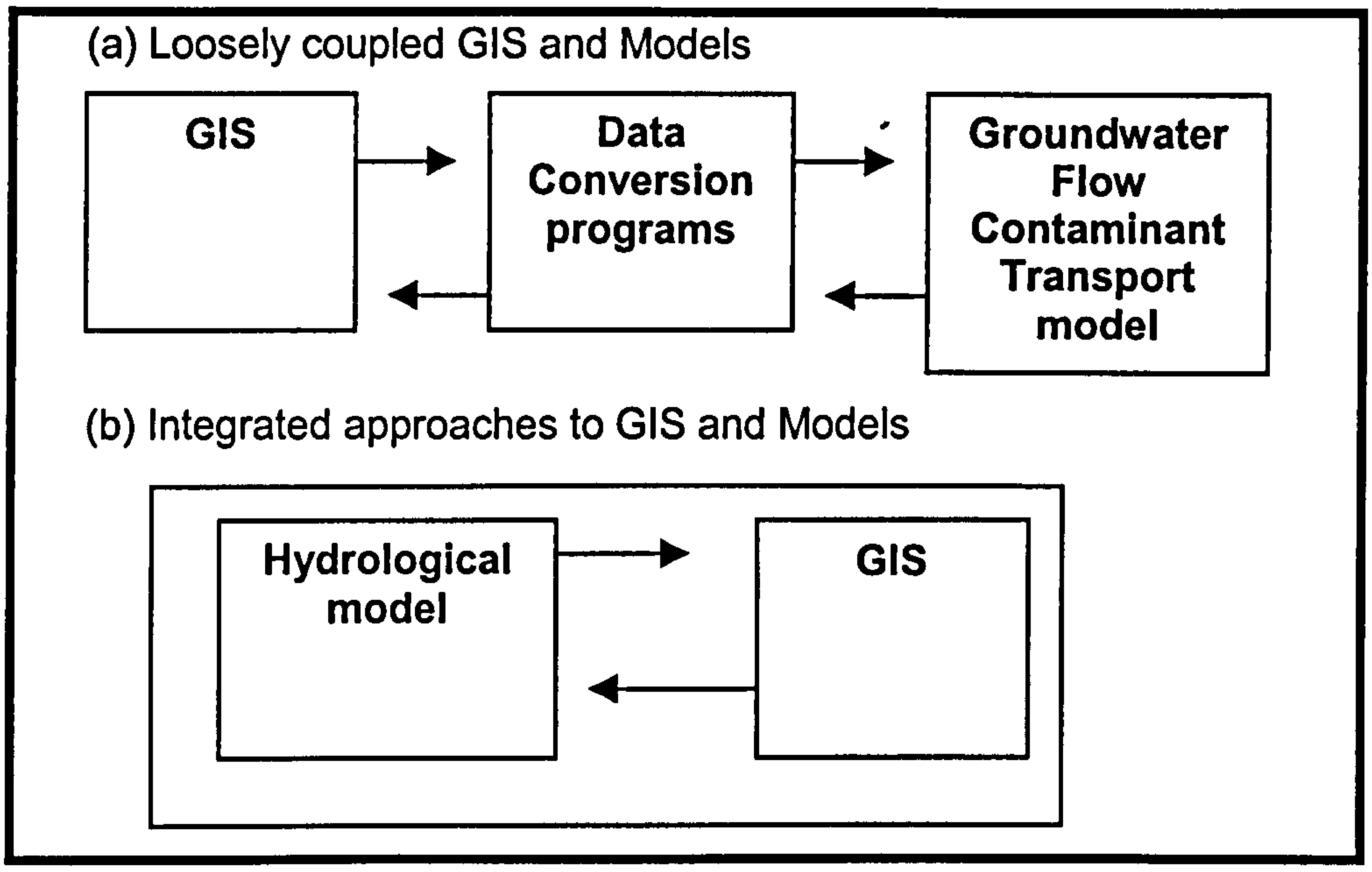


	Observed	Sc. 1	RES SC1
Bh 2a	1.23	1.02	0.21
Bh 4a	1.08	1.3	-0.22
Bh 5a	3.99	3.97	0.02
Bh 6a	4.76	4.56	0.2
Bh 13	2.46	2.29	0.17
Bh 8	4.46	4.28	0.18
Bh 10	1.87	1.79	0.08
Bh 12	1.68	1.44	0.24



RESIDUALS
= (OBSERVED – SCENARIO 1)

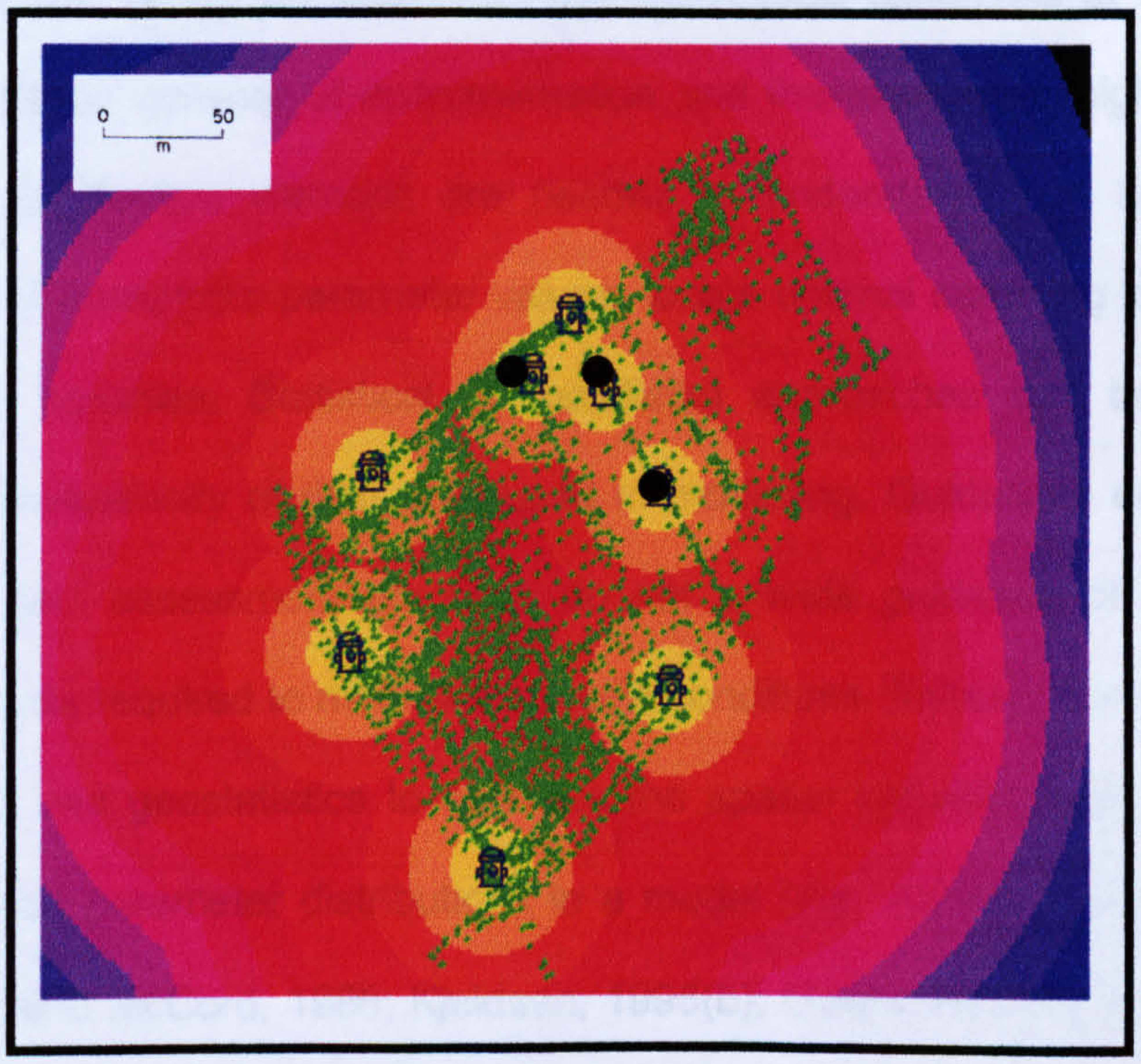
Figure 5.4 GIS and hydrological models links through (a) loosely coupled, and (b) integrated approaches (base on Maidment, 1993; McDonnell, 1996)

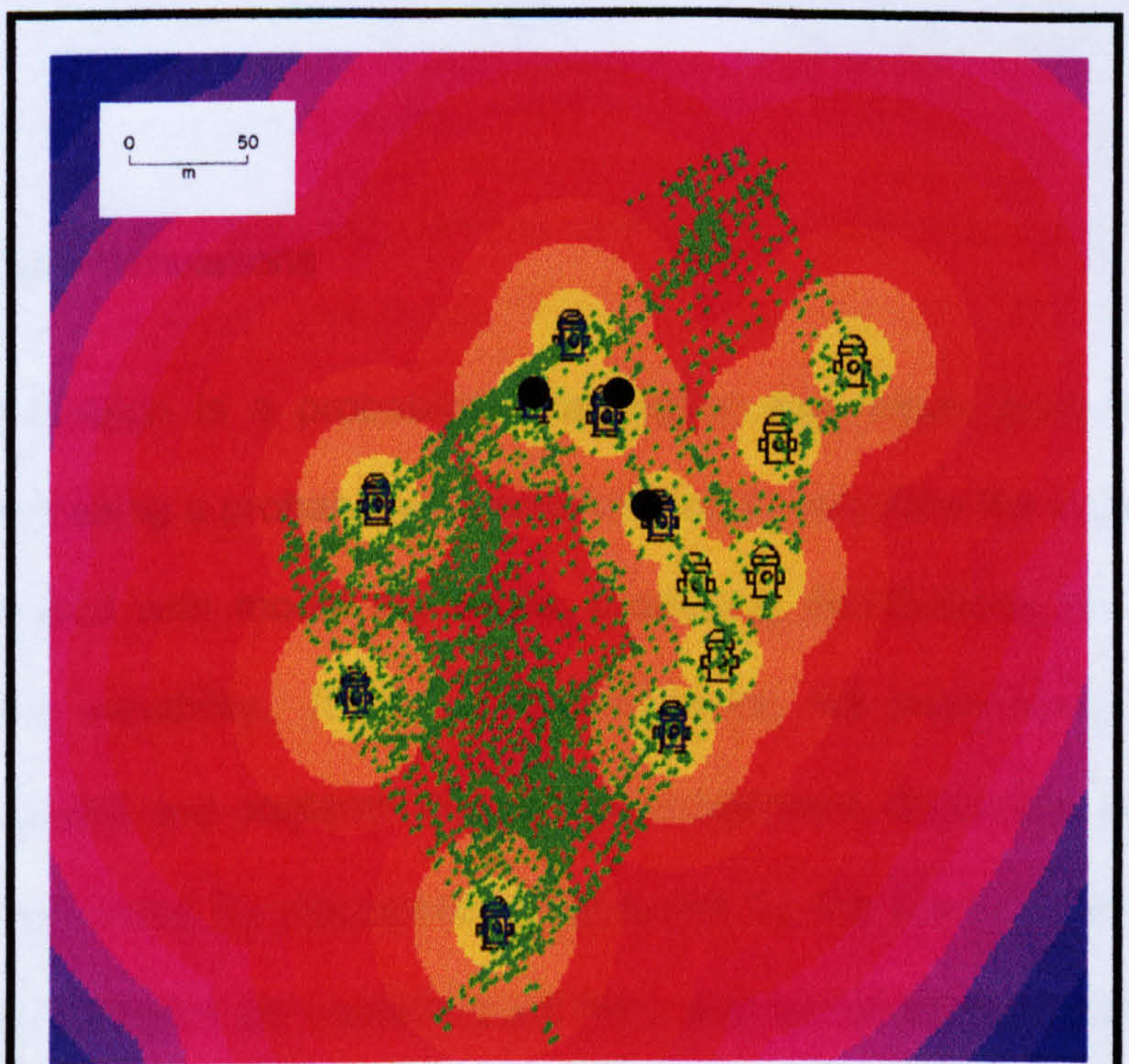


5.7.3 GIS and Contaminant Risk Modelling

Once the model has been constructed and calibrated, GIS can play a substantial role in assisting in the evaluation of the level of risk posed by existing contaminant sources, paths, or contaminant field concentrations. In this capacity, it is used to produce models that map and spatially analyse field data. Such models include weighted models, characteristic models, risk assessment maps, and cluster detection maps. Weighted models are noted in Fisher (1993) comparing the relative toxicity of chemicals in an area; identifying the vulnerability of indicators and estimating contaminant distributions. Characterisation models are discussed in Hooker *et al,* (2000). This type of model can characterise the exposure path acknowledging environmental variables that impact contaminant transport and risk level. Cluster detection maps, illustrated in Figure 5.5, are effective when detecting variability in field samples. Such models are effective for parameter distribution estimates, and comparative assessment of model scenarios (Fisher, 1993; Harris *et al,* 1993; and McDonnell, 1996).

Figure 5.5 Cluster maps produced from kriging analyses in which 8 (top) and 13 (bottom) groundwater sampling points were used to show areas of missing data





Legend:

- Circular contours represent 10 m distances away from each sampling point in which kriging analyses estimated that the first contour zone around each sample point could represent groundwater levels at the closest sampling point
- ● = reference point, representing calibration points 5, 6 and 13

5.8 Geostatistics

Sampling strategies, aimed at assessing hydro-geological conditions at a landfill site require a very dense network of instrumentation and therefore have high operational costs. More cost-effective methods are needed to understand field heterogeneity, provide ranges of acceptable parameter values, locate optimal sampling locations, and identify sample densities. Statistical (probabilistic) approaches can be applied to evaluate the heterogeneity of field data used in modelling. Guidelines outlining good modelling and field assessment practices do not provide guidance on defining the density of field data required to meet site-specific conditions. Instead, they promote the use of statistics and geostatistics to estimate the spatial variability between existing sample points and parameter distributions in a model (e.g. ASTM, 1993(b-c); Beven, 1993; Goodrich and McCord, 1995; Kjeldsen, 1998(b); Golder Associates, 2000). Two approaches that will be discussed in this section are the Monte Carlo technique and geostatistics.

5.8.1 Monte Carlo Simulations

Monte Carlo simulation is a probabilistic estimation that takes point-values of key model parameters using a probability density function to calculate the uncertainties and variability associated with model outcomes, producing distributions of risk reflecting uncertainty and/or variability. It has been used in the risk assessment models with applications to landfill and contaminated land management. Examples of models that are highly recommended for risk-based assessment by the Environment Agency are the CLEA model, ConSim for contaminated land and LandSim for landfill application (Environment Agency, *cited* 2003(c)). Monte Carlo techniques are wide spread, especially with in hydrological field and modelling applications (e.g. Goodrich and McCord, 1995 and Golder Associates, 2000). Monte Carlo simulations are used to randomly select pre-defined ranges of possible input values for hydrogeological parameters, landfill liner specifications and leachate quality in LandSim. The results give a range of output values. The distribution of these values reflects the uncertainty inherent in the input values (Golder Associated, *cited 2002).*

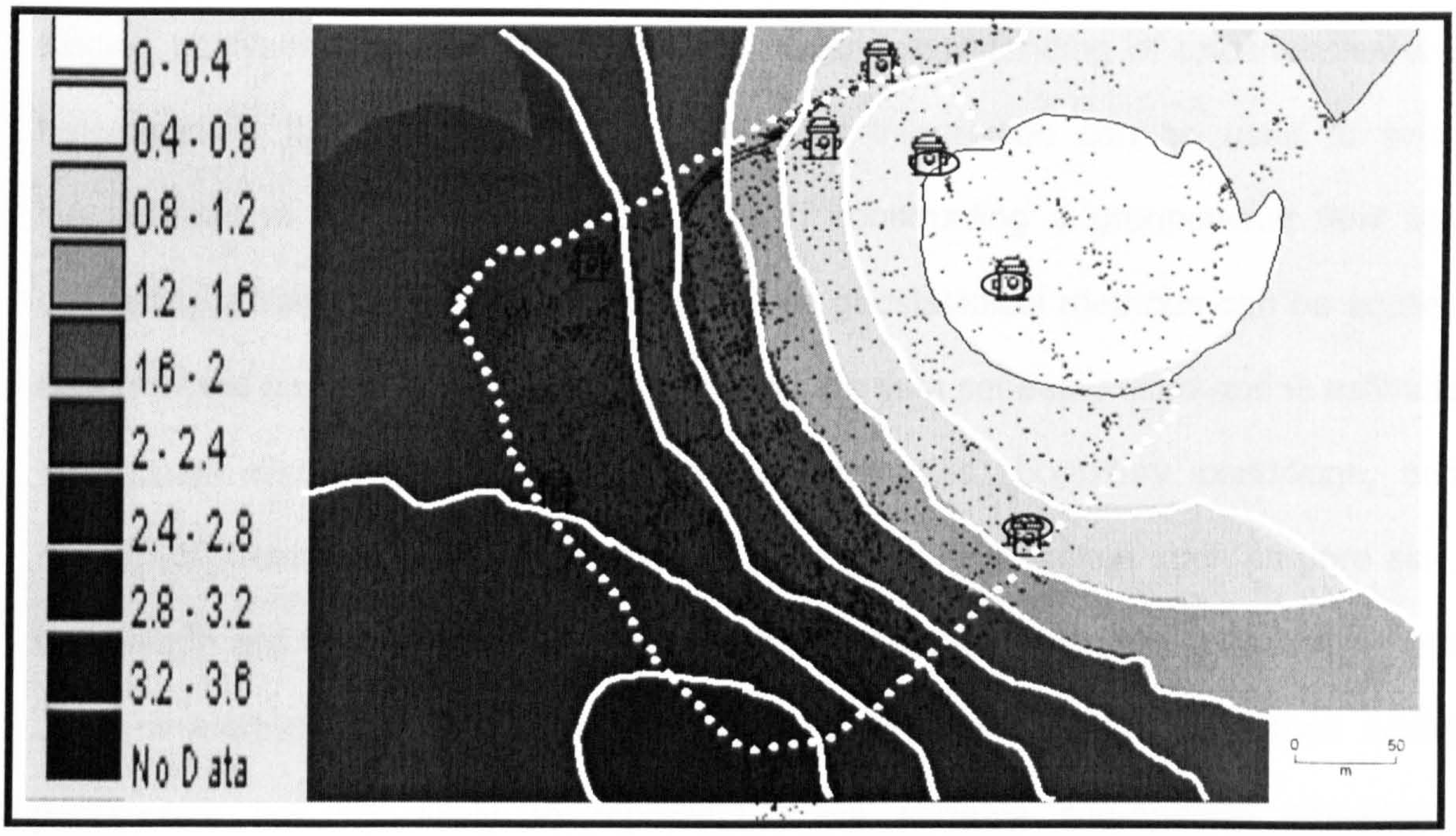
5.8.2 Geostatistics and Site Assessment

Geostatistics is a reliable approach to identifying spatially distributed sampling locations. It is widely applied in environmental sciences because many environmental data sets are based on information that has been sampled over larger areas, often leaving large unsampled locations. In relation to the risk assessment of landfill sites, geostatistical methods are used to evaluate field data in order to provide a better conceptual understanding of field data and site conditions to:

- Better understand uncertainties of field samples collected during the site assessment;
- Evaluate the heterogeneity of field characteristics;
- Identify optimal sampling strategies (sample locations and sample densities);
- Validate existing sampling strategies.

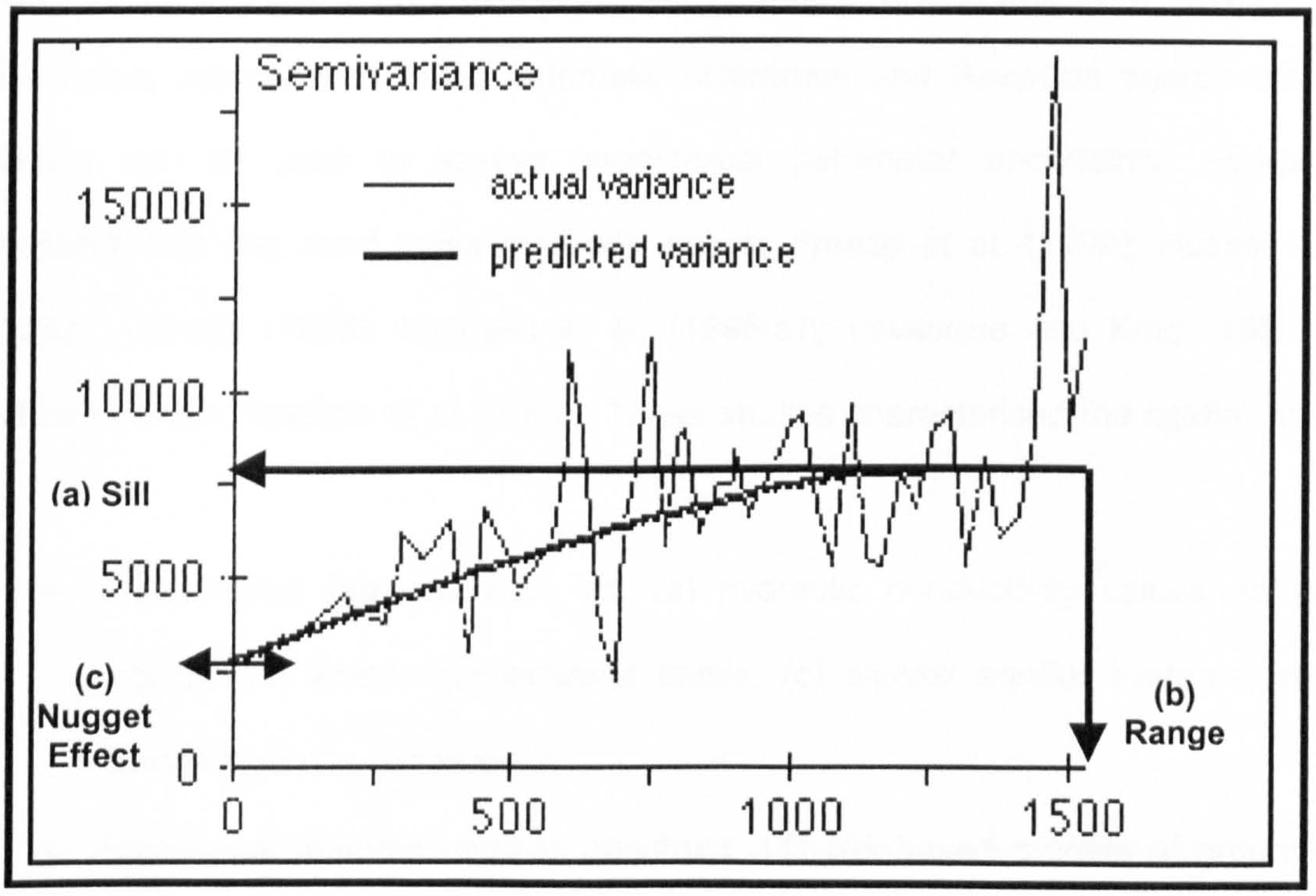
Kriging is one of the more popularly used geostatistical approaches (e.g. Myers, 1993; Melching, 1995; Sorooshian and Gupta, 1995; and Donald and McBean, 1997). In recent years, technological advances have integrated kriging models into GIS software packages, making integrated GIS data and modelling evaluations practical and easy to use. Results are presented as variance histograms, kriged variance maps (e.g. Figure 5.6) and as a variogram. The variogram plots semi variances over distances and can be used to interpolate the scale, and pattern of correlation between sampling points in a data set. The main properties of the variogram are (a) the sill; (b) the range; and (c) the nugget affect (Figure 5.7). The sill is the height of the variogram and is related to the extent of the area covered by the sampled points. An interesting feature is the range, which is the distance to the sill and is related to the maximum distance between sampling locations. The nugget effect is the positive y-intercept on the variogram. It indicates any distinct jumps in sample variance but it is also an indicator of sampling error. The kriging variance provides a measure of prediction error (David, 1977, p.4).

Figure 5.6 Spherical kriging of groundwater levels using landfill borehole data



Legend:
Contours (grey to black) represent groundwater levels in m AOD varying from 3.6: 0 m AOD
⬭ = reference point, representing calibration points 5, 6 and 13
.... = landfill edge

Figure 5.7 A variogram showing: the sill, range and the nugget effect



5.8.3 Geostatistics and Modelling

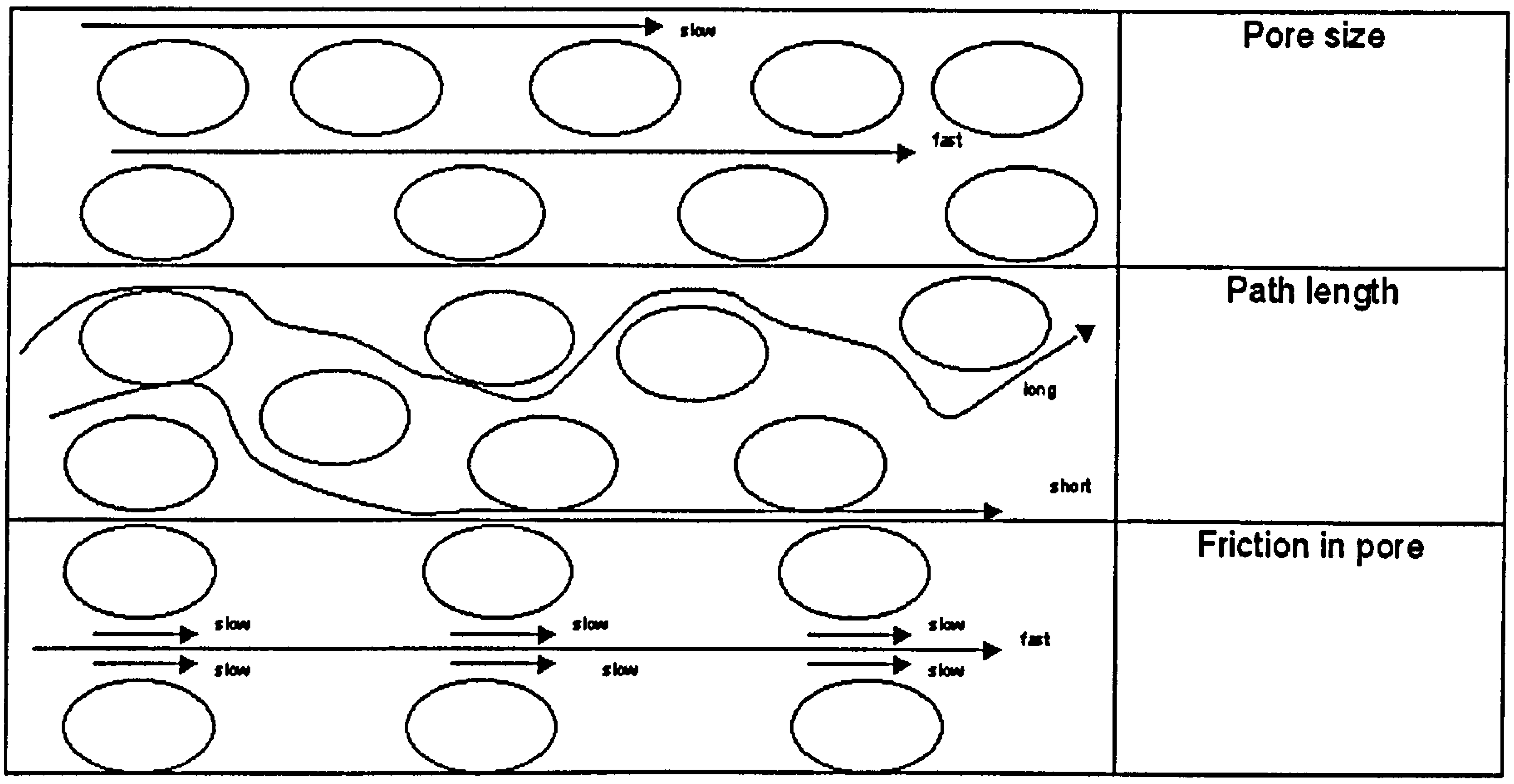
When integrated with GIS and hydrological models, kriging is a useful tool with which to identify the lack of coexistence in environmental data sets, locating gaps in field data. Kriging analyses can give the modeller a better understanding of uncertainties and heterogeneity that may exist in the field. This information can be used to avoid inaccuracies in the conceptual model. When constructing a groundwater flow and contaminant transport model, kriging and other geostatistical methods can be applied during model construction to better understand the data set being used and to estimate the spatial distribution of model hydraulic properties, boundary conditions, and contaminant concentrations in the model. Since geological factors such as pore size, path length and friction in pore spaces (shown in Figure 5.8) influence groundwater and contaminant transport, point measurements and regional estimations of such model parameters are not likely to represent these variable site-specific conditions. Understanding site conditions and the variability in the data set being used for model construction is one way to avoid inaccurate site assumptions. Several studies that have demonstrated the impact of data-derived hydraulic parameters on model results include R. L. Stollar in Fetter (1999), p.75; ASTM, (1993(a)); and Zheng *et al*, (2000). As an alternative approach to using automatic calibration and Bayesian search methods, kriging can be used to assess geophysical parameter uncertainty. Hydrological research that has used these methods include Freeze *et al*, (1990); Hosseini *et al*, (1993); Corona (1998); Kjeldsen *et al*, (1998(a)); Levesque and King (1999); and Ribiero (1999); Theiken *et al*, (1999). These studies characterised the spatial variation in:

- Hydrological features such as: (a) hydraulic conductivity values across an aquifer, (b) karstic groundwater levels, (c) alluvial aquifer systems, and (d) landfill leachate composition.
- Geological features used to construct 3-D risk-based models of groundwater flow and contaminant transport.

- Vegetation cover, deriving stressed and contaminated forest canopies from healthy forest canopies using airborne multispectral remote sensing data at several pixel resolutions (0.25m, 0.5m and 1.0m).

The combined applications confirm the ability of kriging when evaluating the spatial characteristics of site conditions.

Figure 5.8 Geologic factors (e.g. pore size, path length, friction in pore spaces) can influence subsurface contaminant migration and groundwater flow (adapted from Fetter, 1999, p.52)



5.9 Summary

This chapter has reviewed the role of groundwater flow and contaminant transport models as tools that are used in the site assessment and risk assessment of landfill sites and other types of contaminated land. Models are valuable to interdisciplinary teams of professionals and stakeholders dealing with landfill assessment and remediation. As a tool of contaminated land risk assessment, models contain four main areas of uncertainty that need to be addressed in order to avoid poorly constructed models, and more importantly, inaccurate simulations and assumptions that can cause remedial measures to fail.

Four main types of uncertainties that have been discussed are: (1) uncertainties or errors in the modelling code, (2) inadequate field data available during model construction, (3) inaccurate conceptual models, and (4) good modelling practises and professional judgement made by the person building the model. All four are important to the risk assessment of landfill sites because these uncertainties are not always evident in the risk analysis or they are not communicated clearly in modelling or risk assessment reports.

In order to strengthen the main areas causing predictive failure and model uncertainty, several areas are addressed in the literature that has been reviewed in Chapters 4 and 5 in order to improve model accuracy. The literature findings can be summarised into six points that are a direct reflection of the research objectives outlined in Chapter 1:

(a) Collecting spatially distributed field data that form more accurate conceptual models of site conditions (Overall research aim);
(b) Using innovative assessment technologies to collect distributed field data that compliments and validates insitu data sets and adds conceptual certainty to the conceptual and mathematical models (Objective 1);
(c) Using geostatistical modelling to understand variability in existing data sets (Objective 2);
(d) Using GIS as a platform for integrating data sets, conceptualising site conditions, performing spatial analyses of these data sets, and producing geostatistical and risk assessment models (Overall research aim and Objective 1);
(e) Calibrating and conducting sensitivity analysis using manual and automatic calibration techniques (Objectives 3, and 4);
(f) Encouraging modellers to adhere to good modelling practices (Objectives 4, and 5).

In order to implement these initiatives, the focus of attention needs to be placed upon standardising professional behaviour during three phases: the site assessment, model construction and results-communication to stakeholders. Regulation (in the form of professional certification, specialised training courses, adhering to standards or guidelines) will not mean that correct remedial decisions will be made in every site-specific situation. It also does not guarantee more accurate model predictions. Instead, such regulation attaches greater certainty to the steps taken to conceptualise complex site-specific conditions. It assigns quality assurance standards to the environmental assessment of landfill sites and assigns higher levels of accuracy to professionals who assess, model and present risk assessments of landfill sites and waste dumps.

The research aim and objectives of this thesis also suggest taking an integrated 3-D approach to assessment and modelling of landfill sites. Such an approach provides greater confidence to model results, decreasing the level of inherent uncertainty in remedial decisions. Three new approaches to collecting and assessing site conditions have been presented in chapters 4 and 5: kriging, GPR and remote sensing methods. These mirror Objectives 1 and 2 in which new methods of site assessment and geostatistical modelling may provide more effective information about site conditions and sampling locations. This chapter has also focused upon the important role that field data and modelling practices have upon model construction and model results. These are reflected in Objectives 3, 4 and 5 that test the influence and sensitivity of: (a) field data collected during the site assessment, (b) parameters defined during model construction, inferred from modelling practises; and (c) site assumptions made during model construction and their influence on model results.

A great deal of responsibility therefore lies in the hands of the individuals who conduct the landfill site assessment, construct the risk estimation model, and present the model results to stakeholders. Objectives 1 and 2 of the research are focused upon assisting individuals who conduct the landfill site assessment. Objectives 3, 4 and 5 are focused on the modeller and their simulations in context of the landfill assessment. The five research objectives have been tested by conducting six investigations. The methods used in each of the research investigations and a description of each study site are in the preceding chapters. The overall project aim (reflected in the five objectives) is to increase the accuracy of data collected and models constructed and used for landfill risk assessments. Figure 5.9 lists and cross references the five research objectives with the topics addressed in the literature review and with the six project investigations that were undertaken. It is important to note that the above listed approaches (a-f) and objectives (1-5) do not guarantee the model's structural accuracy. Instead, they promote honesty and professional ethics when communicating with stakeholders and encourage detailed landfill site assessments.

Figure 5.9: Objectives 1-5 are cross referenced with the topics addressed in Chapters 2-5 and the research investigations found in Chapters 8 & 9.

Inv. 6				X	
Inv. 5				X	
Inv. 4		0	0 X	0	Chapter 5
Inv. 3	X 0				Chapter 4
Inv. 2	X				Chapter 3
Inv. 1	X	X			Chapter 2
	Objective 1	Objective 2	Objective 3	Objectives 4& 5	

Objectives 1-5 introduced in Chapter 1

Legend:
Inv. = Investigation
X-axis left side = Investigations 1- 6 found in chapters 8 & 9
X-axis right side = Literature review in Chapters 2 - 5
X = Objectives 1-5 which are reflected in Investigations 1-6 found in Chapters 8 & 9
0 = Chapters 2-5 which provide background literature for each of Objectives 1-5
Objective 1 = To test two relatively new multi-spatial field methods that have been tested in other field of science or on other types of contaminated sites; Objective 2 = To test whether geostatistical modelling could assist in defining site-specific sampling strategies; Objective 3: To test the influence of different field data sets when constructing and simulating groundwater flow and contaminant transport conditions in a 3-dimensional model, Objective 4 = To test the influence of modelling practises when constructing a model; Objective 5 = To test the influence of field conditions assumptions when constructing a 3-D model simulating groundwater flow and contaminant transport conditions.

CHAPTER 6: RESEARCH METHODS

6.1 Introduction

This chapter will review the methods that were implemented during this research project. This chapter will also discuss how the methods employed relate to the research objectives and how the overall methodology evolved during the project's lifespan as a response to the logistical and operational factors, as well as to the promising initial results. It will explain the methods that were used, justify their use and their advantages and review the limitations of their use.

6.2 Linking Research Objectives and Thesis Investigations

The focus of the research was to assess risk-estimation models in the context of site assessment outcomes. The research was carried out to reflect the five research objectives, involving a total of six investigations. The investigations comprised several interdisciplinary applications, e.g. geostatistics, geophysics, remote sensing, groundwater flow and contaminant transport modelling, which have been applied to improve the uncertainties inherent in the site assessment of contaminating landfill sites. The first three investigations demonstrate the use of innovative site assessment methods that compliment direct methods and provide detailed information about landfill conditions. Investigations 4,5 and 6 conducted three sensitivity analyses using 3-D groundwater flow and contaminant transport models to test the influence of data sets on (a) model results and (b) the inherent site assumptions made during model construction.

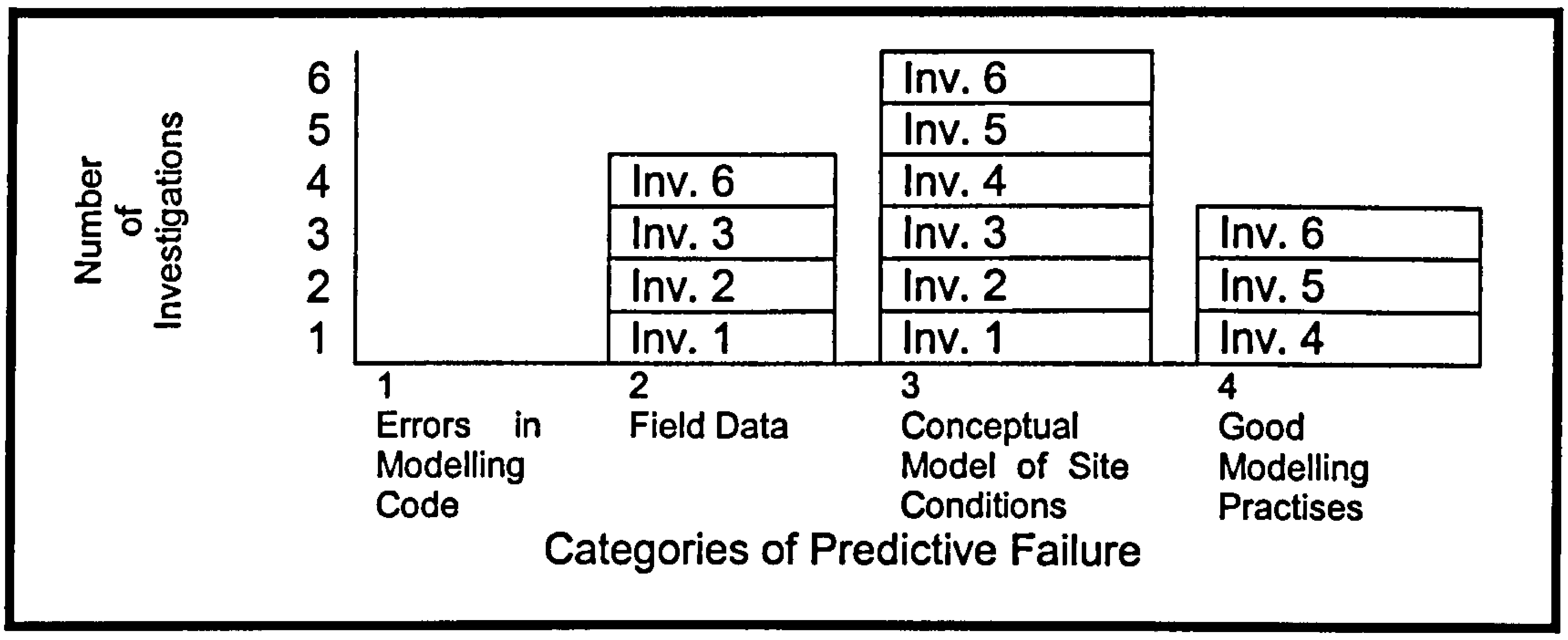
The five research objectives outlined in Chapter 1 were subject to change during the different stages of this project. During the early stages of this project, there were initially two research objectives (a) to test the influence of different field data sets when modelling landfill leachate migration and (b) to assess the potential of two novel geophysical remote assessment methodologies for assessing the location and effects of landfill leachate. After the initial literature review (which comprised of an extensive summary of modelling literature, UK waste management policies, landfilling practises and risks, site assessment guidelines and modelling standards as discussed in Chapters 2-5) it was apparent that the quality of data used for model construction had significant implications for the accuracy and validity of model assumptions and outputs. As a consequence the emphasis of the modelling aspects of the research focuses upon the influences of (a) field data limitations and errors (referred to in Chapter 1 as Objective 3), and (b) modelling assumptions on the construction of landfill leachate models (referred to in Chapter 1 as Objectives 4 and 5).

The second initial objective was also subject to a refocusing of research effort. The novel nature of the two geophysical remote assessment methodologies meant that a limited pilot study was the most appropriate approach. The logistical difficulties caused by having to deploy the necessary field equipment on a loan basis from two different NERC equipment pools greatly restricted the operational deployment of the equipment. In addition, the very high cost of acquiring a multi-date, multi-spatial resolution airborne remote sensing data set meant that it was only practical, within the temporal and logistical limitations of a PhD research project, to study one landfill site intensively.

A range of assessment, analysis and sampling methodologies were employed at the three different landfill sites. To emphasise the utility and limitations of these methodologies the site investigations have been organised to explore five objectives that reflect the accuracy of a landfill site assessment and leachate model simulations. The six investigations aim to improve the data collected during the site assessment, the conceptual understanding of site conditions at a landfill site and the accuracy of field assumptions made during model construction and calibration, e.g. Figure 6.1.

Investigation 1 used kriging within GIS to identify new groundwater sampling locations at landfill sites. Investigation 2 applied GPR to map subsurface hydrogeologic and contaminant features at landfill sites. Investigation 3 applied field-based and airborne remote sensing techniques to measure the spectral reflectance of stressed vegetation and soil, inferring and mapping locations of leachate-stressed vegetation along landfill edges. It gave insight into historical and spatially distributed landfill conditions. Investigation 4 focused upon the implications of additional field data on model results. Data collected during the site assessments at the three landfill sites were used to construct 3-D models of each landfill. Investigations 5 and 6 assumed that grid size and hydraulic conductivity values are modeller-controlled parameters and will reflect modelling outcomes, as they are model parameters inferred from site assessment data.

Figure 6.1: Investigations 1- 6 aimed at addressing three of the four common categories of modelling error (originally listed in Table 5.1)



Legend:
Inv. = Investigation

The nature of the results from the different investigations largely dictated the thesis structure. The summarised findings fall into two categories of research objectives, which aim to improve the accuracy of landfill site assessments, and of risk-estimation models, e.g. Table 6.1. The investigations also address some of the categories of predictive failure as well as common assumptions found in groundwater models. The methods that were used to conduct investigations 1 - 6 are listed in Table 6.2. Many were used in more than one investigation or at more then one study site. The study sites are described in detail in Chapter 7 and are labelled as 'SSA', 'SSB' and 'SSC' in this chapter, abbreviations for study site A, B, and C. In order to avoid repetition, the methods will be discussed in three sections: the methods applied in all the investigations; the methods applied in three investigations; and the methods that were applied in only one of the investigations. Table 6.3 lists the three sections and the methods described in each section in order to assist the reader in locating the described methods.

Table 6.1 Linking investigations 1- 6 to each research study site and then to the categories of predictive failure listed in Table 5.1

RESEARCH OBJECTIVE ADDRESSED	INVESTIGATION CONDUCTED	SSA	SSB	SSC	ASSUMPTION OR FIELD CONDITION IN TABLE 5.1	UNCERTAINTY ADDRESSED
2	<u>INVESTIGATION 1</u> Kriging model in GIS	X	X	--	E, H	FIELD DATA CONCEPTUAL MODEL
1	<u>INVESTIGATION 2</u> Ground Penetrating Radar	X	--	X	A, B, D, E, H	FIELD DATA CONCEPTUAL MODEL
1	<u>INVESTIGATION 3</u> Remote sensing methods measuring the spectral reflectance of stressed vegetation	X	--	--	C, D, H	FIELD DATA CONCEPTUAL MODEL
4 and 5	<u>INVESTIGATION 4 and 5</u> GWFCT modelling several scenarios of each study site using increasing scales of detailed information derived from the preliminary and detailed studies and from investigations 2 and 3	X	X	X	N/A	CONCEPTUAL MODEL GMP
3	<u>INVESTIGATION 5</u> GWFCT modelling testing the sensitivity of modeller-selected grid densities. These grid sizes were inferred from field data collected during the site assessment	X	X	X	N/A	CONCEPTUAL MODEL GMP
3	<u>INVESTIGATION 6</u> GWFCT modelling testing the sensitivity of modeller-selected hydraulic conductivity values. These values were inferred from field data collected during the site assessment	X	X	X	N/A	CONCEPTUAL MODEL GMP

Legend:
(A) = homogeneous layers, (B) = Isotropic transport in cells & zones, (C) = No dispersion or diffusion occurs other then what is specified as the source,
(D) = Constant contaminant and hydrogeologic flow properties, (E) = Constant hydraulic conductivity across model layers and zones, (F) = Sorption approximated by linear isotherm,
(G) = Unknown parameter values estimated from regional averages, past measurements, published values etc., (H) = Routine up scaling and down scaling of field data into model.
SSA = Study Site A, SSB = Study Site B, SSC = Study Site C
GWFCT = Groundwater Flow and Contaminant Transport model
Objective 1 = To test two relatively new multi-spatial field methods that have been tested in other field of science or on other types of contaminated sites;
Objective 2 = To test whether geostatistical modelling could assist in defining site-specific sampling strategies;
Objective 3 = To test the influence of field data when constructing a 3-D model simulating groundwater flow and contaminant transport conditions;
Objective 4: To test the influence of modelling practises when constructing a groundwater flow and contaminant transport model
Objective 5: To test the influence of data assumptions made during model construction.

Table 6.2 Matching investigations 1-6 to methods applied in each investigation

	Objectives 1 and 2 : New Innovative Methods for the Site Assessment			**Objectives 3, 4 and 5: Modelling Practices and Field Data During Model Construction**		
	OBJ. 2	OBJ. 1	OBJ. 1	OBJ. 3	OBJ. 4 & 5	OBJ. 4 & 5
	Inv. 1	Inv. 2	Inv. 3	Inv. 4	Inv. 5	Inv. 6
Study Site Used In Each Investigation	A, B	A, C	A	A,B,C	A,B,C	A,B,C
Methods Applied						
Preliminary Study • Desk study • Walk-over study • Aerial photo interpretation	X	X	X	X	X	X
Detailed Study • Soil quality • Groundwater quality • Geology • GIS Conceptual Model	X	X	X	X	X	X
Kriging	X					
Ground Penetrating Radar		X				
Remote sensing methods measuring the spectral reflectance of stressed vegetation			X			
GWFCT Model Construction				X	X	X
GWFCT Model Calibration and Validation				X	X	X
GWFCT Model Sensitivity Analyses				X	X	X

Legend:
A = Study Site A, B = Study Site B, C = Study Site C
Inv. = Investigation; OBJ. = Objective 1-5; GWFCT = Groundwater Flow and Contaminant Transport model
Objectives 1-5 = see Legend Figure 6.1

Table 6.3 Research methods organised into three sections

Section 6.3 Methods Applied In all the Investigations	**Section 6.4 Methods Applied In two or more Investigations**	**Section 6.5 Methods Applied In one Individual Investigation.**
PRELIMINARY STUDY • Desk study • Walk-over study • Aerial photo interpretation	GWFCT Model Construction	Kriging model in GIS
DETAILED STUDY • Soil quality • Groundwater quality • Geology	GWFCT Model Calibration and Validation Model	Ground Penetrating Radar
GIS Conceptual model	GWFCT Model Sensitivity Analyses	Remote sensing methods measuring the spectral reflectance of stressed vegetation

6.3 Methods Applied in All the Investigations

Three methods were applied in all investigations: the preliminary study, the detailed study and the GIS modelling to construct conceptual models and integrated databases of site conditions.

6.3.1 The Preliminary Study

The preliminary study was one of the initial steps taken to identify potential or existing hazards posed at the three landfill sites. The objectives of the study were as follow:

- to collect background data about each landfill site
- to evaluate historical groundwater and leachate quality records
- to evaluate the nature of contamination and its possible paths of migration
- to define whether a source-path receptor linkage could be established.

The study consisted of two parts – the desk study and the walk-over study. The desk study was an historical review of archived information such as maps, plans, local authority records, hydrogeologic, climate and contaminant data, consultant surveys, and aerial photographs. It provided valuable information with which to begin constructing conceptual GIS models of each landfill. The walk-over study verified and added to the findings of the desk study, evaluating aerial photographs of each site to infer levels of land changes and establish whether the site had previously been a local or regional source of contamination. Initial soil, surface water, groundwater, and leachate quality samples were also collected around each site, verifying them with historical records.

These initial samples were taken at assessment points that already existed at each landfill site, and at leaking areas at Site A and Site C (identified during the desk study). The preliminary study was conducted because it is part of the standard steps that need to be taken when assessing land conditions. The advantage of the study was that it provided a conceptual understanding of hydrogeologic conditions, landfill and leachate migration issues, which were needed to identify hazards. This information also assisted in constructing initial GIS models of leachate and groundwater fluxes. The disadvantages faced during the study were that historical records were often incomplete. GIS was used to address the issue of missing data at all three sites. Contour maps using available groundwater levels and leachate concentrations were constructed and compared with other site records to infer historical landfill behaviour and potential paths of leachate migration. Literature used as a guideline when conducting the preliminary study included Vincent (1994) which described methods of integrating historical hydrological data with other data collected; Barrett and Curtis ((1999) p.133-143) which outlined methods for deriving field data from aerial photographs; and ASTM (2000(a)) which provided detailed guidance on conducting a preliminary assessment.

6.3.2 The Detailed Study

A detailed study was conducted at each site following completion of the preliminary study. It had five objectives:

- to verify the hazards identified in the preliminary study
- to collect detailed information about soil, contaminant and hydrogeologic conditions at each landfill
- to begin constructing groundwater flow and contaminant transport models of each site in order to better understand leachate and groundwater fluctuations
- to establish whether the source-path and receptor were present.

Three methods were applied during this phase of assessment:

(a) measuring groundwater levels in boreholes

(b) sampling water quality in boreholes to infer leachate quality around each site

(c) sampling surface water near each landfill to infer landfill influences on water quality.

Collecting the borehole and surface water data was part of the standard landfill assessment program outlined by the landfilling license at each site. The existing assessment points at each landfill were used as sampling points. This was done to verify historical data sets with newer data sets collected during the detailed study. Additional groundwater, soil and vegetation samples were taken along landfill edges at Site A in order to further investigate ecosystem changes that were identified using aerial photographs from 1990–1999. Access to Site B was limited (due to the fact that it is located in Croatia) therefore the landfill's existing sample points were used. Borehole and surface water sampling was used to assess and monitor hydrogeologic conditions and contaminant behaviour. They are widely and highly recognised because they provide quantitative information about contaminant concentrations or hydrogeologic conditions at a specific place and time. The data derived using these methods validated the findings of the preliminary study providing evidence of a source-path and receptor linkage. However, such methods have two disadvantages. Firstly, they provide information that does not always represent the heterogeneous landfill conditions. Secondly, sampling errors can alter the sample results. Some examples of sampling errors that were detected during the detailed study were: (a) inaccurate field instruments used when measuring borehole groundwater levels; (b) inappropriate conditions when transporting leachate samples from the field to the laboratory in which the chemical concentration of contaminants was altered; (c) improper methods of borehole purging which altered the results being measured. These sampling errors were corrected wherever possible by repeat sampling. Literature used as a guide during the detailed study included Smith (1990) which was used to identify and classify marshland vegetation that was found growing along landfill edges; Department of Environment (1995) which provided detailed instructions for measuring and monitoring surface and groundwater around landfill sites; and ASTM (2000(b)) which provided detailed instructions for conducting a detailed study. The study was a valuable part of the risk assessment at each study site. Comprehensive information about landfill conditions was collected during this phase, verifying the initial conceptual model and providing information needed for the hazard assessment and risk estimation at each site.

6.3.3 GIS Databases and Modelling

GIS was used during the preliminary and detailed studies conducted at each study site for two reasons. Firstly, the tool has been widely used in other environmental management applications therefore there was an opportunity to explore its application at landfill sites. Secondly, it was able to integrate different scales of data collected during the preliminary and detailed study, constructing multi-layer maps of each landfill. Twelve types of field data were integrated into each landfill GIS: background and measured groundwater levels; leachate quality measurements; surface water quality measurements; landfill gas measurements; regional hydrogeologic and geological maps; landfill site maps; GPS points which created digital elevation models of each landfill; aerial photographs of each site; multi-spectral remote sensing data sets; regional land use maps; and sampling locations at each landfill. These integrated data sets, maps and models significantly improved the conceptual understanding of subsurface and regional conditions at each site. ArcView GIS v.3.2 was used, having two objectives:

- to build a GIS database for each study site in order to integrate different data sets
- to use GIS as a platform for conceptualising and calibrating groundwater flow and contaminant transport models.

It was used because it is popularly used in both academic and industry related applications. It was also available without charge in the Geography Department at the University of Hull, from where the research was based.

GIS have several advantages. It is a data management tool in which data sets were updated and re-modelled without difficulty, producing real-time maps and models of site conditions. This made it much easier to identify the source-pathway receptor elements at each study site. It also allowed for spatial and quantitative analysis during calibration and sensitivity analysis. This allowed the modeller to visualise changes in the model with each parameter change. Much of the work associated with setting up model parameters involved creating database files and assigning properties to the elements in the grid. The spatial analysis in GIS allowed for physical aquifer properties to be grouped, helping assign zones of constant parameter values in the models. During calibration water table levels and contaminant concentrations were simulated using borehole data to create contour maps to create maps of existing conditions. These were overlaid with MODFLOW-generated contour maps of water table levels. Used with caution, this was a helpful tool to spatially compare the measured and modelled groundwater patterns. The overall application of GIS in calibration and sensitivity analysis proved invaluable when testing conceptual assumptions about geophysical conditions within each model domain.

In the context of improving landfill site assessments, GIS also had its disadvantages. Extensive amounts of field data were needed to create accurate integrated maps and models of site conditions. Contour maps and spatial models of site conditions were initially misleading due to inadequate data sets in the database. This was addressed by adding all the available field data into each landfill database and interpreting GIS models with care. This inaccuracy leads to the second disadvantage when using GIS. As a tool, it is easy to use and can create misleading maps and models. The use and interpretation of GIS databases and models therefore require trained professionals. This is especially important in contaminant risk assessment applications in which misleading GIS models could lead to significantly negative consequences and remedial inaccuracies. Literature used as a guide for GIS-based modelling included: Fisher (1993) who described the steps in constructing 3D GIS models as part of hazardous waste site investigations; Goodchild *et al* (1993) who outlined the key areas of uncertainty that need to be addressed when building GIS models; Harris *et al* (1993) who described methods that can be used to integrate GIS models with 3-D finite element models when monitoring pollutant fluctuations; Ferrier and Wadge (1997) who used GIS and a knowledge-based system to assist in conducting geological analysis of basins; Theinken *et al* (1999) who presented several ways in which GIS could be integrated with hydrological models; and Hooker *et al* (2000) who provided detailed information about GIS applications in land management.

6.3.4 Use of GPS to Map Sampling Points and Topography

GPS was used to determine the geographical location of field samples collected at Site A. The geographic addresses for Sites B and C were obtained by digitising site maps. Data positions collected at Site A were marked and stored in the GPS until they were downloaded to disk later. Data was recorded in latitude and longitude format. The raw data was converted into ASCII format using a proprietary internal program developed for GPS analysis performed within the University of Hull Geography Department. This ASCII format data was then used spatially to address the data sets within the GIS. The advantage of this technology in view of the site assessment is that is provides geographic addresses to every sampling and topographic point in the field. This allows for GIS databases to integrate, map and model data sets. When compared to data sets derived from digitised maps, GPS is far more accurate, with position errors estimated to be less then one metre (Brasington *et al*, 2000). This improves the inherent accuracy of landfill GIS databases and models.

6.4 Methods Applied in Three of Six Investigations

This section reviews methods that were applied in three of the investigations (investigations 4, 5 and 6). They were used during the construction, calibration and sensitivity analysis of four groundwater flow and contaminant transport models using data from Sites A, B and C. A description of the four models and their parameter values can be found in Table 6.4(a and b).

6.4.1 Model Construction

Modelling of groundwater flow and contaminant transport was undertaken using data collected during the site assessment. There were three parts to model construction: (a) construction of the conceptual model; (b) selecting the appropriate modelling code; and (c) setting up model parameters to match site conditions.

(a) The Conceptual Model

Since innovative methods of field assessment were used in the construction of each conceptual model, the accuracy of this model was considered a focal point in translating field conditions into mathematical form. The conceptual models were constructed using the ASTM (1993(a-b)) standards, which outlined the steps for constructing a site-specific conceptual model. The advantages of using this method in context of the site assessment were twofold. Firstly, the documents adhere to good modelling practices and clearly outline the steps needed to build a conceptual hydrogeologic model. Instructions for model construction are easy to understand and implement. Secondly, the document is issued by a reputable organisation recognised for its quality assurance measures, ensuring that the standards are well tested and verified. The disadvantages are that model construction is an ongoing process in which calibration and sensitivity analyses continually question conceptual assumptions about groundwater and contaminant flow. Following the individual steps outlined in the standard is time consuming, a disadvantage for limited project budgets and timelines attached to modelling of landfill risks.

(b) Computer Code Selection

As stated, the objective of model construction was to test the sensitivity of data sets and modelling practices. As a result, Visual MODFLOW with MT3D MS modelling packages were used. The selection of this computer code was based on its popularity within industry in simulating groundwater flow and contaminant transport scenarios. It was also selected for its cost effectiveness in relation to the budget set for this research project. An additional and important factor was that the code was capable of simulating geophysical conditions found at each study site. The software was easy to use, with additional resources (training sessions, customer service information and published literature) to assist in model construction. The modelled results were also compatible with ArcView GIS, making modelled and measured groundwater comparisons an easy task providing an estimate of error for every simulation that converged. This was a useful tool for measuring the level of accuracy when comparing measured and modelled groundwater and ammonia concentrations for each study site. Despite these strengths, there were five disadvantages to using this software. Firstly, the software assumed saturated three-dimensional groundwater flow contaminant transport. The modeller therefore had to accept these modelling conditions and omit landfill areas where unsaturated flow was present. Secondly, the software offered advective dispersive contaminant transport conditions using a finite difference grid. The modelling therefore had to assume that leachate transport (represented as ammonia concentrations in the models) was based on advection and dispersion. Thirdly, the finite difference grid also influenced contaminant transport, in that MT3D calculated contaminant transport from the centre of one grid node to the centre of the next grid node. From a risk assessment perspective, model predictions may carry an error uncertainty if the grid size of a model is not carefully constructed to represent the area being modelled. The fourth limitation was that it is not exclusively intended for landfill applications. As such, heterogeneous landfill conditions were difficult to construct. For example, the software manual encourages modellers to group and zone hydrogeologic properties. This contradicts the heterogeneous nature of landfill sites. The last limitation is that the model consisted of over 300 input parameters. Not all of the parameters needed to be addressed during model construction; however, many were impossible to measure under field conditions, therefore requiring the modeller to assume values. In context of simulating site-specific landfill conditions, this is a major disadvantage because many of the model's parameters need to be assumed, adding unnecessary assumptions into the model domain. Despite these limitations, the Visual MODFLOW and MT3D proved to be robust tools for modelling landfill based groundwater and contaminant transport around the three study sites.

(c) Setting Up Model Parameters

In order to ensure accuracy during model construction, the methods from several ASTM standards were used (ASTM 1993(a-b); 1994(a-b)) providing information for site-specific groundwater flow model construction. These were used because of their good reputation among groundwater modellers. The advantage of using these methods is that they outline specific steps to ensure greater certainty and accuracy in model construction. These steps are as follows: (1) identifying representative model dimensions; (2) discretising the model domain into a representative mesh; (3) identifying accurate boundary conditions; (4) noting initial conditions; and (5) identifying hydraulic properties within the modelling code that represent field conditions. The disadvantage of these steps is that they do not guarantee model accuracy or conversion. This was experienced on several occasions during model construction, in which the guidelines were used to define hydraulic properties within each model; however, the model often behaved differently from the observed field conditions. This was a learning process in which the modeller had to understand the software being used, and adhere to the good modelling practices. Tables 6.4 (a and b) cite the parameter values that were used in each of the landfill site models. Of the listed parameters, Visual MODFLOW showed to be significantly sensitive to grid size, boundary conditions and rates of hydraulic conductivity. The values and ranges for these parameters were derived from field data collected during the site assessment. Increasing or decreasing these parameter values significantly altered model results during calibration and sensitivity analyses.

6.4.2 Model Calibration

The objective of calibration was to adjust hydraulic parameters, boundary, and initial conditions so that results resembled field conditions. Model parameter ranges were determined from field data and published literature. Both qualitative and quantitative calibration techniques were used. These included calculating residuals and identifying spatial and temporal correlation in data. By integrating qualitative and quantitative approaches, the modeller ensures measured and modelled conditions are similar and also verifies that hydrogeologic processes simulate field conditions. The ASTM (1996) standard outlined the steps to take during qualitative calibration of groundwater flow models. The advantage of the qualitative approach is that it compares changes in general flow features and distinct hydrogeologic conditions using a variety of scenarios. This assisted in verifying parameter values and boundary conditions that would have otherwise been overlooked if only quantitative calibration was used. The standard was useful in that it outlined industry-accepted uncertainties of non-uniqueness and human error, and explained four steps needed to minimise the impact of non-uniqueness in calibration. This guidance was very beneficial during model construction since the landfills being modelled had heterogeneous field conditions, whilst the Visual MODFLOW model had a large number of input parameters. The following four steps were taken to minimise non-uniqueness: (1) establishing calibration targets: all the models, due to their risk-estimation intention, needed to have a high to medium level of accuracy; (2) identifying parameters that needed calibration; (3) matching modelled groundwater levels with measured levels to verify whether the model was able to simulate historical conditions; and (4) producing different scenarios of calibrated models with multiple hydrogeologic conditions. This allowed for several conceptual and remedial scenarios to be tested when evaluating landfill risks.

Table 6.4(a): Parameters that were used in the construction of groundwater flow models using study sites A, B and C

Model Fundamental Components	**Site A Large-Scale Model Scenario 1**	**Site A Large-Scale Model Scenario 2**	**Site B Large-Scale Model Scenario 1 High Sava River Levels**	**Site B Large-Scale Model Scenario 2 High Sava River Levels**	**Site A Small-Scale Model Scenario 1**	**Site A Small-Scale Model Scenario 2**	**Site A Small-Scale Model Scenario 3**	**Site C Small-Scale Model Scenario 1**	**Site C Small-Scale Model Scenario 2**	**Site C Small-Scale Model Scenario 3**
Spatial and Temporal dimensions	3-D Steady State and Transient (489 Days)	3-D Steady State and Transient (489 Days)	3-D Steady State and Transient (18250 Days)	3-D Steady State and Transient (18250 Days)	3-D Steady State	3-D Steady State	3-D Steady State	3-D Steady State	3-D Steady State	3-D Steady State
Modelling Grid	10 x 10m grid 107 rows 23 columns 5 layers Calibration data: 11.98: 11.99	10 x 10m grid 107 rows x 96 columns 6 layers Calibration data: 11.98: 11.99	25 x 25m grid 76 rows x 85 columns 8 layers Calibration data: 01.98: 12.99	25 x 25m grid 76 rows x 85 columns 8 layers Calibration data: 01.98: 12.99	0.5 x 0.5m grid 49 rows x 40 columns 6 layers Calibration Data: 11.98: 11.99	0.5 x 0.5m grid 49 rows x 40 columns 5 layers Calibration Data: 11.98: 11.99	0.5 x 0.5m grid 49 rows x 40 columns 5 layers Calibration Data: 11.98: 11.99	0.5 x 0.5m grid 68 rows x 68 columns 5 layers Calibration Data: 11.97: 12.99	0.5 x 0.5m grid 68 rows x 68 columns 5 layers Calibration Data: 11.97: 12.99	0.5 x 0.5m grid 68 rows x 68 columns 5 layers Calibration Data: 11.97: 12.99
No Flow Boundaries	Layer 1	Layer 2	Layer 1	Layer 1	N/A	N/A	N/A	N/A	N/A	N/A
Boundary Conditions	General Head Boundary = 5.6:1.31 m Conductance in m/d = 3:1 (different values depending on site location)	General Head Boundary = 10:1.31 m Conductance in m/d = 5:1 (different values depending on site location)	General Head Boundary = 102 m Conductance in m/d = 2.44 River Bottom Elevation 102 Conductance m/d = 0.36	General Head Boundary = 104.5 m Conductance in m/d = 2.5 River Bottom Elevation 102 Conductance m/d = 0.5	General Head Boundary = 2.45 m Conductance in m/d = 2.40	General Head Boundary = 2.45 m Conductance in m/d = 2.40	General Head Boundary = 2.45 m Conductance in m/d = 2.40	Constant Head = 12 m	Constant Head = 12 m	Constant Head = 12 m

Table 6.4(b): MT3D parameter values that were used to construct contaminant transport simulations in Visual MODFLOW using data from study sites A, B and C

MT3D 1.5 Parameters	Site A Large-Scale Model Scenarios 1 and 2	Site B Large-Scale Model Scenarios 1 and 2	Site A Small-Scale Model Scenarios 1 - 3	Site C Small-Scale Model Scenario 1 - 3
Initial Concentration for L. Dispersion	10 mg/L	75 mg/L = L1, L2, L3	10 mg/L	10 mg/L
Dispersion Per Layer has 3 values: Longitudinal dispersion Transverse dispersion Diffusion Coefficient (m/d)	L1 = 20, 0.2, 1 L2 = 10, 0.1, 0.5 L3 = 10, 0.1, 0.5 L4 = 10, 0.1, 0.5 L5 = 5, 0.05, 0.15	L1 = 0.01 L2 = 0.1 L3 = 0.01 L4 = 0.1 L5 = 0.01 L6 = 0.1	L1 = 10, 0.1, 0.1 L2 = 20, 0.2, 1 L3 = 10, 0.1, 0.5 L4 = 10, 0.1, 0.5 L5 = 10, 0.05, 0.15	L1 – L5 = 20, 0.2, 0.02
Constant Concentration	30 mg/L for unlined landfill parts 100 mg/L for lined landfill parts	30 mg/L for unlined landfill parts for L1, L2, L3, L4 100 mg/L for lined landfill parts for L1, L2, L3, L4	30 mg/L	300 mg/L
Recharge Concentration	5 mg/L at unlined high flux Bh 13 area	N/A	Method of Characterisation (MOC) 1st Order Euler algorithm	Method of Characterisation (MOC) 1st Order Euler algorithm
MT3D Calculation Parameters	Method of Characterisation (MOC) 1st Order Euler algorithm	Method of Characterisation (MOC) 1st Order Euler algorithm	365 days	5000 days
Simulation Time	365 days	18250 days	GCG Solver Preconditions = Jacobi Dispersion Tensor = Lump	GCG Solver Preconditions = Jacobi Dispersion Tensor = Lump
Number of Calibration Boreholes	8	10	5	4

Legend:
L1 - L6 = model layer 1 – model layer 6
Bh = borehole
Dispersion / Layer = 0.2, longitudinal / Layer = 20, Transverse Dispersion = 0.2, Diffusion Coefficient = 1 (m/d)

Visual MODFLOW automatically calculates residuals, and graphs measured and modelled groundwater levels, producing an estimate of root mean square error (RMSE). This was a helpful guide during the tedious process of manual calibration. However, this should not be the only tool used to measure the effectiveness of calibration because this only uses calibration points allocated by the modeller. Given the fact that modelling is non-unique, other calibration techniques also had to be used. The PEST non-linear parameter estimation model was used as a tool of automatic calibration. It was more successful with individual sensitive hydraulic parameters than with specified groups of hydraulic parameters. The tool provided reliable parameter ranges that were used to evaluate model reactions. However, users should attend training sessions prior to using the model because it is important for the user to understand how the model works, how the numerous parameters interact and how easy it is to produce incorrect results. A large amount of time was lost due to lack of training, which resulted in poorly estimated parameter ranges. The manual calibration process was ongoing and lasted up to several weeks for each model scenario. It was the easiest way for the modeller to understand better the modelling software being used and the site being modelled. The downside is that it consumed a lot of time, and required a lot of repetitive and often frustrating estimations. When models are used in a landfill risk-assessment context, an integrated calibration approach provides the most effective results. ASTM standards provided structure and guidance in this stage. Using a combination of calibration tools (including manual and automatic, qualitative and quantitative) is the most effective option of addressing both non-uniqueness and model uncertainty.

6.4.3 Model Sensitivity Analysis

A sensitivity analysis was conducted to analyse changes in input variables and their effect on model output. This analysis was used on two occasions during the research. Firstly, during model construction in order to better understand parameter behaviour in site models. Secondly, during investigations 4, 5 and 6 in which grid size, values of hydraulic conductivity and different data sets were tested. The steps and analysis used in sensitivity analysis were taken from two published guidelines: CAMASE (2000) and ASTM (1994). Both documents provided guidelines for analysing the sensitivity of parameters and outlined the steps that need to be taken when calibrating groundwater and contaminant transport models. The disadvantage of these guidelines is that they are written about groundwater and pesticide leaching models. It would have been beneficial to have a modelling guideline that gave insight into landfill processes or parameters. Such guidelines were not available at the time of assessment.

In the first sensitivity analysis (during model construction) three types of sensitivity analysis were conducted for each study site model: (a) one-at-a-time; (b) local; and (c) factorial. The first analysed the response to one parameter change at a time, while keeping the other parameters unchanged. The second analysis looked for local reactions to model responses. Both were valuable methods during the initial phases of construction as they assisted in better understanding model behaviour. The third analysis, factorial analysis, was effective for investigating parameter interaction.

During investigations 4, 5 and 6 various combinations of the three analyses were applied. Investigation 4, which evaluated changes in grid size, focused on one-at-a time analysis, in which grid sizes were changed, keeping other parameters equal and unchanged. This was effective because it showed how each model responded to these changes. Investigation 5 (testing conductivity values) used factorial analysis. Conductivity value ranges were defined in which the lowest and highest values were analysed. It was an effective method because is showed how the different parameters interacted. Investigation 6 used a combined approach, one-at-a-time and local. The additional data sets had to be evaluated for their effect on small-scale changes and other parameters to infer model behaviour.

6.4.4 Verification and Validation

Verification of each site model was conducted to ensure that the models simulated historical field conditions. There were four steps: (1) the software code was checked for errors; (2) distributions of parameter values were reviewed; (3) modelled and measured data sets were compared; and (4) the RMSE was verified, ensuring that it was below five percent. Literature that provided guidance during this phase of model construction was Sorooshian and Gupta (1995) and Golder Associates (2000). The former discussed the role of verification and calibration when building a model while the latter provided instructions for groundwater flow model validation.

Validation requires an entire calibrated and verified model domain to be tested against other measured data sets. It is an important step in risk-based modelling because it evaluates process parameters and the accuracy of simulations. It is the main tool with which to prove that a risk-based model can simulate site-specific conditions with some confidence. It is a difficult step because many modelling codes are not capable of dealing with environmental factors or cyclic conditions within a single scenario. It is common practice to take a data set, e.g. groundwater levels measured monthly from January to December, and split it into two. The first part of the data set, e.g. January to June, is then used to calibrate the model and the second part, e.g. July to December, is used to validate the model results. This method was used to validate the Site A and Site B models because both had more then 18 months of groundwater data. The advantage of taking such an approach was that validation could be conducted, despite the fact that data were limited. It is interesting to note that modellers are encouraged to conduct some form of validation to infer a level of certainty in their results. However, validated models are not essential when presenting model results to stakeholders and decision-makers, as was noted in literature such as Visual MODFLOW (1999); ASTM (1993(a-b)); (1994a-c); (1995); (1996). Validation of the smaller-scale models simulating problem areas at Sites A and C was not carried out because there was not enough field data. Instead the models were only calibrated to match measured field conditions. In the case of the Site A model, there was only one borehole that could have been sampled at higher frequencies. At Site C there were four possible boreholes that could have been sampled at higher frequencies. Data for validation should have been collected at these points after every rain event. This was not done for two reasons. Firstly, the need for such detailed data was only recognised after the model had been constructed. It was too late in the project's timeline to begin measuring because other parts of the research had to be addressed. Secondly, such measurements would have required additional instrumentation and landfill staff support, which was not available at that period in the research.

6.5 Methods Applied to Single Investigations

This section will discuss the methods that were applied in only one of the six investigations. They relate to the kriging models applied in investigation 1, the GPR survey used in investigation 2, and the remote sensing instruments measuring the red edge position of stressed vegetation, used in investigation 3.

6.5.1 Kriging

The principles used in kriging are of a geo-statistical nature in which data sets can be evaluated to determine the level of heterogeneity in field conditions or the level of uncertainty in a data set (discussed in detail in section 5.8). In context of a landfill site assessment, kriging can provide insight into the distribution, fluctuation, and flow of groundwater and leachate. It can also provide insight into the level of accuracy in the distribution of field samples.

Kriging was used in investigation 1 for two reasons. Firstly, because previous studies showed that it could be applied in landfill sites, e.g. Kjeldsen (1998(a)), groundwater investigations, e.g. Ribiero (1999), and as a cost-effective tool to identify new locations for ore exploration drilling, e.g. David (1977, p.89). However, previous studies have not evaluated its ability to improve locations and distribution of existing and new data sampled during risk assessment. Secondly, kriging was applied at Sites A and B as these sites had appropriate available field data. At Site A, eight boreholes were already present, with bi-monthly groundwater level measurements. At Site B, 12 piezometers were also already present, with daily groundwater level measurements recorded over a one-year period. A disadvantage that was faced when modelling was that more sample points at both sites would have increased the accuracy of the analyses.

There were several advantages to using kriging analysis, in which the software was an extension within ArcView GIS. Firstly, the data needed for kriging was easily imported or already part of the GIS databases. Secondly, the analysis produced maps of kriged groundwater levels that could immediately be compared with other GIS models and data sets. Kriging results could be related back to risk assessment objectives at each site. The disadvantage was that the software code developer did not provide an explanation of the model's parameters. As a result, modelling of each study site took several weeks and required extensive calibration and sensitivity analysis to better understand the software's capabilities.

6.5.2 Ground Penetrating Radar

GPR maps subsurface features using electromagnetic responses. It is capable of detecting groundwater levels, mineral deposits, geological features, and buried features. It was selected because previous studies demonstrated that the instrument could identify hydrogeological features, and delineate buried features and plumes, e.g. Davis and Annan (1989), Forde (1996), Trenholme and Bentley (1998). There was an opportunity for a new application at landfill sites, in which GPR data would be used to investigate site conditions and aim to calibrate and improve contaminant transport models. The GPR was also used because it was available from the NERC equipment pool.

Investigation 2 used GPR to map near-surface landfill conditions. Previous studies successfully used this instrument to map groundwater levels, contaminant plumes and geological strata. In context of the landfill site assessment, the investigation objective was to evaluate whether this type of radar could provide large scales of field information for the site assessment and risk-based modelling. Literature that outlines GPR best practice suggests that the most effective GPR results are obtained when the survey has a clear objective and when background information is known about the surveyed area (Annan, 1992). Survey locations at Sites A and C were identified based on these two factors. The survey objectives (at both sites) were to verify whether leachate migration, groundwater levels, and geological layers could be identified and mapped along leaking edges at both sites. At the Site A model, landfill cap thickness and waste depth were also part of the survey objective. At Site C, the survey also aimed to determine whether the containment cut-off wall was leaking. The PulseEKKO 100 and 1000 GPR systems from Sensors and Software Inc. were borrowed from the NERC equipment pool (University of Edinburgh). The advantage of these brand name radars was their well-known reputation and the customer service information that was of great assistance prior to data collection and during data processing stages.

Broad ranges of transmitters (antennas) were used in the investigations. Although this prolonged the time needed to set up and conduct each survey, it allowed for each problem area to be surveyed at greater depth. The investigations conducted in August 1999 and June 2000 used 50, 100, 200, 225, 450 and 900 MHz transmitter antennas using the fixed offset profile mode, where the transmitter and receiver antennas were separated by a fixed distance and moved across the area of interest in regular steps. The larger frequency antennas were effective at Site A while the shorter frequency antennas were effective at Site C. The difference was due to the different subsurface materials and conditions at each site. A technical limitation faced during surveying was that survey lines could only be 30m in length because the extension cord linking the radar to the power source was this length. It was addressed by planning 30m survey lines along the leaking landfill edges.

Input parameters that needed to be considered carefully before data collection and during data processing were: frequency; time window; time sampling interval; line location and spacing; antenna orientation; antenna separation (m); frequency (MHz); sampling interval; number of stacks; points and step size (m). Table 6.5 describes the parameters that were used for each transmitter antenna. Other parameters that were determined based on the site-specific conditions at each survey line were the permittivity, electrical conductivity, velocity and attenuation. These parameters needed to reflect subsurface materials. This was difficult because values (listed in Table 6.6) have not been established for landfill conditions. In order to overcome this lack of data, soil and geological information about each survey line were used to derive parameter values. The calibration of unprocessed digital radar data was done using the EKKO 1000 EKKO_TOOLS and Slicer 3D software packages, which were highly recommended by experienced NERC staff. The EKKO 1000 package produced 2-D black and white subsurface images while the Slicer 3D produced 3-D full colour images. Both were easy to use and provided impressive results that were helpful when the survey results were presented to the landfill managers. The disadvantage of the Slicer 3D software was that it produced multi-colour images that were easy to misinterpret. Therefore most of the subsurface cross-sections presented in this thesis used the EKKO 1000 program. Once analysed, the GPR data was interpreted in four ways: (1) Comparing results of similar GPR studies, e.g. Peretti *et al* (1999), Sauck *et al* (1998) and Trenholm and Bentley (1998). (2) Comparing GPR data with geological records from nearby boreholes. This was an important process that validated depths shown in GPR cross-section images. At Site A, the results from a seismic refraction investigation were also used. Each survey line was relatively close to the recorded geological profiles. However, additional geological information would have helped to verify geological strata along the leaking edges. (3) Discussion of the results with landfill managers. This was helpful because they understood the subsurface and landfill conditions very well. (4) Review of the results with geologists at the University of Hull and with the director of Sensors and Software Inc. This integrated approach was useful because it allowed the results to be evaluated by qualified specialists.

Table 6.5: Maximum sampling intervals assigned for different antenna frequencies (Cited in Annan and Cosway, 1992)

Transmitter Antenna Frequency MHz	Maximum Sampling Interval (ns)
1000	0.17
500	0.33
200	0.83
100	1.67
50	3.3
25	8.3
10	16.7

Table 6.6 GPR parameter values that are commonly used for geologic materials (Cited in Annan and Cosway, 1992)

MATERIAL	Permittivity - K	Electrical Conductivity - 'mS/m	Velocity – v m/ns	Attenuation – dB/m
Air	1	0	.3	0
Distilled Water	80	.01	.033	2x10-3
Fresh Water	80	.5	.033	.1
Sea Water	80	3x10^3	.01	10^3
Dry Sand	3-5	.01	.15	.01
Saturated Sand	20-30	.1-1	.06	.03-.3
Limestone	4-8	.5-2	.12	.3-1
Shale	5-15	1-100	.09	1-100
Silts	5-30	1-100	.07	1-100
Clay	5-40	2-1000	.06	1-300
Granite	4-6	.01-1	.13	.01-1
Dry Salt	5-6	.01-1	.13	.01-1
Ice	3-4	.01	.16	.01

6.5.3 Remote Sensing

Remote sensing instruments that measure the spectral reflectance of earth surfaces have been used to estimate annual crop growth, monitor ecosystem changes and calculate rates of coastal erosion. Investigation 3 used airborne and field-based instruments to measure the spectral reflectance of stressed vegetation using the 'red edge position', measured between 690 and 740nm. Such methods have effectively mapped contaminated land that had homogeneous vegetation cover, e.g. Jago *et al*

(1999). The investigation evaluated whether these methods could be applied at landfill sites to provide distributed data that were needed for the risk-based landfill site assessment. Six types of data were collected and analysed in this investigation. They included:

1) assessment of contaminated soil and vegetation
2) assessment of chlorophyll concentration
3) remote sensing using field-based spectroradiometers to map vegetation stress
4) remote sensing using the CASI airborne sensor to map vegetation stress
5) use of GPS to locate and map sampling points and topography.

The objectives, advantages and disadvantages and sampling locations for each of these methods will be discussed further.

(a) Assessment of Contaminated Soil and Vegetation

Biochemical sampling of vegetation, soil and surface water was undertaken at Site A to assess whether a link could be made between vegetation stress, ecosystem changes and pathways of near-surface leachate migration. The objective of the biochemical sampling was to:

- identify leachate concentrations in areas adjacent to the landfill
- delineate the extent of leachate seepage off-site
- link possible plant growth patterns and ecosystem changes to off-site leachate seepage.

Samples of leachate, surface water, vegetation and soil were collected in January and April 2000, focusing upon two areas where leachate presence was evidenced. The samples were collected randomly:

- 11 leachate and surface water samples
- 51 foliar samples (digging out the entire plant – root to tip)
- 19 soil samples (collected at the same location as leachate and foliar samples).

The soil samples were taken with a small hand-auger at a depth of 30cm. The sampling plan and methods used were similar to those used by Strub *et al* (1998) and Jago *et al* (2000). The advantage of this integrated sampling was that it could verify whether there was a link between landfill leachate and contaminated soils and vegetation. The disadvantage was that it did not represent other parts of the site where leachate migration and ecosystem contamination might also have been present.

Heavy metals were used as leachate indicators because the region surrounding Site A had several industries and agricultural fields that could have been potential contributors to high nutrient concentrations in the sampled materials. Inductively coupled plasma mass spectroscopy was used to analyse samples. The method was effective because it was able to confirm the presence of heavy metals in soil, vegetation and leachate samples. This was important for the risk assessment in that it confirmed that a source (the landfill), pathway (soil and groundwater) and receptor (stressed vegetation) were present. A list of heavy metals that were identified in all three types of samples is listed in Table 6.7.

Table 6.7: Types of heavy metals that were identified in soil, vegetation, and leachate samples at Site A

Rb	Te	Er	Hg	Co	Ru	Se	Sb	V	Nb	Ce	Lu	Th	Zn	Cd	Tb	Nd	W	Sm	Re	Dy	Pt	Zr	Cr
Be	Cs	Tm	Tl	Ni	Pd	Eu	Os	Mn	Mo	Pr	Ta	U	As	Ti	Au	Cu	Ag	Gd	Ir	Se	Sb	Yb	Pb

An eco-toxicological study of the leaking landfill edge was also conducted, having three parts. The first part classified the types of vegetation (reeds, cattails, and grasses) that grew in the vicinity of the leachate leaking area. The second part analysed all data sets to create a map showing sensitivity to leachate. The third integrated this sensitivity map into the GIS. This integrated study provided a link between the landfill as a source of contamination, and the ecosystem surrounding the landfill, as a receptor. The study was beneficial in context of the site assessment because it linked biological conditions observed at the landfill, e.g. reeds, cattails and lush green grass that grew all year long, with leachate migration. The analysis showed that the nutrient-rich leachate seeping from Cells 10 and 5 had triggered the creation of leachate-induced marshlands at the edge of the landfill. Smith (1990) classified the different types of wetland vegetation was used. This was used as a guideline when classifying vegetation types growing along the edge of Site A.

(b) Assessment of Chlorophyll Concentration

The objective was to measure chlorophyll concentrations in vegetation around the leaking landfill in order to establish whether the vegetation showed signs of stress. Samples were taken at three locations: (a) near leaking areas of the landfill; (b) at areas north of the landfill; and (c) up to 500m from the landfill at locations where leachate presence was evident. Chlorophyll and its derivatives were extracted using acetone, which was an effective method because the spectrophotometric analysis differentiated healthy and stressed samples. Since the analysis was based on measuring the optical density of chlorophyll concentration, the results could also be compared with airborne and field-based spectral reflectance measurements. Foliar samples, field reflectance and airborne CASI images of the site were collected in the same week in April 2000. This combined data allowed for a pollutant linkage to be established by comparing optical densities of the three data sets. Literature that was used as a guideline during laboratory analysis included: Curran *et al* (1990) and Curran *et al* (1991) who discussed measurement errors and methods of calculating the red edge; as well as Jago *et al* (1999) who estimated canopy chlorophyll concentrations from field and airborne Spectra on agricultural and contaminated plots of land. All three publications were used to develop field sampling strategies and data analysis methods for data collection at Site A.

(c) Remote Sensing Using Field-based Spectroradiometers

A field-based spectroradiometer was used to measure vegetation stress around the landfill site utilising variations in the location and dimension of the red edge position of spectral reflectance. In context of the site assessment, the instrument was used to link leachate migration with vegetation stress, testing whether the spectroradiometers could be used as a field instrument for measuring vegetation stress around a landfill site. The instrument was also used as a form of ground-truthing, collecting surface reflectance, in support of the multi-spectral remotely sensed CASI data set. Ground spectra were measured at 10 different locations at Site A using the GER3700 and GER1500 Spectroradiometers on loan from the NERC equipment pool at the University of Southampton. Table 6.8 outlines the information about each of the spectroradiometers that were used. Spectroradiometric data was collected in April 1999, August 1999 and April 2000. This allowed the data sets to be compared for seasonal and annual trends. The methods outlined in Jago *et al* (2000) for CASI and ground spectra integration and interpretation were followed in order to avoid errors during data collection.

Spectra were initially processed using proprietary software provided by the NERC and were then analysed in Excel, specifically focusing upon the Normalised Difference Vegetation Index (NDVI) and Red Edge Position of spectral reflectance. The NDVI equation is the most widely used index for remote sensing of vegetation. In context of the site assessment (it uses radiance or reflectance from the red channel of the electromagnetic spectrum (660 nm) and a near-IR channel (860 nm)) NDVI was calculated for different spectral reflectance areas around the site. This data was then used to calibrate remotely sensed CASI airborne images (Strub *et al*, 1998; Bo-Cai Gao, 1998)

Equation 1: Normalised Difference Vegetation Index Equation

$$NDVI = (IR - R) / (IR + R)$$

Where:

	NDVI	= Normalised Difference Vegetation Index
	IR	= Infra Red
	R	= Red

Table 6.8: Information collected to infer levels of contaminated land (Based on GER, 1996 (a-b); NERC, *cited* 2000)

	GER 1500	GER 3700	CASI
Sample Date	April 2000 10-14 GMT	April 1999, August 1999 10-14 GMT	April 2000 10-14 GMT April 1999 10-14 GMT August 1999 10-14 GMT
Study Site	Site A	Site A	Site A
Spectral Range	300 - 1100nm	300 - 2500nm	400 - 1100nm
Number of Field Samples Collected	150	430	N/A
Channel	N/A	N/A	13, 48, 72 (depending on band configuration)
Sampling Locations at Site A	Top of Cell 1 Gypsum Lagoon Gypsum Lagoon Moss Leachate Wetland Leachate Pool	Paved Road Gypsum Lagoon Gypsum Lagoon Moss Leachate Wetland Leachate Pool	Site A

(d) Remote sensing Using CASI to Map Vegetation Stress

The Compact Airborne Spectrographic Imager (CASI) was used to collect high-resolution spectral producing CASI images. The data contain spatially explicit information on the absorption features associated with canopy biochemistry. This data was used to estimate the red edge-chlorophyll concentration relationship for vegetation canopies. A similar study was conducted by Jago *et al* (1999). The objective for Site A was to identify locations of stressed vegetation that could be linked with leachate migration and ecosystem changes to infer levels of land contamination. CASI flights were conducted in on April 9, 1999 and September 6, 1999. The April data set had eight flight lines: three flight lines with 13-band setting and five flight lines with 48-band setting. The September data set consisted of two flight lines, one with 13 bands, and one with 72 bands. CASI imagery collection was acquired at a nominal spatial resolution of 1000m.

The empirical line technique was used to radiometrically correct data using ground spectra that coincided with the CASI data acquisition (Ferrier, 1999). The imagery was geometrically rectified using a nearest neighbour re-sampling algorithm. To assess the sensitivity of the CASI data in identifying "stressed" and "non-stressed" vegetation, field spectra were convolved to the CASI bandwidths for the 13, 48 and 72-band setting to review spectral profiles. The slope (first derivative) was also calculated for each CASI band setting (13, 48 and 72) to see whether there was an increase or change in slope when comparing "stressed" and "non-stressed" vegetation. An analysis of the raw radiance CASI data was also carried out to identify whether any spatial patterns representing the "stressed" vegetation was identifiable at different stages in the vegetation growing cycle and to determine the sensitivity of the number of bands in the detection of the "stressed" vegetation. A Minimum Noise Fraction (MNF) transform was applied to the whole CASI data set (Green *et al*, 1988). This produced a set of principal component images ordered in terms of decreasing signal quality.

The CASI data set had several positive features in context of providing broader scales of field information needed for the site assessment. Firstly, it provided airborne images of field conditions that could not be inferred through other methods of field assessment. Secondly, data sets were easily integrated with other field-based data sets, e.g. field-based chlorophyll concentrations and field-based spectroradiometer readings, in GIS to provide multi-spatial maps of landfill conditions.

6.6 Limitations Faced By the Investigations

There were two main challenges faced during each of the investigations. The first problem was that although a lot of information was collected about each study site, there were many gaps and assumptions made about field conditions. This lack of data is a common problem faced when conducting a landfill site assessment. To an extent, the lack of data simulated the inherent uncertainties faced in assessments. There are many remedial landfill projects based on models and evaluations that have many unvalidated geophysical assumptions (e.g. Jurkovic, 1995; Radenkova Yaneva *et al,* 1995; Fatta *et al,* 1997; Kjeldsen *et al,* 1998(a); Ahel *et al,* 1999; Splajt *et al,* 1999). The need for more detailed information at each study site was experienced during all the investigations. For the modelling investigations (investigations 1, 4, 5 and 6), the lack of data meant that some model parameters had to be assumed or inferred from regional estimate values or published literature. The problem of missing geophysical data during landfill model construction is a challenge that is commonly faced by modellers. However, the limitation provided an opportunity for investigation 6 to be integrated into the modelling objectives (the investigation tested different data sets to determine their influence on contaminant simulation). For the field investigations, the lacking data meant that geological profiles used in the GPR investigation were not necessarily representative of survey line conditions. In the remote sensing application, historical records of leachate quality at higher frequencies would have helped to establish clearer links between the landfill leachate and contaminant pathways.

The second problem was that there were not enough standards or guidelines dealing specifically with kriging, GPR or modelling practices when looking at risk-based landfill site assessments. This challenge was frequently experienced during the development and planning of investigations. There are two reasons for the lack of clear guidelines. Firstly, the six investigations cover a very wide spectrum of interdisciplinary topics that include hydrology, geology, contaminant transport, ecology, biology, chemistry, geophysics, geostatistics, 2-D and 3-D environmental modelling and knowledge of British, Canadian and American landfilling, contaminated land, modelling and site assessment practices and legal frameworks. Given the diversity of topics, some form of 'data mining' was necessary. The second reason, however, is that the investigations addressed new approaches to landfill site assessment and challenged the robustness of site assessment and modelling practices. The instruments and techniques presented in investigations 1, 2 and 3 are new techniques, which have not been widely applied to landfill sites. This, on the one hand, proves that the investigations are a new contribution to applied landfill management. On the other hand, it also explains why the author had to research other industries and other fields that have applied these techniques, in order to find guidelines and explanations. In the kriging investigation this problem was experienced by the fact that kriging had been applied in hydrogeologic, river management and ore exploration studies, to name a few. A lot of time was spent learning about these topics, in order to apply kriging to the field in question. In the case of GPR, this problem was faced with the instrument's input parameters. Literature did not give insight into acceptable values for subsurface landfill conditions. During data collection critical assumptions about conditions had to be made in order to assign the closest pre-defined input parameters for relative permittivity, electrical conductivity, velocity, and attenuation. Surprisingly, for investigation 3, there was a significant amount of information that did provide guidance during the data collection and analysis phases, e.g. Strub *et al* (1998), Barrett and Curtis (1999, p.101-111), Jago *et al* (1999); however, a very large amount of time was spent waiting for the CASI data to arrive.

The lack of adequate modelling standards was a major difficulty that was faced during the modelling investigations. In the initial phases of model construction, the software manual, training sessions and university level courses seems adequate for model construction. However, this began to change when the author realised that there were two very different uses of groundwater and contaminant flow models. One group of modellers considers them to be tools with which to better understand and test subsurface conditions. The other groups of modellers use such models for risk estimation applications. It was at this point that a literature review of modelling standards was conducted. The literature review found that ASTM had a large number of standards that applied to groundwater flow models, with some focus upon contaminant modelling such as pesticide and NAPL transport, but guidance for landfill applications was difficult to find. In recent years (2001- 2003) several contaminant and landfill modelling guidelines have been issued by the Environment Agency (e.g. McMahon *et al,* 2001 (a-c); Whittaker *et al,* 2001; DEFRA 2002(d)). However, most of the modelling was completed by the time these documents were issued.

6.7 Chapter Summary

This chapter has discussed the methods that were applied in investigations 1 – 6, identifying their objectives as well as the advantages and disadvantages in terms of the research question. It has also reviewed the main challenges that were faced during the development and implementation of these investigations. In the next chapter, the three study sites will be described in order for the reader to gain a better conceptual understanding of the conditions and risks faced at each site.

CHAPTER 7: STUDY SITE DESCRIPTIONS

7.1 Introduction to Study Sites

7.1.1 Introduction

This chapter will describe the three study sites that were used to conduct investigations 1 to 6. The description of each site will include its geographical location and history as well as a description of the geological, hydrological and landfill conditions. The chapter aims to:

- describe the hydrogeological conditions at each study site
- describe the landfill structure and leachate migration occurring at each study site
- provide the reader with a conceptual model of each study site in order to better understand the results presented in Chapter 8.

7.1.2 Criteria for Site Selection

The three sites were selected because the landfill management companies at all three sites were willing to provide the researcher with several years of historical landfill data as well as access to each site in order to conduct field studies. Sites A and C were selected early on in the project because Humberside Wastewise Ltd. (a waste management company in East Yorkshire) initially funded the research project through the 'Enventure' funding program. Site B was selected based on two facts: an extensive amount of historical data was available to the researcher. Secondly, both academic and waste management institutions in Croatia were willing to co-operate and work with foreign researchers in dealing with a local waste management and water quality problem. All three landfill sites were operational at the time of research, having older and unlined as well as lined and contained landfill areas. As stated in Section 1.6, the sites had four similarities, they were all the sites were based on strata with sand-clay lenses, they contained unlined buried landfill cells, they were identified to be leaching off site and they all posed risks to local soil and groundwater resources.

7.2 Study Site A

7.2.1 Description of Site A

Site A is located in north east England, 1.5 km from the Humber Estuary (Figure 7.1(a-b)). The site has been opened since 1988, receiving domestic and industrial waste from surrounding regions. The landfill covers approximately 32 ha and is surrounded by flat agricultural and industrial land (Entec, 1996(b)). The site elevation ranges from 5-15m above sea level, containing 15 cells. Cells 1-6 are the oldest, in which waste was placed onto unlined silty alluvium. They were filled with 3 to 5m of waste and capped with 1.5m of silty clay alluvium. Cells 7-11 are engineered to the depth of the boulder clay, in which local boulder clay met regulation standards and was therefore used as landfill liner (Department of Environment, 1995). Cells 11-15 were under construction at the time of the assessment. The unlined parts of the landfill (cells 1-6) continually experienced leachate migration off-site (Figure 7.1(c)). The effect has been local soil contamination in which a toxic wetland began forming in 1997 adjacent to the site, showing indications of vegetation stress and alteration to the local ecosystem. The unlined landfill cells, as the source of contamination, have deteriorated local soil quality and ecosystem health as outlined in lines along the landfill edges in Figure 7.1(c).

A preliminary and detailed study of site conditions was conducted at this landfill from 1999 through to 2001. The data collected were used in all the investigations (1-6).

Figure 7.1(a) Location of Site A in north east England

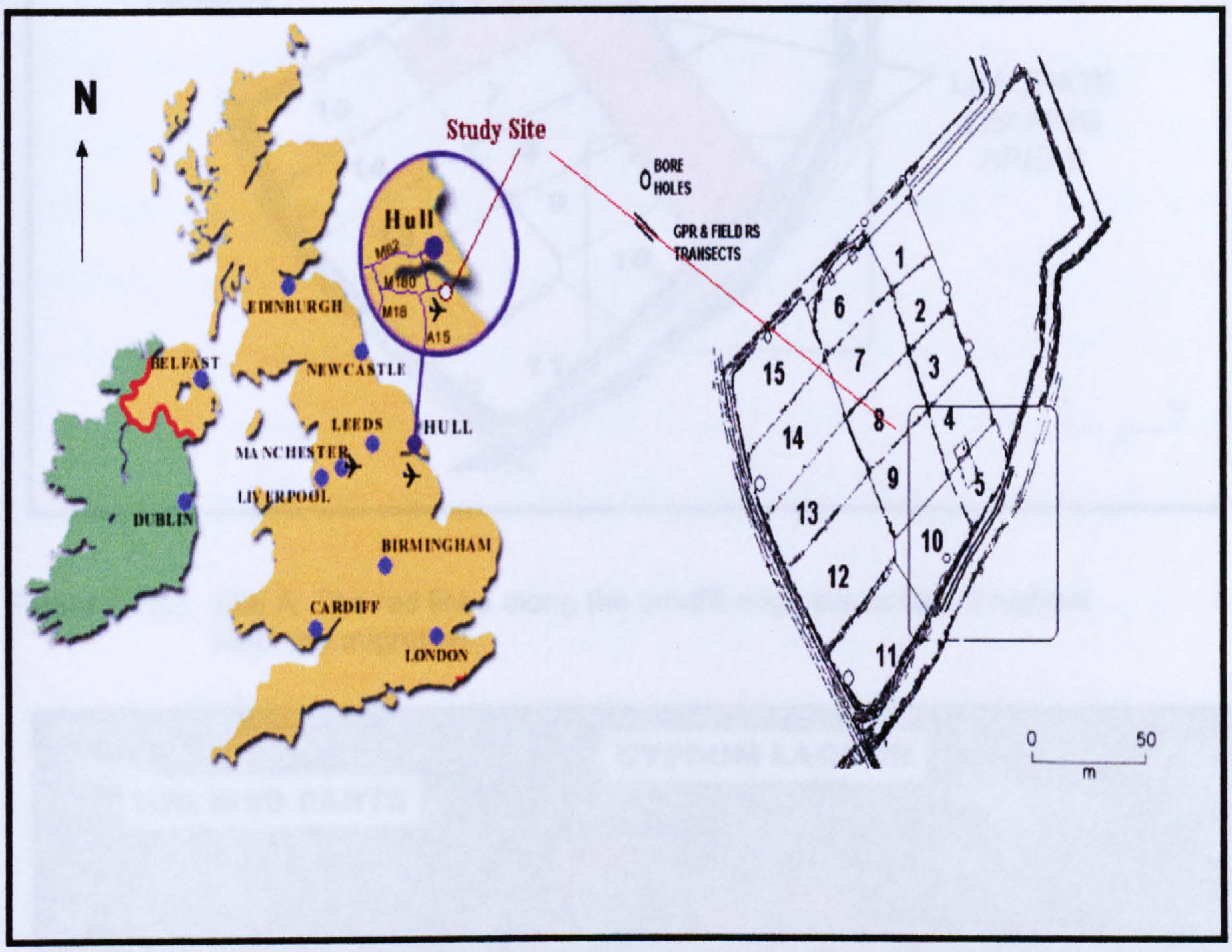


Figure 7.1(b) Location of Site A in north east England, 1.5 km from the Humber Estuary

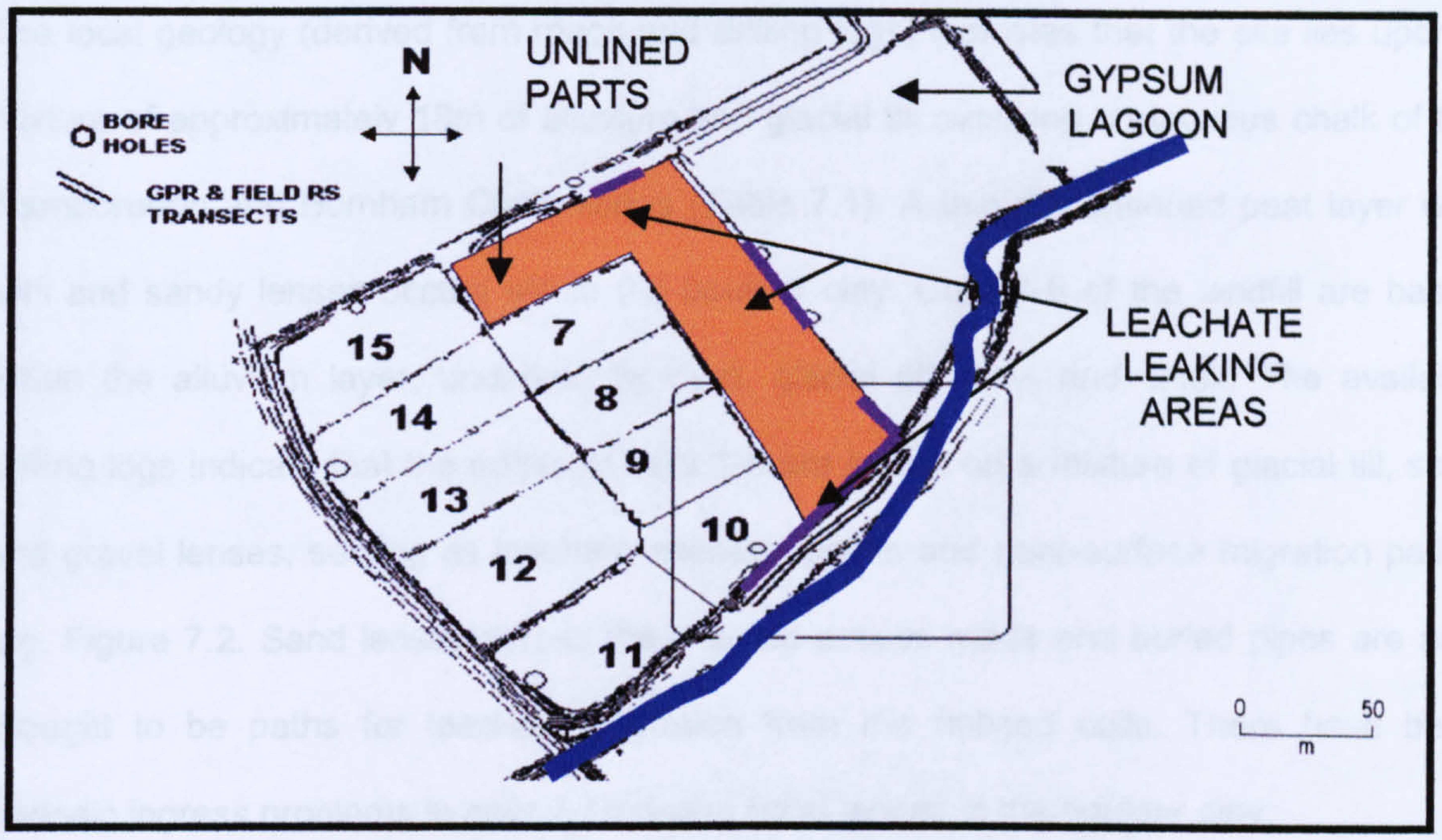
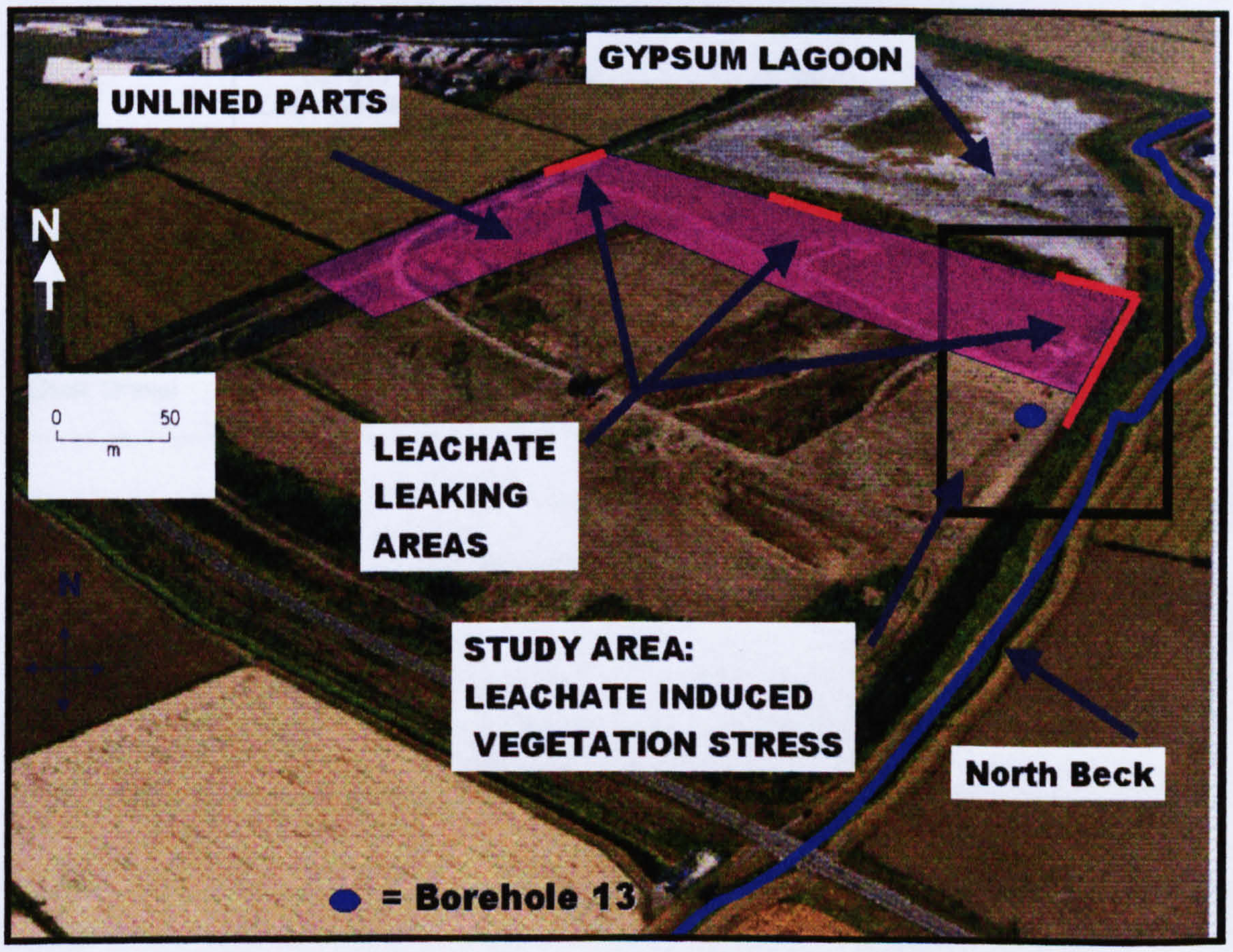


Figure 7.1(c) Site A: The red lines along the landfill edge are areas of highest leachate migration



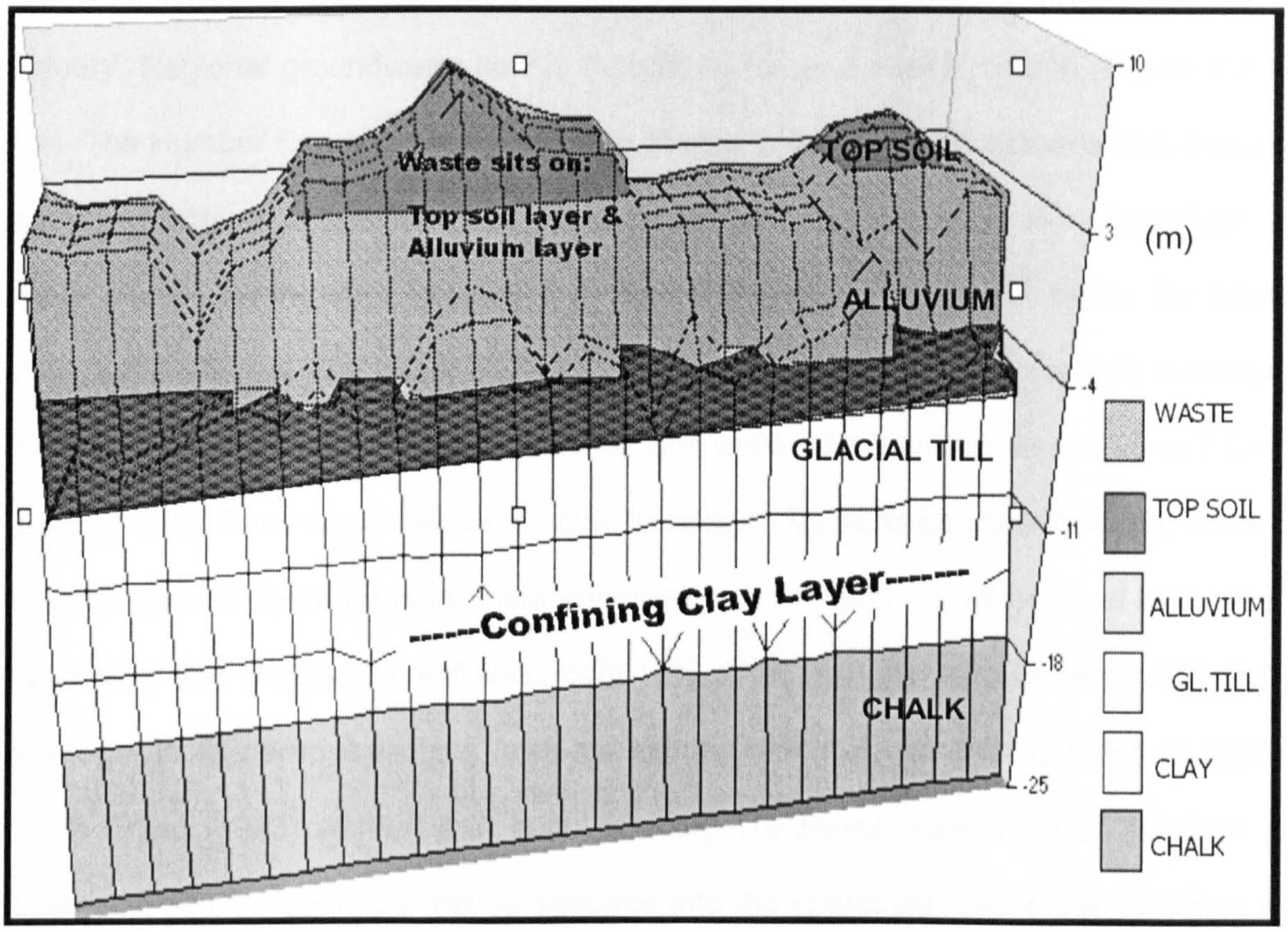
7.2.2 Geology at Site A

The local geology (derived from maps and drilling logs) indicates that the site lies upon a mixture of approximately 18m of alluvium and glacial till overlying cretaceous chalk of the Flamborough and Burnham Chalk series (Table 7.1). A thin discontinued peat layer with silts and sandy lenses occurs within the boulder clay. Cells 1-6 of the landfill are based within the alluvium layer, underlain by peat, glacial till, clay, and chalk. The available drilling logs indicate that the edges of cells 1-6 are based on a mixture of glacial till, sand and gravel lenses, serving as leachate seepage points and near-surface migration paths, e.g. Figure 7.2. Sand lenses across the site, old access roads and buried pipes are also thought to be paths for leachate migration from the unlined cells. There have been periodic ingress problems in cells 7-10 due to sand lenses in the boulder clay.

Table 7.1 Summary of hydrogeologic information for the area near Site A taken from four different site assessments conducted from 1990-1996

Material	Cited in Entec (1996(b))	Cited in Entec (1993)	Cited in AIG Consult. (1992)	Cited in C.J. Smith (1992)
Soil	N/A	0.7 - 2.0	N/A	N/A
Estuarine Alluvium	0 - 7: 11.0	0.8 - 1.4 2.7 - 7.0	0 - 2.0	0 - 3.0
Glacial Deposits	7: 11 - 15.3	7.0 - 11.5	2.0 - 13.0	3.0 - 11.0
Chalk Gravel	11.5 - 15.3	11.5 - 15.3	Unknown	11.0 - 14.0

Figure 7.2 Geological layers under Site A (depth in metres)



7.2.3 Hydrology at Site A

(a) Surface Water Flow

The North Beck drain flows from west to east towards the Humber Estuary, along the south edge of cells 10 and 5 (this is shown in Figure 7.1(c) as the blue winding line on the right side of the photograph). It is fed by surface runoff and recharge and is confined by an embankment on either side of the drain. An accumulation of landfill leachate and runoff has developed along the southern edge of cells 5 and 10, running parallel to the North Beck drain. Aerial photographs confirmed that this accumulation fluctuates after rain events, and has caused vegetation stress along the southern perimeter of the site, e.g. Figure 8.36. A detailed study of ecological conditions was conducted, confirming a high concentration of landfill leachate in surface water, soil, and local vegetation.

(b) Groundwater Flow, Sinks, and Sources

The local topography gently slopes from north west to south east toward the Humber Estuary. Regional groundwater flow is thought to follow a similar course (Figure 7.3 and 7.4). The Humber Estuary is not thought to directly affect landfill hydraulics as the boulder clay layer acts as an aquitard, confining landfill leachate migration to near-surface and upper alluvial layers while keeping the regional flowing groundwater below the boulder clay at chalk levels. The upper alluvium layer is unsaturated but will transmit recharge to the lower silts and sands that are saturated with regional transmissivities of up to 7.9 m^2/d (Entec, 1993). Sand lenses within the clay have been found to be water bearing. Table 7.2 lists site-specific hydraulic data; meteorological data for 1985 - 1998 is found in Table 7.3, suggesting that effective rainfall was about 17% of the total precipitation with relatively low recharge for this area. Leaching from the unlined cells 1-6 was initially noted in 1993, in which Entec (1993) warned that such leaching conditions could lead to seepage into underlying alluvial layers as well as seepage into the contained cells if the cell lining was weak.

Table 7.2 Hydraulic conductivity (m/s) and transmissivity (m^2/d) from different historical site investigations of Site A

Material	Cited in Entec (1996(b)) in m/s	Cited in Smith (1992) in m/s	Cited in Entec (1993) in m^2/d	Cited in AIG (1992) in m^2/d
Silt	$3.2x10e^{-7}$: $2.4x10e^{-8}$	$0.1x10e^{-03}$	7.9 : 0.4	7.9
Clay	$2.5x10e^{-10}$: $9.3x10e^{-10}$	$0.1x10e^{-09}$	0.005 : 0.15	N/A
Chalk	N/A	$0.11x10e^{-03}$	N/A	N/A
Chalk Gravel	N/A	$0.1x10e^{-02}$	N/A	N/A

Table 7.3 Annual mean rainfall for Site A: (a) = amount in mm/yr. cited in Smith (1992); and Environment Agency (1999); (b) = effective rainfall cited in Smith (1993)

Year	85	86	87	88	89	90	95	96	97	98
(a)	611	687	671	617	540	685	500.2	503	574	365
(b)	96	150	143	143	3	123	N/A	N/A	N/A	N/A

Figure 7.3 Locations of boreholes used for monitoring groundwater levels and leachate quality and the direction of leachate fluctuations

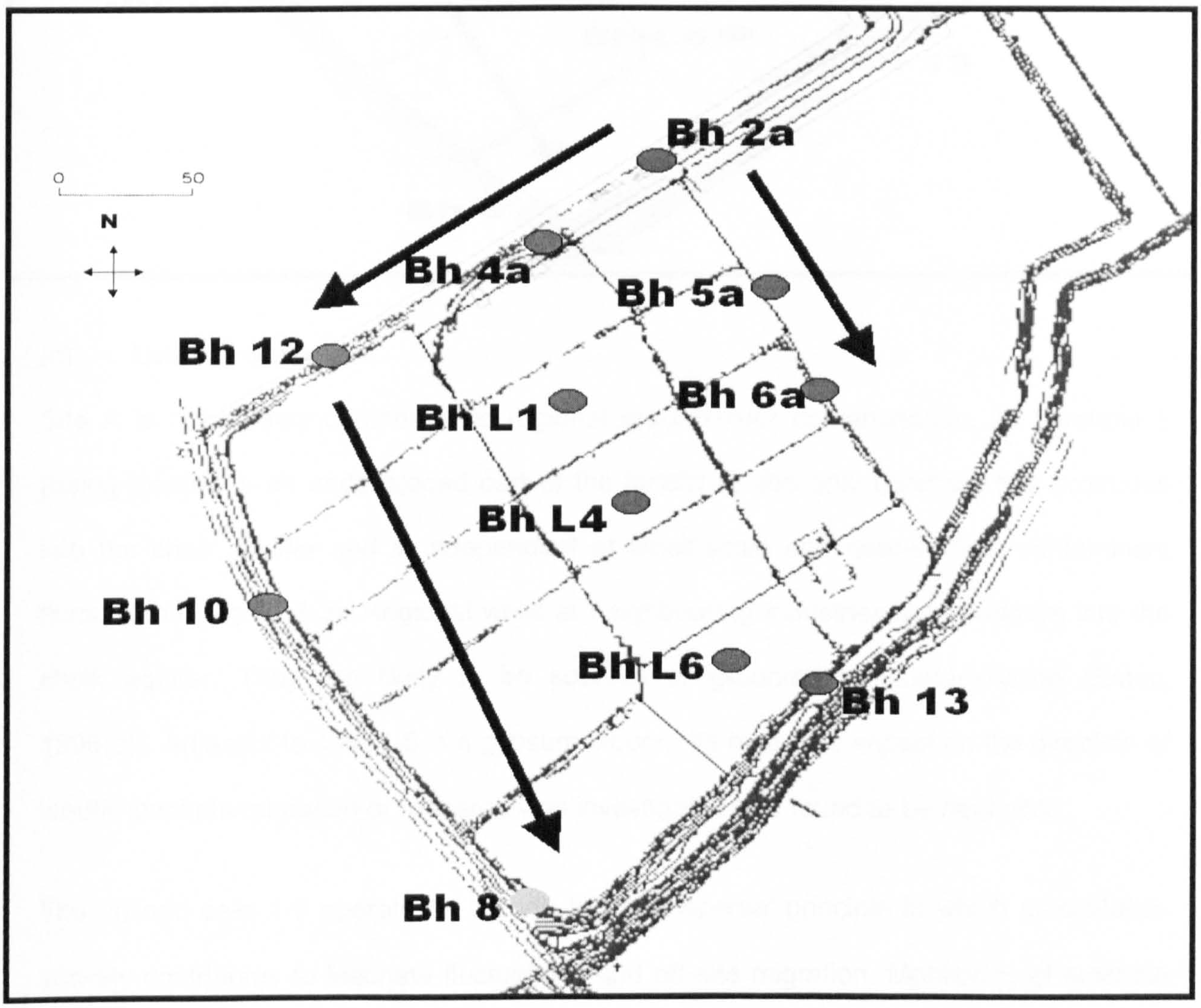
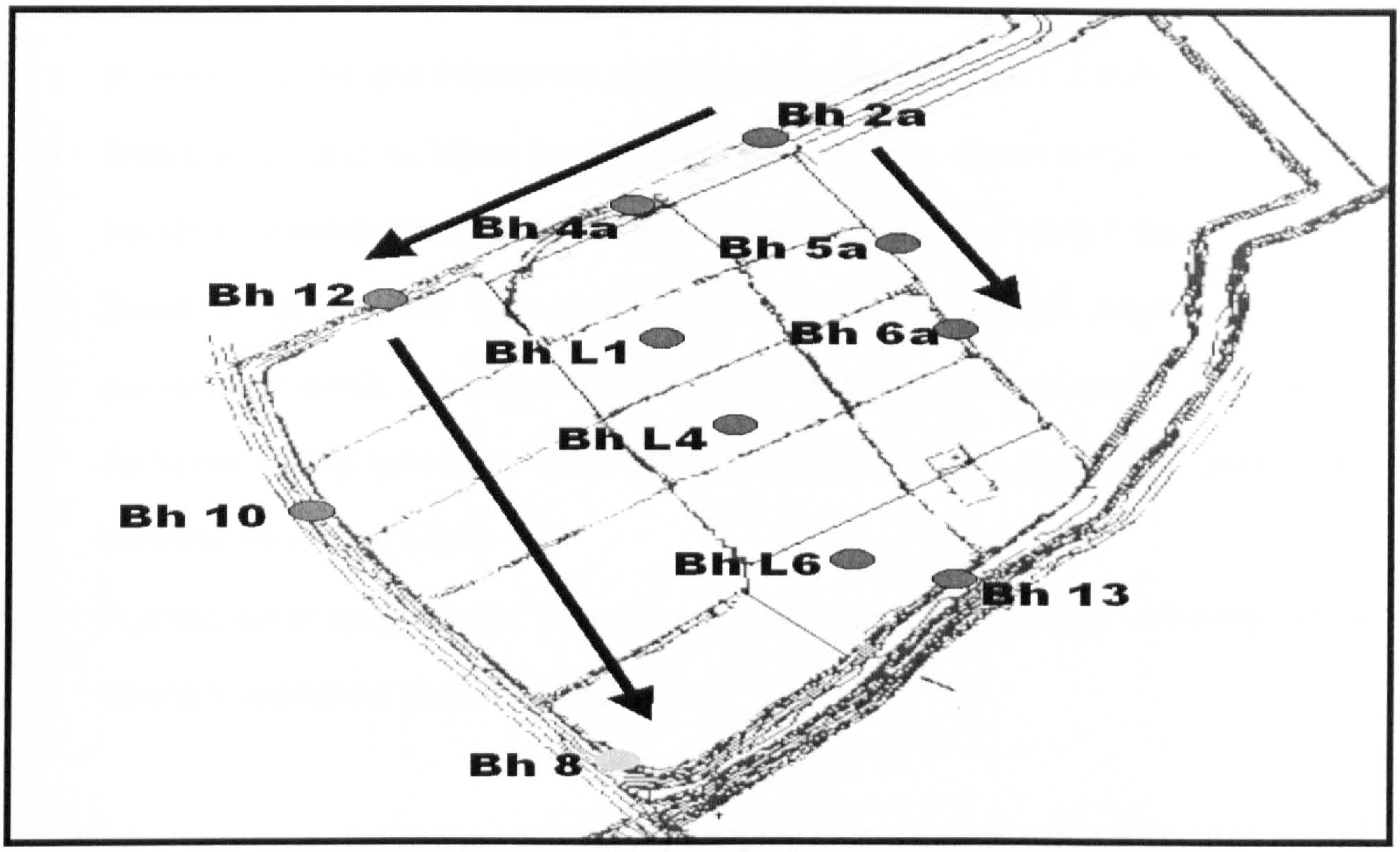


Figure 7.4 Hydrological boundaries and flow conditions around Site A and the direction of water flow (groundwater and the North Beck)



(c) Landfill Hydraulics

Site A is not a likely contributor to regional groundwater contamination, as borehole 8 (being located in an undeveloped part of the landfill) is the only borehole that protrudes into the chalk aquifer and is independent of small-scale and near-surface contaminant fluctuations (Figure 7.3). Regional wells at neighbouring industries also protrude into the chalk aquifer. They are likely to be sources of groundwater contamination (Entec, 1996(b)). Adjacent to cells 1-5 is a gypsum lagoon. Its hydraulic impact on the direction of landfill leachate migration or recharge was investigated and found to be negligible.

The unlined cells 1-6 operate on the 'dilute and disperse' principle in which precipitation actively contributes to leachate fluctuations and off-site migration. Monitoring of leachate in these cells began in 1992. The data shown in Figure 7.5 suggests that leachate in cells 1-6 moves north to south east. Leachate concentration varies drastically in cells 1-7 but is relatively constant in the engineered cells 7-11. Near-surface hydraulic fluctuations around the landfill can be summarised as follows:

- Boreholes 12, 10, and 8 follow the same hydraulic regime (Figure 7.5(a))
- Boreholes 2a and 4a follow the same hydraulic regime. Borehole 12 is independent but shows some delayed similarities in hydraulic fluctuations (Figure 7.5(b))
- Boreholes 2a, 5a and 6a seem to be independent hydraulic regimes in which groundwater levels in borehole 6a are constant and high at all times. Borehole 5a fluctuates slightly but seems to have a constant input source at a lesser degree than borehole 6a (Figure 7.5(c))
- Fluctuations at boreholes 6a, 13 and 8 are independent of each other with borehole 13 having a substantial amount of fluctuations.

Figure 7.5(a) Near-surface groundwater fluctuations around Site A showing that boreholes (Bh) 12, 10, and 8 follow the same hydraulic regime

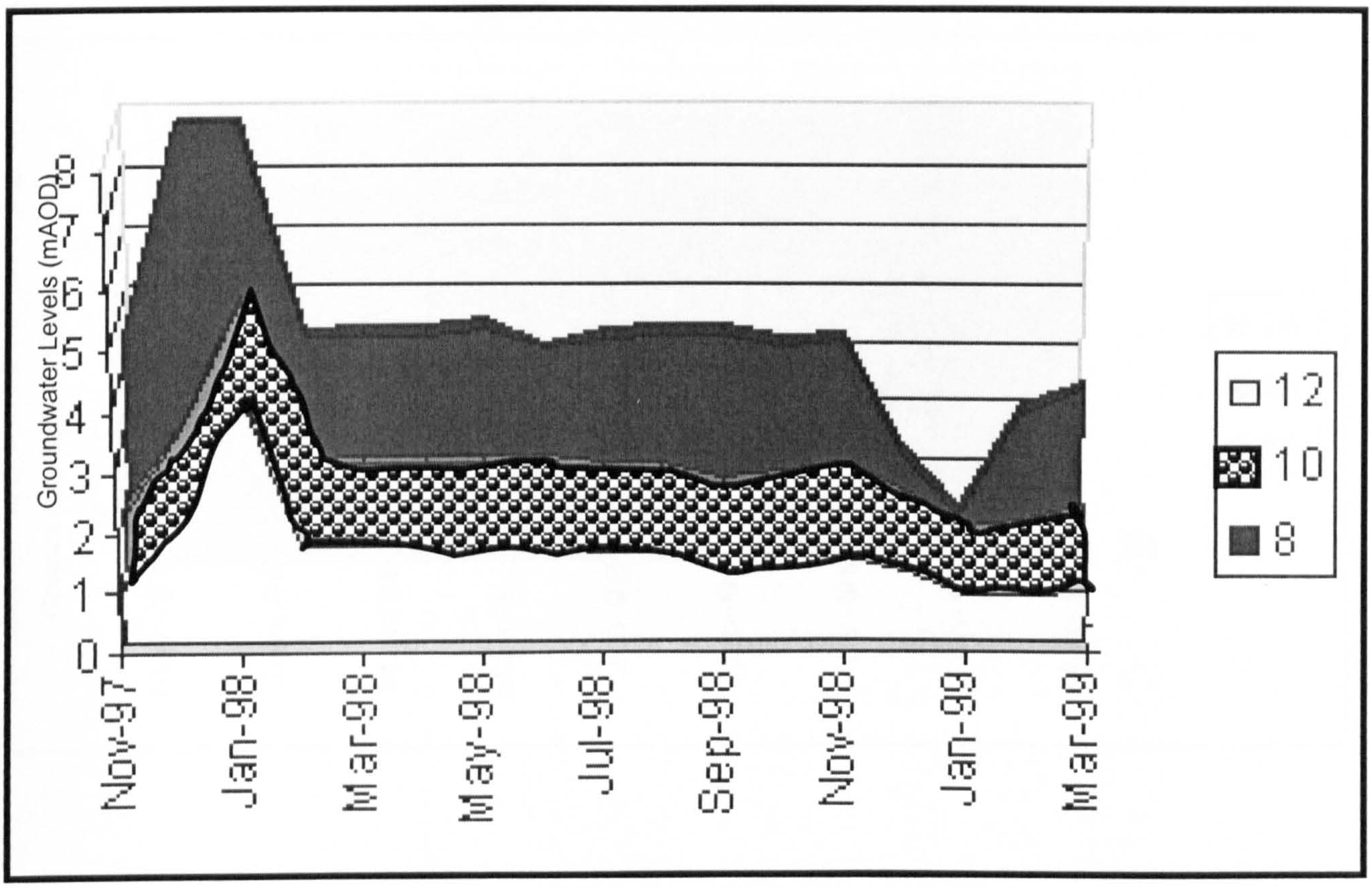


Figure 7.5(b) Groundwater fluctuations showing that boreholes 2a and 4a follow the same hydraulic regime while Bh 12 shows some delayed similarities

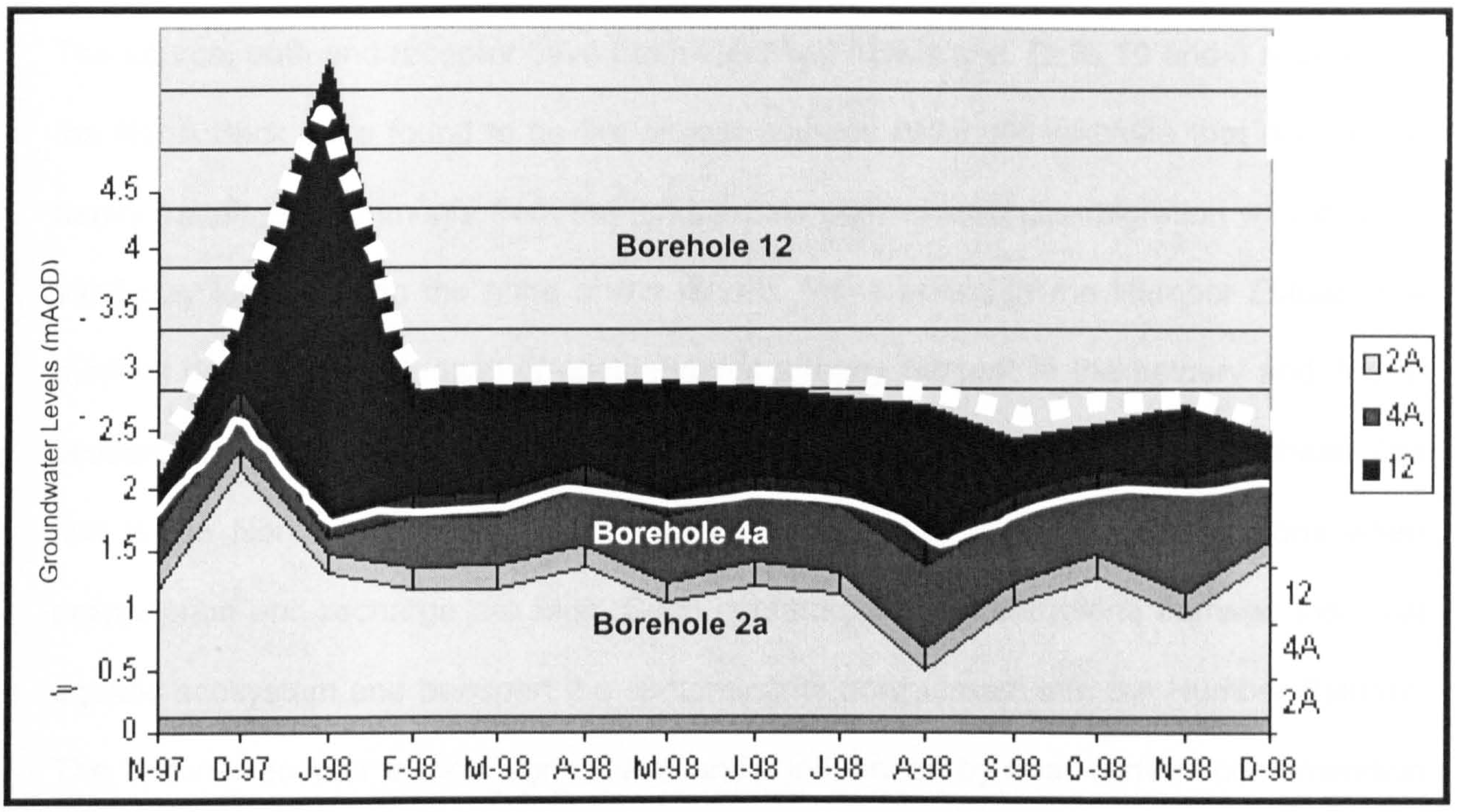
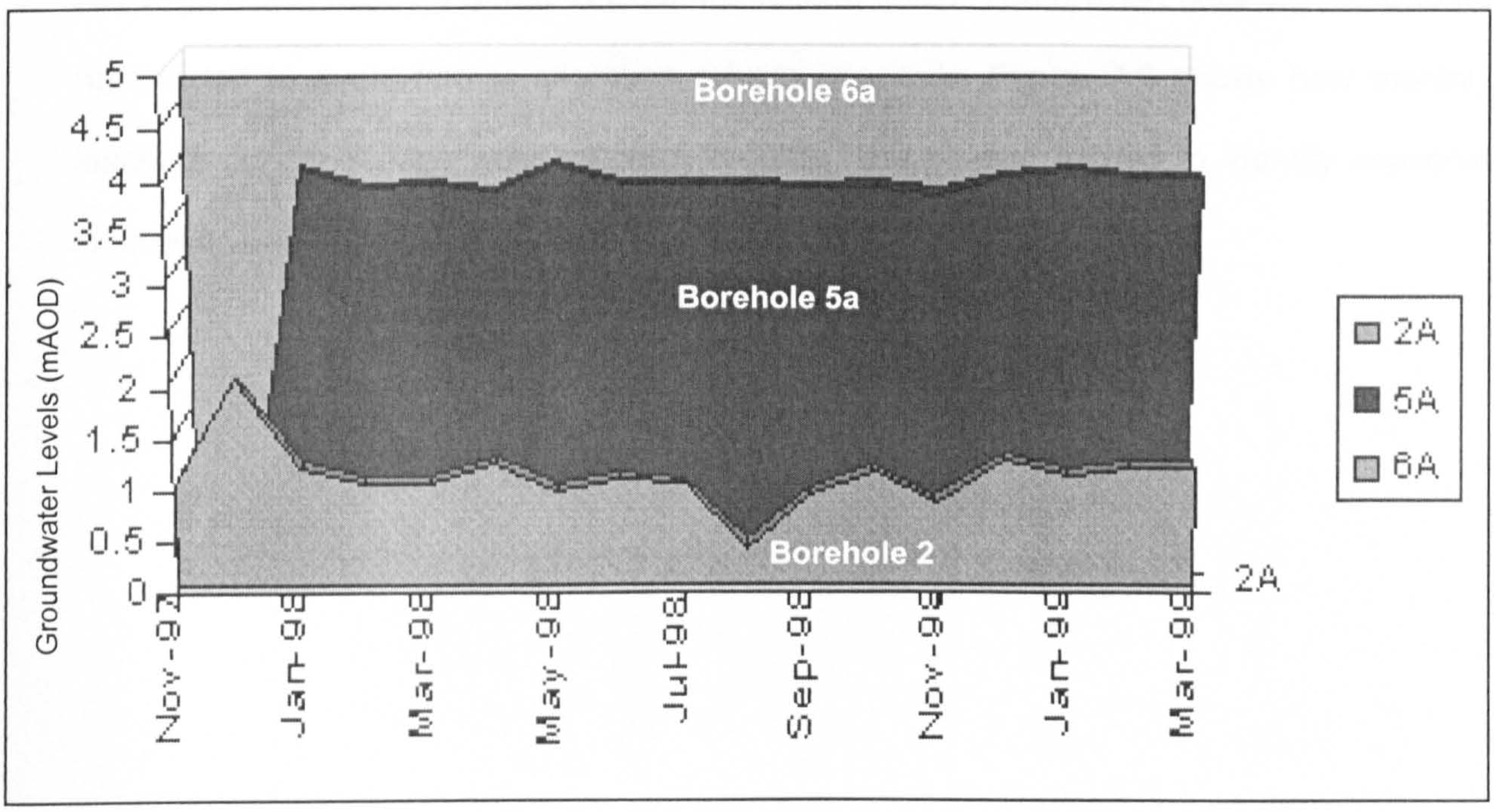


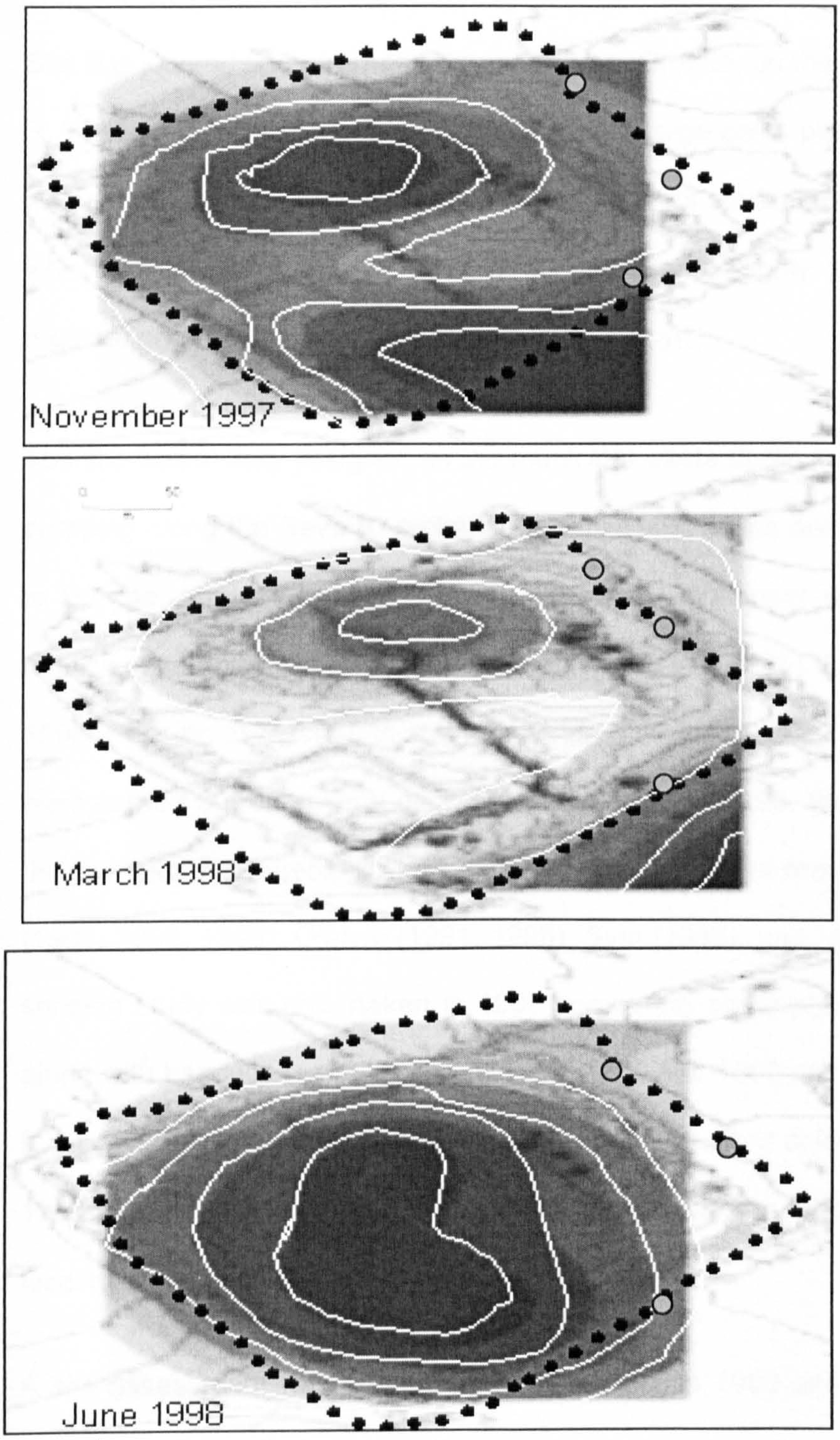
Figure 7.5(c) Groundwater fluctuations showing that Bh 2a, 5a and 6a have independent hydraulic regimes



7.2.4 Risk-Based Assessment of Site A

The source, path and receptor have been identified at this site. Cells 10 and 5 adjacent to the North Beck were found to be the largest sources of landfill leachate that discharged heavy metals and ammonia from the landfill. The path of leachate migration was through sand-clay lenses along the edge of the landfill. Risks posed to the Humber Estuary are minimal due to the high level of contaminants already present in the estuary and due to dilution factors. However, there are three other receptors influenced by the leachate. The first is the North Beck drain, which receives leachate in periodic concentrations when precipitation and recharge are high. Such contaminant concentrations damage the local aquatic ecosystem and transport the contaminants downstream into the Humber Estuary. The second receptor is local agricultural lands, influenced by near-surface contamination of groundwater supplies, contaminating agricultural fields surrounding the landfill site. The third receptor is near-surface soil and groundwater contaminated by leachate flushing and recharge which poses a threat to local agricultural fields. The unlined cells 1-6 need to be remediated to avoid further migration off-site into soils. Figure 7.6 shows how monthly leachate concentrations were mapped in GIS. These maps helped to identify seasonal fluctuations and directions of contaminant transport.

Figure 7.6 GIS-based risk assessment using measured ammonia concentrations to map areas of highest leachate fluctuations, identifying areas with highest potential for leachate migration



Legend:
The darker areas have higher leachate concentrations while the lighter areas have lower leachate concentrations in mg/l of N (Dark grey – white = 4.7 - 0.5 mg/l of N)

○ = reference point, representing calibration points 5, 6 and 13

■ = 4.7 mg/l of N

□ = 0.5 mg/l of N

7.3 Study Site B

7.3.1 Description of Site B

Site B is located 6 km from the City of Zagreb, Croatia, on the south east bank of the Sava R. The site is located within a groundwater recharge zone, planned as a long-term potable water supply source for the region. It is upstream of several municipal pumping stations, in which waste was dumped for over 30 years onto highly permeable alluvium causing regional groundwater contamination (Figure 7.7(a)).

The site was initially assigned as the municipal waste dump in 1965 in order to control wild dumping along the Sava R. embankment. In 1995, it was one of the largest waste dumps in Europe, covering some 1500 x 400m and containing some five million tonnes of municipal, industrial and hazardous waste (Jurkovic, 1995). Landfill remediation began in 1998. Periodic assessments and monitoring of hydrogeologic and geochemical conditions were carried out on several occasions from 1986 to 1998. Studies that have investigated the complex hydrogeologic and geochemical conditions around the landfill include Ahel (1991, 1998, 1999); Gjetvaj (1991, 1998); Svel (1998); and Mikac *et al*, (1999). A remote sensing study was undertaken in 1995 comparing aerial photographs from 1968 - 1989 along with Landsat TM images from 1984, 1990 and 1992 and SPOT P images from 1994 (Olujic, 1995). The study identified regional fault lines and calculated waste quantities from 1968 through 1994 but did not give much insight into the heterogeneity of the regional or local hydrogeologic system (Figure 7.7(b)).

A site assessment was conducted at this landfill in 1999 and 2000. Data collected were used in modelling that was conducted in investigations 1, 4, 5 and 6.

Figure 7.7(a) The landfill location in proximity to Zagreb city centre, the local pumping stations, residential areas and the Sava R.

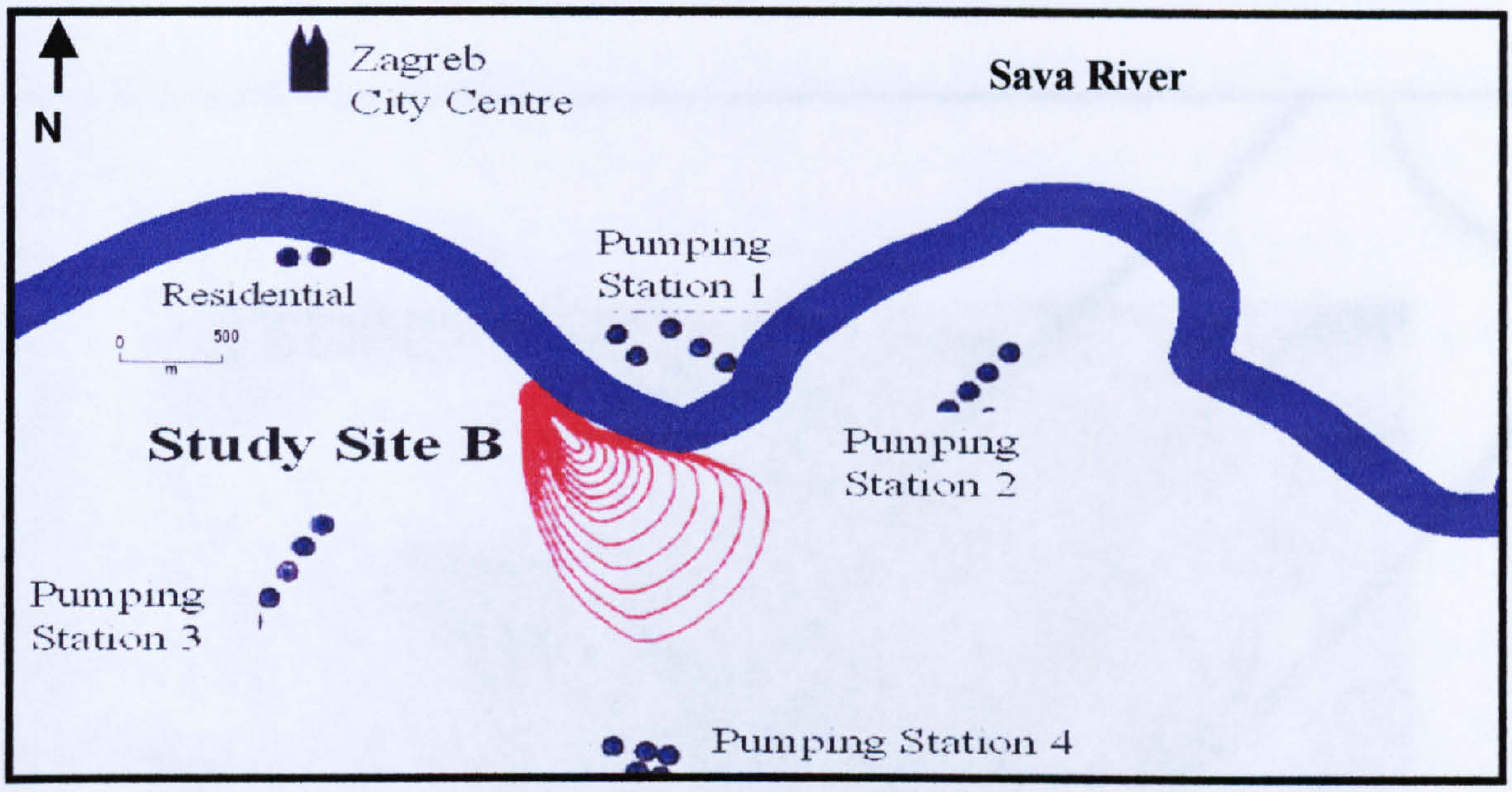


Figure 7.7(b) A cross-section description of regional geology and the site's location in relation to the city centre

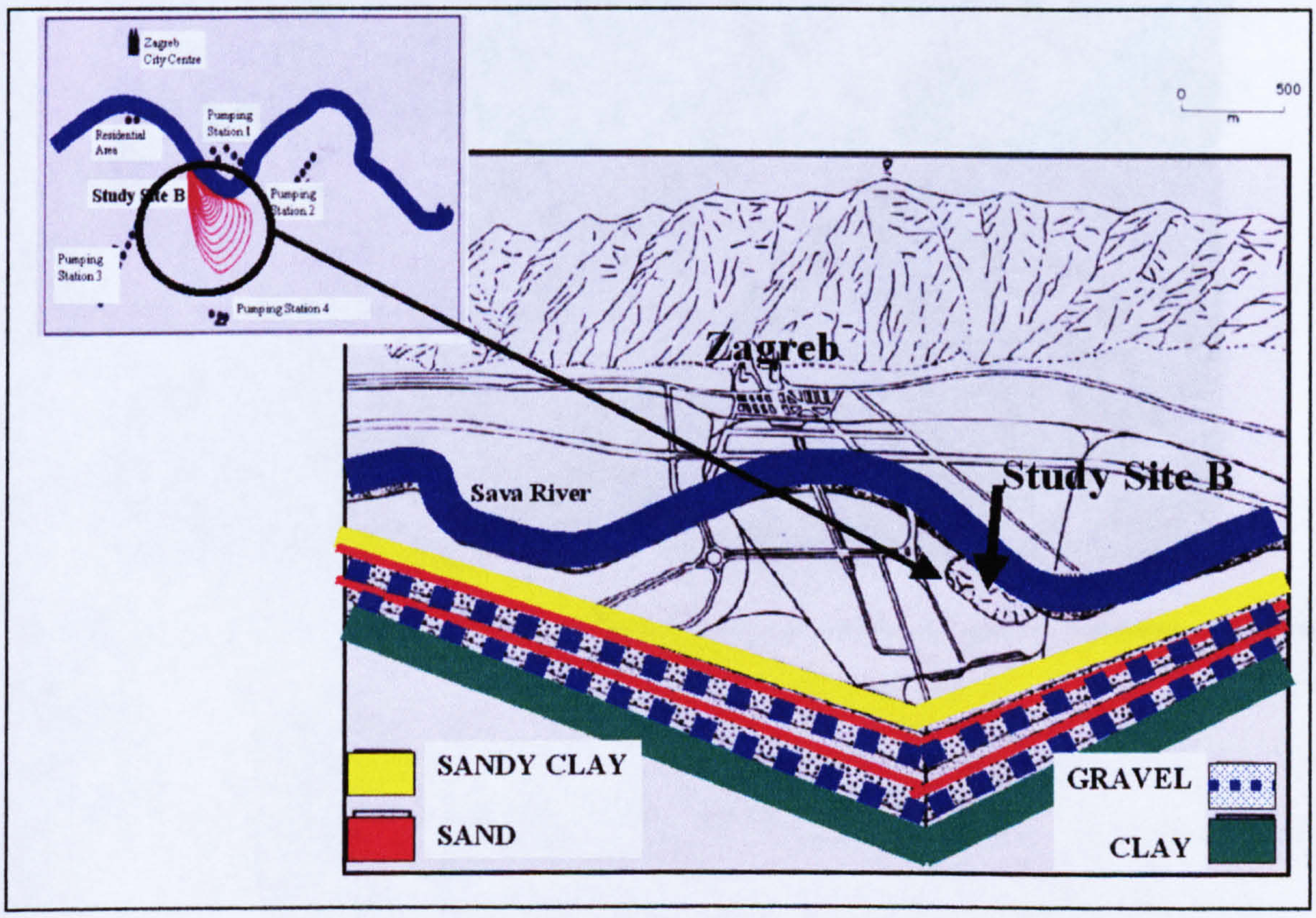


Figure 7.7(c) Remote sensing images of Site B using Landsat TM images (cited in Olujic, 1995)

Legend:

Landfill edge 1995

Direction of regional groundwater flow

Sava River

Locations of buried waste identified used remote sensing images

(1) Waste buried 1968-1977

(2) Waste buried 1977 - 1984

(3) Waste buried 1984 - 1992

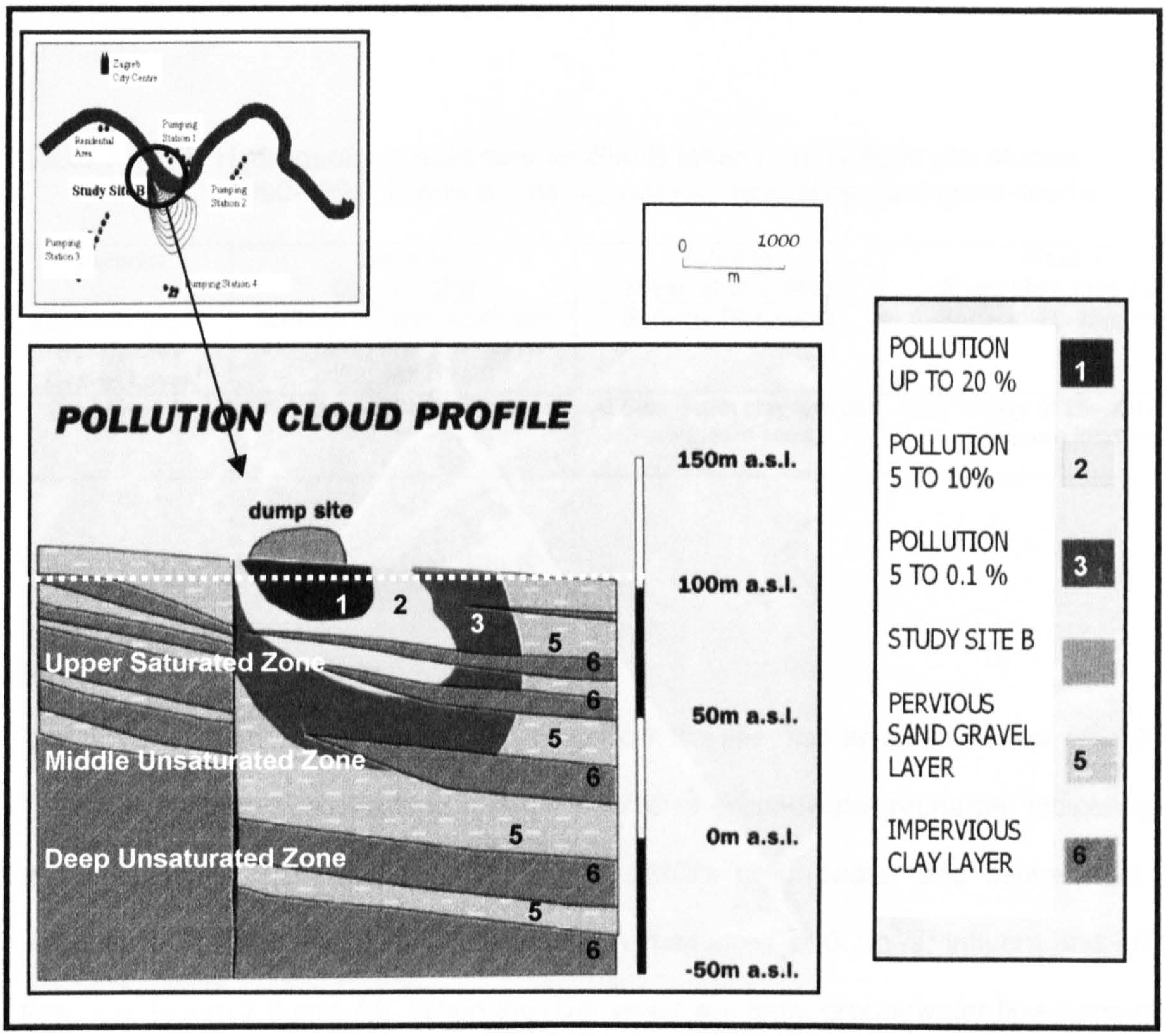
An initial daily groundwater monitoring program was conducted between 1995 and 1996 (Svel, 1998). The data confirmed findings of previous assumptions about flow directions, velocities and fluctuations. In 1998 27% of the site was remediated with completion expected in 2010. The remediation project comprised of sectioning the wild dump into several cells, initially digging out the old buried waste, section by section and then lining and engineering each designated area, bailing the old waste, and re-burying the waste.

The new depth of the landfill is somewhat controversial, planned at 110 – 107 m AOD. The depth of the unlined dump (prior to remediation) varies, estimated as deep as 100 m AOD (see Figure 7.8). The unlined landfill depth and its influence on regional groundwater quality were determining factors when designing the site's remedial actions. The data for these factors were defined by numerous scientific studies such as those done by Ahel (1990) and Olujic (1995), to mention a few. The remediation is based on a conceptual model that assumes that: (a) the regional groundwater level will not rise higher then 105 m AOD and (b) the landfill's contaminant plume does not migrate under the adjacent Sava R.

This model is highly disputed, as the Petrusevec Pumping station is located on the opposite site of the river. The pumping station may very well influence leachate migration from the existing leachate plume or from potential leaks in the remediated site, since the landfill is located on the left bank of the Sava R., while the pumping station is located on the right bank. This hypothetical situation is possible since leachate migration and fluctuation are highly dependent upon the river levels that affect the direction of groundwater flow (Figure 7.8). The Jakusevec landfill poses a real threat to Zagreb's future potable water supplies, as it is located upstream and adjacent to several regional water pumping stations. Critics of the planned remediation claim that the conceptual model on which the remediation is based is over simplified and does not account for the heterogeneous regional hydrogeology (based on interviews with experts at the

Department of Civil Engineering Department, University of Zagreb; Institute Rudjer Boskovic, University of Zagreb; and ZGOS d.o.o. Waste Management Company).

Figure 7.8 Cross-section conceptual model of regional geology and contaminant plume under Site B.



Legend:
100% pollution = landfill site which is assumed to have 100% pollution, measuring ammonia concentrations
Zone 1 = Up to 20% pollution
Zone 2 = 5 to 10 % pollution
Zone 3 = 5 to 0.1 % pollution
Zone 5 = Pervious sand gravel layer
Zone 6 = Impervious clay layer

7.3.2 Geology at Site B

The Site is located on a heterogeneous and highly permeable alluvium aquifer that is approximately 90m thick. Under the landfill is an irregular pattern of 0.5 to 3m thick clay lenses within a sand and gravel alluvium layer that is 30-50m thick. In order to simplify the geologic conditions within a conceptual model (Figure 7.8), the regional hydrogeology can be spatially lumped into three zones - the upper unsaturated zone, the middle saturated zone (depths up to 50m below surface), and the deep subsurface zone (depths from 50 - 100m below surface).

Table 7.4 Hydrogeologic summary for Site B taken from various site studies (1990-1998) showing heterogeneity in describing geological depths

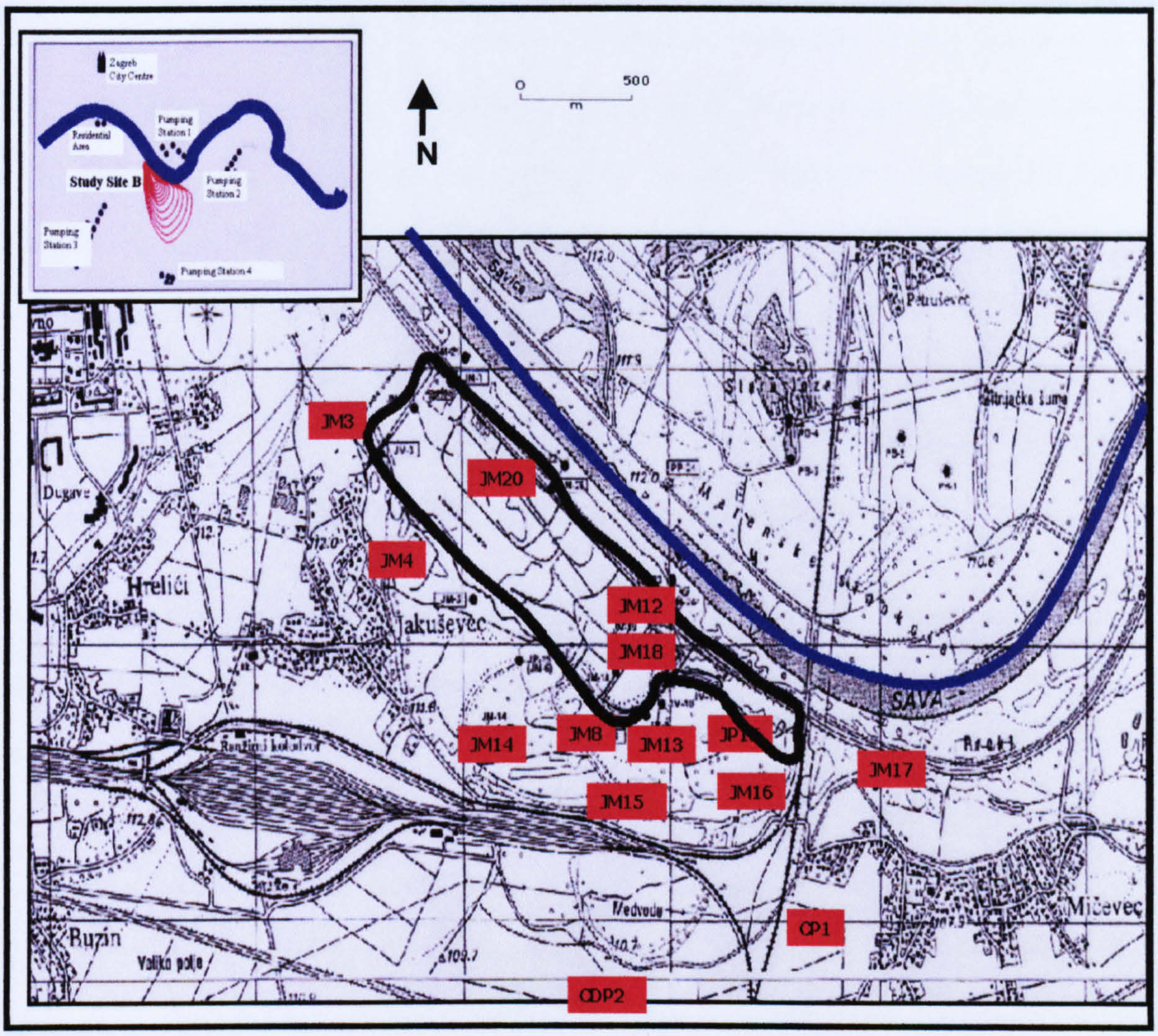
Material	Cited in Olujic (1995)	Cited in Mikac et al, (1998)	Cited in Ahel (1991,1998,1999)
Alluvium Layer Sandy-clay Gravel Layer	Surface: 40–50m depth with increasing thickness when going south east	Surface: 50m depth	Surface: 45–75m depth
Clay Lenses	Below 50m depth	At 50m depth clay lenses are 0.50m wide	Clay lenses at 25–50m depth, unknown location

7.3.3 Hydrologic Framework at Site B

(a) Surface Water Flow

The Sava R., which is only 50 - 200m away from the site, has an average flow of $200m^3/s$. Infiltration into the subsurface is a major source of groundwater recharge, indicating that the river has a substantial impact on the landfill's groundwater and contaminant flow velocities and directions. Infiltration has been measured at $0.7m^3/s$, influent and effluent flow, e.g. Figure 7.8 and 7.9. When the river levels are high, groundwater flow turns south, flowing downstream from the landfill. During low river levels, groundwater flow is redirected toward the river (Mikac *et al*, 1998). During most periods of the year, the Sava remains at mid levels, with groundwater flowing south east. The river levels drop only during dry seasons. As a result, remediation has assumed that the Sava R. will remain at medium to high levels. This assumption however carries a great deal of uncertainty as river levels have dropped by some 5m in recent years due to upstream users.

Figure 7.9 Site plan of Site B, the Sava R. and monitoring piezometers



Legend:
JM = piezometers

(b) Groundwater Flow, Sinks, and Sources

The regional groundwater gradient flows from north west to south east as shown in Figure 7.10. Since groundwater levels fluctuate in sympathy with river levels, the permanent saturated groundwater layer is located at 103m AOD. When Sava R. levels are high, groundwater levels rise to 105m AOD, bringing the water table to 2m below the base of the unremediated landfill site. This fluctuating 2m layer between the landfill base and the water table layer is believed to be a narrow and organically rich layer that contains the highest accumulation of landfill contaminants. Each increase in groundwater levels causes contaminant flushing into subsurface layers in which the Sava R. levels dictate both horizontal rates of dispersivity and strong rates of vertical infiltration.

When the river is at mid levels, groundwater velocity is variable, measuring up to 5m/d in near-surface layers and up to 23m/d in deeper layers. This also affects rates of horizontal and vertical dispersion and infiltration. Under average river levels, groundwater flow and contaminant transport from Site B moves in a south east direction. Groundwater monitoring was conducted over 365 days from February 1995 to February 1996 using spatially distributed piezometers located around the site (Figure 7.9). The groundwater monitoring conducted in 1998 confirmed minimum and maximum groundwater levels. Table 7.5 lists the different values of hydraulic conductivity and transmissivity that have been measured in several studies (Ahel et al, 1998; Mikac et al, 1998; Svel, 1998; Gjetvaj, 1999).

Table 7.5 Hydraulic conductivity and transmissivity values taken from various published studies

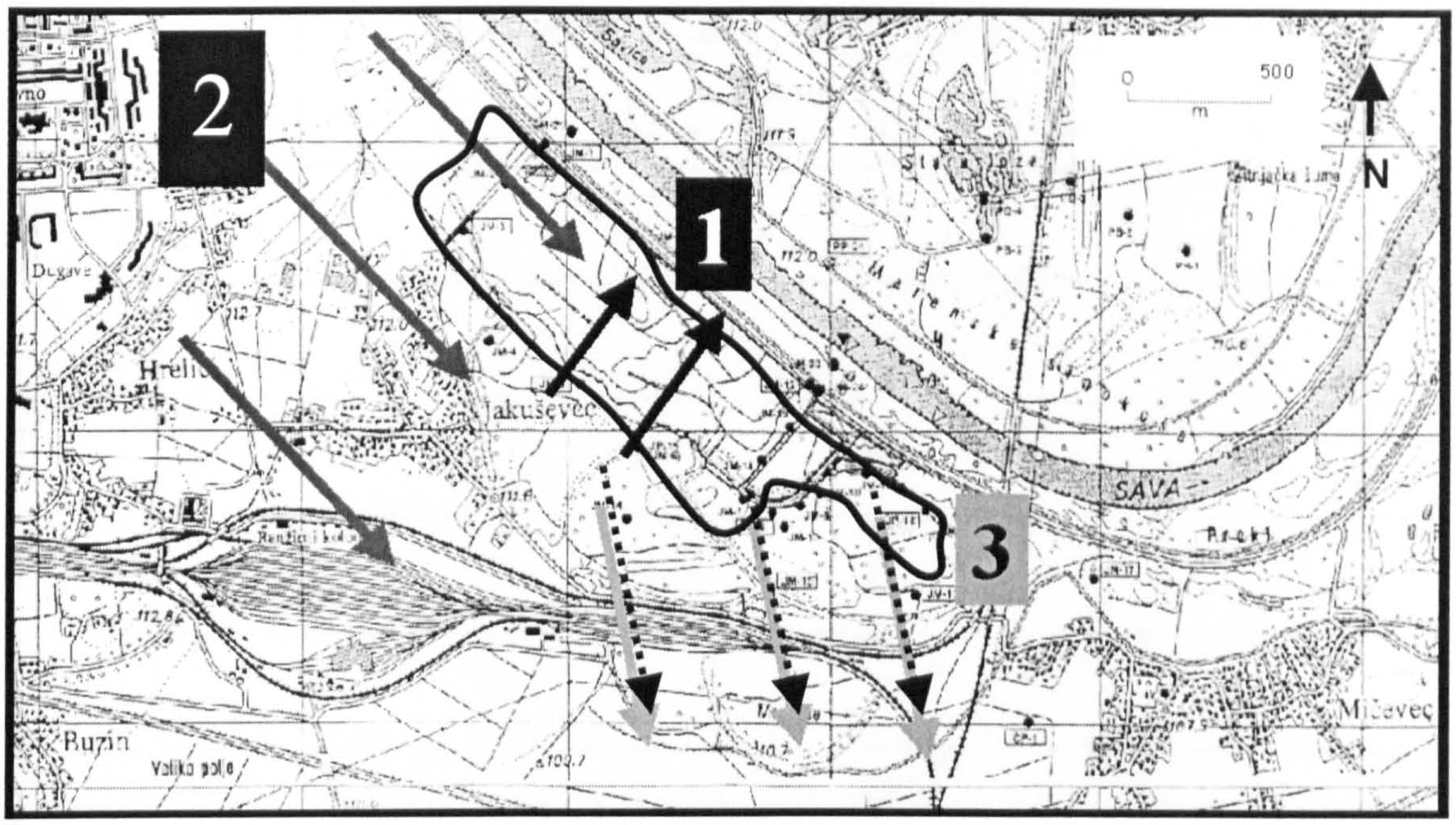
Material	Cited in Ruzic (1992)	Cited in Gjetvaj (1999)	Cite in Svel (1998)
Transmissivity (m^2/sec)	< 0.04 – 0.4	N/A	N/A
Kh (m/s) Alluvium Sand	N/A	Kx & Ky = $1 \times 10e^{-2}$ Kz = $6.7 \times 10e^{-6}$	Kx, Ky, Kz = 0.001
Kh (m/s) Alluvium Gravel	N/A	Kx & Ky = $1.5 \times 10e^{-2}$ Kz = $1 \times 10e^{-3}$	Kx, Ky, Kz = 0.0001 : 2.5×10^{-3} : 8×10^{-3}
Kh (m/s) Clay	N/A	Kx & Ky = $1x10e^{-7}$ Kz = $6.7 \times 10e^{-8}$	Kx, Ky, Kz = $1 \times 10e^{-11}$

Legend:
Kh, Kx, Ky, Kz = m/s
Transmissivity = m^2/d

(c) Landfill Hydraulics

Samples of groundwater, soil and waste were collected during investigations conducted from 1990 through 1998 taken at depths of between 10 and 60m (Figure 7.11). The results show that the contaminant plume is highly variable with concentration and contaminant type with depth. Figure 7.9 shows the piezometer locations. Ammonia is a reliable indicator of site-specific leachate migration although other contaminants such as nitrate, iron, manganese, sulphate, sulphide and methane were also present (Ahel et al, 1998; and Mikac *et al*, 1998). Contaminant concentrations vary with depth. This is shown in Figures 7.11 (a) and (b) indicating that ammonia concentrations in the subsurface are greatly influenced by the heterogeneous hydrogeologic conditions.

Figure 7.10 Description of groundwater flow around Study Site B

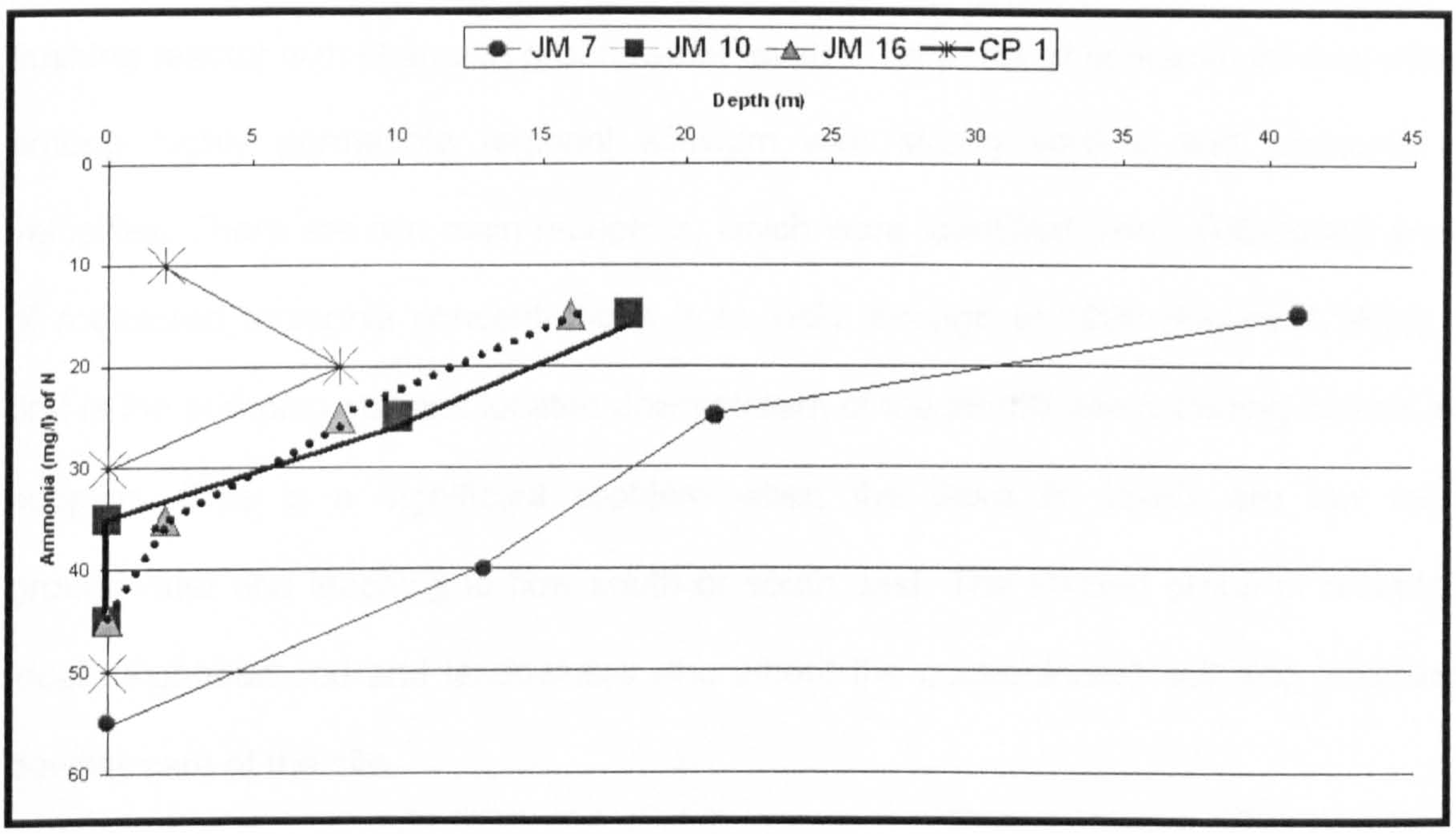


Legend:

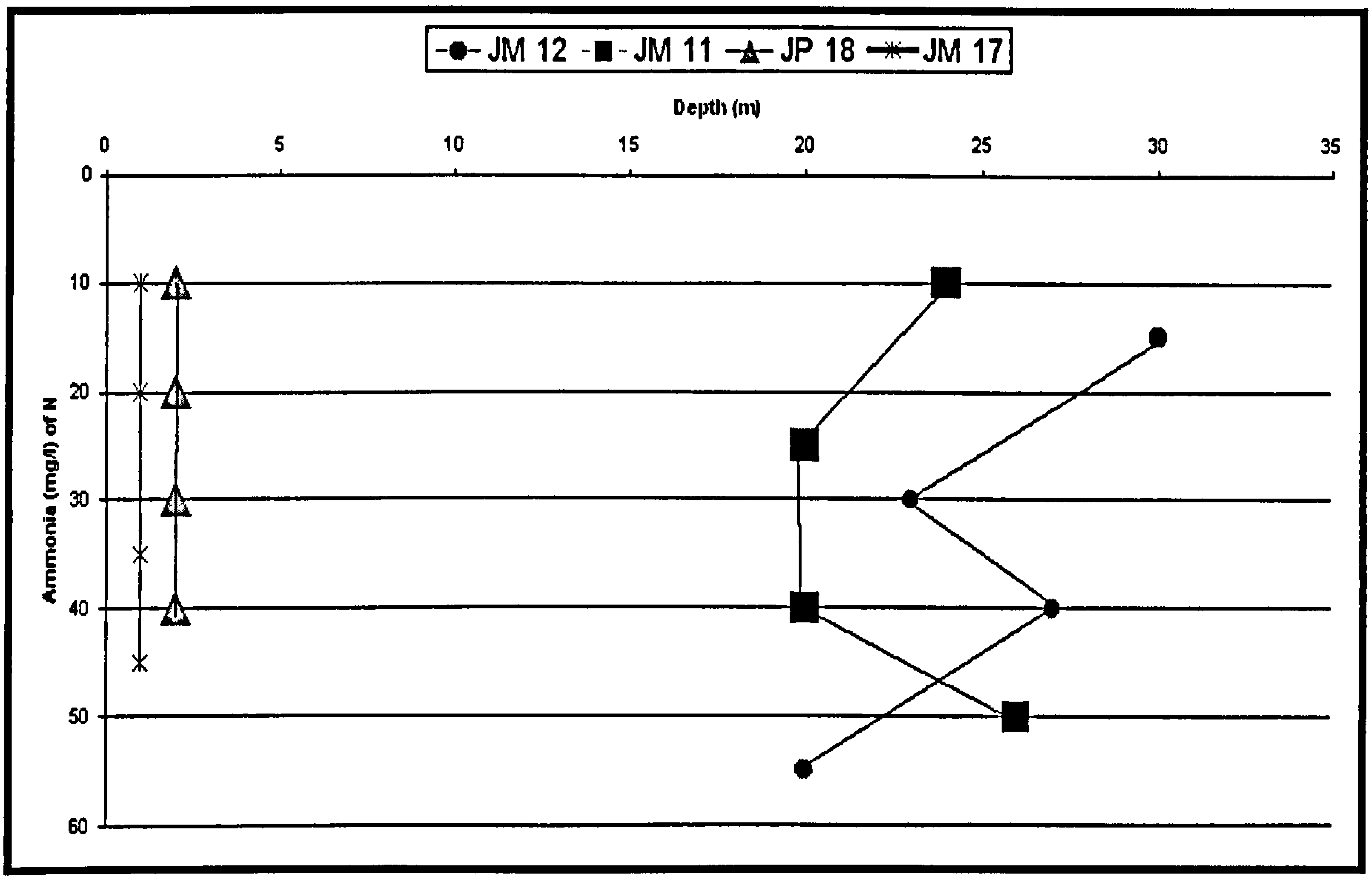
(1) North bound groundwater and contaminant flow when Sava R. is low;
(2) South east bound ground-water and contaminant flow when Sava R. is at mid levels;
(3) South bound groundwater and contaminant flow when Sava R. levels are high

Figure 7.11 Graphs showing leachate concentration variability with depth (adapted from Mikac *et al*, 1998; Ahel *et al*, 1998)

(a) Ammonia concentrations with depth at monitoring points JM 7, JM 10, JP 18 and CP1



(b) Ammonia concentrations with depth at monitoring points JM 12, JM 11, JP 18 and JM 17



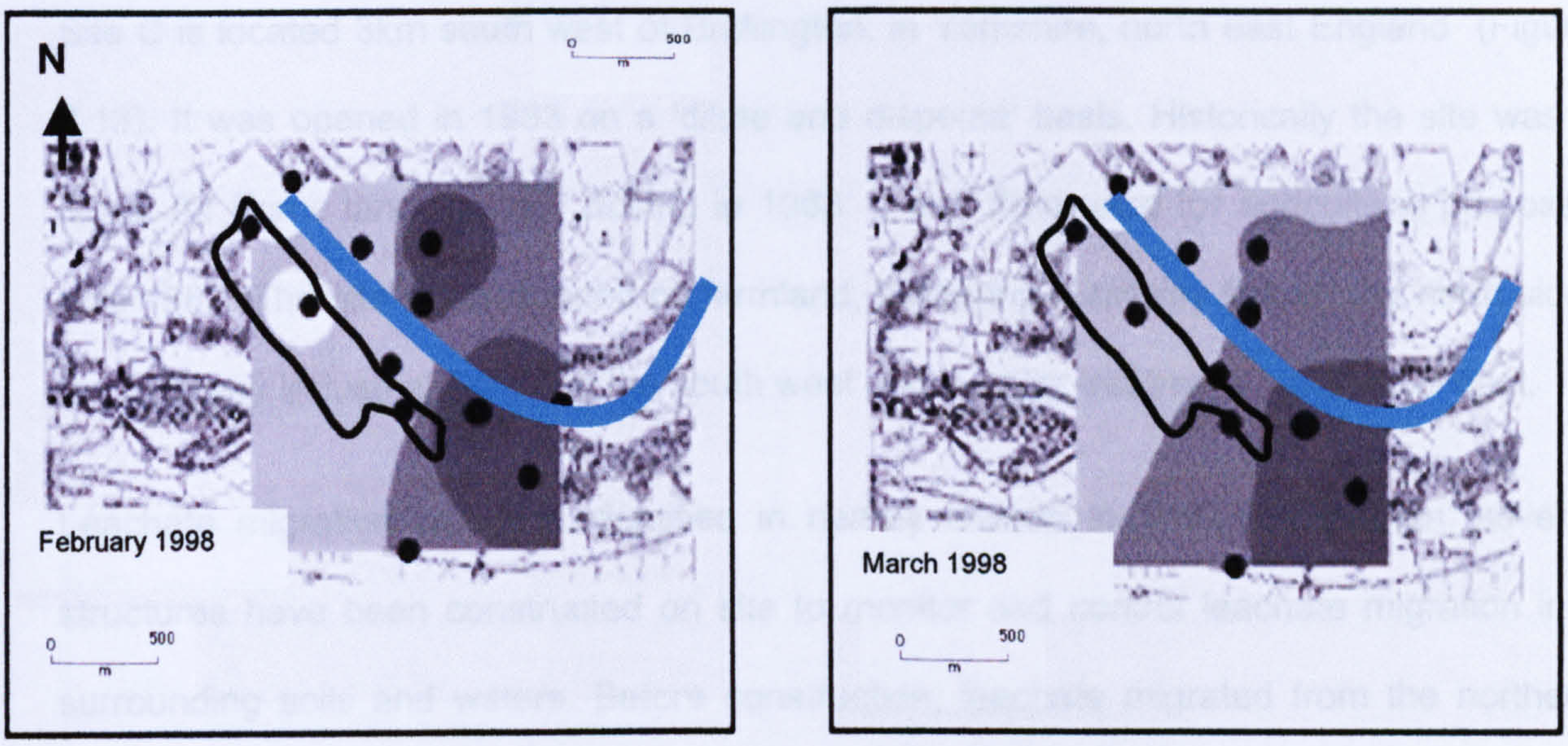
7.3.4 Initial Risk Models: Site B

The site assessment identified that a source, a path and several receptors exist. The source is the landfill and the unsaturated clay lens located beneath the landfill, acting as a flushing reactor with changing groundwater levels. The paths of contaminant migration are among highly permeable regional alluvium with strong vertical and horizontal flow velocities. There are two main receptors, which were identified using GIS-based analysis of measured ammonia concentrations from 1992 through to 1998 (Figure 7.14(b)). The first is the pumping stations located down stream of the landfill site providing current water supplies. This is a significant problem when the Sava R. levels are low causing groundwater and leaching to flow south or south east. The second group of receptors is local neighbourhood and landowners who inherit the contaminated soil and groundwater downstream of the site.

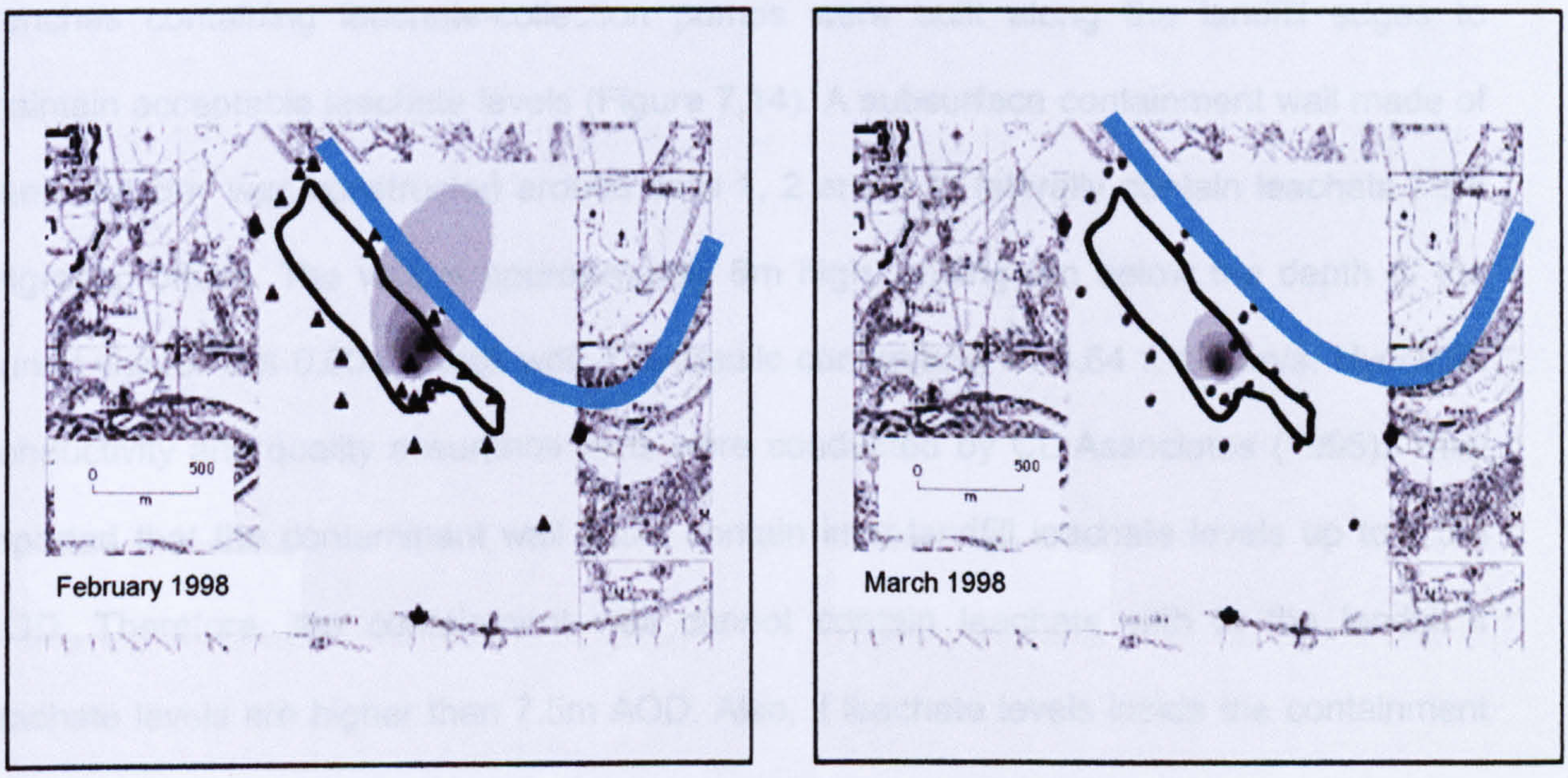
The conditions found at this study site are an example of risks posed by older landfill sites, since the site has been a source of local and regional groundwater contamination for over three decades in which the leachate has contaminated potable water resources. The site's remediation assumes that groundwater and contaminant flow will continue migrating in a south and south eastern direction (Figure 7.14(a)). This is a critical assumption as the river levels have fluctuated and decreased greatly in recent years. If this assumption proves false, then the city's future potable water supplies may be threatened by the landfill's leachate.

Figure 7.12 GIS-based analysis of groundwater levels and ammonia concentrations

(a) Groundwater levels in February and March 1998 (m AOD) showing that in 1998 groundwater levels flowed south and south east putting downstream pumping stations at risk



(b) Ammonia concentrations in February and March 1998 (mg/l of N) showing that concentrations vary monthly, indicating that local hydrological factors play a significant role in contaminant migration from the landfill



Legend:

Black ■ = sample points

= landfill edge

= Sava River

(a) = white – dark grey = 100m - 104m AOD

(b) = white - dark grey = 120 - 450 mg/l of N

7.4 Study Site C

7.4.1 Description of Site C

Site C is located 3km south west of Bridlington, in Yorkshire, north east England (Figure 7.13). It was opened in 1983 on a 'dilute and disperse' basis. Historically the site was a Royal Air Force landing strip, closing in 1963. It was then used for agricultural purposes until 1983. The site is surrounded by farmland, bordering a railway line on the north side, the Carnaby Industrial Estate to the south west and a water treatment plant to the east.

Leachate migration was first identified in nearby ditches in 1992. Since then, several structures have been constructed on site to monitor and control leachate migration into surrounding soils and waters. Before construction, leachate migrated from the northern and eastern perimeters of the site into the surface and groundwater. Remediation of the site occurred between 1994 and 1996 in which drains, boreholes, and subsurface trenches containing leachate-collection pumps were built along the landfill edges to maintain acceptable leachate levels (Figure 7.14). A subsurface containment wall made of bentonite clay was constructed around cells 1, 2 and 3 to laterally contain leachate from migrating offsite. The wall is approximately 6m high, ending 2m below the depth of the buried waste. It is 0.60cm thick with a hydraulic conductivity of 8.64 x 10-4m/s. Hydraulic conductivity and quality assurance tests were conducted by CL Associates (1995). They reported that the contaminant wall could contain inter-landfill leachate levels up to 7.5m AOD. Therefore, the containment wall cannot contain leachate with in the landfill if leachate levels are higher then 7.5m AOD. Also, if leachate levels inside the containment wall are higher then those outside the wall, a hydraulic gradient will be created, resulting in leachate migration across, under, or over the contaminant wall (CL Associates, 1995; Entec, 1996(a)).

A preliminary and detailed study of site conditions was conducted from 1998 through to 2000 at this site. Data collected during this period were used to conduct investigations 2, 4,5 and 6.

Figure 7.13 Set up and location of Site C

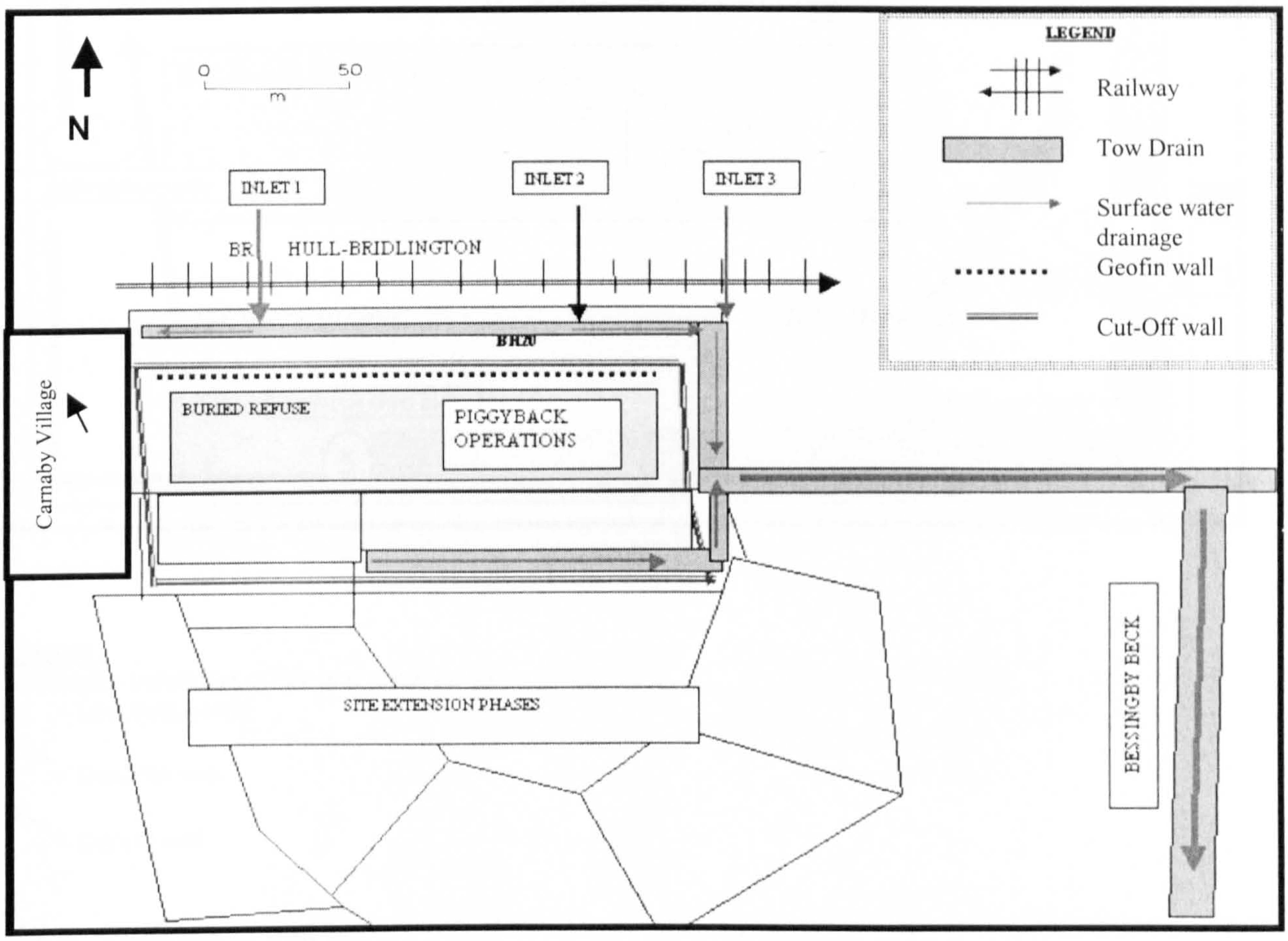
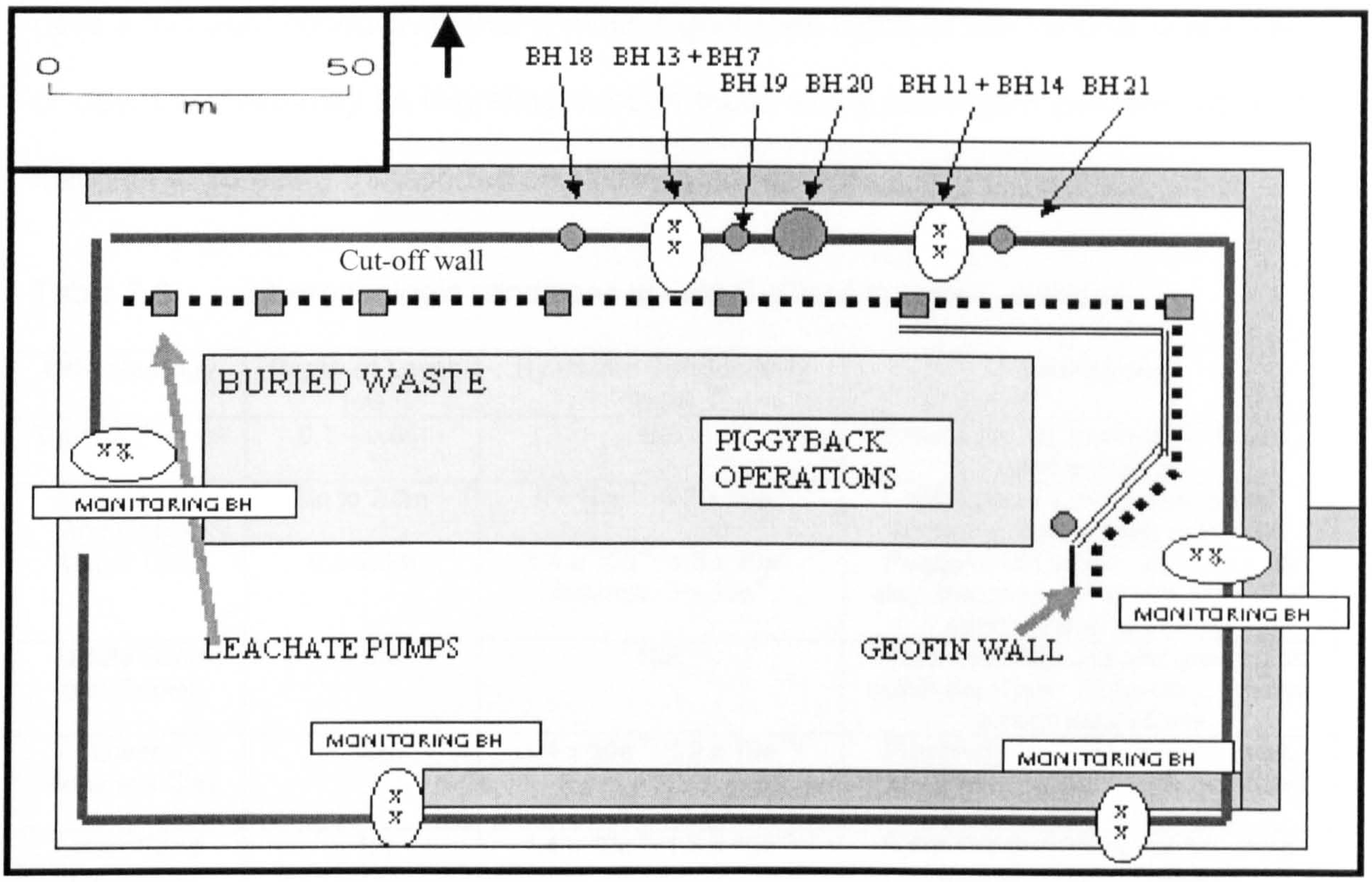


Figure 7.14 Monitoring boreholes and remedial infrastructure at Site C



Legend:
Monitoring boreholes = 'Bh' and show as 'xx'
□ = Leachate pump
= Geophin wall
= Cut-off wall

7.4.2 Geology at Site C

The region has a low-lying topography ranging from 7-10m AOD. Site topography ranges from 7-16m AOD with maximum heights in cells 1,2 and 3 where current landfill operations are taking place (Figure 7.13). The geologic sequence is shown in Table 7.6. Three large lenses are believed to exist on site. Their exact location and size are unconfirmed, however, piezometric monitoring found constant hydraulic head levels throughout the drift which indicates that the lenses are in hydraulic continuity with each other. The piezometric head within the drift varies between 7m AOD in summer to 8.5m AOD in winter. The presence of gravel and sand lenses is of extreme hydrologic importance. These materials have a hydraulic conductivity that is much higher then those of surrounding Quaternary Till or clay. Leachate may be migrating through these easily permeable patches, allowing the leachate to be easily transported off site by groundwater.

Table 7.6 Hydrogeologic conditions at Site C (Cited in Entec, 1996(a))

Drift Geology	Depth of Layer (m)	Hydraulic Conductivity (m/d)	Description
Topsoil/ Subsoil	0.1 – 0.6m	N/A	Brown clay, occasionally flinty soil, peat in places
Upper Sand and Gravel	Up to 2.0m	$5 \times 10e^{-3}$: $6.8 \times 10e^{-3}$	Silty/clayey sand or flint gravel occurring across most of the site
Upper Clay	0.3-2.6m	$1.4 \times 10e^{-5}$: $1.3 \times 10e^{-4}$ Average: $3 \times 10e^{-5}$	Purple or red brown, stoneless silt clay. Becomes stony with increased sand content and depth
Middle Sand and Gravel	<1.0m	N/A	Water bearing sand and gravel lens within the Upper Stony Clay, present across most of site
Lower Stoneless Clay	~ 1.5m	$1.4 \times 10e^{-5}$: $1.3 \times 10e^{-4}$	Purple or red brown, stoneless silt clay. Becomes stony with increased sand content and depth
Lower Sand and Gravel	~ 1m	$1.4 \times 10e^{-5}$: $1.3 \times 10e^{-4}$	Sand and gravel deposit occurring mainly in north, probably discontinuous lenses
Lower Stony Clay	0.9 – 11.8m	$1.4 \times 10e^{-5}$: $1.3 \times 10e^{-4}$	Purple or red brown, stoneless silt clay. Becomes stony with increased sand content and depth
Blue or Chalky Clay		N/A	Grey green to white clay cobbles and flints. Weathered chalk
Chalk	–7m AOD to –9m AOD	N/A	Transition from chalky clay to chalk bedrock

7.4.3 Hydrologic Framework at Site C

(a) Surface Water Flow

The site C is located in the Bessingby Beck surface water catchment. Three ditches (the north and south ditches lead into the eastern ditch) and four sewage outlets feed into this catchment. The western ditch (shown in Figure 7.13), drains into underground drainage networks. The eastern channel leads to Moor Lane that drains into the Bessingby Beck, draining into Auburn Beck. These drainage systems are assumed to absorb much of the infiltration and surface drainage due to high levels of regional water tables. The Auburn Beck drains into Bridlington Bay at Auburn Sands and Fraithsthorpe Beach. These beaches are potentially threatened by upstream drainage as they are designated bathing beaches meeting EC Bathing Water Directives (Entec, 1996(a)).

(b) Groundwater Flow, Sinks, and Sources

The regional aquifer is the Flamborough Chalk with a hydraulic gradient of 0.004 (Figure 7.15). Clay bands confine chalk groundwater levels. Overlying this is Quaternary Till with a hydraulic gradient of 0.0002 (Entec, 1996(a)). Regional rainfall information is listed in Table 7.7.

Table 7.7 Rainfall and Evaporation for Site C (Cited in Entec, 1996(a))

Average precipitation	659.9mm/year (1960 - 1990 average)
Potential evapotranspiration	573.3mm/year
Actual evapotranspiration	497.0mm/year
Effective precipitation	163.0mm/year
Regional Hydraulic Gradient – clay, sand and gravel	0.0002 – shallow gradient toward south east
Regional Hydraulic Gradient – Flamborough chalk	0.004 – slow gradient toward south east

(c) Landfill Hydraulics

The water quality around Site C has been monitored closely since 1992. In recent years leachate migrated south east in the direction of the regional groundwater flow and locally in the direction of the greatest hydraulic gradient. The containment wall on most parts is an effective hydraulic barrier, isolating the landfill from its surroundings (Figure 7.15). Construction of the wall in 1996 improved groundwater quality. Monitoring boreholes on both sides of the wall indicated that the structure is an effective hydraulic barrier except near boreholes 18 and 20, identified in Figure 7.16, where leachate migration continues to occur. These boreholes are located along the northern perimeter of cell 1, on the outer side of the containment wall. They are within a highly permeable sand and gravel lens (depth of 7.25m) indicating that the increased leachate concentrations shown in Figure 7.16 indicates a weak spot in the containment wall. Splajt *et al* (1999) conducted a site assessment modelling leachate migration across the wall showing that the number of leachate pumps and their pumping capacity was below site specific requirements.

7.4.4 Initial Risk Models: Site C

The pollutant linkage for Site C was initially established in 1994, prior to remediation. The source of contamination is the unlined landfill site, the paths of migration are sand lenses through which leachate flows. The receptors are the local waterways that flow downstream, potentially threatening coastal ecosystems, bathing beaches and water quality. Since remediation, the site has experienced small scales of leachate migration impacting local receptors such as the Bessingby Beck, and agricultural fields adjacent to the site.

Figure 7.15 Directions of regional groundwater flow at Site C

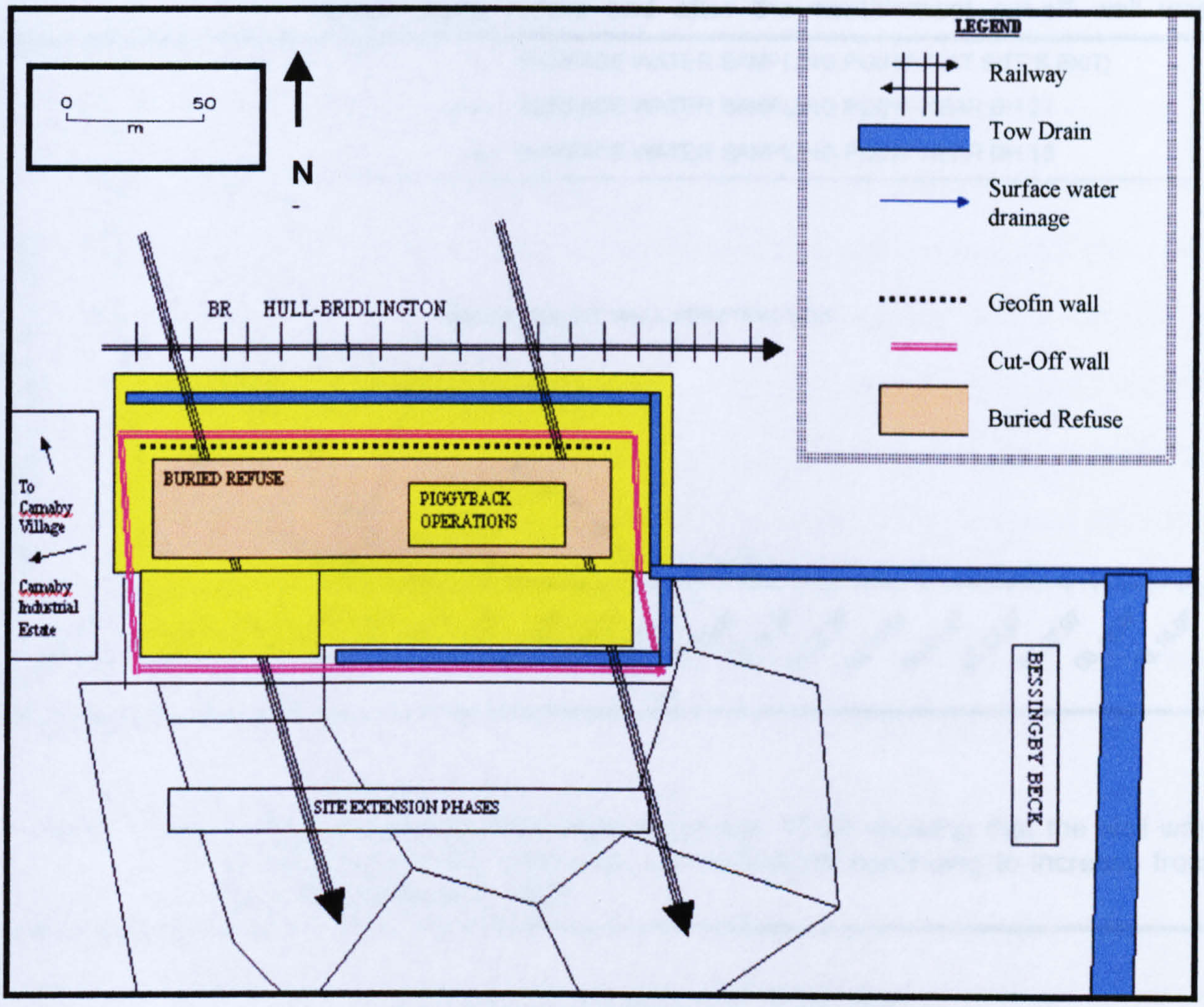


Figure 7.16(a) Leachate concentrations in surface water samples near the landfill edge before and after the containment cut-off wall was

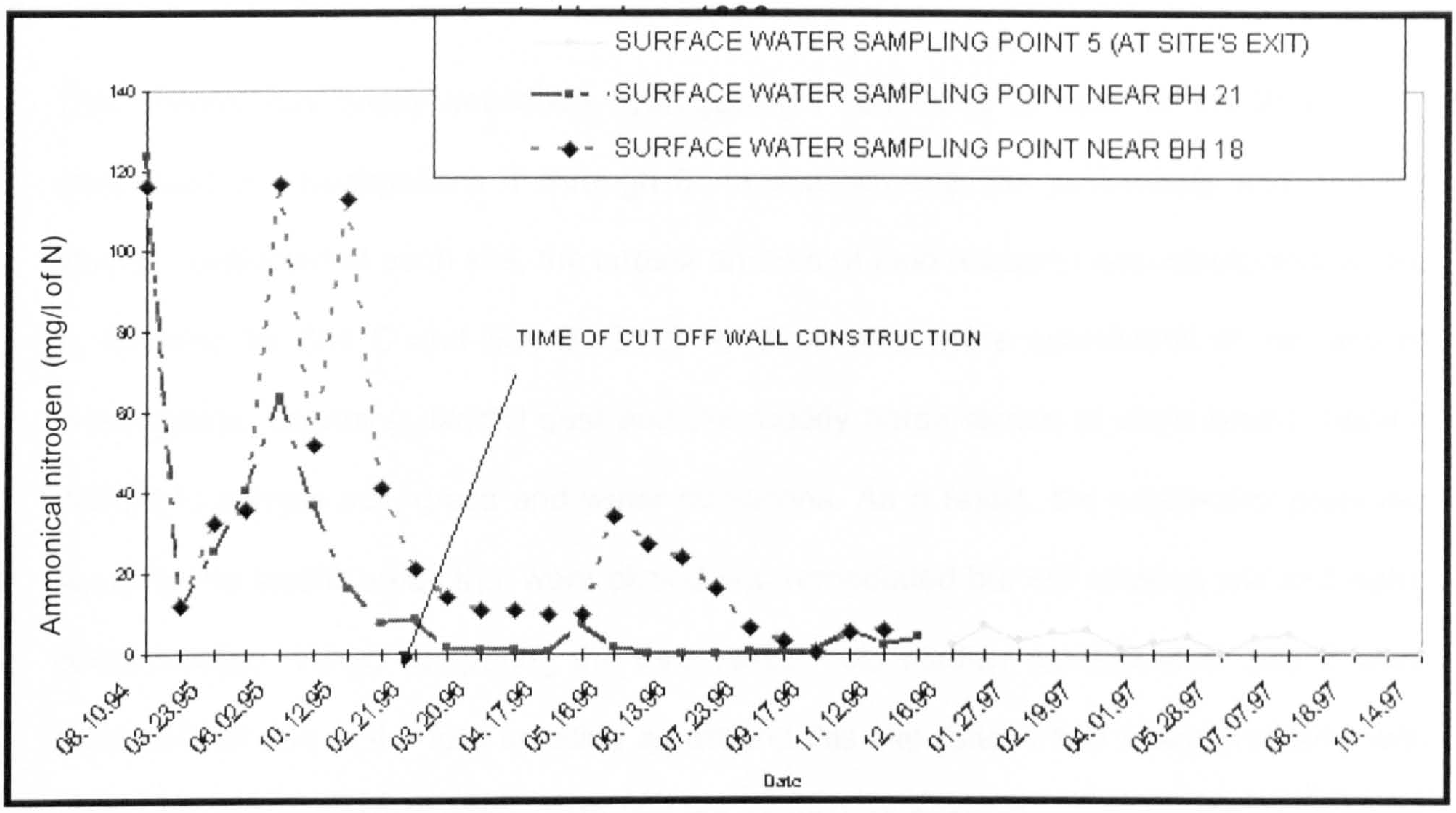
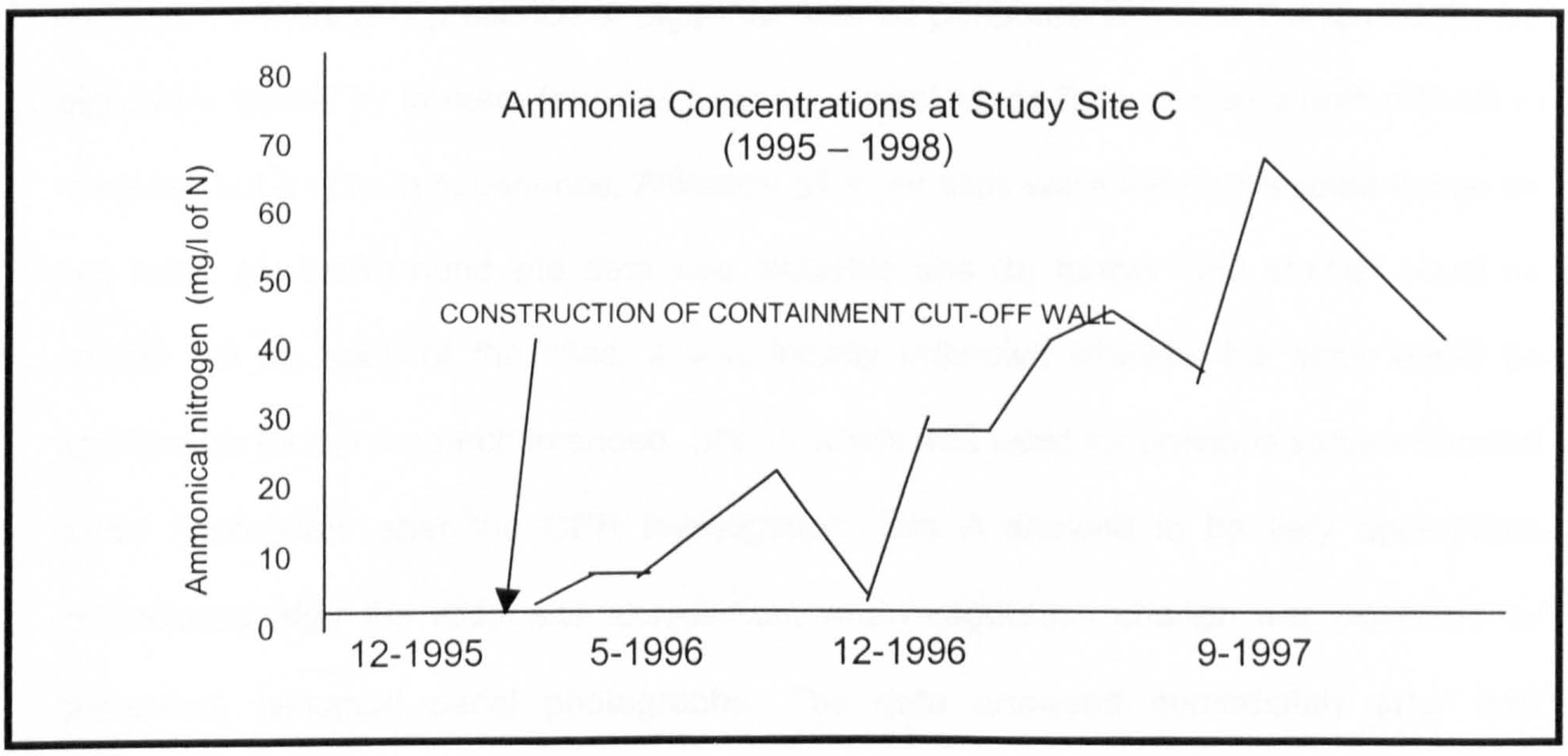


Figure 7.16(b) Ammonia concentrations near boreholes 18-20 showing that the wall was constructed in early 1996, with concentrations continuing to increase from May 1996 through to 1998



7.5 Summary

This chapter has briefly described hydrogeologic conditions at each of the three study sites used in investigations 1 through 6. In summarising the preliminary and detailed studies conducted at each site, the largest amount of field research was conducted at Site A, followed by Site C and Site B. Since all three sites were operational at the time of investigation. Leaching, landfill dust and the muddy harsh terrain at each landfill made it difficult to sample soil, grass and water conditions. As a result, the researcher preferred studying the landfill areas that were closed and remediated but still causing soil and water contamination. When comparing the three sites, field studies conducted at Site C were most difficult due to the fowl smelling air around this site. Site A had fewer problems with air quality but near-surface leachate leaking as well as excessive surface drainage made access to the site's study plots very difficult due to muddy conditions. Site B was the cleanest of the three field sites (from an air quality and mud perspective) however the excessive nettles and presence of pigs that roamed peripheral areas of the landfill (these pigs were owned by farmers from the near-by village) made field sampling both difficult to carry out but a unique experience. Although all three sites were initially selected based on two facts: (a) background site data was available and (b) further field studies could be carried out on each of the sites, it was initially unknown whether the sites would be appropriate for the research intended. Site C, which was used for previous studies showed to be appropriate after the GPR investigation. Site A showed to be very appropriate immediately after the initial site assessment when vegetation change was observed by comparing historical aerial photographs. The data analysed immediately after field collection using the GPR and field-based spectrometer again confirmed that Sites A and C were appropriate for the research. As the project developed, there was a need to find an additional site that would have an appropriate amount of historical data that could be used in geostatistical modelling. Access to data collected for Site B was discovered by coincidence, in networking with other landfill leachate researchers. Although it took about 18 months to collect all the necessary data required for the Site B models; the modelling results also showed that the site was appropriate for the intended research objectives. Figure 7.17 outlines the research objectives and the study sites that were appropriate for each of the objectives.

Figure 7.17: Objectives 1-5 are cross-referenced with Study Sites A, B and C

SSC	X		X	X
SSB		X	X	X
SSA	X	X	X	X
	Objective 1	Objective 2	Objective 3	Objectives 4 & 5

Objectives 1-5 introduced in Chapter 1

Legend:
X = Objectives 1-5 which are tested using data from Study Sites A, B and C
SSA = Study Site A, SSB = Study Site B, SSC = Study Site C
Objective 1 = To test two relatively new multi-spatial field methods that have been tested in other field of science or on other types of contaminated sites;
Objective 2 = To test whether geostatistical modelling could assist in defining site-specific sampling strategies;
Objective 3 = To test the influence of field data when constructing a 3-D groundwater flow and contaminant transport model;
Objective 4 = To test the influence of modelling practises when constructing a groundwater flow and contaminant transport model.
Objective 5 = To test the influence of data assumptions when constructing a groundwater flow and contaminant transport model.

The summary, the aim of this chapter was to provide the reader with a conceptual model of each study site in order to better understand the hydrogeologic and landfill conditions that were assessed using innovative methods in investigations 1, 2 and 3 and modelled in investigations 4, 5 and 6. The results of these 6 investigations are found in Chapters 8 and 9.

CHAPTER 8: NEW SITE ASSESSMENT METHODS - RESULTS

8.1 Introduction

The following two chapters present the findings and results of investigations 1 through 6. This chapter presents the results of investigations 1, 2 and 3 relating them to the first two research objectives that evaluate whether the field data derived from innovative field and geostatistical methods are useful in risk assessment of contaminating landfill sites. Results of kriged groundwater models are presented for investigation 1. Investigations 2 and 3 use ground penetrating radars and remote sensing methods to produce GIS maps of contaminant conditions around a landfill site. Chapter 9 presents modelling results from investigations 4, 5 and 6. These relate to research objectives 1,2, and 3 which evaluate whether (a) the different scales of field data collected during the site assessment and (b) modelling practises and (c) geophysical assumptions during model construction affect model simulations and influence the accuracy of assessing risks at contaminated landfill sites.

Since the results all relate to the risks posed by landfill sites, there are several terms that are used throughout chapters 8 and 9, needing clarification in context of the results. First is the term 'spatial data sets' in which spatial refers to three-dimensional information about site conditions. The second term is 'model' used in investigations 1, 4, 5 and 6. In investigation 1 this term refers to kriging models that tested different sampling strategies at Sites A and B. In investigations 4, 5 and 6 the term refers to site-specific groundwater flow models that simulate contaminant transport. The third term frequently used is 'scenario'. For investigation 1 this referred to the different sample patterns that were tested using kriging models. For investigations 4, 5 and 6 the term referred to different hydrogeological situations set-up in the model as well as the different parameters and circumstances that were being evaluated.

8.2 Investigation 1: Kriging

As discussed in sections 5.8.2 and 5.8.3, kriging can assist in both the site assessment and risk estimation process. It can be used to locate new sampling locations and identify the distance needed between samples to make the distribution of sample points representative.

Investigation 1 used 'Ordinary' kriging to model groundwater levels at Study Site A and Study Site B. Two data sets for winter and summer groundwater levels were used for each site. (Site A used March and August 1998 data while Site B used February and September 1998 data). The aim was to test whether kriging could identify optimal sample locations over a given area. Four models were produced for each data set, evaluating data sets that used twelve sample points, seventeen sample points, twenty-one sample points and twenty-nine sample pionts.

8.3 Investigation 1: Kriging using Study Site A

8.3.1 Background

Site A is a municipal landfill site in north east England. Kriging was considered an appropriate approach because this site originally had only eight sampling points from which the site assessment could infer groundwater quality and form a conceptual model of the site's hydrogeology. These points were used as the initial points from which kriging analyses were conducted.

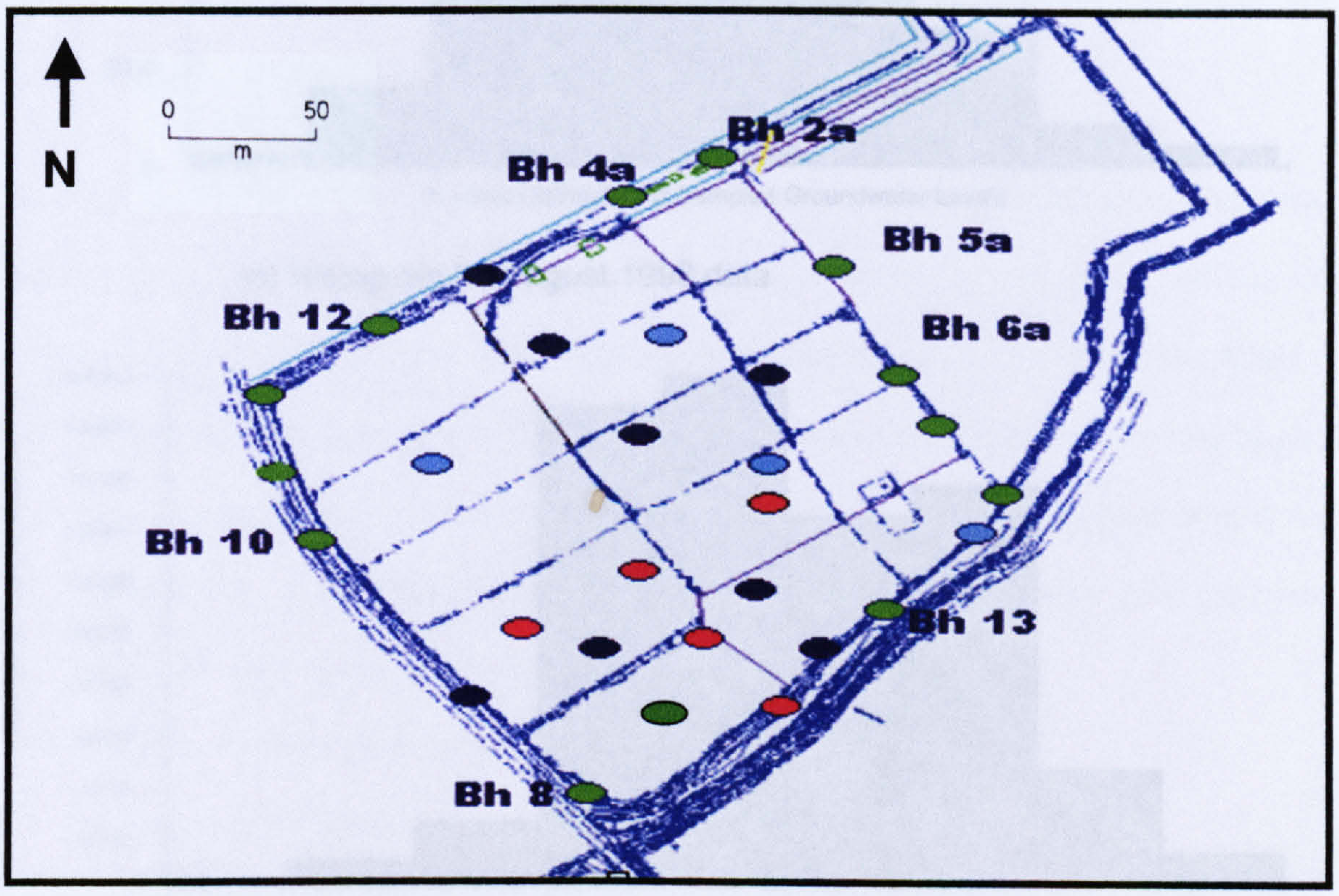
8.3.2 Introduction: Investigation 1 at Study Site A

Groundwater levels detected during winter (March 1998) and summer (August 1998) months (shown in Table 8.1) were used to construct kriging models using ordinary kriging. A site map of sample points describing the kriged sampling locations and model scenarios is shown in Figure 8.1. The first step was to characterise the spatial continuity of sampling using histograms and statistical analysis shown in Figure 8.2 and Table 8.2.

Table 8.1 Piezometer groundwater levels (m AOD) from Site A used for kriging analysis

	Water Levels	(m AOD)
Bh	**March 1998**	**August 1998**
2a	0.97	1.65
4a	0.88	1.3
5a	0.41	0.4
6a	0.02	0.10
8	3.53	3.52
10	2.69	2.85
12	2.69	2.85
13	0.97	2.8

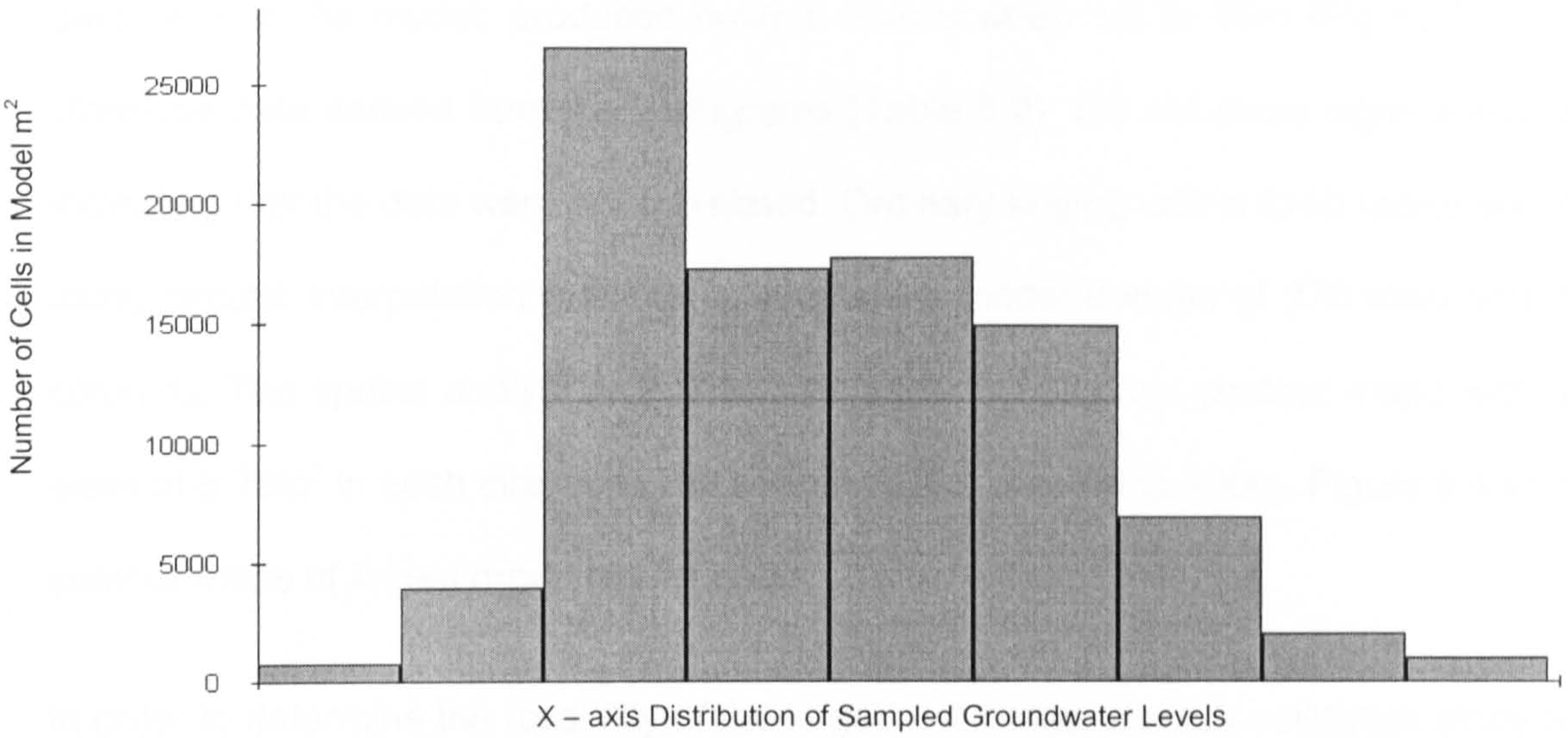
Figure 8.1 Measured and kriged groundwater sampling points around Site A



Legend:
Green = scenario 1 - 12 sample points
Blue = scenario 2 - 17 sample points
Red = scenario 3 - 21 sample points
Black = scenario 4 - 29 sample points

Figure 8.2 Histograms of groundwater levels measured across Site A in (a) March 1998 and (b) August 1998 showing both data sets had a similar distribution

(a) Histogram for March 1998 data



(b) Histogram for August 1998 data

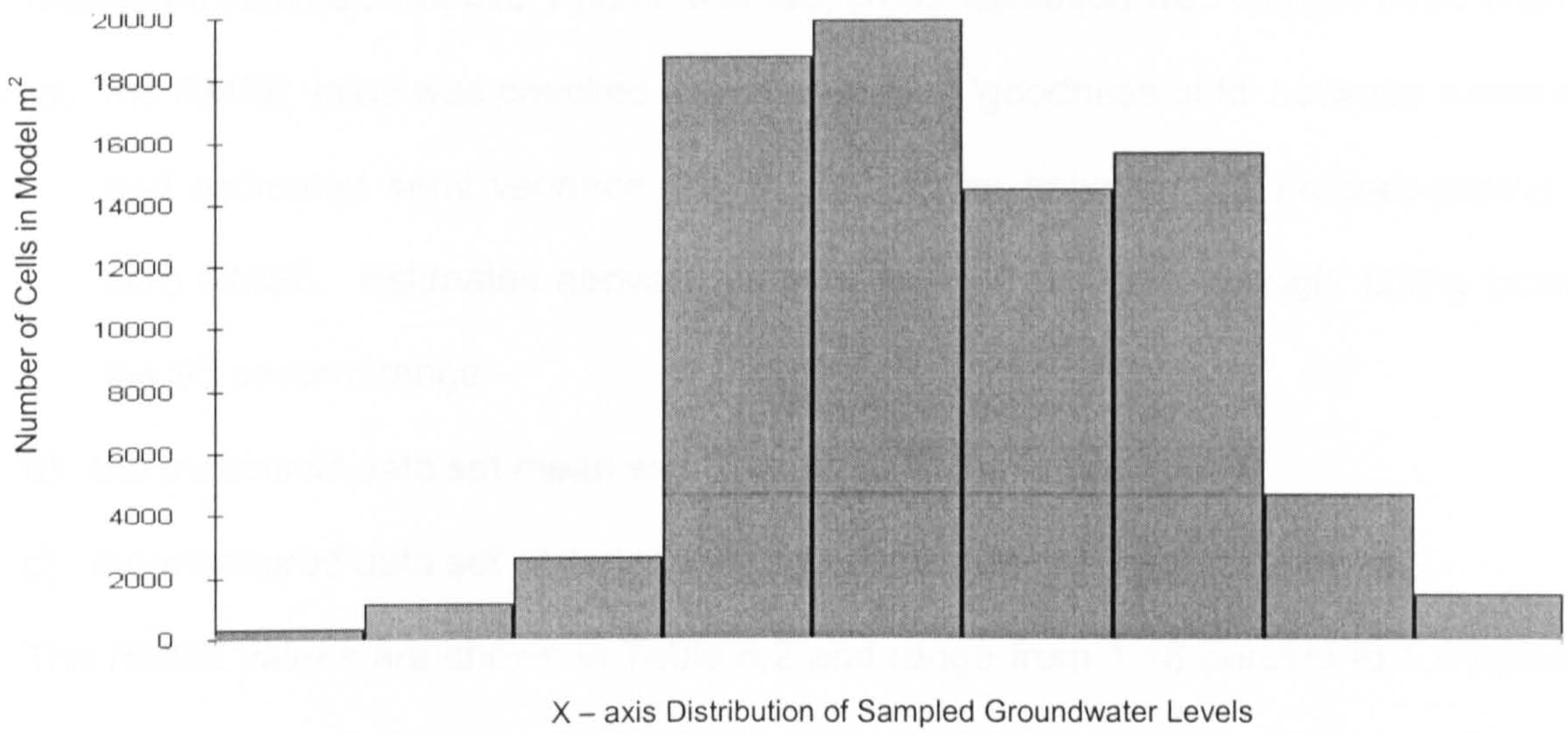


Table 8.2 Field sample statistics and variogram statistics

Statistical Data	Ground Water Level March 1998	Ground Water Level August 1998
Number of sample points	8	8
Mean value of sample points (m)	1.5175	1.92125
Variance of sample points	1.619279	1.64547
Variogram Nugget Effect	0.12	0.3
Variogram Sill (m)	2.0	4
Variogram Range (m)	650	650
Variogram RMSE	1.5	1.16

8.3.3 Methods, Modelling and Data Analysis

Kriging was used to construct variograms for March and August 1998 data sets (ArcView GIS version 3.2). Sensitivity analysis found that the lag distance, as an input parameter in the model, produced optimal results when set to 85m (Figure 8.3). The statistical data derived from the variograms (Table 8.2) did not show signs anisotropy indicating that the data were not correlated. Ordinary kriging with a fixed radius was set using circular interpolation methods to produce a model domain of 276 rows and 250 columns. The spatial analyst extension was used to construct contour maps with grid sizes of $6.75m^2$ in each direction. The search radius was set to 100m. Figure 8.4 shows contour maps of kriged groundwater levels.

In order to determine the reliability of the kriged estimates, a cross-validation study was conducted. It is impossible to check the accuracy of all kriged estimates without the measured field value at every point. Instead, cross-validation was done in three ways:

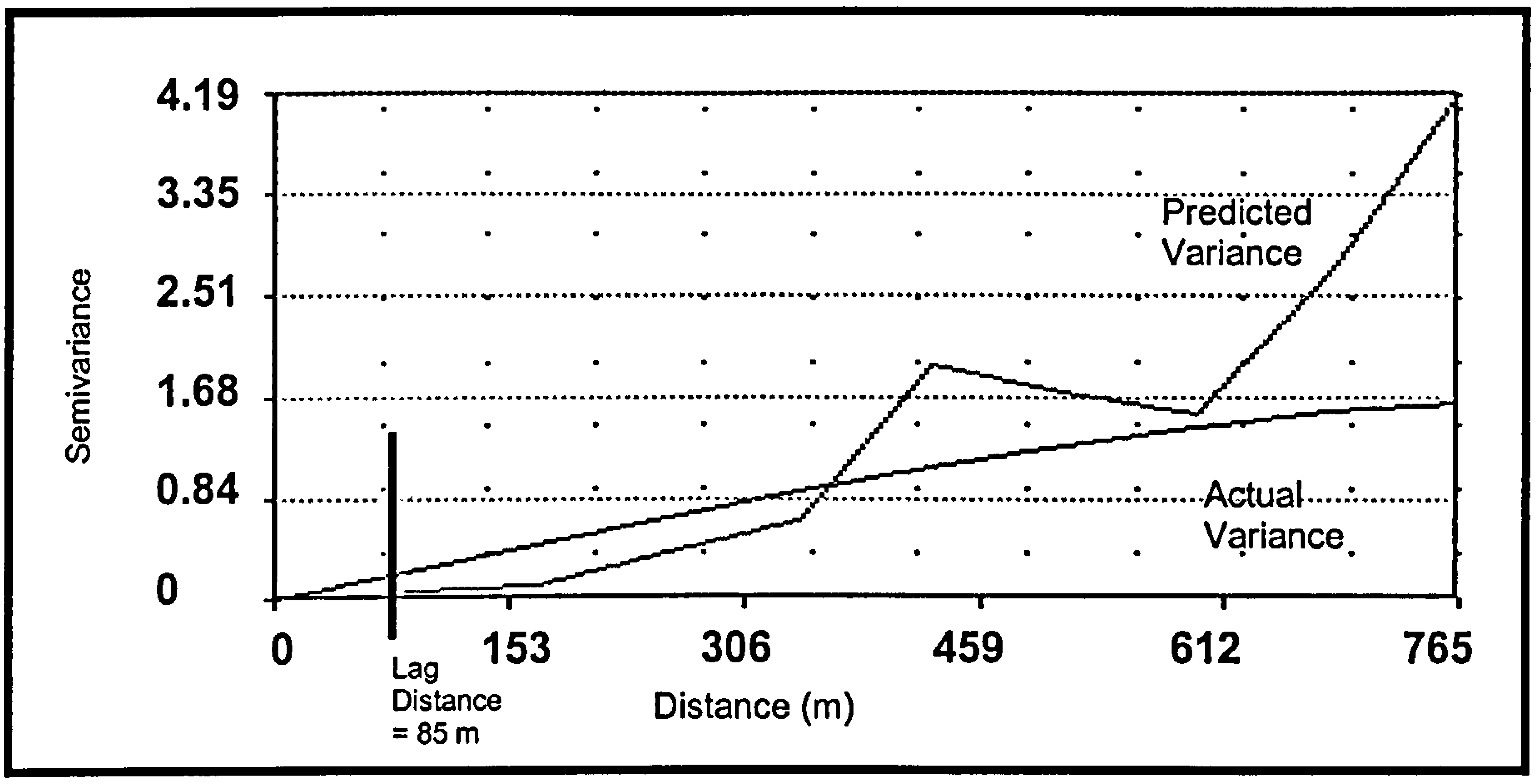
a) the RMSE value was checked as a measure of 'goodness of fit' between measured and estimated semi variance. Values should be between 0-5 percent aiming for zero RMSE. Estimates above 5 percent are not accurate enough, falling outside the 95 percent range
b) the measured data set mean was compared to the kriged mean
c) the measured data set variance was compared with the kriged variance.

The RMSE values are shown in Table 8.2 and range from 1.16 percent to 1.5 percent. Table 8.3 shows that the measured and kriged means were similar. This is confirmed in Figures 8.5 with graphs of measured groundwater levels against kriged groundwater levels producing regression values of $R^2 = 0.99$

Figure 8.3: Variograms produced using groundwater data from (a) March and (b) August 1998 using circular kriging (X-axis = distance in metres, Y-axis = semivariance)

Legend:
X axis = distance in metres, Y axis = semi variance

(a) Variogram of March 1998 data



(b) Variogram of August 1998 data

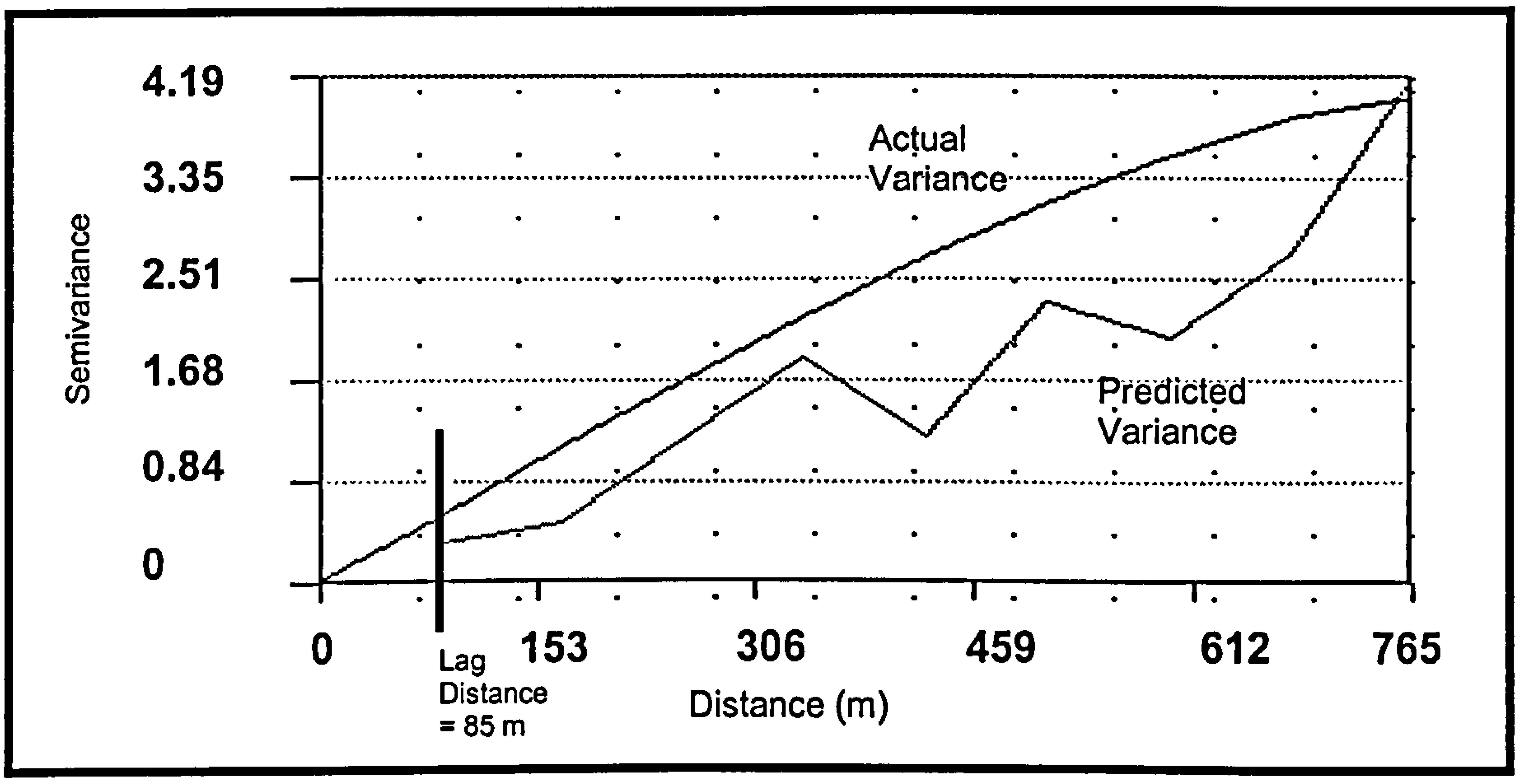
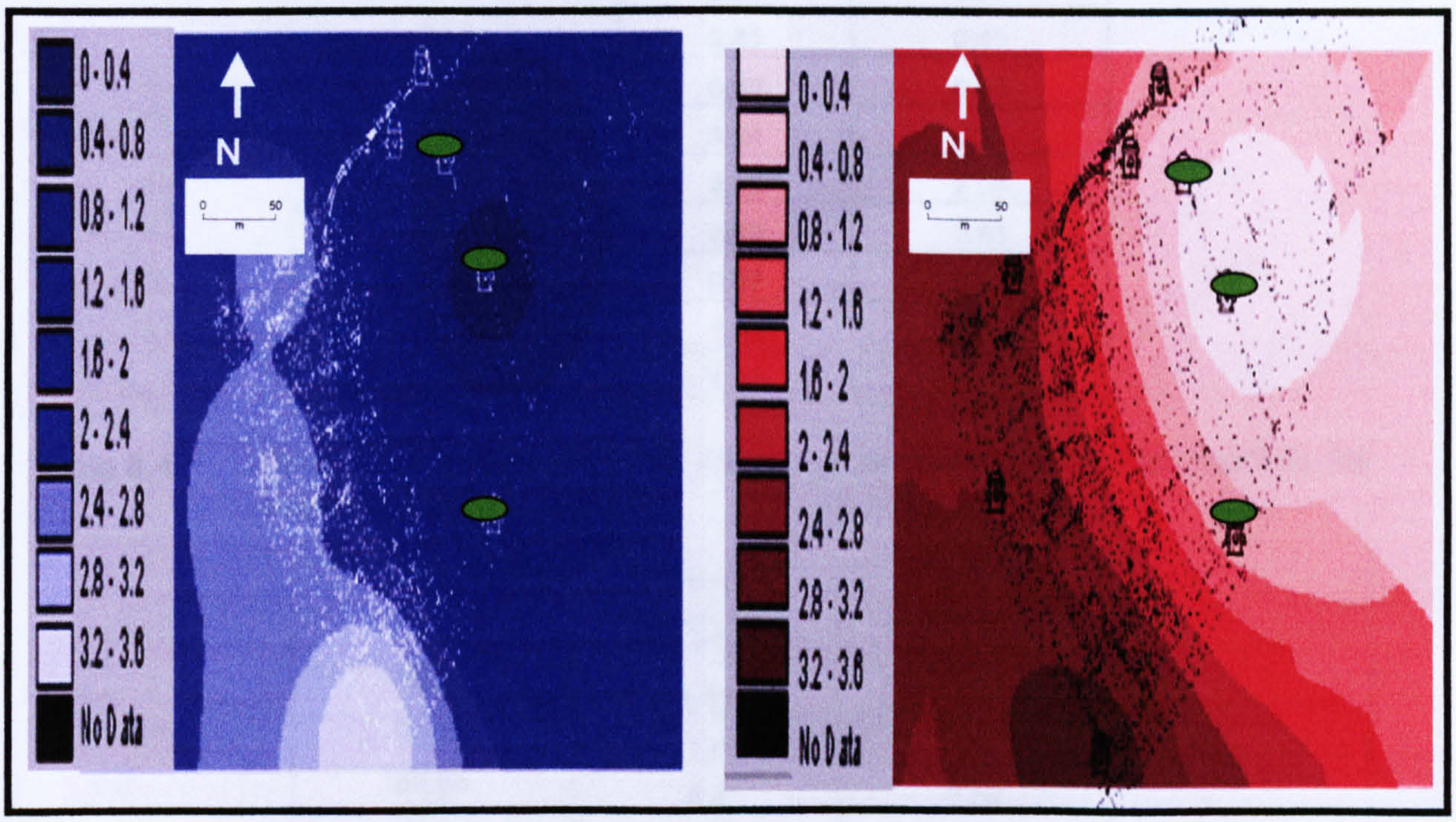
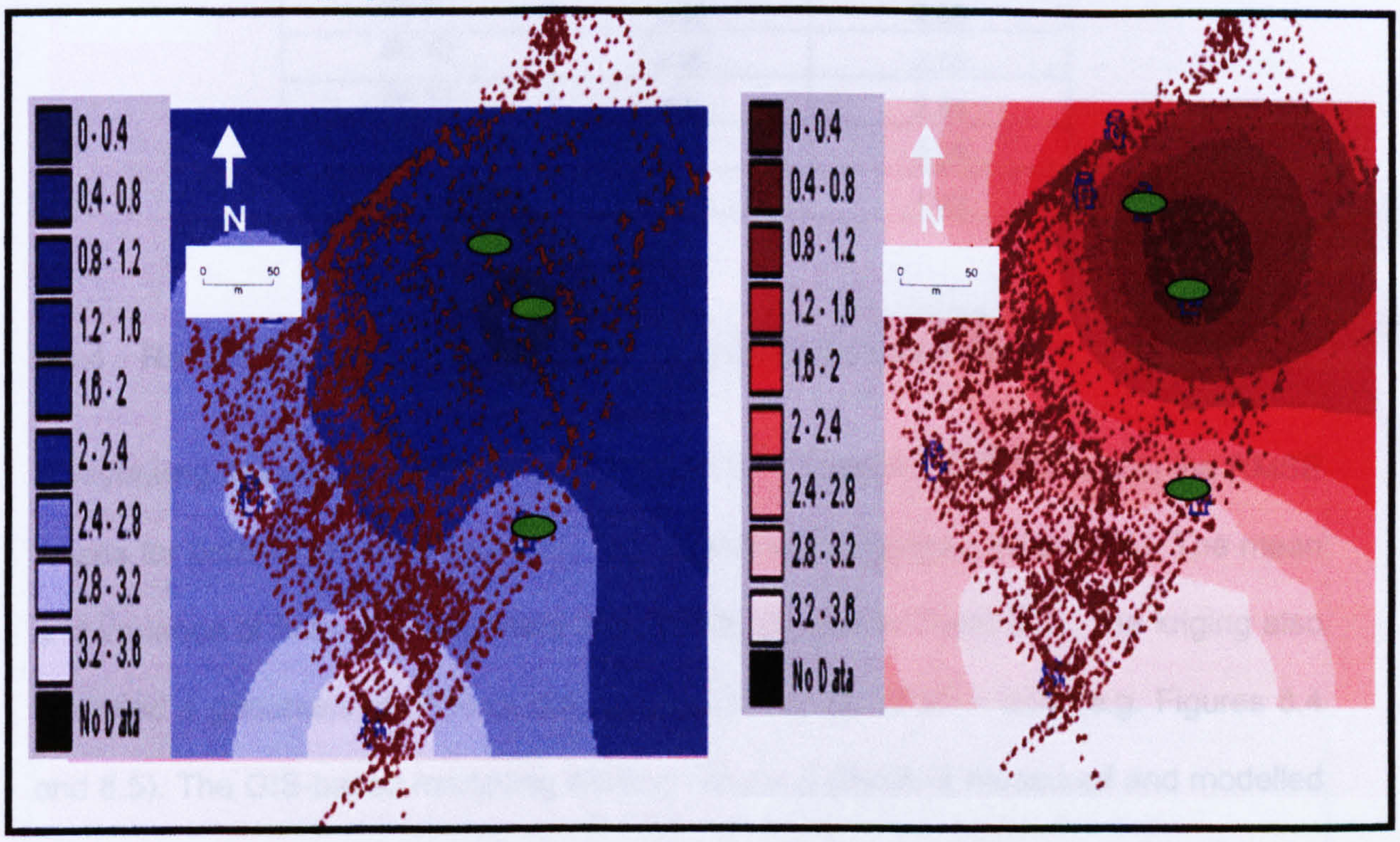


Figure 8.4 Contour maps produced using measured and kriged groundwater levels for (a) March and (b) August 1998 showing that circular kriging effectively simulated groundwater conditions in both periods; (b) August 1998 data – measured and calculated using kriging

(a) March 1998 data – measured (left side) and values calculated using kriging (right side)



(b)August 1998 data – measured (left side) and values calculated using kriging (right side)



Legend:
Colours represent groundwater levels in m AOD
= reference point, representing calibration points 5, 6 and 13

Table 8.3 Comparing measured and kriged groundwater levels (in metres) for March 1998

Bh	Measured March 1998 (m AOD)	Kriged March 1998 (m AOD)
Bh 2a	0.97	0.933
Bh 4a	0.88	0.85
Bh 5a	0.41	0.40
Bh 6a	0.02	0.03
Bh 8	3.53	3.5
Bh 10	2.69	2.68
Bh 12	2.69	2.63
Bh 13	0.97	0.97
Mean (data set)	1.91	1.88
Variance (data set)	1.25	1.23

Table 8.4 Comparing measured and kriged groundwater levels (in metres) for August 1998

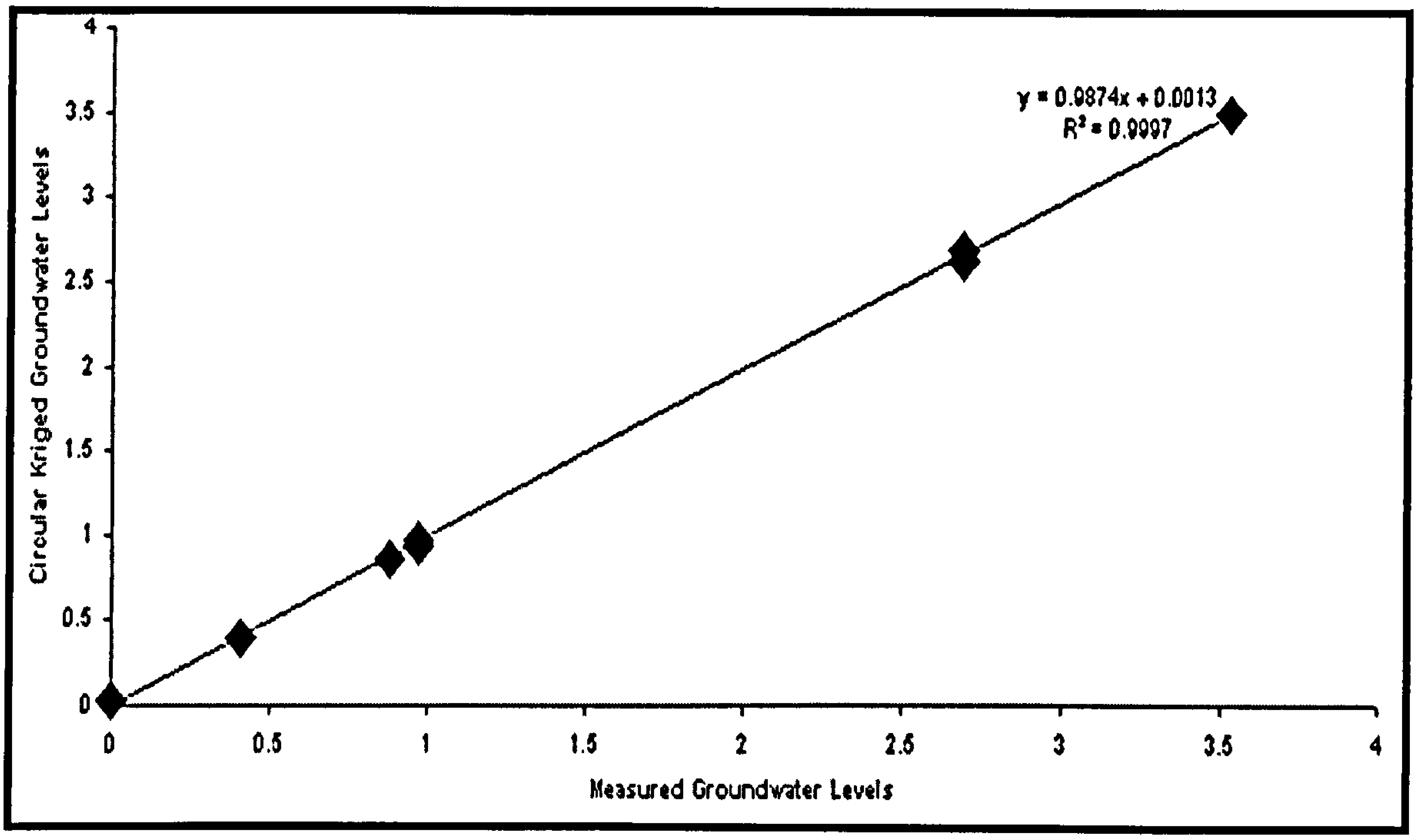
Bh	Measured August 1998 (m AOD)	Kriged August 1998 (m AOD)
Bh 2a	1.65	1.65
Bh 4a	1.3	1
Bh 5a	0.4	0.52
Bh 6a	0.1	0.02
Bh 8	3.52	3.49
Bh 10	2.85	2.85
Bh 12	2.85	2.83
Bh 13	2.8	2.81
Mean (data set)	2.38	2.33
Variance (data set)	1.25	1.28

8.3.4 Results Discussion: Initial Kriging using Eight Sample Points

In reviewing the variogram information (Figure 8.3, Table 8.2 and Table 8.3) the RMSE values for both kriged data sets were within acceptable ranges (below 5%). The mean and variance of both data sets were also similar (shown in Table 8.3). The kriging also exhibited a goodness of fit producing regression curves of R^2 = 0.99 (e.g. Figures 8.4 and 8.5). The GIS-based modelling allowed further analysis of measured and modelled differences to be compared (e.g. Figure 8.6). The two maps show zones of the highest differences between the measured and kriged groundwater levels. The circled areas shown in dark green and dark brown are zones of greatest difference and both are areas of the site where few samples were taken.

Figure 8.5: Kriging exhibited a goodness of fit when graphing measured against modelled groundwater levels: (a) March 1998 and (b) August 1998 data in which R^2= 0.99

(a) March 1998 kriging and measured groundwater levels, R^2=0.9997



(b) August 1998 kriging and measured groundwater levels, R^2=0.9995

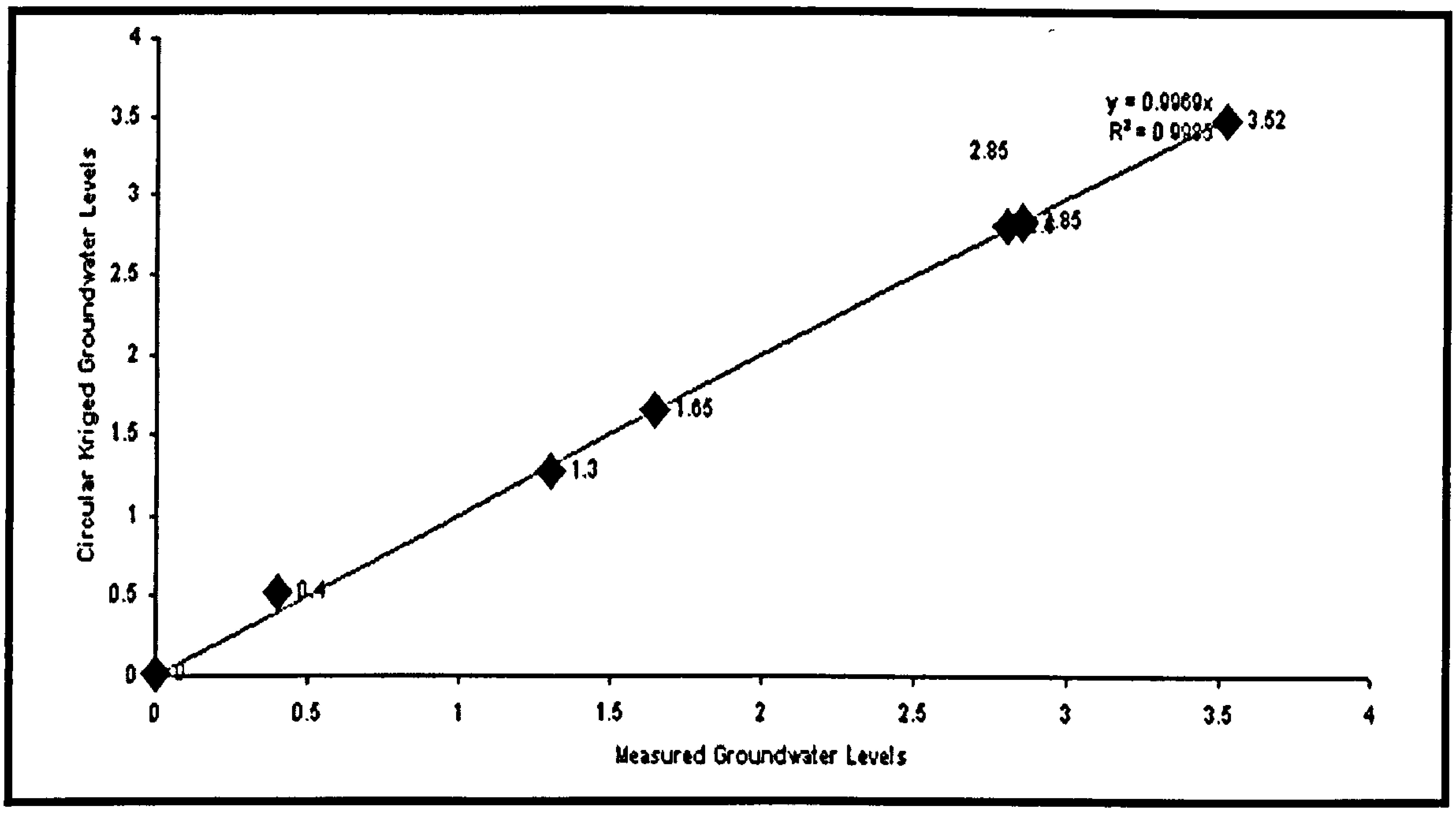
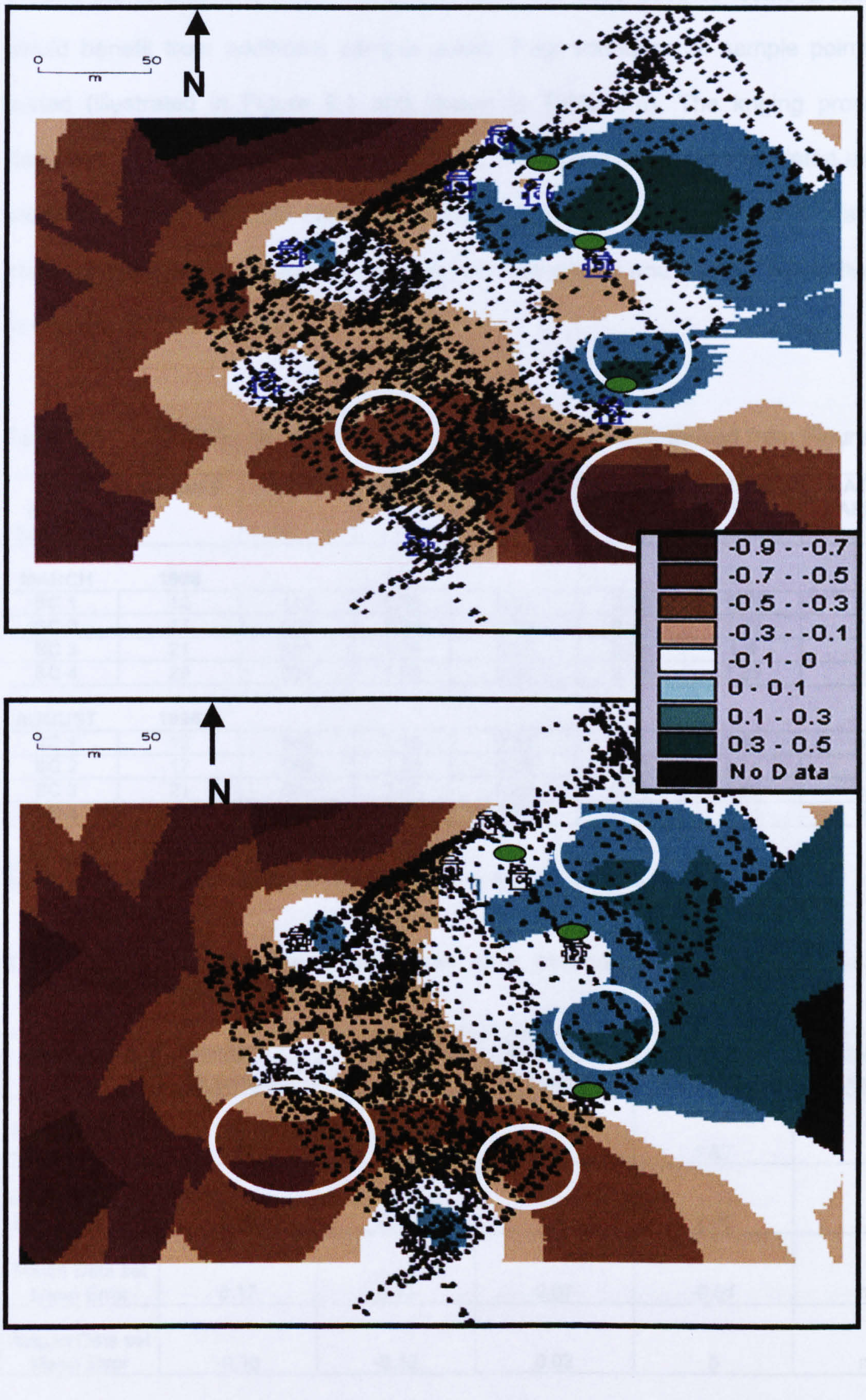


Figure 8.6: Zones of largest differences in (a) March and (b) August 1998 using circular kriging

Legend:
Colours represent differences in kriged and measured groundwater levels in m AOD
= reference point, representing calibration points 5, 6 and 13



8.3.5 Evaluating Sample Numbers and Sample Locations: Scenarios 1-4

The kriging results presented in sections 8.3.3 and 8.3.4 (which used eight sampling points) were used to derive new sampling locations for measuring groundwater levels around the landfill. The maps shown in Figure 8.6 were used to locate areas which would benefit from additional sample points. Four scenarios of sample points were tested (illustrated in Figure 8.1 and shown in Table 8.6). The kriging procedures described in section 8.3.3 were also applied. This investigation was validated using 29 piezometers and borehole samples collected in March and August 1998. Variogram statistics are listed in Tables 8.5 and 8.6 with variograms and contour maps illustrated in Figures 8.7, 8.8 and 8.9.

Table 8.5 RMSE, sill, variance, nugget effect, and range derived from Figure 8.7

M onthly Scenario	Sample #	RANGE	SILL	NUGGET	RMSE percent	DATASET MEAN (m AOD)	DATASET VARIANCE
MARCH	1998						
SC 1	12	410	0.55	0.20	0.95	1.69	0.66
SC 2	17	560	1.30	0.29	0.31	1.73	0.70
SC 3	21	650	1.16	0.20	0.31	1.8	0.65
SC 4	29	620	1.05	0.26	0.11	1.87	0.62
AUGUST	1998						
SC 1	12	560	1.18	0.33	0.5	2.19	0.89
SC 2	17	560	1.25	0.28	0.21	2.14	0.84
SC 3	21	620	1.35	0.25	0.21	1.99	0.67
SC 4	29	620	1.10	0.28	0.13	2.01	0.67

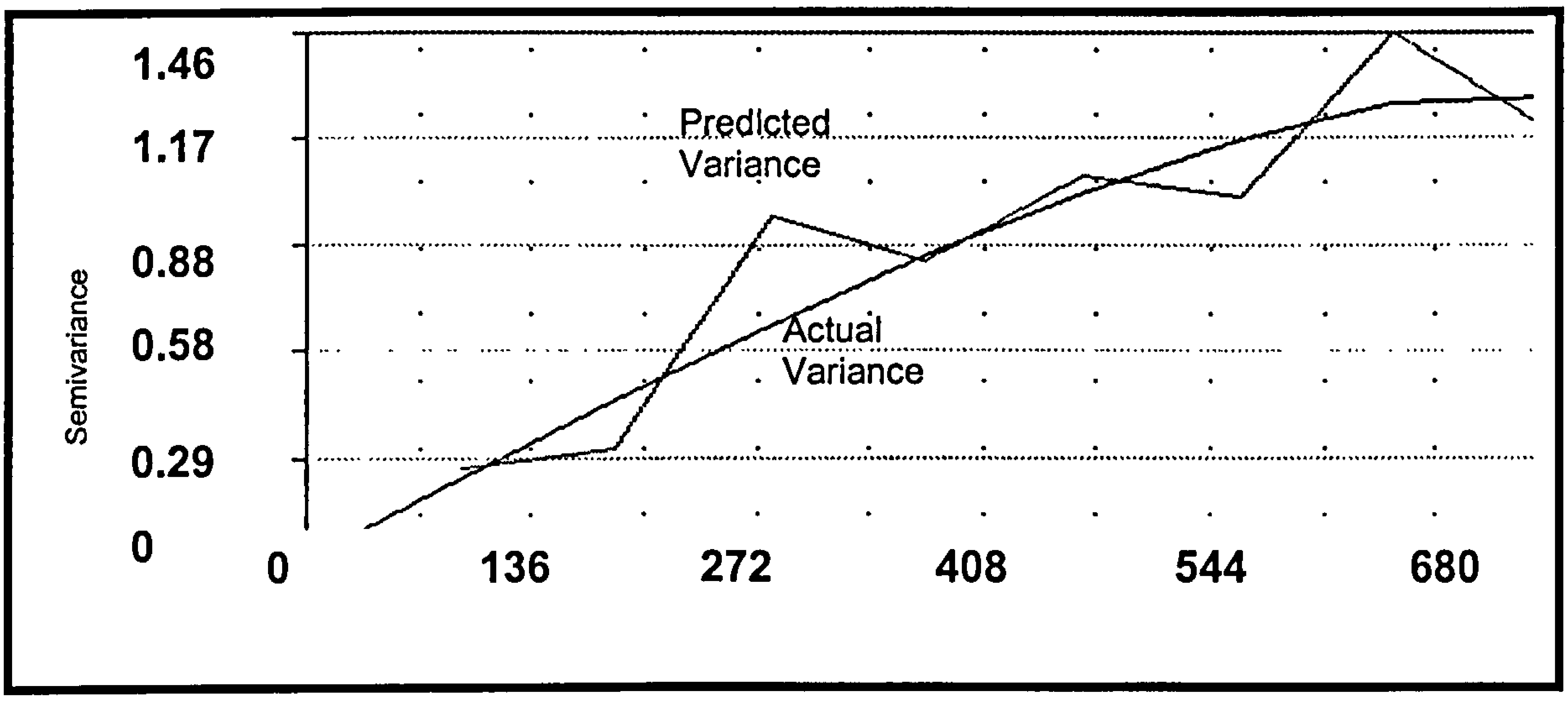
Legend:
SC 1= scenario 1, SC 2= scenario 2, SC 3= scenario 3, SC 4= scenario 4

Table 8.6 Data set mean and mean error between interpolated and measured values for each scenario

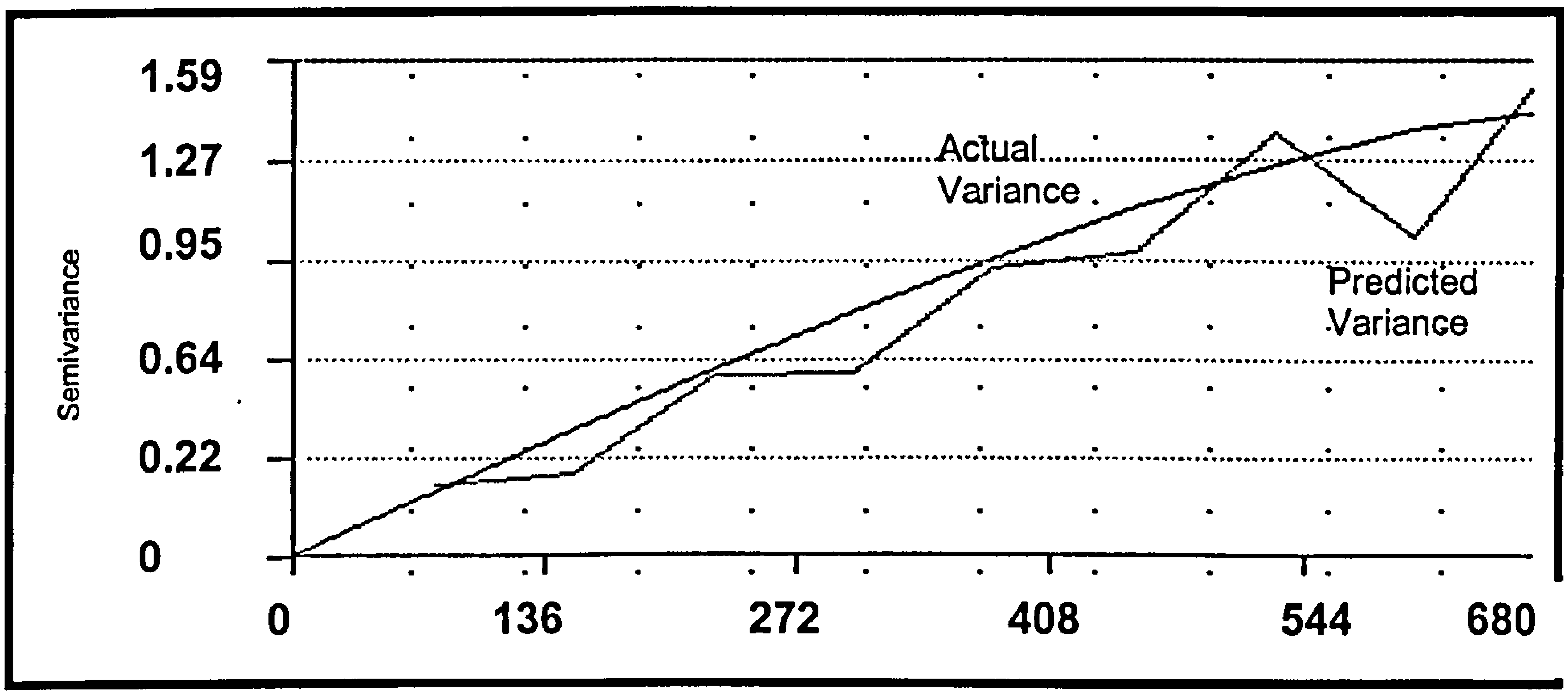
Study Site A	Scenario 1	Scenario 2	Scenario 3	Scenario 4	Measured
	12 samples	17 samples	21 samples	29 samples	29 samples
March Data set Mean (m AOD)	1.69	1.73	1.8	1.87	1.86
August Data set Mean (m AOD)	2.19	2.14	1.99	2.01	2.01
March Data set Mean Error	0.17	0.13	0.07	-0.01	n/a
August Data set Mean Error	-0.18	-0.13	0.02	0	n/a

Figure 8.7 Site A variograms using August 1998 groundwater data: (a) 17 sample points; (b) 21 sample points; and (c) 29 sample points

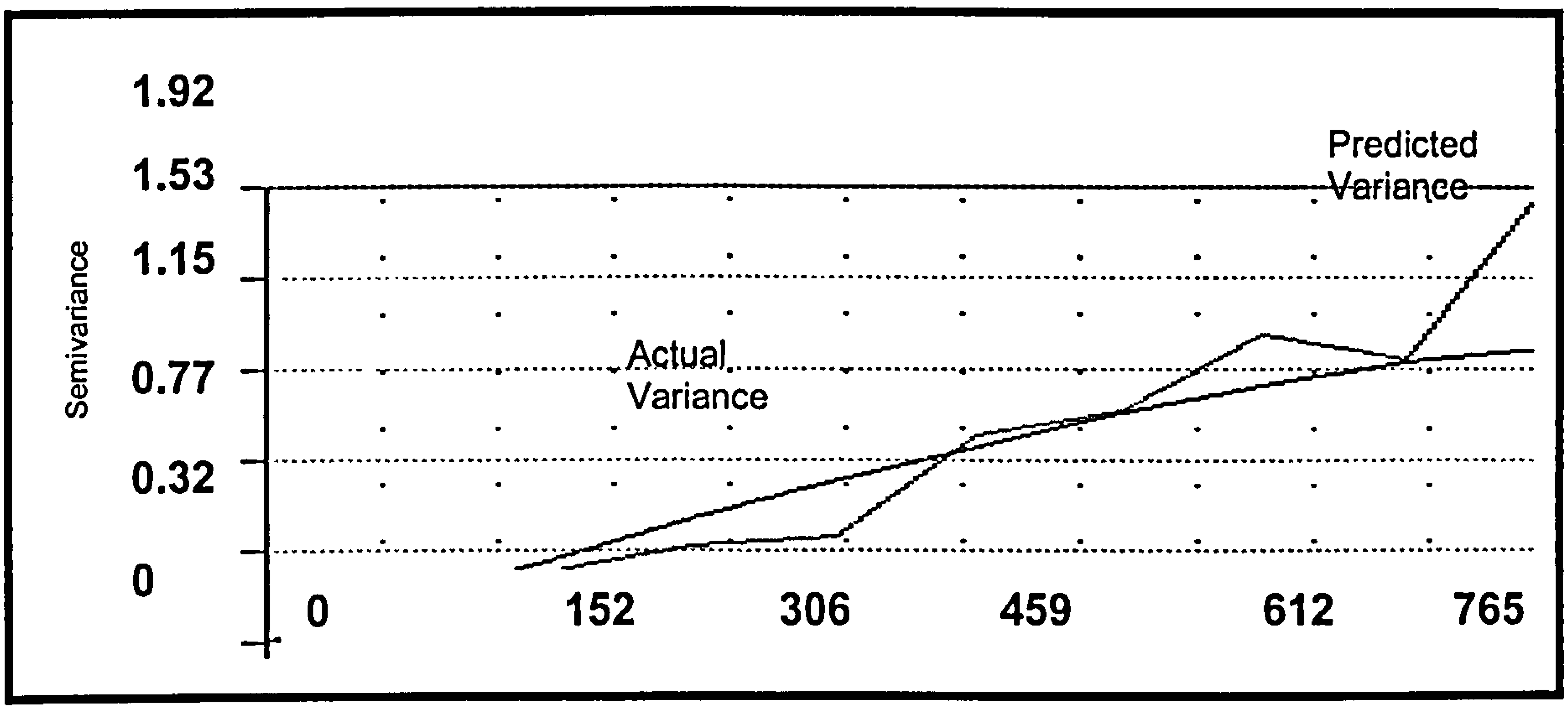
(a) August 1998 variogram of Scenario 3 : 17 sample points



(b) August 1998 variogram of Scenario 4 : 21 sample points

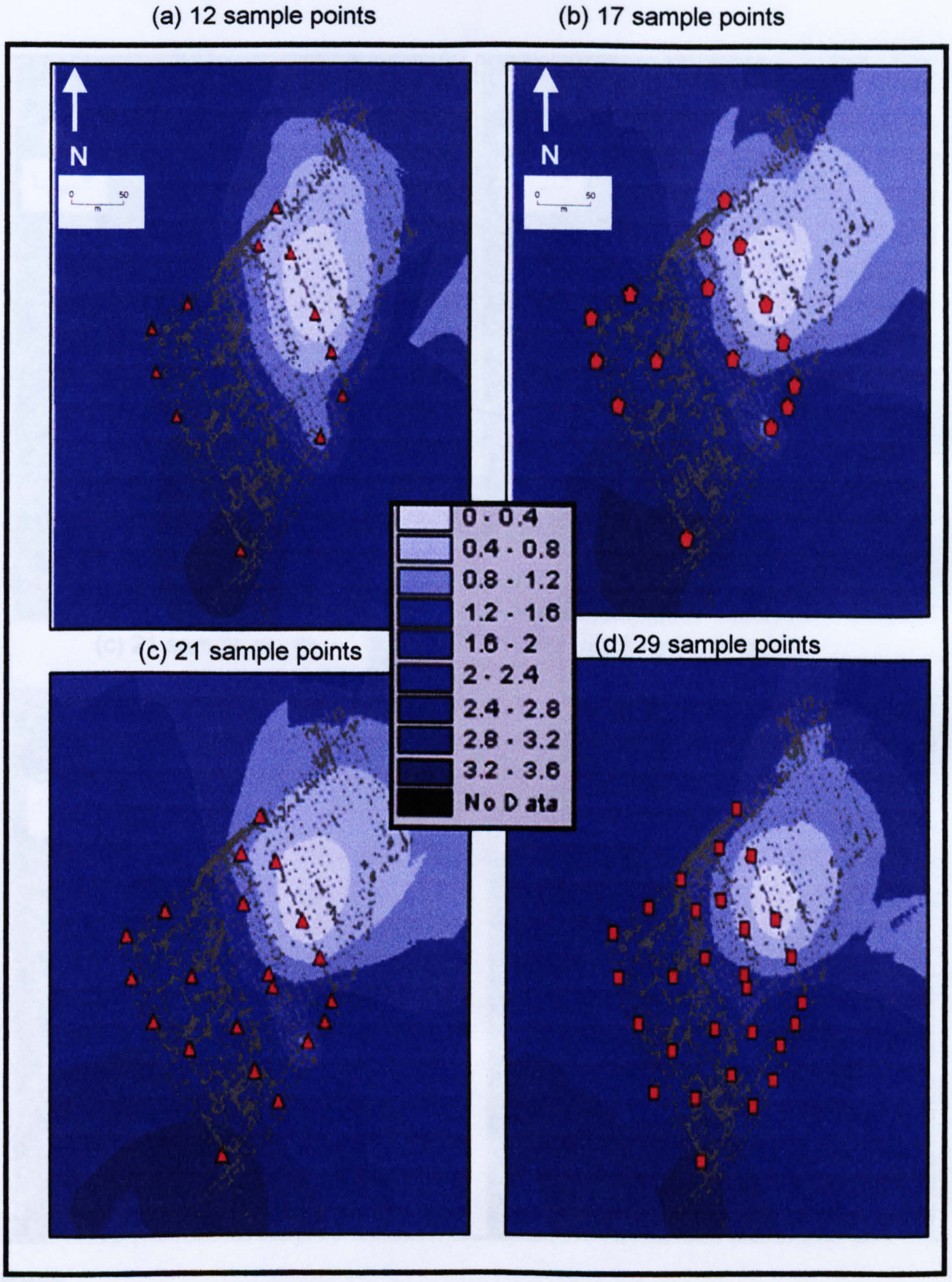


(c) August 1998 variogram of Scenario 4 : 29 sample points



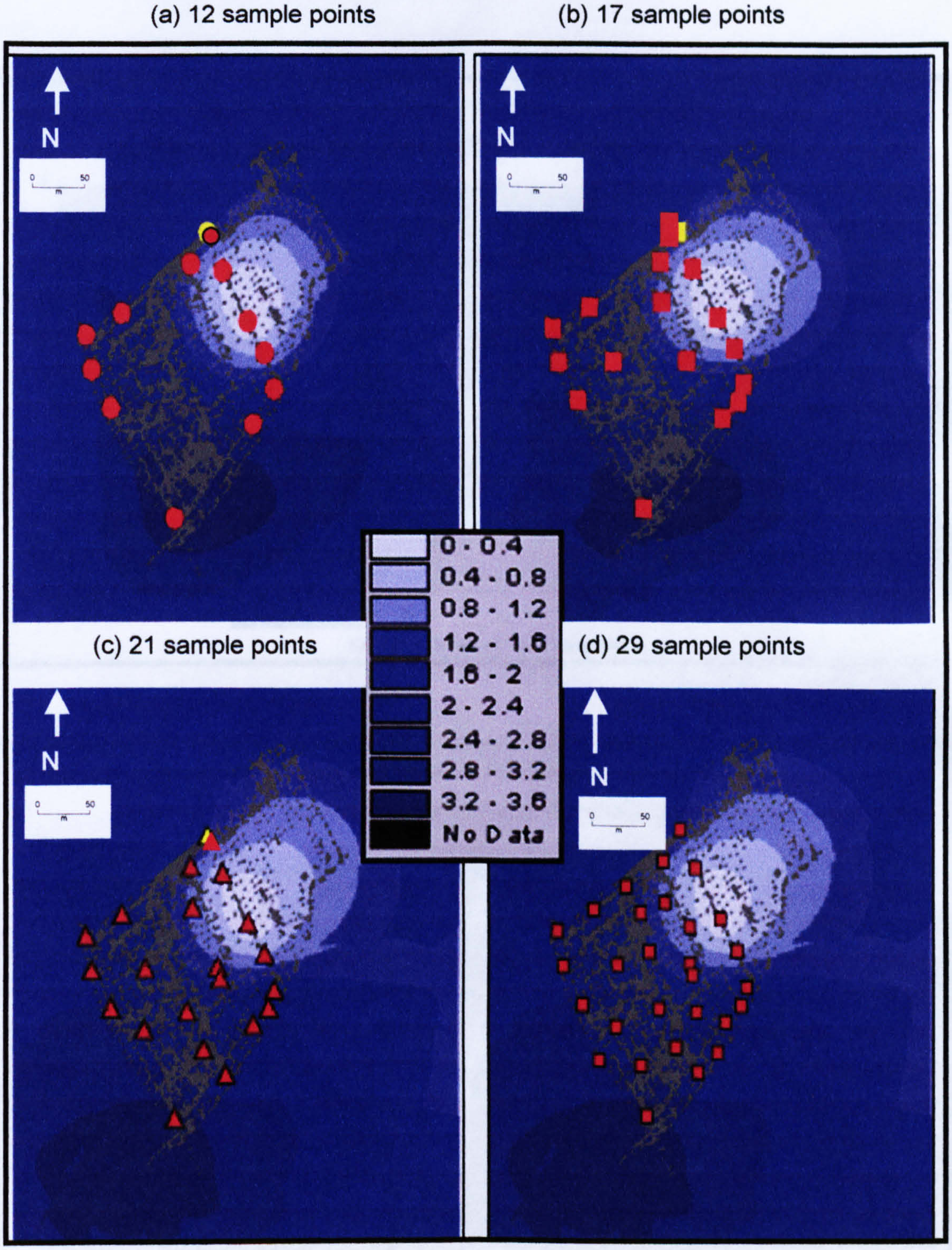
Legend:
X axis = distance in metres, Y axis = semi variance

Figure 8.8 Kriged groundwater levels mapped in GIS using March 1998 using 12, 17, 21 and 29 points (Legend in metres)



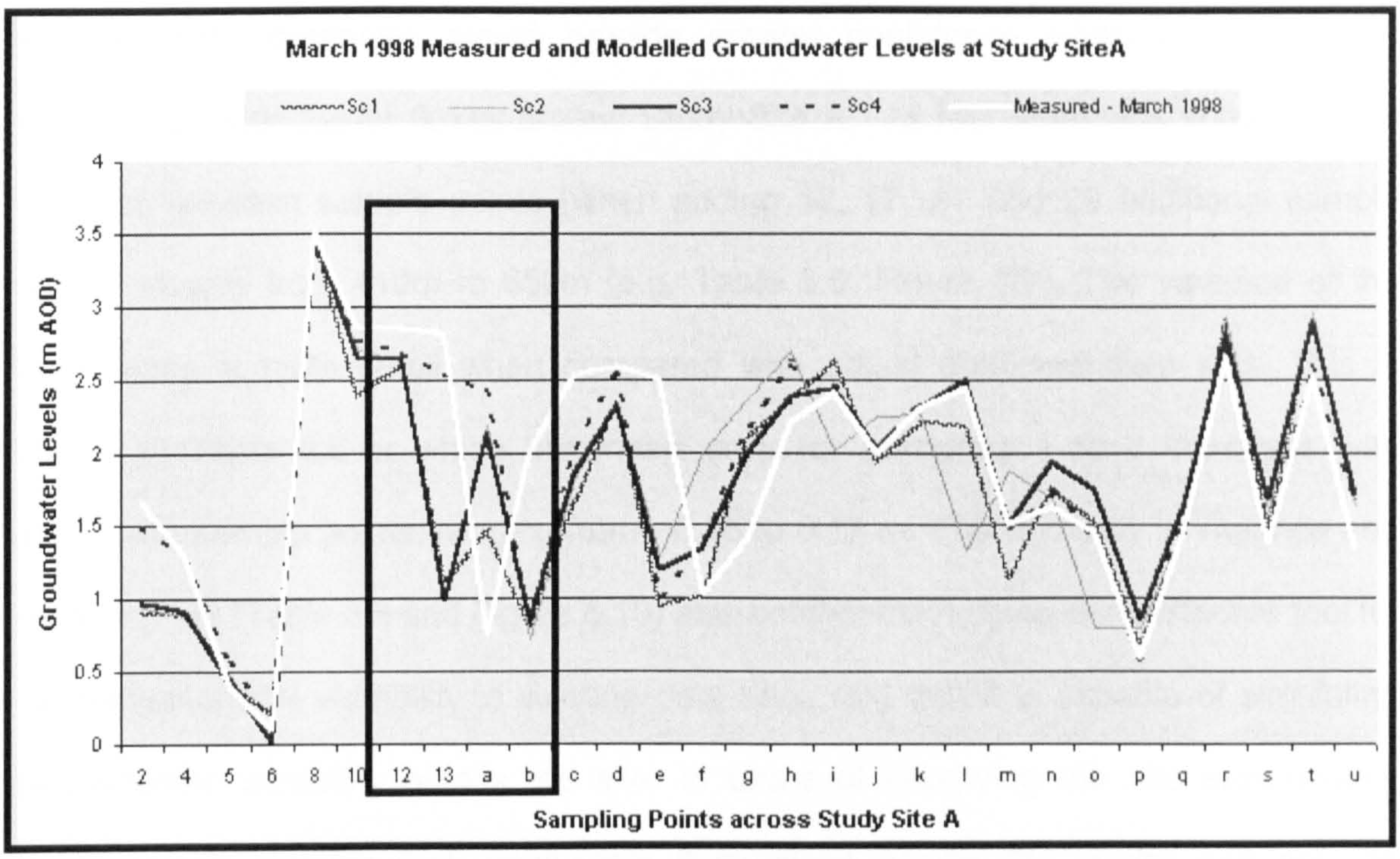
Legend:
Colours represent differences in kriged and measured groundwater levels in m AOD
= reference point, representing the number of sampled points used in each scenario

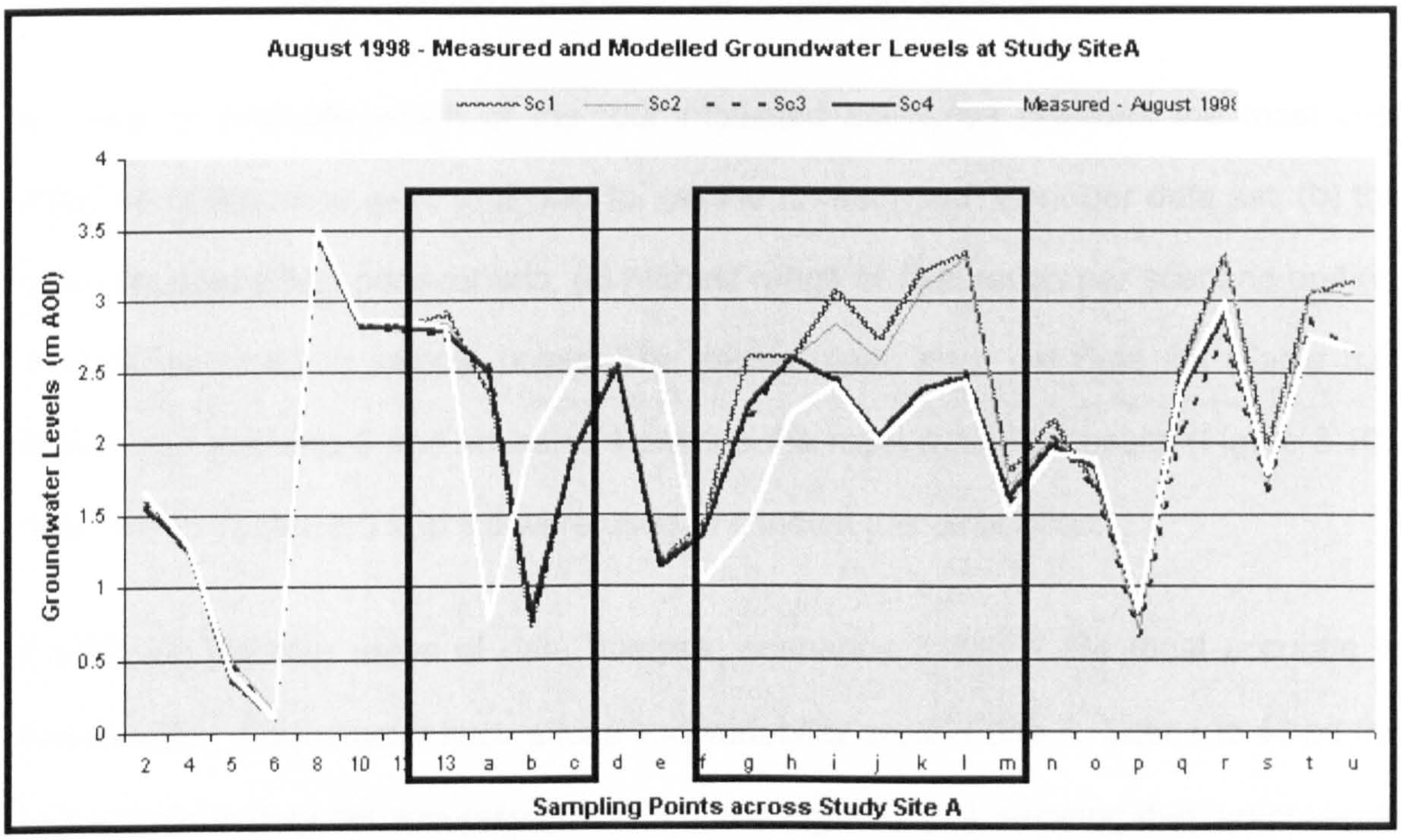
Figure 8.9 Kriged groundwater levels mapped in GIS using August 1998



Legend:
Colours represent differences in kriged and measured groundwater levels in m AOD
= reference point, representing the number of sampled points used in each scenario

Figure 8.10 Measured and estimated groundwater levels compared using 12, 17, 21, and 29 sampling points (scenario 1-4). The boxed areas are sample points with highest error.





8.3.6 Results and Discussion – Comparing Scenarios 1 – 4

Results from the four scenarios show that adding samples to information from the existing eight boreholes would provide greater confidence when assessing and monitoring groundwater fluxes around Study Site A. All four scenarios show that the distance between sample points (when adding 12, 17, 21 and 29 additional sample points) ranges from 410m to 650m (e.g. Table 8.5, Figure 8.7). The variance of the estimations is quite small when compared with actual observed data sets. This is shown in Table 8.6 in which the mean error for scenarios 1 to 4 increases with increasing sample points, ranging from - 0.18 to 0.17 m. The similarity in variance and mean values (Table 8.5 and Figure 8.10) also confirm that kriging is an effective tool for estimating spatial variability in existing data sets, and that it is capable of simulating groundwater variability across the site. In terms of improving the site assessment, these results indicate that small-scale fluctuations across the landfill can only be adequately measured using one of these four sampling scenarios.

In order to evaluate which of the four modelled scenarios provides the most cost effective option, they were analysed for (a) the lowest mean error per data set, (b) the lowest nugget effect per scenario, (c) highest range of field range per scenario and (d) greatest accuracy in sample points. The lowest mean error per data set (Table 8.6) shows that scenario 3 and scenario 4 provide the most accurate results (Figure 8.10). The data in Tables 8.5 and 8.6 were used to conduct this evaluation.

Combining the four areas of data analysis, scenarios 3 and 4 are most accurate in representing groundwater fluctuations and variability around Site A. Scenario 4 had the highest mean data set accuracy (Table 8.6 and Figure 8.10) meaning that it represents the most effective sample distribution for field conditions. However, this finding is misleading since the overall study objective was to provide the most effective sampling pattern. In context of the site assessment, when sampling groundwater conditions at a landfill site, the aim (quite often due to limited assessment budgets) is to have as few sample points as possible, placing them in the most representative locations around a site.

The challenge therefore is determining how many samples are needed and where to place them. In this context, scenario 4 did not meet the study objectives for three reasons:

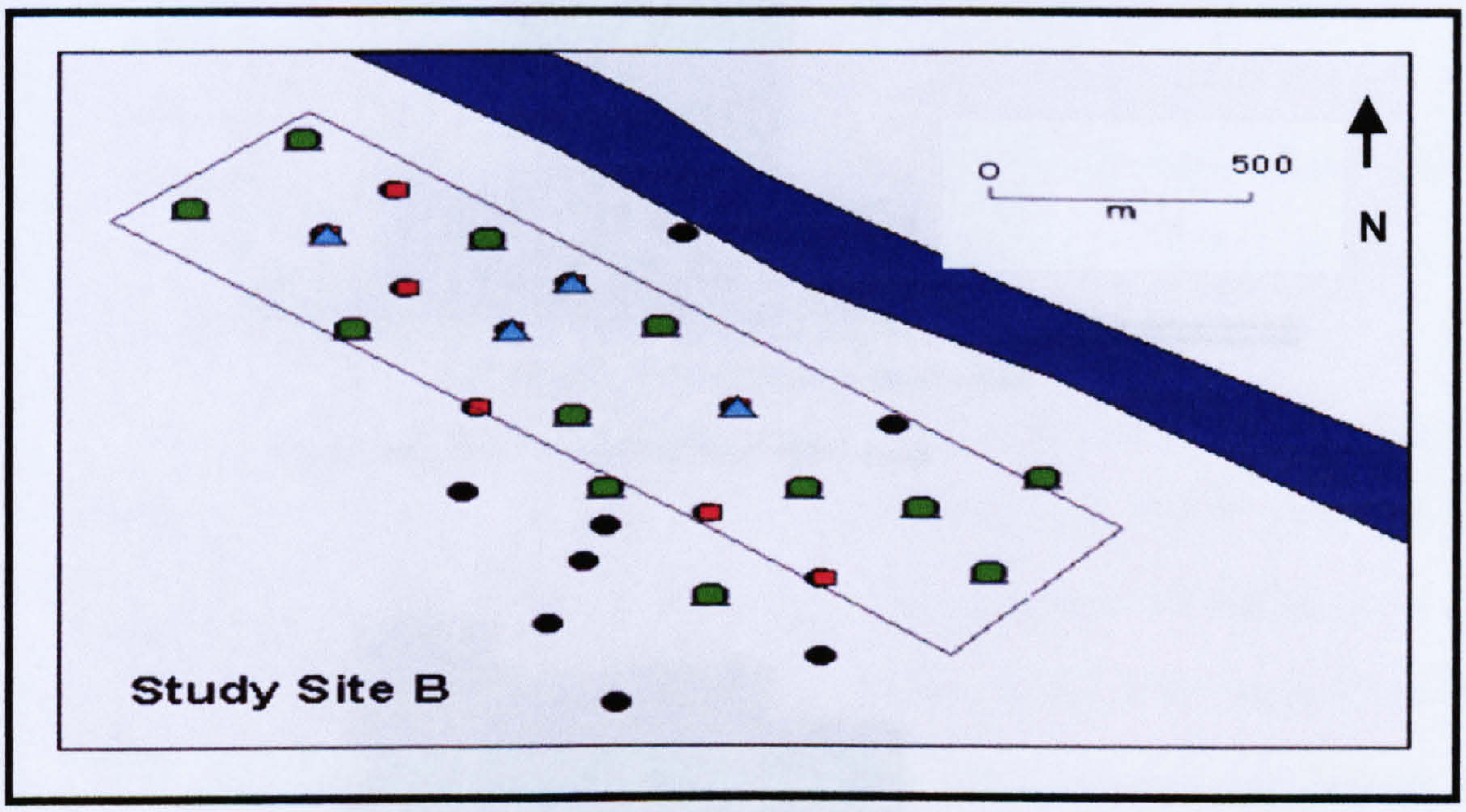
- scenario 3 had a sampling error (nugget effect) that was less than that of scenario 4 (e.g. 0.20 compared to 0.26 for March and 0.25 compared to 0.28 in August)
- scenario 3 offered a larger sampling distance between sampled points (e.g. scenario 4 range = 620m for both data sets while scenario 3 range = 650m for March and 620m for August)
- scenario 3 used 21 sampling points as opposed to 29 sample points in scenario 4.

Since the aim is to use as few sample points as possible to lower the cost of sampling but still provide effective results, scenario 3 provided the more effective sampling pattern for Site A. The only area that scenario 3 was not able to represent was the area around sample points 13, A and B (shown in circled areas of Figures 8.6 and 8.10). Further kriging is needed to determine adequate sampling patterns for this area of the landfill which experiences frequent groundwater fluctuations.

8.3.7 Introduction: Investigation 1 at Study Site B

Site B is a municipal landfill site located in the suburbs of Zagreb, Croatia. A description of geophysical conditions at the site can be found in section 7.3. The kriging objective and methods used at this study site were similar to those used for the Site A data sets (presented in sections 8.3.3 - 8.3.6). The kriging analysis conducted using Site B data sets differs from that conducted at Site A in two ways. Firstly, Site B had 12 piezometers located across the landfill (while Site A had 8 boreholes). Secondly, February and September 1998 groundwater data were used (the Site A model used March and August 1998 data sets). Figure 8.11 is a site map showing piezometer and sample points that were used in four scenario models which used 12, 17, 21 and 29 piezometers. The spatial variability of these data sets was characterised using histograms and statistical analysis shown in Figure 8.12 and Table 8.8.

Figure 8.11: Measured and kriged groundwater sampling points around Site B



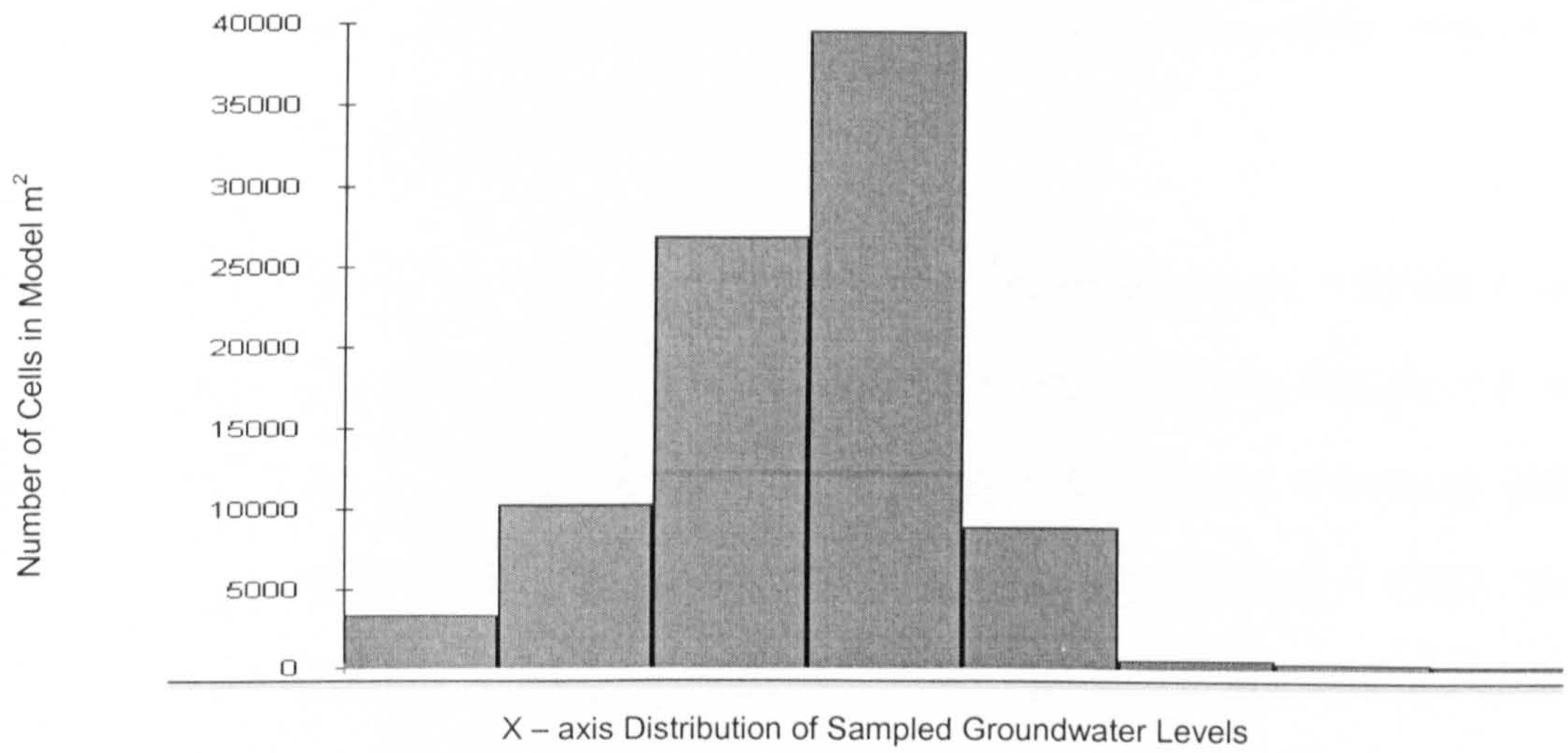
Legend:
Green = scenario 1 - 12 sample points
Blue = scenario 2 - 17 sample points
Red = scenario 3 - 21 sample points
Black = scenario 4 - 29 sample point
Landfill edge

Table 8.7 Piezometer groundwater levels (m) from Site B used for kriging analysis

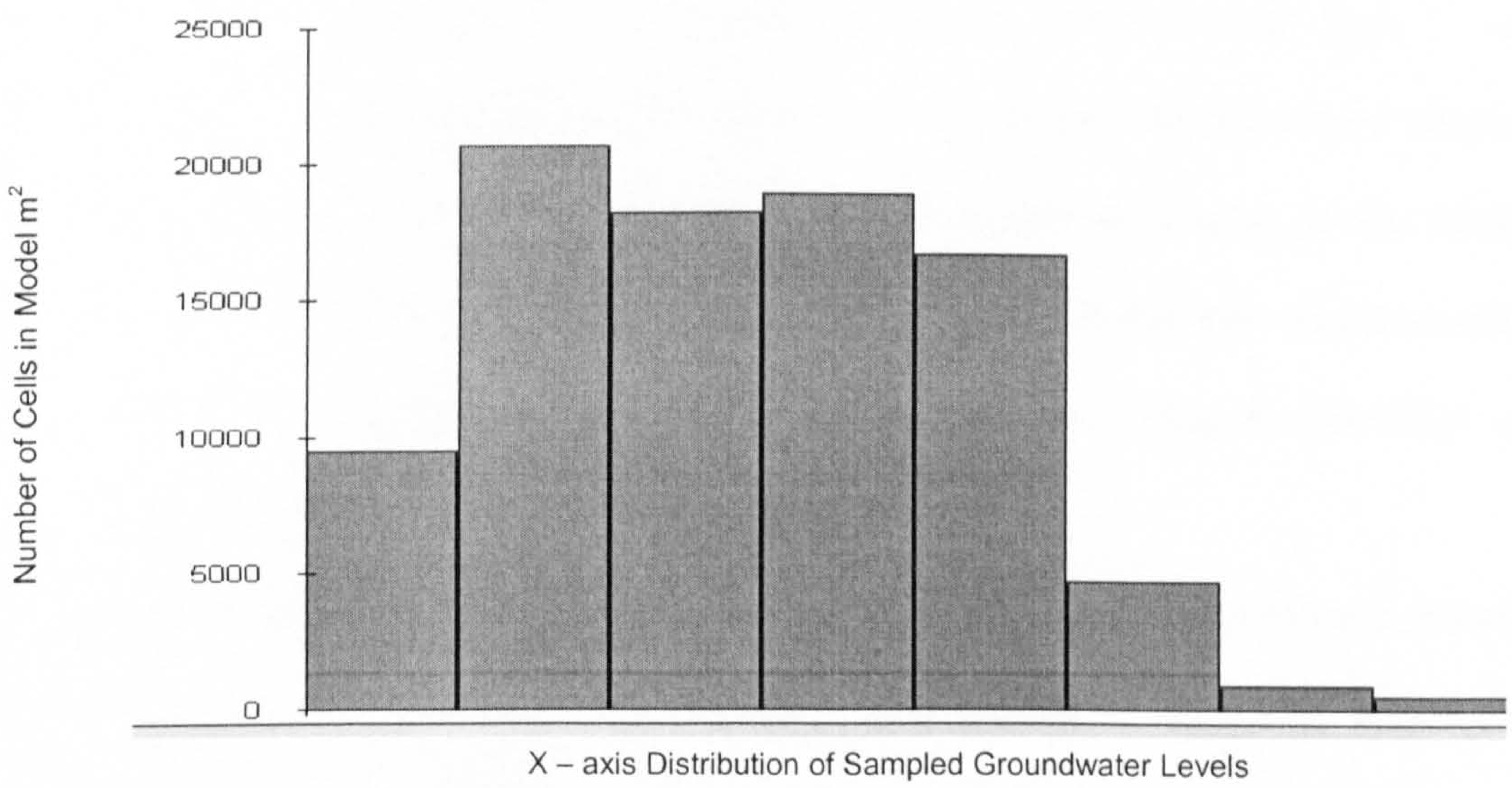
Sample #	February 1998 (m AOD)	September 1998 (m AOD)
1	102.5	103.5
20	102	103
24	100.7	102.6
12	101.6	102.6
16	101.4	102.2
3	101.4	103.4
5	102.4	102.8
8	102	102.4
Cp1	101.6	101.9
17	101.2	101.8
14	102	103
2	102	103

Figure 8.12 Groundwater level histograms of Site B showing that the data sets had different distributions

(a) Histogram for February 1998 data



(b) Histogram for September 1998 data



Legend:
X-axis = distance in metres, Y axis = semi variance

Table 8.8 Field sample statistics and variogram statistics for Site B

Statistical Data	Groundwater Level February 1998	Groundwater Level September 1998
Number of sample points	12	12
Mean value of sample points (m)	101.71	102.48
Variance of sample points	0.375	0.43
Variogram Nugget Effect	0.03	0.03
Variogram Sill (m)	0.28	0.5
Variogram Range (m)	850	950
Variogram RMSE	0.14	0.19

8.3.8 Kriging and Data Analysis

The February and September 1998 data sets were used to calculate variograms. The lag distance was set to 53m. The data derived from the variograms in Figure 8.13 are listed in Table 8.8, and they confirmed that there are no signs of anisotropy. Contour maps were created with each grid cell representing $5.4m^2$ and using a grid containing 250 rows and 363 columns. The search distance was 100m. Figure 8.14 shows contour maps of kriged groundwater levels.

The reliability of the kriged estimates was cross-checked as outlined in section 8.3.3. The data for February and September 1998 are shown in Tables 8.8 and 8.9. The variance values were similar in September, however the February values differed significantly (measured variance = 0.375 while kriged variance = 0.03). This difference is illustrated in Figures 8.14(a) and 8.15, showing different contour maps and correlation graphs (e.g. the February data set had a correlation of 59 percent while the September data sets had a correlation of 95 percent). The high r^2 value for the September data is probably an artefact of the lower measurement resolution for the measured groundwater levels (as seen by the vertical banding of the data). In context of the landfill site assessment, these results confirm the site assessment findings (in section 7.3) which showed the site had highly variable seasonal groundwater fluctuations, especially in winter months.

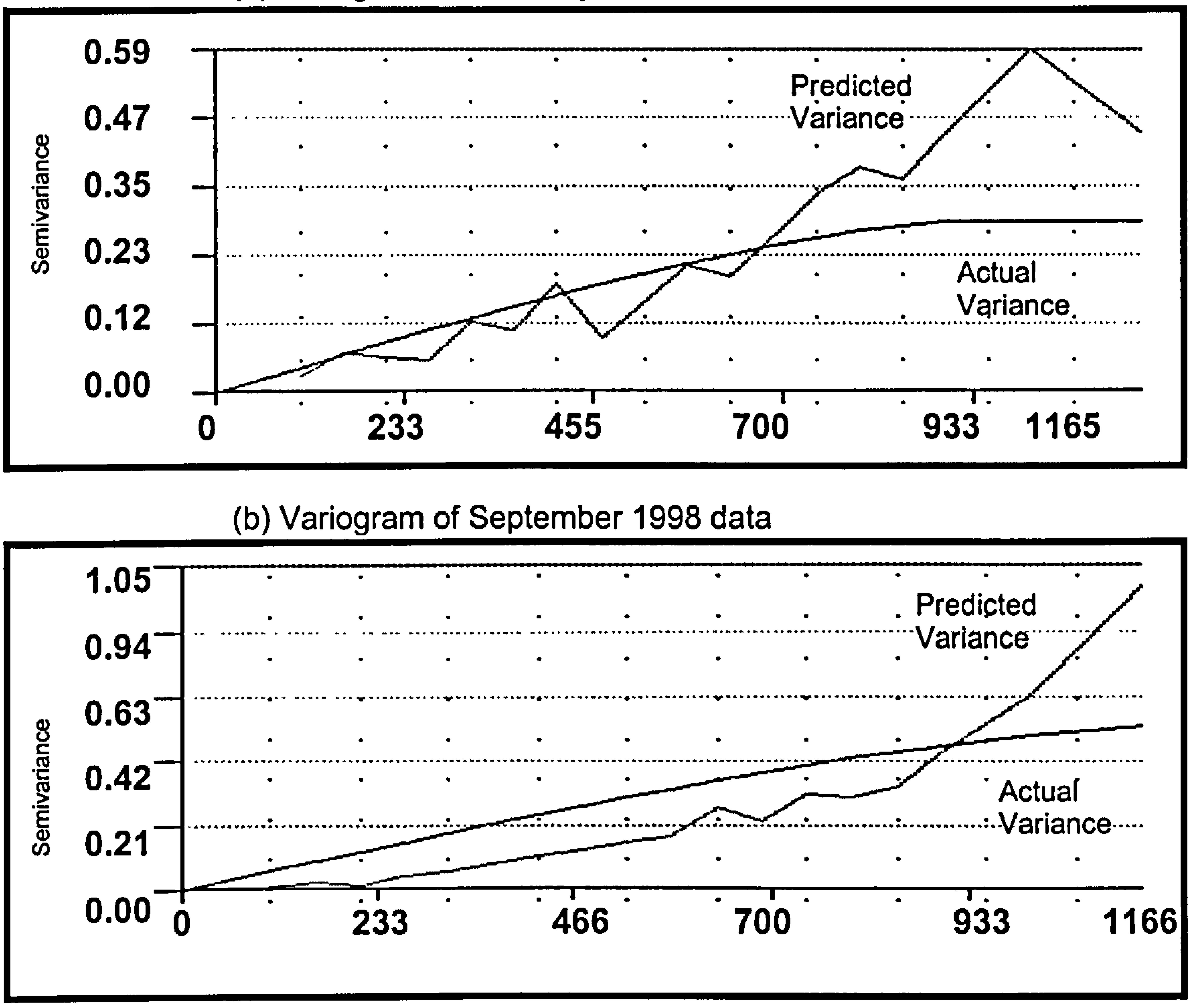
These results indicate that although this sampling distribution (12 piezometers) may be adequate for September groundwater fluctuations, it may not, however, give data representative of February 1998 groundwater conditions. Figure 8.16 was produced in

GIS to identify areas of greatest difference between measured and kriged groundwater contours.

Figure 8.13: Variograms produced using groundwater data from February and September 1998

(a) Variogram of February 1998 data

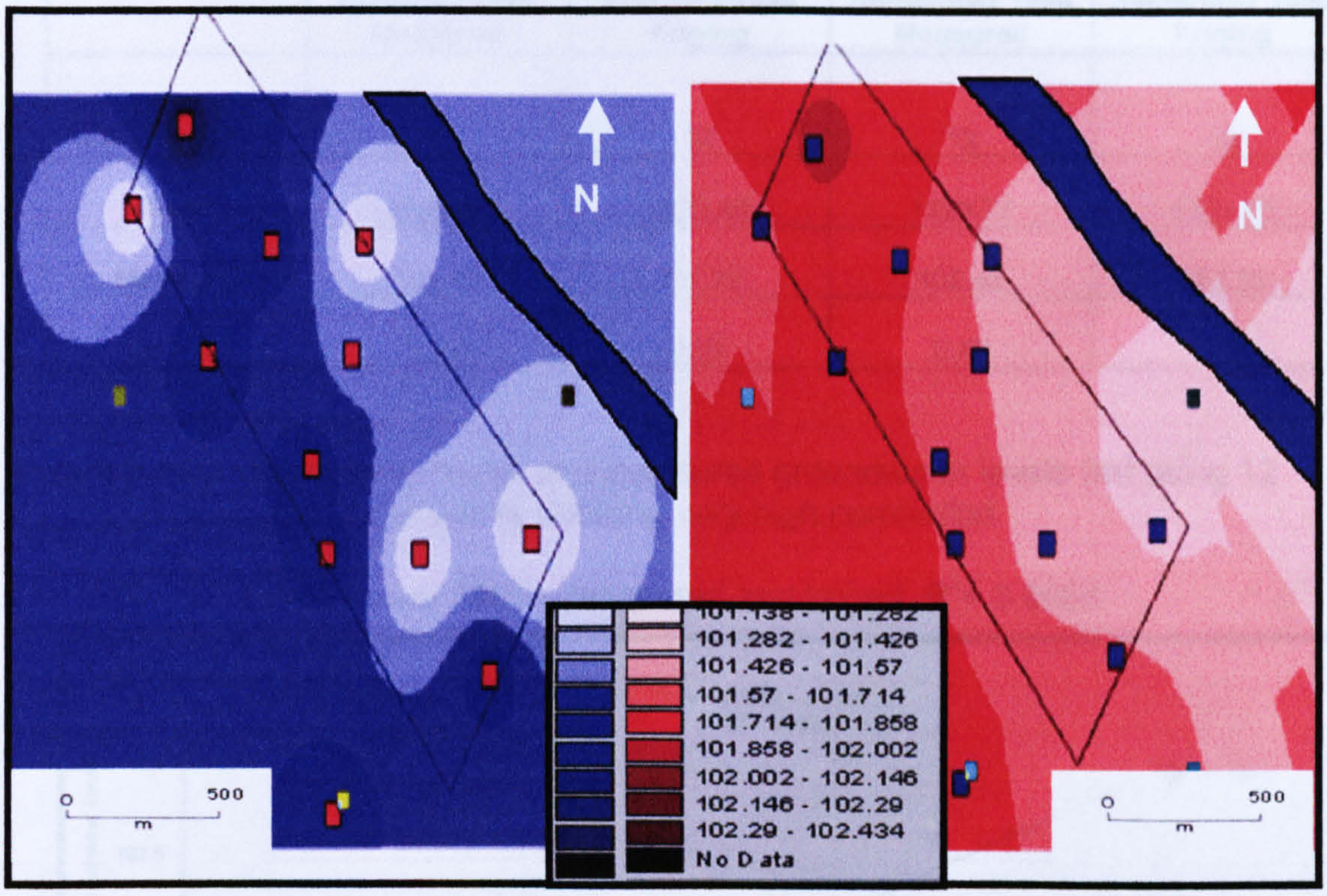
(b) Variogram of September 1998 data



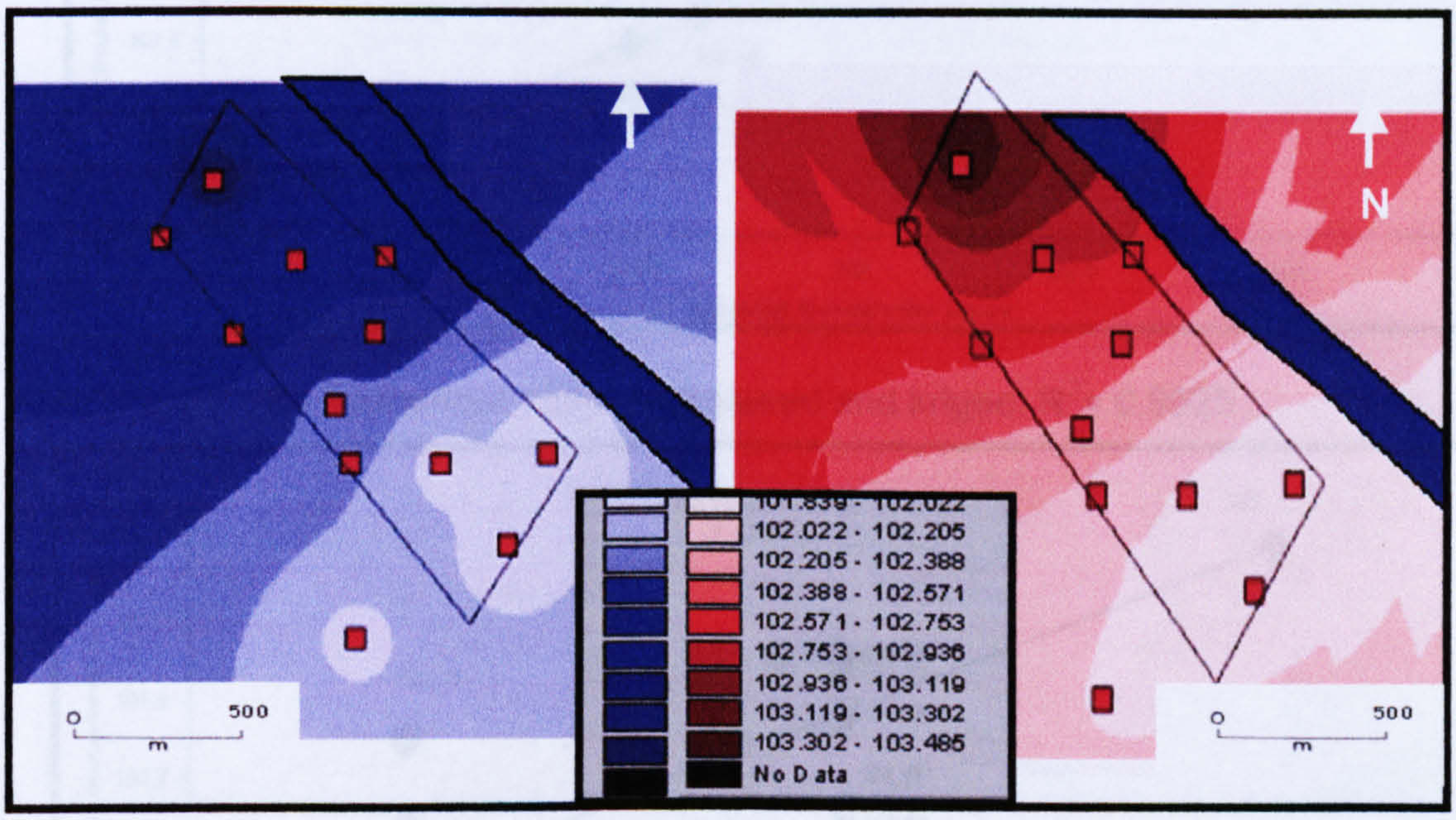
Legend:
X axis = distance in metres, Y axis = semi variance

Figure 8.14 Contour maps produced from measured and kriged groundwater levels

(a) February 1998 data – measured (left side) and kriging (right side)



(b) September 1998 data – measured (left side) and kriging (right side)



Legend:

Black ■ ■ = sample points

□ = landfill edge

▬ = Sava River

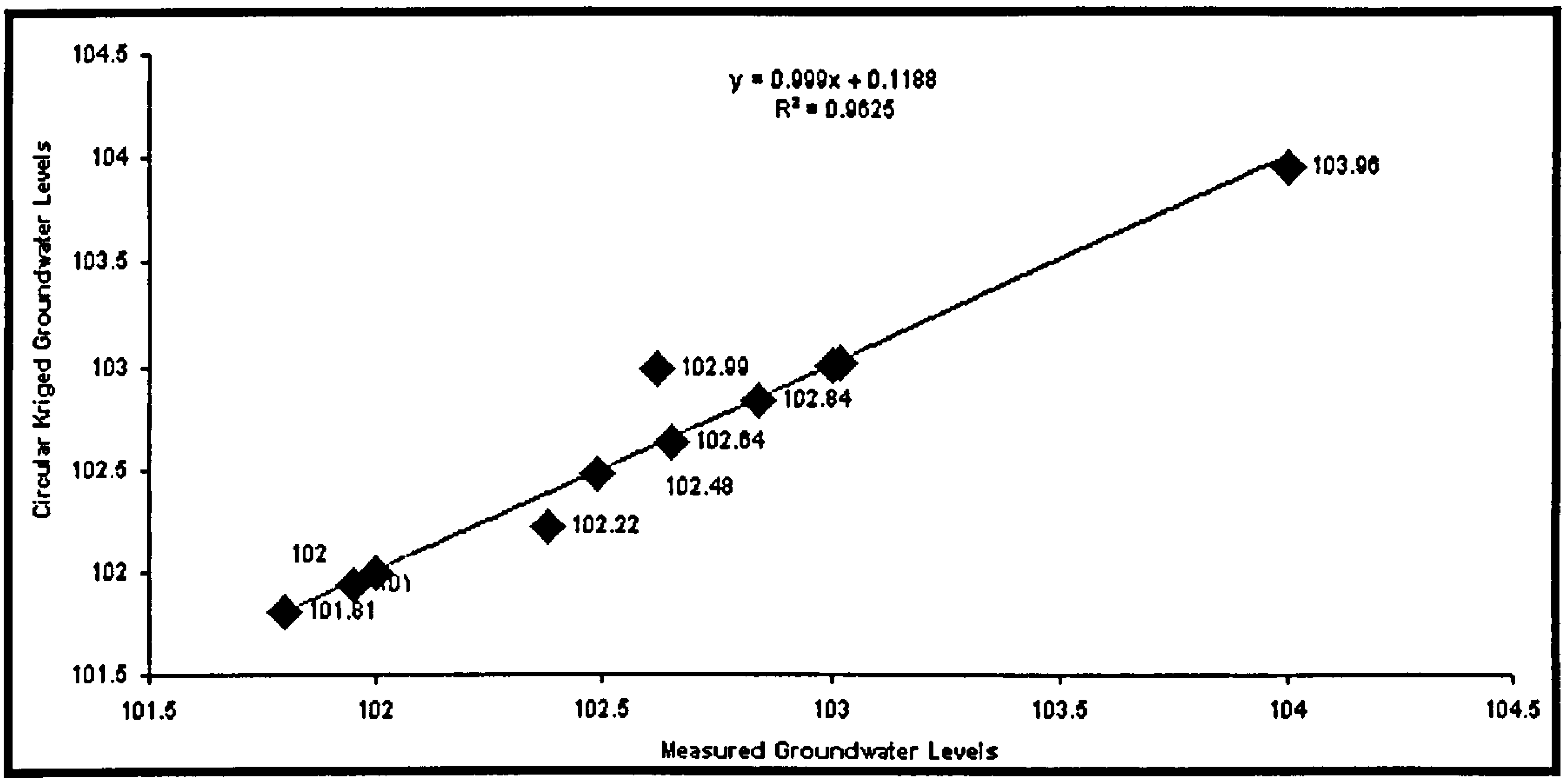
Colours represent differences in kriged and measured groundwater levels in m AOD

Table 8.9 Comparing measured and kriged groundwater levels for February and September 1998

	February 1998 Measured	February 1998 Kriging	September 1998 Measured	September 1998 Kriging
# of sample points	12	12	12	12
Sum	1220.49	1220.62	1229.77	1229.95
Mean (m)	101.71	101.72	102.48	102.50
Variance	0.375	0.03	0.43	0.41

Figure 8.15: Comparing kriged and measured groundwater levels (m) using 12 piezometer points showing very high correlation

(a) February 1998 – measured and kriged, R^2= 0.5998



(b) September 1998 – measured and kriged, R^2= 0.9625

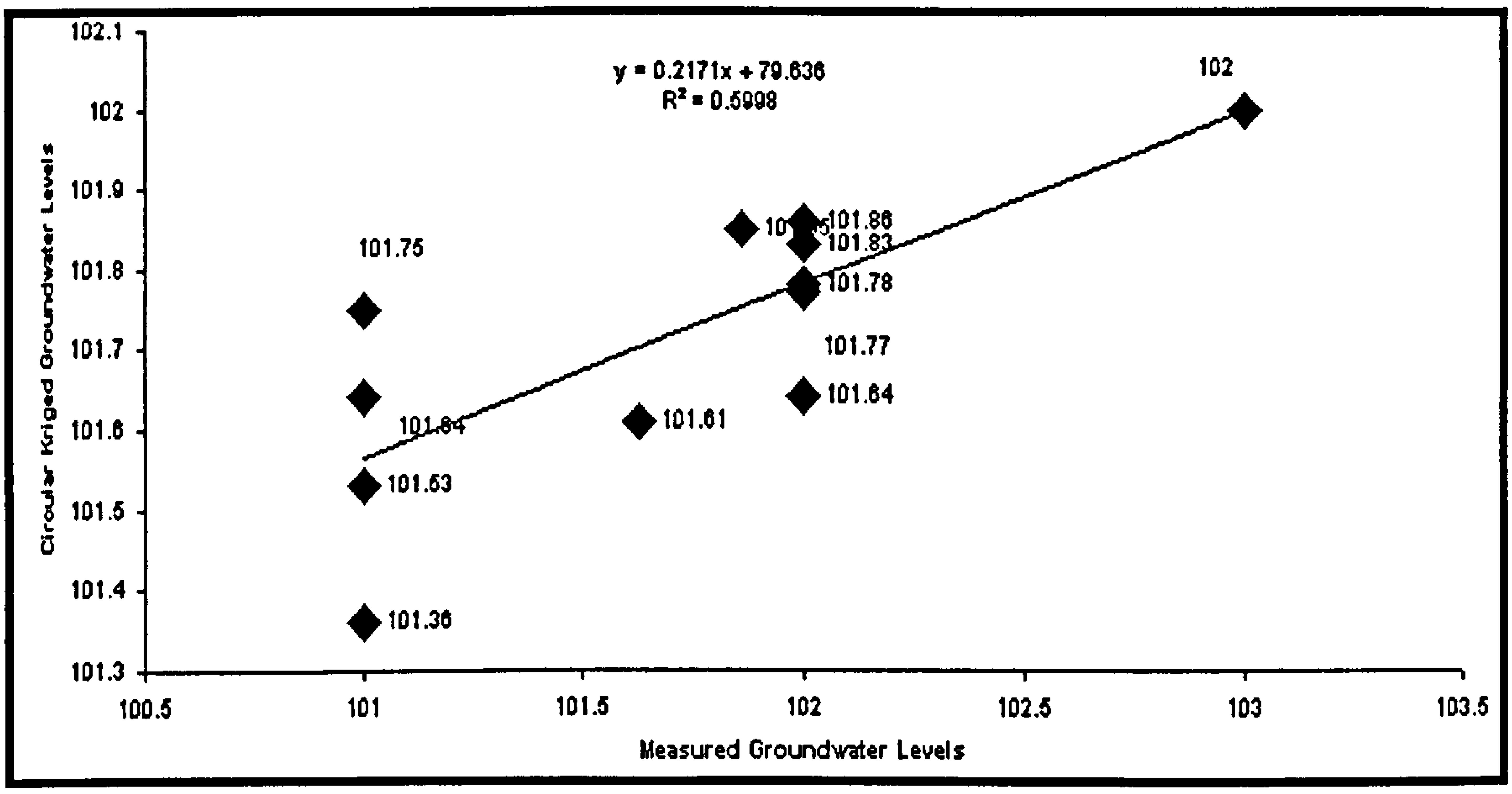
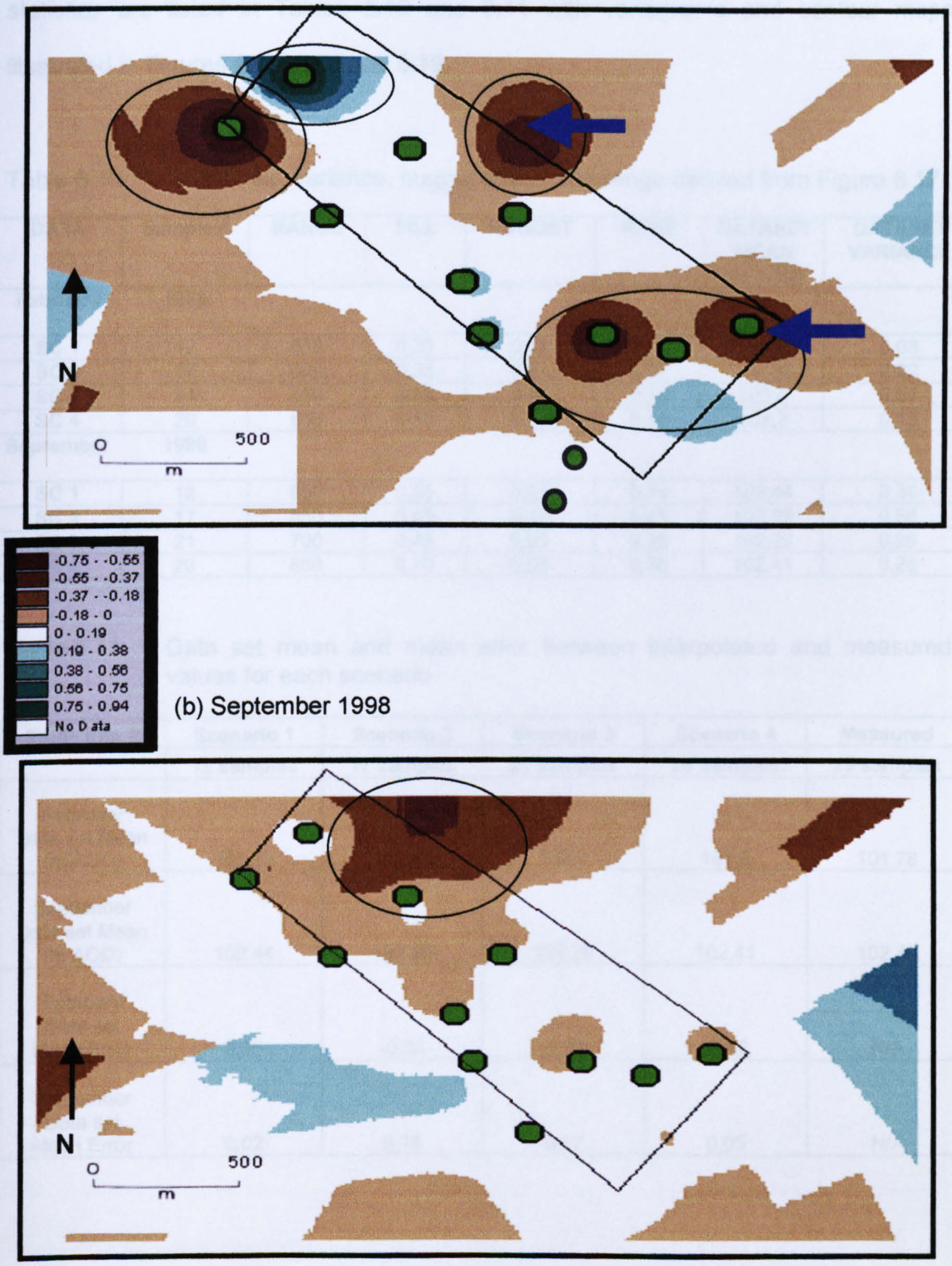


Figure 8.16: Zones of highest predictive differences in February and September 1998 using circular kriging (Legend in metres)

Legend:
Green = sample points
= Landfill edge

(a) February 1998



8.3.9 Evaluating Sample Numbers and Sample Locations: Scenarios 1 - 4

In order to test the findings of the initial kriging investigation at Site B (presented in section 8.3.8), four scenarios that were modelled (illustrated in Figure 8.11) and verified using 29 piezometer value collected in February and September 1998. Variogram statistics are listed in Tables 8.10 and 8.11 with variograms and contour maps illustrated in Figures 8.17, 8.18 and 8.19.

Table 8.10 RMSE, sill, variance, nugget effect, and range derived from Figure 8.17

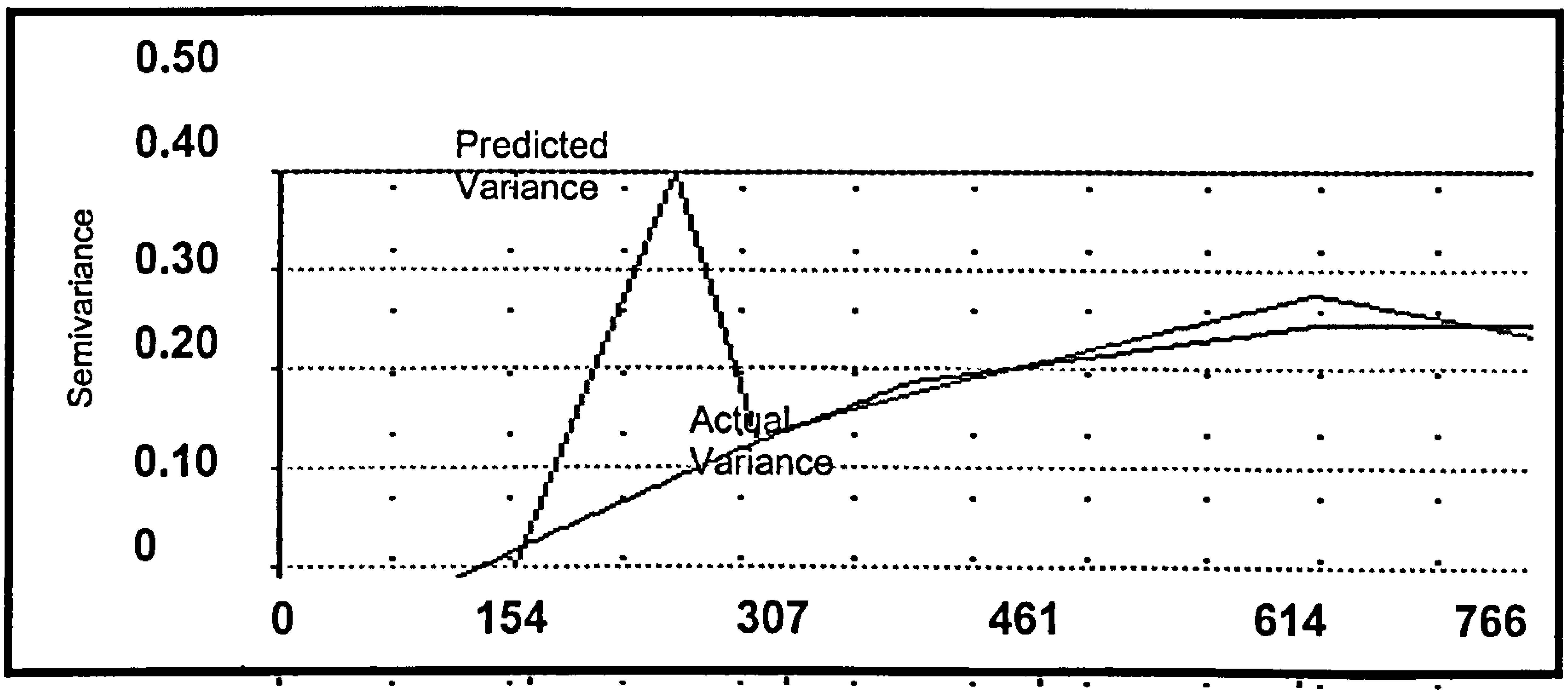
DATA	Sample #	RANGE	SILL	NUGGET	RMSE	DATASET MEAN (m AOD)	DATASET VARIANCE
February	**1998**						
SC 1	12	850	0,35	0,19	0,12	101,73	0,03
SC 2	17	600	0,35	0,11	0,22	101,86	0,22
SC 3	21	450	0,35	0,05	0,18	102,0	0,21
SC 4	29	850	0,45	0,08	0,13	101,8	0,15
September	**1998**						
SC 1	12	950	0,55	0,02	0,19	102,44	0,34
SC 2	17	950	0,45	0,10	0,45	102,28	0,38
SC 3	21	700	0,45	0,03	0,36	102,29	0,26
SC 4	29	850	0,70	0,03	0,30	102,41	0,26

Table 8.11 Data set mean and mean error between interpolated and measured values for each scenario

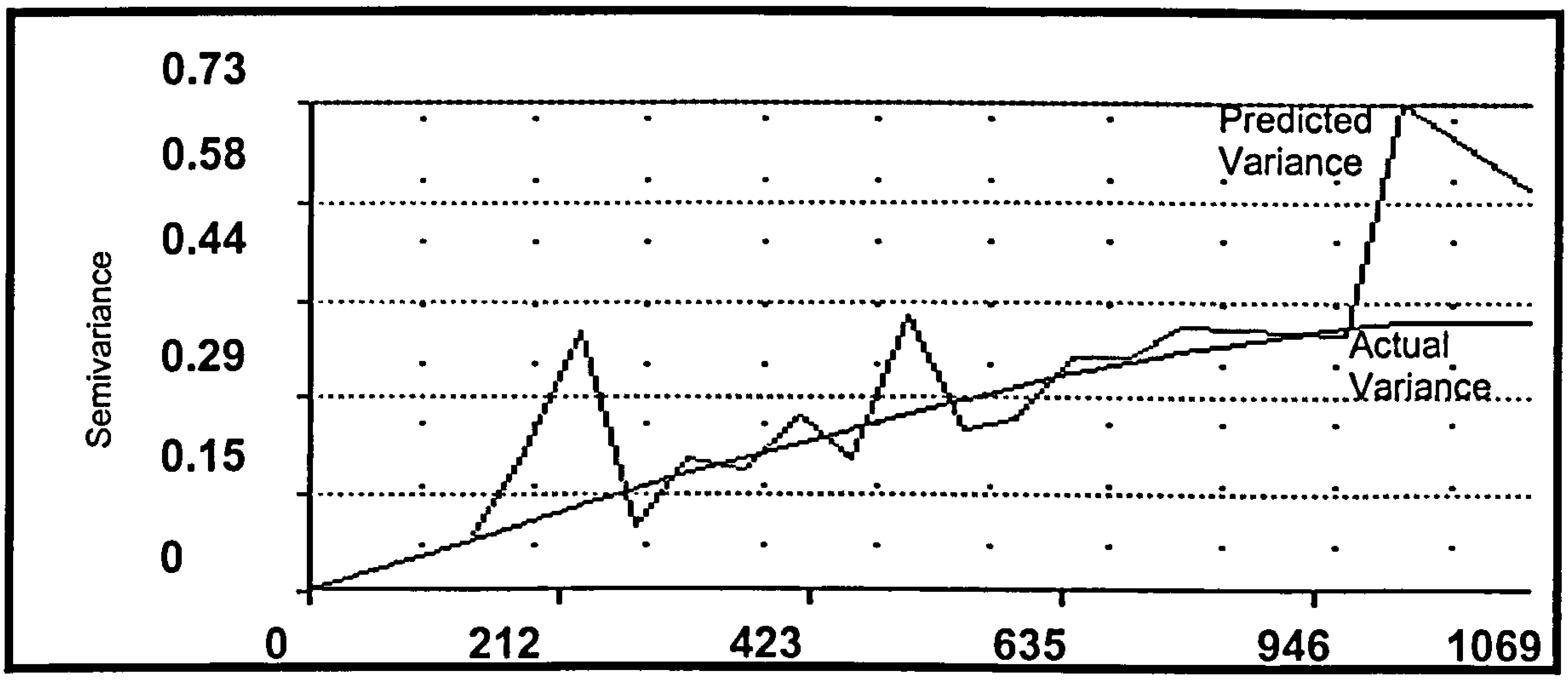
Study Site B	Scenario 1	Scenario 2	Scenario 3	Scenario 4	Measured
	12 samples	**17 samples**	**21 samples**	**29 samples**	**29 samples**
February Data set Mean (m AOD)	101,73	101,86	102,0	101,8	101,78
September Data set Mean (m AOD)	102,44	102,28	102,29	102,41	102,46
February Data set Mean Error	0,05	-0,08	-0,22	-0,02	N/A
September Data set Mean Error	0,02	0,18	0,17	0,05	N/A

Figure 8.17 Variograms produced using September 1998 groundwater data from Site B

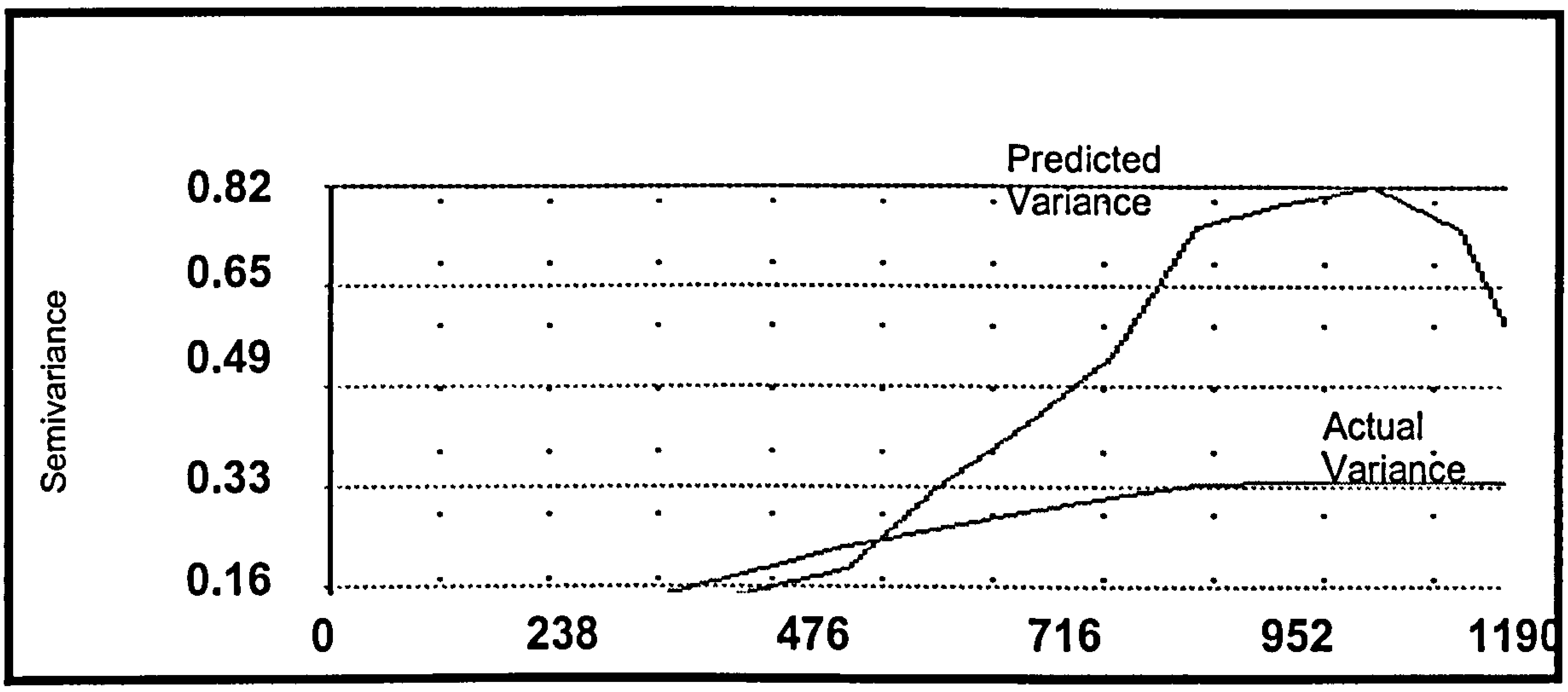
(a) February 1998 variogram of Scenario 3 : 17 sample points



(b) February 1998 variogram of Scenario 4 : 29 sample points



(c) September 1998 variogram of Scenario 4 : 29 sample points



Legend:
X axis = distance in metres, Y axis = semi variance

Figure 8.18 Groundwater level contour maps using February 1998 data (Legend = m AOD, small circles = sample points, rectangle = Site B, blue thick line = Sava River)

Legend:

Black = sample points

= landfill edge

= Sava River

Colours represent differences in kriged groundwater levels in m AOD

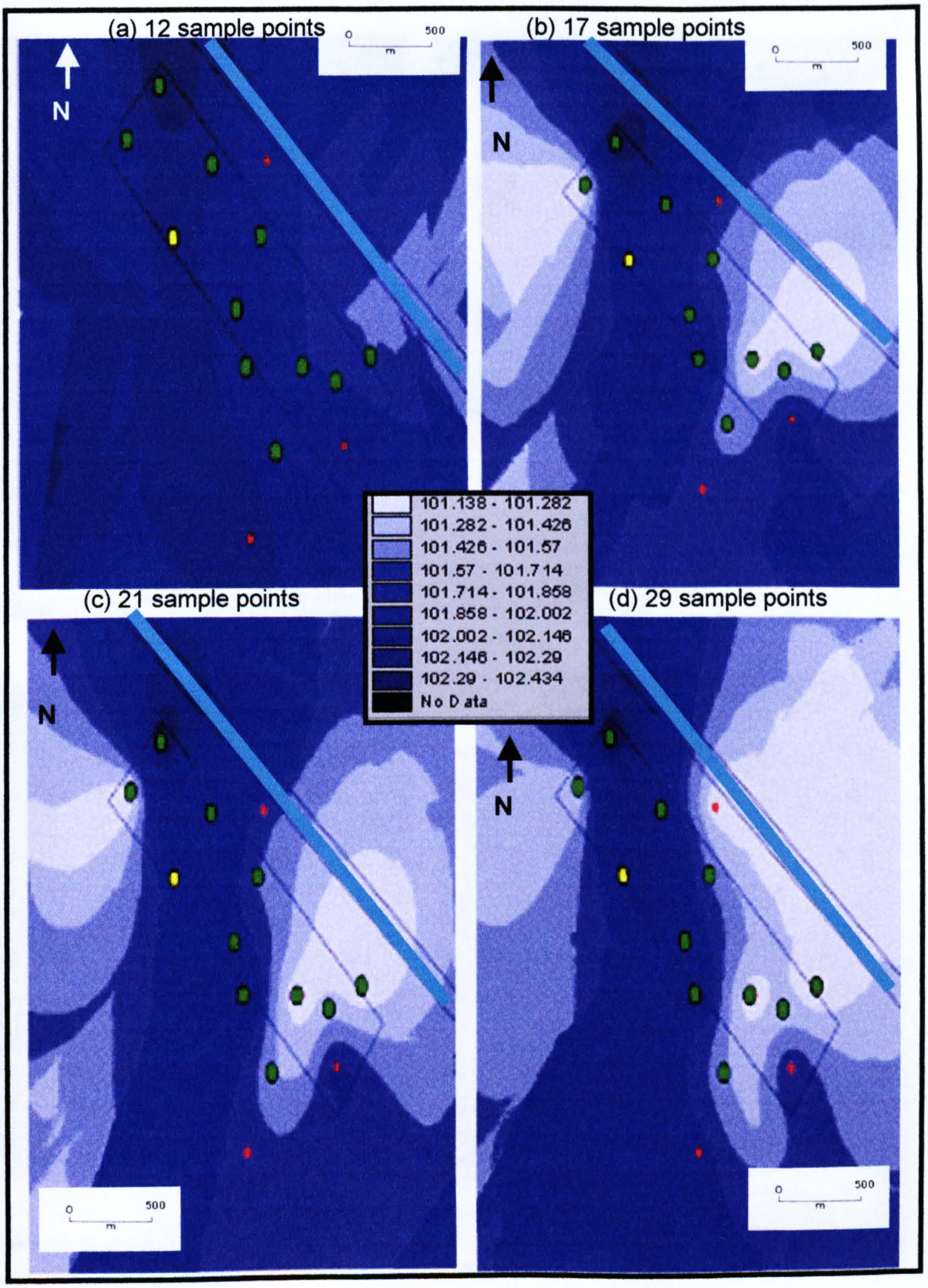


Figure 8.19 Kriged groundwater level contour maps using September 1998 data

Legend:
Black = sample points
= landfill edge
= Sava River
Colours represent differences in kriged groundwater levels in m AOD

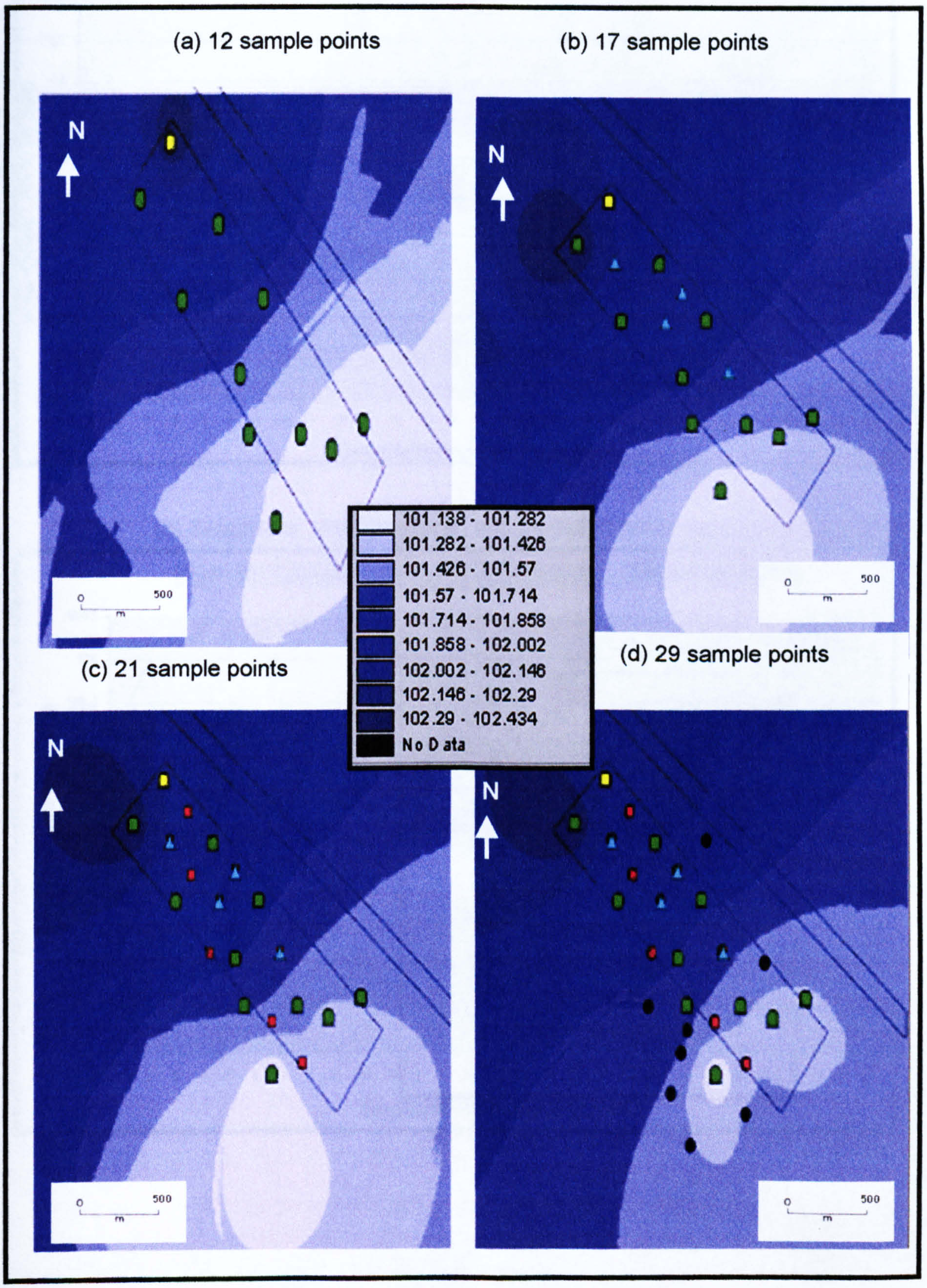
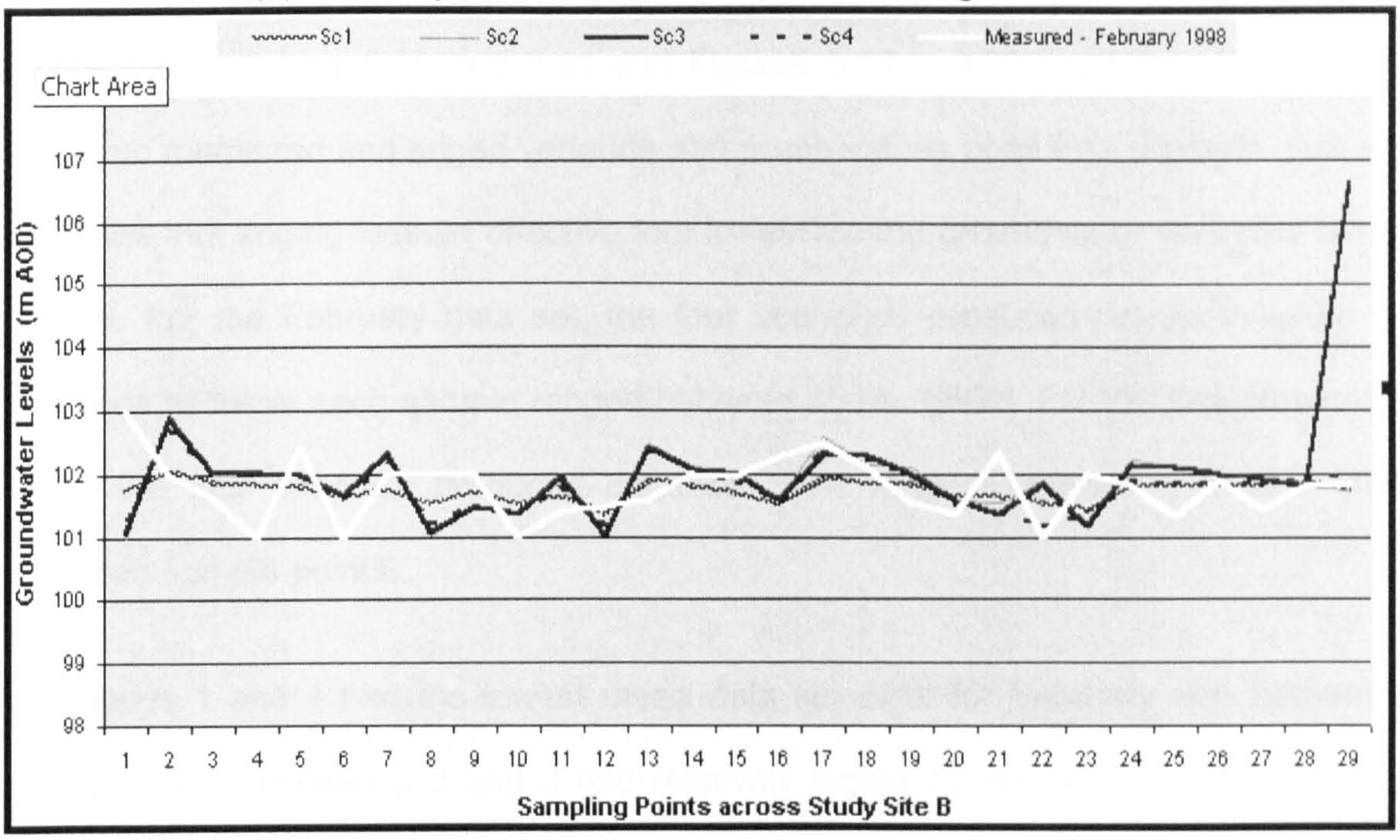
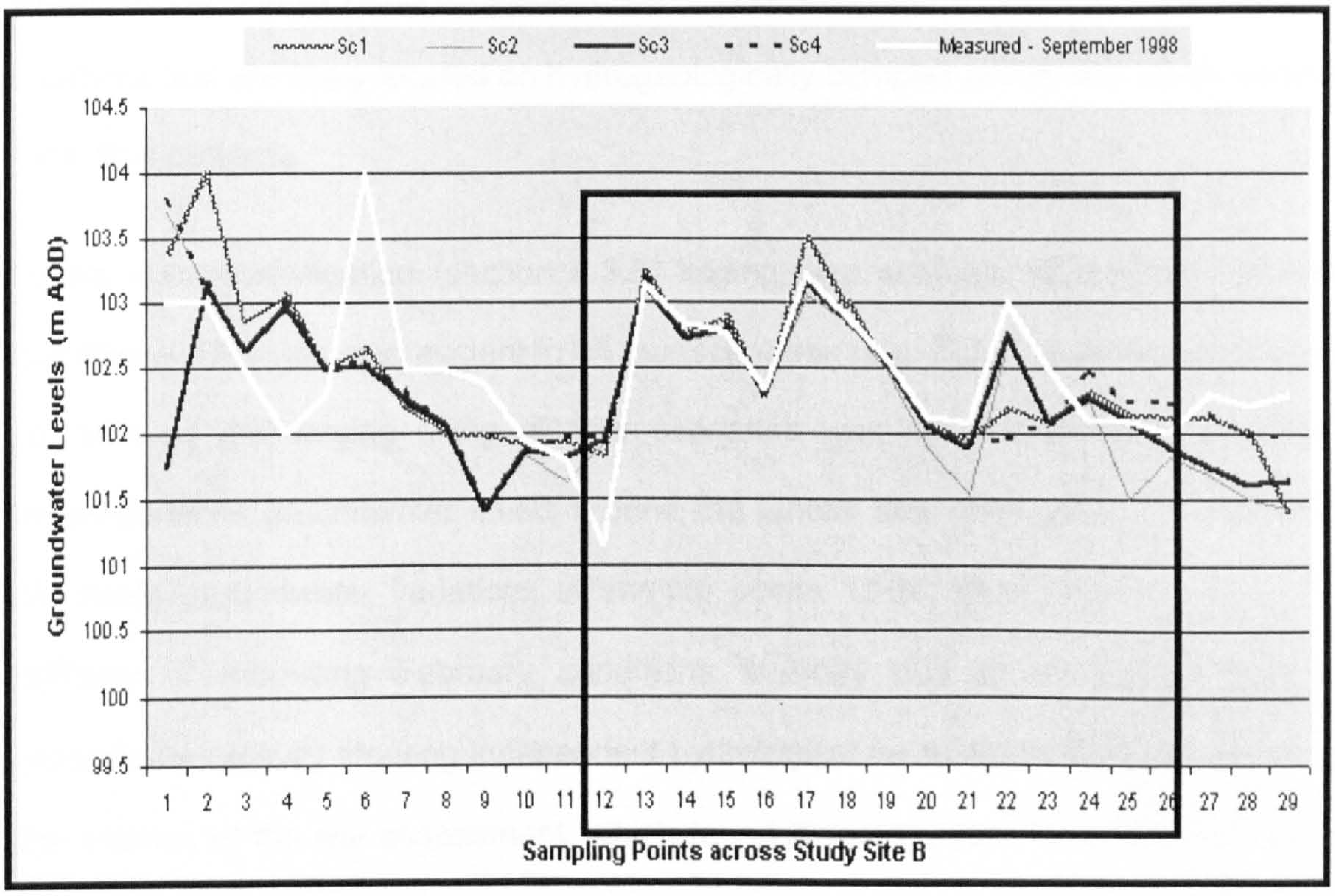


Figure 8.20 Groundwater levels estimated at 29 locations using 12, 17, 21, and 29 sampling points (scenarios 1-4) for February and September 1998 at Site B (boxed areas indicate accurate areas of kriged model)

(a) February 1998 measured and modelled groundwater levels at Site B



(b) September 1998 measured and modelled groundwater levels at Site B



8.3.10 Discussion: Comparing Scenarios 1 – 4

There are several interesting findings from the kriging simulations of Site B. The data presented in Tables 8.10 and 8.11 and Figures 8.17 – 8.20 show that the similarity between measured and kriged variance and mean values of all four scenario data sets confirms that kriging was an effective tool for estimating groundwater variability across Site B. For the February data set, the four scenarios produced results in which the distance between each sample ranged between 450 to 850m. For the September data sets, the four scenario produced distances that ranged between 700 and 950m between sample points.

Scenarios 1 and 4 had the lowest mean data set error for February and September models, while scenarios 2 and 3 had relatively higher comparable errors (e.g. Table 8.11), indicating that scenarios 1 and 4 provided the highest confidence in sampling locations that reflect regional groundwater levels. Scenario 2 and 3 point to sampling locations that are likely located on hydrogeologically complex areas with highly variable local flow patterns.

In the initial investigation (section 8.3.8) kriging was not able to simulate February conditions. This was also evident in all four scenarios (e.g. Figure 8.20) in which graph (a) showed that kriging using all four scenarios was not successful in simulating heterogeneous groundwater levels around the landfill site while graph (b) accurately simulated groundwater variations at sample points 13-26 show in boxed area. The difficulty of modelling February conditions is likely due to the highly localised groundwater activity creating independent hydrological fluctuations. This coincides with the findings of the site assessment, which found that the Sava River (adjacent to the site) influences regional groundwater levels, especially in winter months when groundwater levels fluctuate between 101 and 103 m AOD. In February 1998 groundwater conditions under the landfill varied between 101 and 102m. Regression graphs comparing measured and estimated groundwater levels also confirm the difficulties of kriging February conditions (e.g. Figure 8.15). The February data sets gave a regression value of 59 percent while the September data set produced 96 percent.

The findings conclude that the heterogeneous February conditions cannot be adequately represented using low density sampling regimes such as those described in scenarios 1 to 4. The areas of greatest difference when comparing measured and kriged values are shown in Figure 8.16. In terms of the site assessment, although the four sampling scenarios were not able to identify more robust sampling locations for February 1998 conditions, the findings provided valuable information about the impact that regional hydrology (groundwater and river levels) on landfill groundwater fluxes.

September 1998 kriging simulations were successful. During this period regional groundwater levels were higher than in February, varying between 102 and 103m. Under these conditions the Sava River has a greater impact on groundwater levels and flow directions, directing groundwater into a south and southeastern direction. The kriging simulations presented in section 8.3.8 and 8.3.9 show it to be an effective tool for simulating groundwater conditions during this period. In the initial investigation, the measured and kriged data set mean and variance were quite similar (e.g. Table 8.9 and Figure 8.14). The scenario-based modelling that followed also provided successful results, illustrated in Figure 8.19 and 8.20.

The findings of all four scenarios were compared to determine which of the four sampling patterns would improve the accuracy of groundwater data collected during the site assessment. Scenarios 1 and 4 had the lowest mean data set error (e.g. 0.02 and 0.05 as shown in Table 8.11) Of these, scenario 1 had the lowest value, indicating that it would be the most effective sample distribution for field conditions. However, the February kriging indicated that heterogenous site conditions significantly influenced local groundwater levels. These conditions should not be overlooked as they could have misleading results for risk estimation models and remedial decisions that follow.

Scenario 4 (29 sample points) therefore provides the most effective distribution of sampling locations of the four scenarios modelled. However, further kriging analysis is required since the September kriging simulated only about 50 percent of sample points (points 13 – 26 in Figure 8.20 (b)).

In summarising, there are two conclusions that can be drawn from the kriging application at Site B:

- Given the level of groundwater variability around Site B, kriging showed that none of the four scenarios could effectively represent groundwater conditions at the site in Febraury 1998. Instead, scenario 4 was the only scenario that was able to simulate about half the sampled locations effectively
- More kriging analyses are needed. These analyses should change the distribution of sample points in order to verify whether other sample locations would improve the level of variability. Other monthly data sets (e.g. March and August 1998 or February and September 1999) should also be tested.

Despite the inconclusive finding, the investigation provided useful information that can improve accuracy of the Site B risk assessment. Firstly, kriging provided information about both small-scale and regional factors that influenced groundwater levels. This information would be difficult to infer unless a longer-term groundwater monitoring program was established. Kriging can, therefore, be used during the site assessment to infer information about hydrogeological variations using historical data, therefore improving the amount of data that can be derived from historical measurements. Secondly kriging is most effective when there is a clear objective to the site asessment. For example, if the site assessment objective at Site B was to assess regional groundwater impacts on the landfill, then the sampling patterns in scenario 1 or scenario 4 would have been effective. If the objective was to infer groundwater levels at only one part of the landfill, for example the area covered by sampling points 15 -25 in Figure 8.20, then parts of scenario 4's sampling pattern could be used to conduct this assessment. The main point to stress is that kriging analysis will produce the most effective results if the objective of the site assessment is clearly defined.

8.3.11 Investigation 1: Kriging Conclusions

The purpose of the kriging investigations was to evaluate whether kriging could be used as a tool during the site assessment, to assist in locating new sampling points and evaluating the distribution and effectiveness of existing sample points.

The findings at Site A indicated kriging can be used to assess the effectiveness of existing sampling points. It can also be used to evaluate the number and location of further sampling points. This was illustrated in section 8.3.6. The investigation concluded that scenario 3, in which 21 sampling points are added would provide the most spatially adequate sampling locations if measuring groundwater variability across the landfill. The study findings can be strengthened by continuing with further kriging analyses in areas identified in Figure 8.10, for which estimations were not accurate. Other patterns of sample distribution should also be investigated in order to verify the findings of this study.

The investigation conducted using the Site B data sets found kriging to be an effective method of improving the understanding of patterns of groundwater flow around the landfill. However, estimations conducted using Site B produced results that differed from those found in Site A. Despite the differences, the results were useful as they indicated the flexible application of kriging as a site assessment tool. In the case of Site B, kriging simulations were not accurate, however the analysis provided spatial information about the level of variability in groundwater flow across the site. The inability to simulate conditions also confirmed previous site assessments which identified that seasonal and regional factors are important elements that influenced landfill groundwater levels. Further analyses (focusing upon the sampling pattern in scenario 4) would strengthen the findings of this investigation. An important conclusion of this investigation was that kriging, when used to improve the understanding of data sets collected during the site assessment, needs to have a clear objective from the start of the investigation.

The investigation conducted using both Sites A and B confirmed that kriging is an effective tool with which to verify whether existing or planned sampling locations represent heterogeneous groundwater conditions at a landfill. The information provided by kriging can confirm or provide insight into geophysical site conditions around such sites. It is also an effective tool for identifying zones of a landfill that may require further sampling or may be influenced by localised hydrogeological conditions. The most powerful feature of kriging is its ability to optimise further sampling locations by estimating the maximum number of sampling points needed, providing site-specific sampling strategies, depending on the scope and risks being evaluated during a site assessment.

8.4 Investigation 2: Ground Penetrating Radar

Another approach to minimising the geophysical uncertainty during the site assessment is to use several monitoring technologies that provide different scales of information on site conditions. Ground Penetrating Radar (GPR) is one such method that can provide non-intrusive and spatially distributed information about subsurface conditions which can then be integrated with other data sets in GIS to produce layered maps of site conditions. Risk estimation models can then use this information for model calibration and validation. Both study Sites A and C were used to test GPR as a method of identifying groundwater levels and leachate near unlined landfills.

8.4.1 Application of Ground Penetrating Radar at Site A

The study objectives were to map and locate:

1) Leachate-groundwater levels and near-surface contaminant plume paths
2) Near-surface geologic features
3) Landfill depth, cap thickness, buried waste boundaries and
4) Spatial variations in subsurface features that could be added to the site-specific groundwater flow and contaminant transport model.

Figure 8.21 shows the survey location conducted at Site A. The survey lines were 30m in length placed along leachate-leaking edges of the site, at cells 1, 3, 5 and 10. Investigations were conducted under sunny and dry conditions in August 1999 and June 2000. The survey location was based on two sources of information. Firstly, the findings of the ongoing field assessments (conducted from January 1999 to June 2000) and secondly on contour models produced in GIS using groundwater and leachate concentration data sets. All the site assessment findings were compiled into a GIS database which used models that identified 'hot spot' areas of groundwater recharge and leachate fluctuation, e.g. Figure 7.6. These findings were validated using aerial photographs and field data. GPR was then used to investigate these 'hot spot' areas.

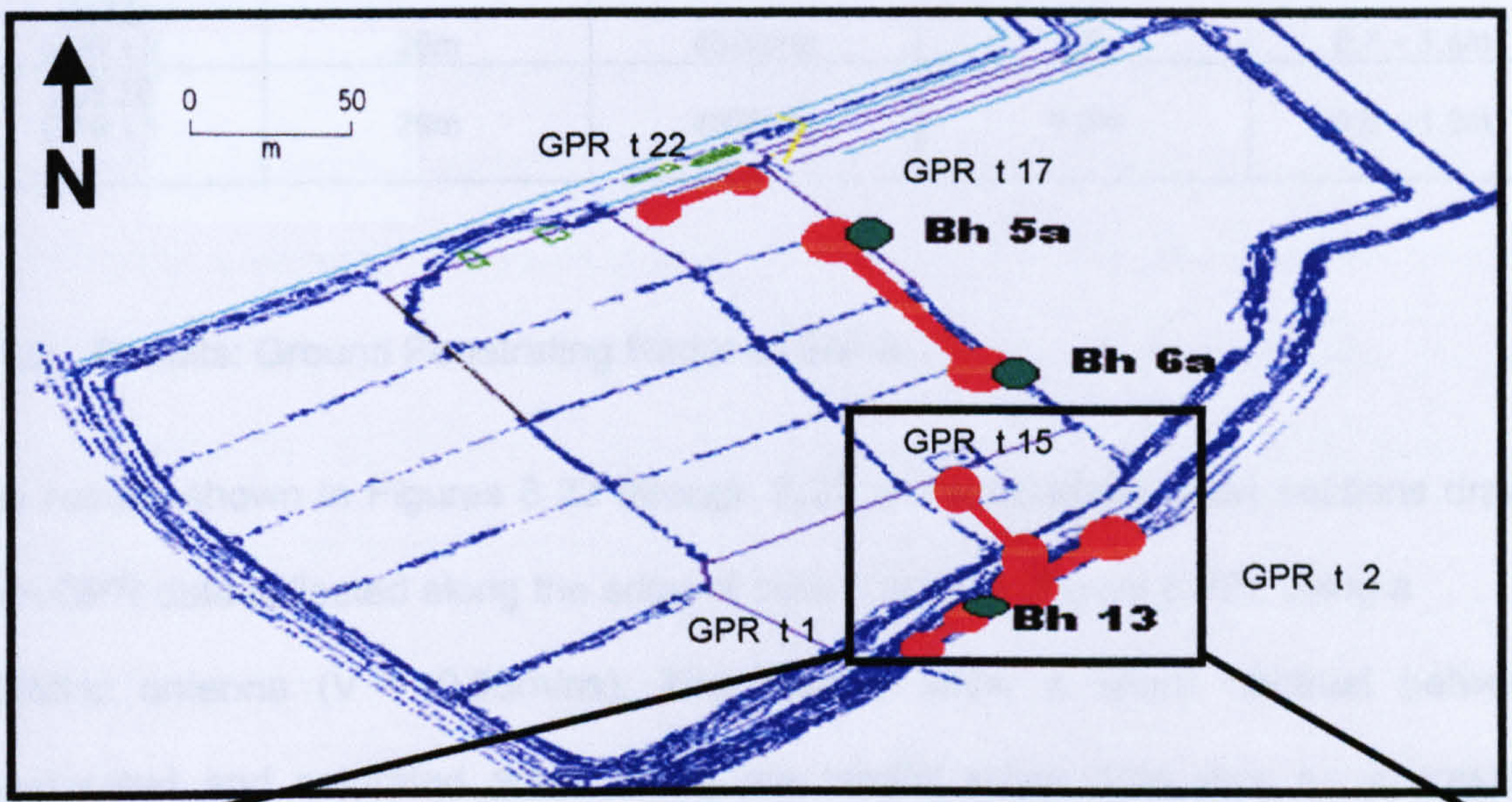
The local geology was interpreted from regional geology maps, consultant reports, and geological profiles taken from borehole drilling logs, e.g. Figure 8.29. The parameters used to collect and analyse GPR data are listed in Table 8.12. Cells 5 and 10 were of particular interest (Figures 8.21 and 8.22) since site assessment findings and GIS modelling indicated that this area served as a path for off-site leachate migration.

8.4.2 Data Processing Methods

Analysis was carried out using a variety of site assessment information. The 450 MHz antennas provided the clearest image of site conditions along the edge of cells 10 and 5 using a velocity of 0.06m/ns (V = 0.06 m/ns) showing a depth of 4m (Table 8.12). A sensitivity analysis of the velocity parameter was conducted testing values between 0.06 - 0.12m/ns. The results showed that the depth of the saturated-unsaturated interface was depressed by 2.5m when velocity values (V) were increased from 0.06 to 0.12m/ns. The cross-sections were validated at 0.06m/ns using geological maps of the site, drilling logs from boreholes 13, 5a and 6a and the chart of typical GPR subsurface patterns (e.g. Figure 4.6) from van Heteren *et al* (1994) in Smith and Eccles (1998).

Figure 8.21 (Top) Site A showing GPR survey locations conducted in August 1999 and June 2000 showing cells 5 and 10 in which leachate seepage and recharge occurred frequently

Legend:
Borehole = Bh
GPR transect names = t1, t2, t15, t17 and t22
= GPR transect



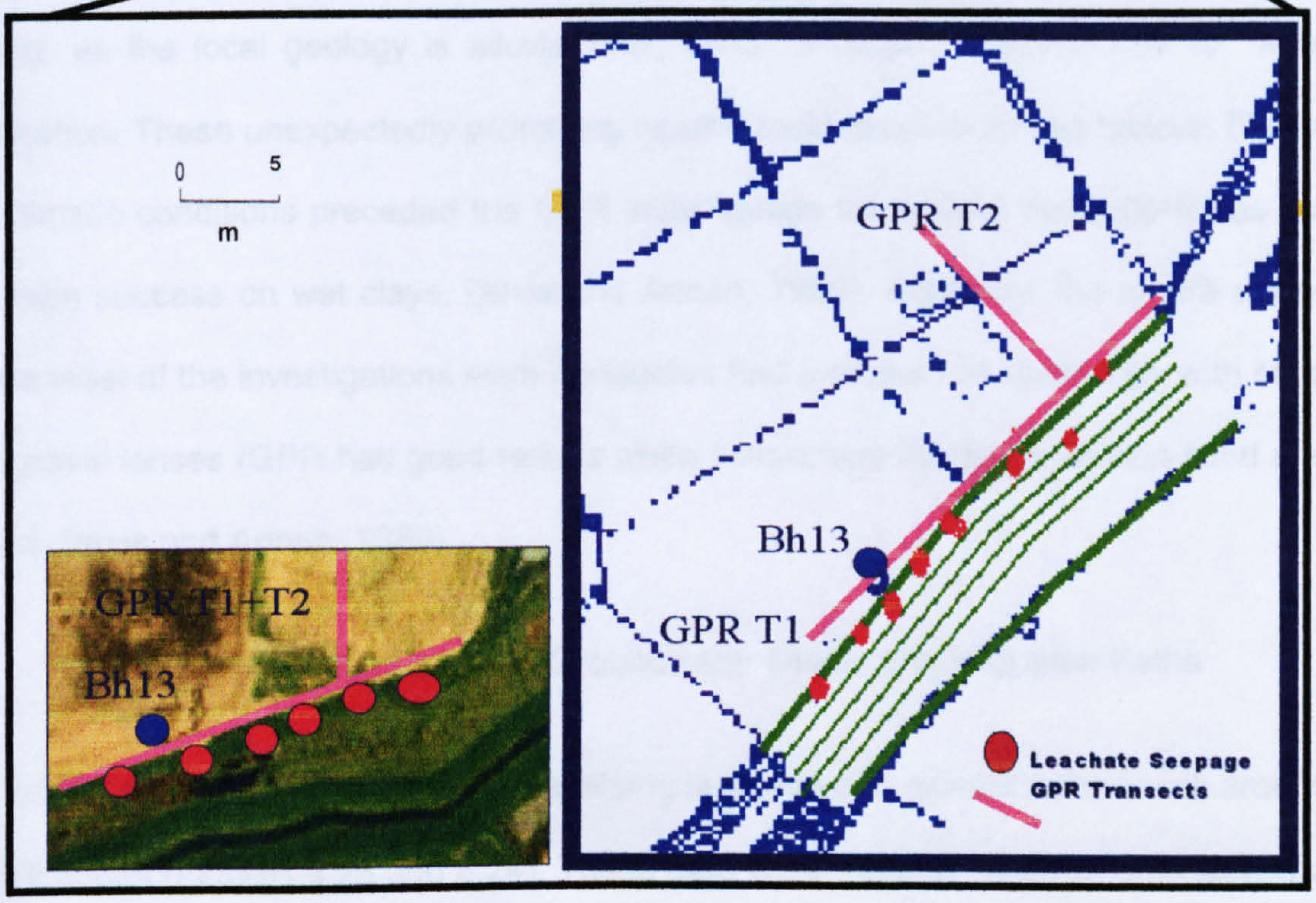


Figure 8.22 (Bottom) GPR survey lines (pink lines, red dots indicate leachate seepage points along the landfill edge) Monthly groundwater levels measured at Site A were spatially analyses in GIS to identify aeas where Ground Penetrating Radar survey could be conducted

Table 8.12: Locations and information about each GPR transect collected at Site A as shown in Figures 8.21 and 8.22

Cell and Transect Name	Transect Length	Antenna Used	Depth of GPR Image	Leachate-Water Level
Cell 1				
GPR t 22	30.4m	225MHz	3.5m	1.5 – 3m
	28	225MHz	6.4m	2.4 – 5.6m
Bh 5a & Bh 6a				
GPR t 17	30m	225MHz	4m	0.5 – 2m
GPR t 15	28.5m	225MHz	4m	1.5 – 2.5m
Cell 5				
GPR t 2	26m	450MHz	2m	0.7 – 1.4m
Cell 10				
GPR t 1	29m	450MHz	1.6m	0.8 – 1.2m

8.4.3 Results: Ground Penetrating Radar at Site A

The results shown in Figures 8.23 through 8.27 are subsurface cross-sections drawn from GPR data collected along the edge of cells 5 and 10 (Figure 8.22), using a 450MHz antenna (V = 0.06m/ns). The images show a sharp contrast between unsaturated and saturated areas along the landfill edge. This was an interesting finding, as the local geology is alluvial clay, which is usually unfavourable for radar application. These unexpectedly promising results could result from two factors. Firstly, dry climatic conditions preceded the GPR investigation for several days (GPR has not had high success on wet clays, Davis and Annan, 1989). Secondly, the landfill edges where most of the investigations were conducted had a mixture of sandy clay with sand and gravel lenses (GPR had good results when subsurface materials contain sand and gravel, Davis and Annan, 1989).

1) Objective 1 Results: Leachate-Groundwater Levels and Migration Paths

The investigation was successful in identifying leachate and groundwater levels around landfill edges (Figures 8.23 and 8.24). These data were used to calibrate qualitatively a groundwater flow and contaminant transport of the site, verifying simulated groundwater levels in the GPR-investigated areas, e.g. Figure 8.26.

2) Objective 2 Results: Near-Surface Features

The method was successful in delineating subsurface landfill features, e.g. Figure 8.24, including estimating cap thickness and waste depth along unlined parts of the landfill. The depths were validated by comparing images at cells 10 and 5 with preliminary study results, and images taken at other boundary areas of the landfill. Investigations in June 2000 used lower frequency antennas (the EKKO 100 GPR was used with 200 and 100MHz antennas). These investigations also confirmed landfill cap thickness and waste depth. The combined findings provided structural information that was not documented in the site's historical records, providing qualitative information, which was useful during the construction of a groundwater flow model.

3) Objective 3 Results: Geologic Features

GPR identified and confirmed sand-gravel lenses along the landfill edges that serve as paths for off-site leachate migration. Data from across the site were compared to identify similarities in GPR data collected at leachate-leaking parts of the landfill. Geological profiles of these areas confirmed sand-gravel lenses as potential paths of migration, e.g. Figure 8.25. The location of these lenses was mapped and used to accurately distribute hydraulic conductivity and related hydrogeological parameters in the site's groundwater flow model.

4) Objective 4: Conceptual and Groundwater Flow Modelling

The GPR images provided valuable hydrogeological and geophysical information that helped validate site conditions, increasing the understanding of hydrogeological conditions, which also increased the accuracy during groundwater flow model construction. The data derived from these investigations helped to optimise the models through:

- identifying areas of higher conductivity, e.g. Figure 8.25, by locating sand-gravel lenses along the landfill edges that act as leachate migration pathways
- identifying real-time groundwater-leachate levels, e.g. Figures 8.23 and 8.24. These data were used to establish parameter ranges for model boundary conditions, e.g. Figure 8.26
- providing previously unknown structural data about subsurface landfill conditions, e.g. Figures 8.24 and 8.25
- providing three types of modelling information. Firstly, quantitative information important for model construction. E.g. Figures 8.26 and 8.27 show the transfer of waste depth, groundwater levels and sand-gravel locations mapped by GPR into the model domain. Secondly, qualitative data for model calibration, e.g. Figures 8.25, and 8.26. Thirdly, a range of values for hydraulic conductivity parameters derived from GPR images that identify sand-gravel lenses.

Although the investigations did not provide quantitative values of hydraulic conductivity or exact measurements of groundwater-leachate levels, they did improve the conceptual accuracy of regional and local hydrogeological conditions at Site A.

Figure 8.23: A cross-section interpreted using GPR data, identifying groundwater and leachate levels along edge of cells 10 & 5 (the survey used a 450 MHz antenna, over a 30 m transect)

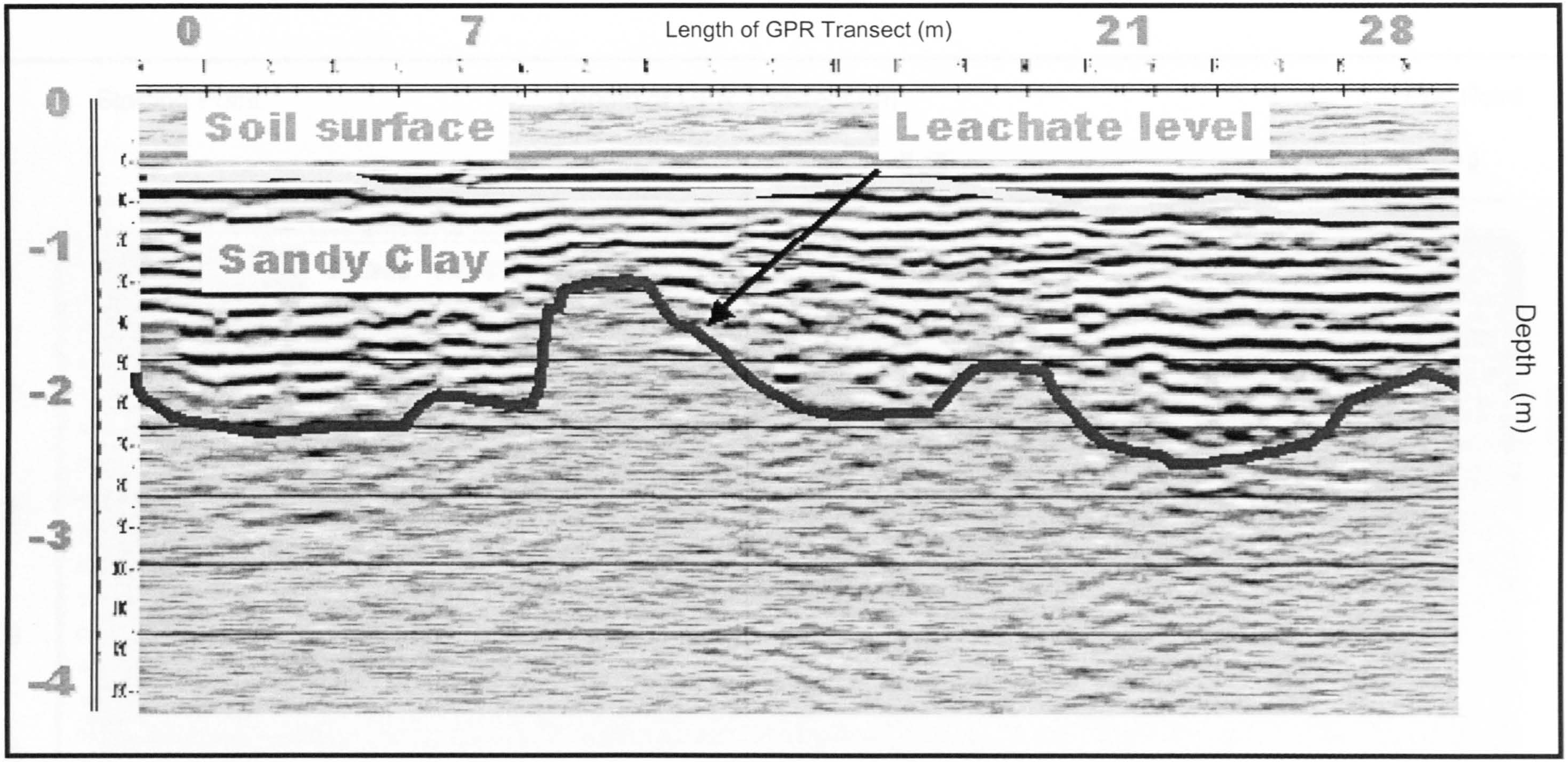


Figure 8.24: A cross-section view of the landfill edge at cell 5, near leachate breakout points showing leachate levels at cell 5, and measuring landfill cap thickness and height of buried waste (A 450 MHz antenna was used)

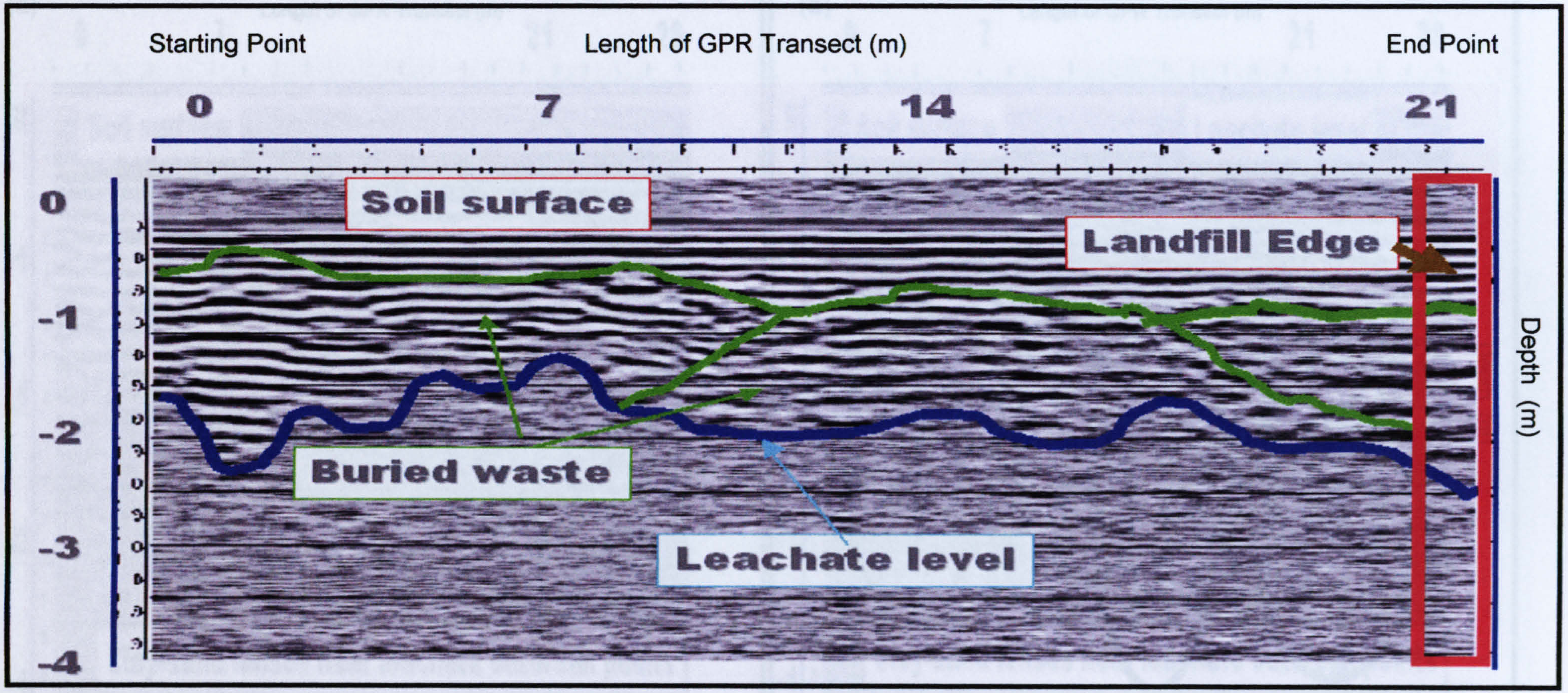


Figure 8.25: Objective 3 aimed to identify sand-gravel lenses along the landfill edges: (a) shows the unlabelled GPR image; (b) shows the labelled GPR image (A 450 MHz antenna was used)

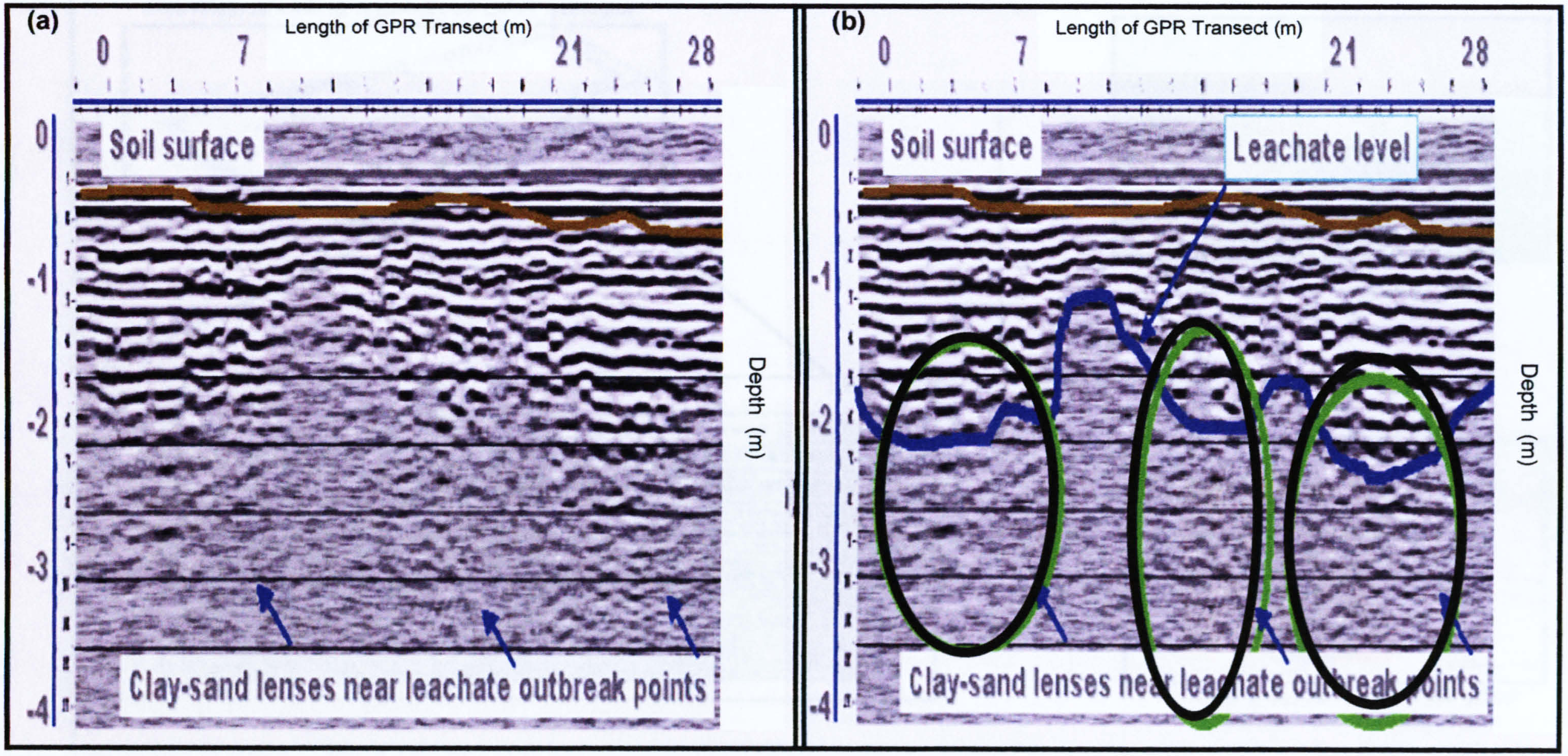


Figure 8.26: Objective 1 aimed at estimating leachate levels within unlined parts of the landfill by comparing reflection differences at cells 10 and 5 with historical site maps, other survey results and other landfill GPR images

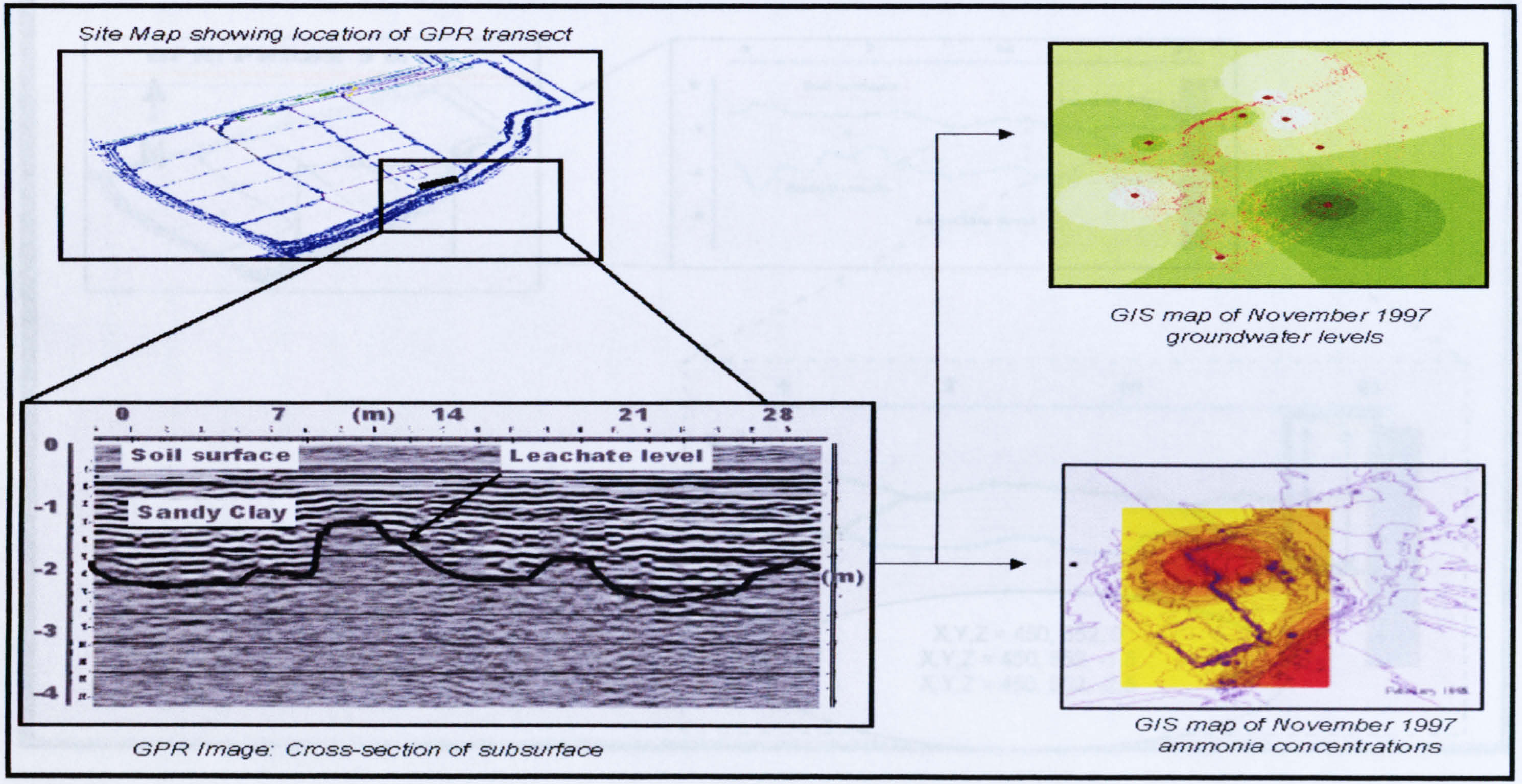
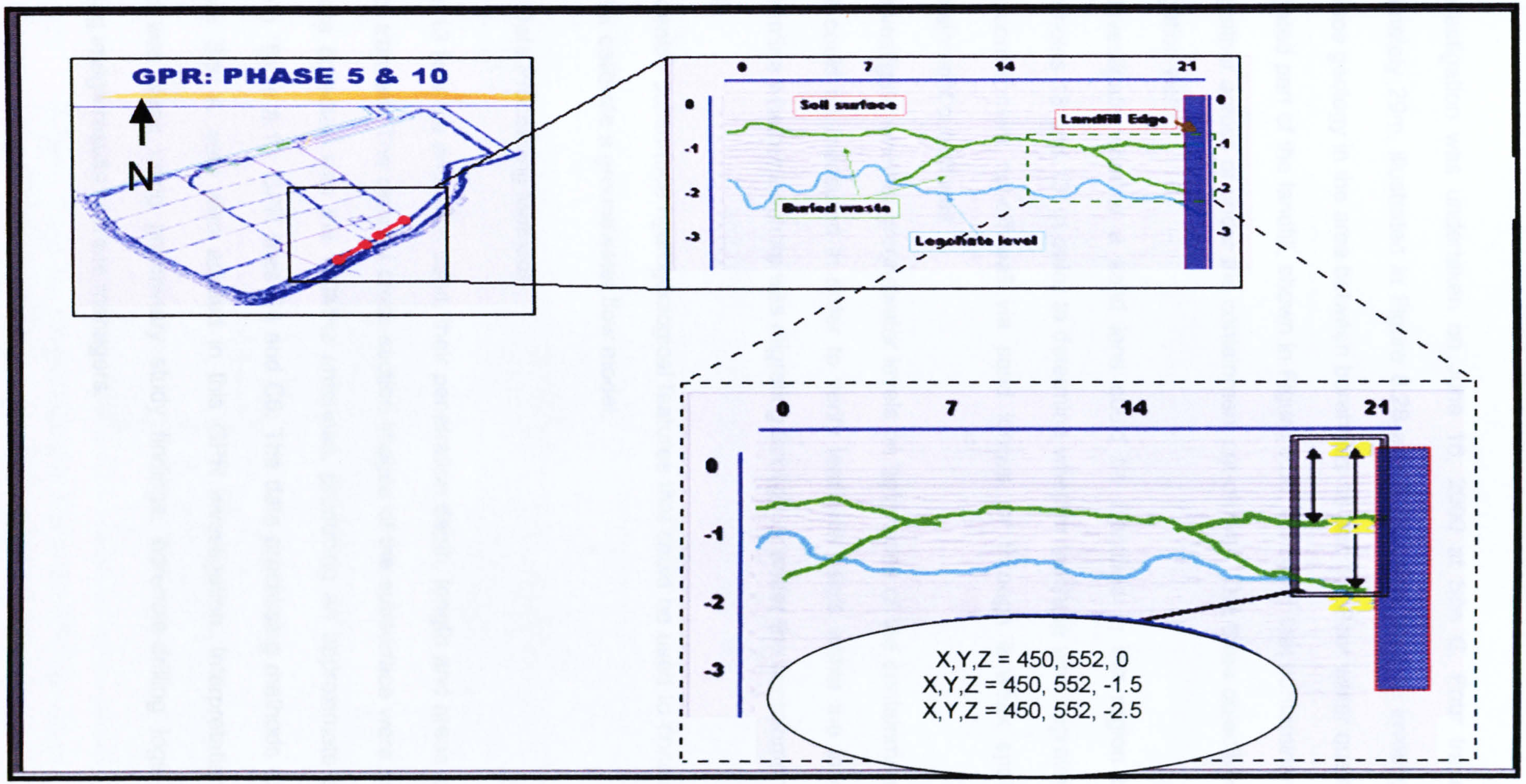


Figure 8.27: Objective 2 aimed at estimating cap thickness and waste depth along unlined parts of the landfill in which geographic locations (using Global Positioning System) were assigned to each survey line



8.4.4 Applications of Ground Penetrating Radar at Site C

This investigation was undertaken on June 16, 2000 at Site C. Four transects of approximately 29m, illustrated in Figure 8.28 and 8.30, were used to investigate the subsurface geology in the area between boreholes 18 and 13. Poor water quality in the investigated part of the landfill, shown in Figure 8.30, indicated that contaminants were moving either across or under the containment cut-off wall. The three objectives of this investigation were:

1) to investigate whether a sand lens could be identified in the region between boreholes 18 and 13, in order to determine whether leachate was migrating under the containment cut-off wall via sand lenses, or through a weak spot in the containment cut-off wall
2) to investigate whether groundwater levels on both sides of the containment cut-off wall could be measured in order to verify leachate levels within the landfill and determine whether leachate was migrating through or under the containment cut-off wall
3) to identify subsurface hydrogeological features that could be used to construct and better calibrate a groundwater flow model.

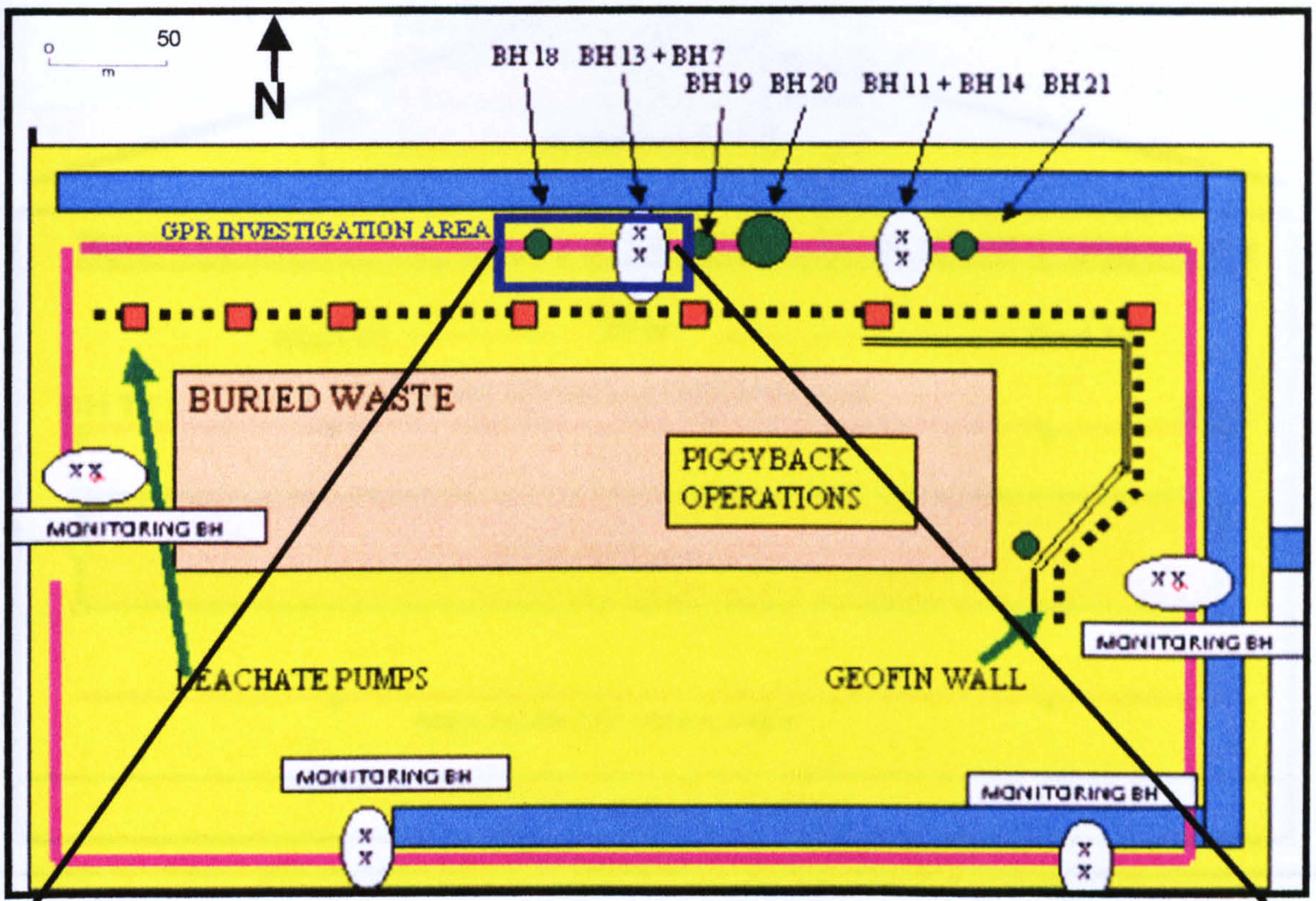
8.4.5 Data Processing Methods

Table 8.13 lists the antennas used, their penetration depth, length and areas identified for each transect. The clearest cross-section images of the subsurface were produced from data collected with the 100MHz antennas, producing an approximate depth of 10m, e.g. Table 8.13, GPR lines C4 and C6. The data processing methods explained for study Site A were also applied in this GPR investigation. Interpretation of the images was done using preliminary study findings, borehole-drilling logs and by reviewing image results with site managers.

Figure 8.28: (Top) Map of contaminant monitoring across Site C and GPR survey lines

Legend:

= leachate pump
= geofin wall
xx = monitoring boreholes
= monitoring boreholes (BH)
= GPR investigated area



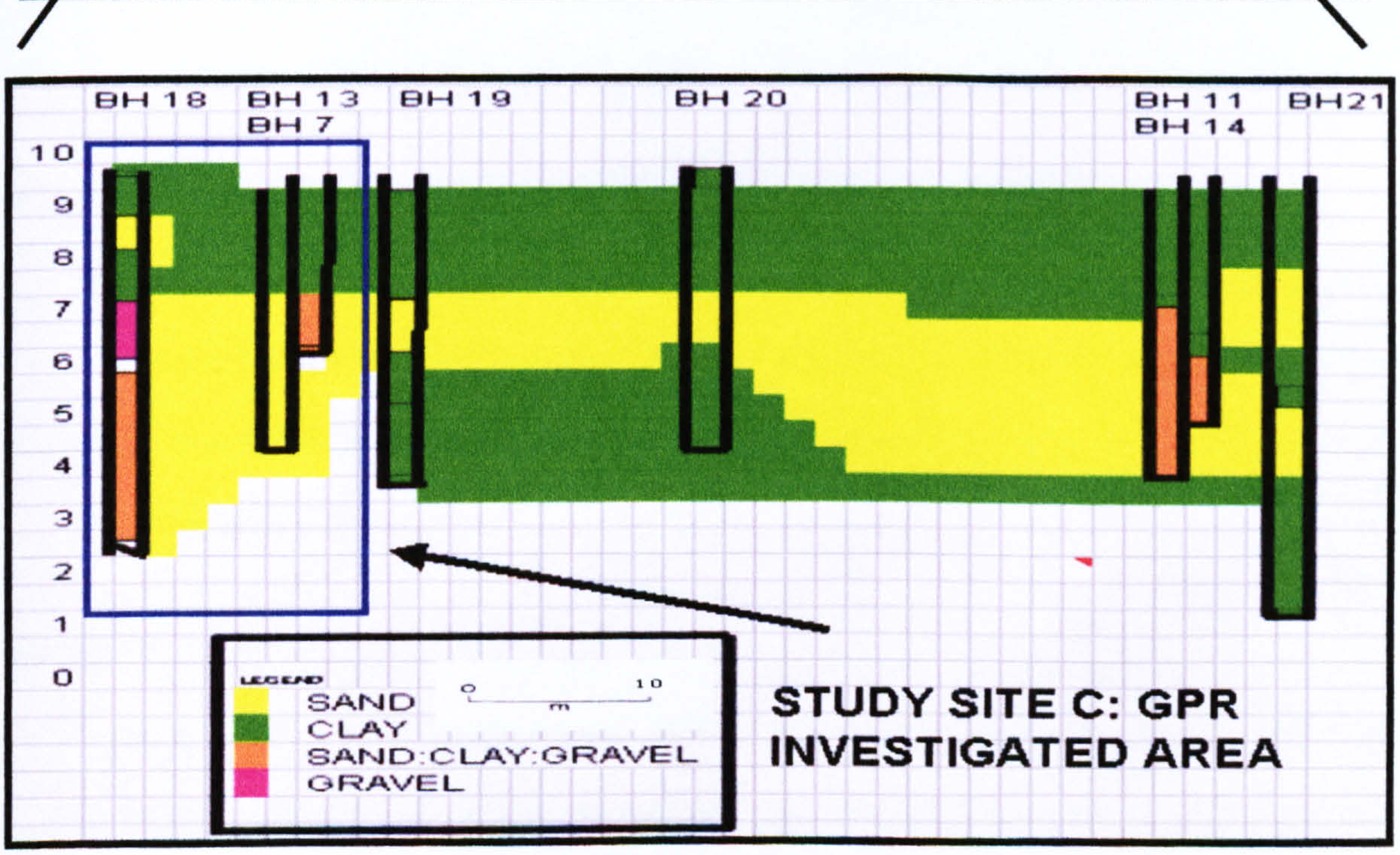


Figure 8.29: (Bottom) Cross-section geological profile of the GPR surveyed area at Site C

Figure 8.30: Close up aerial view of GPR investigated area at Site C

Legend:
Starting and ending point of GPR transect = Start Pt/ End Pt
Boreholes used to calibrate GPR parameters and validate cross-section images = BH 18 and BH 13
GPR transect names = CAR3 – CAR 8

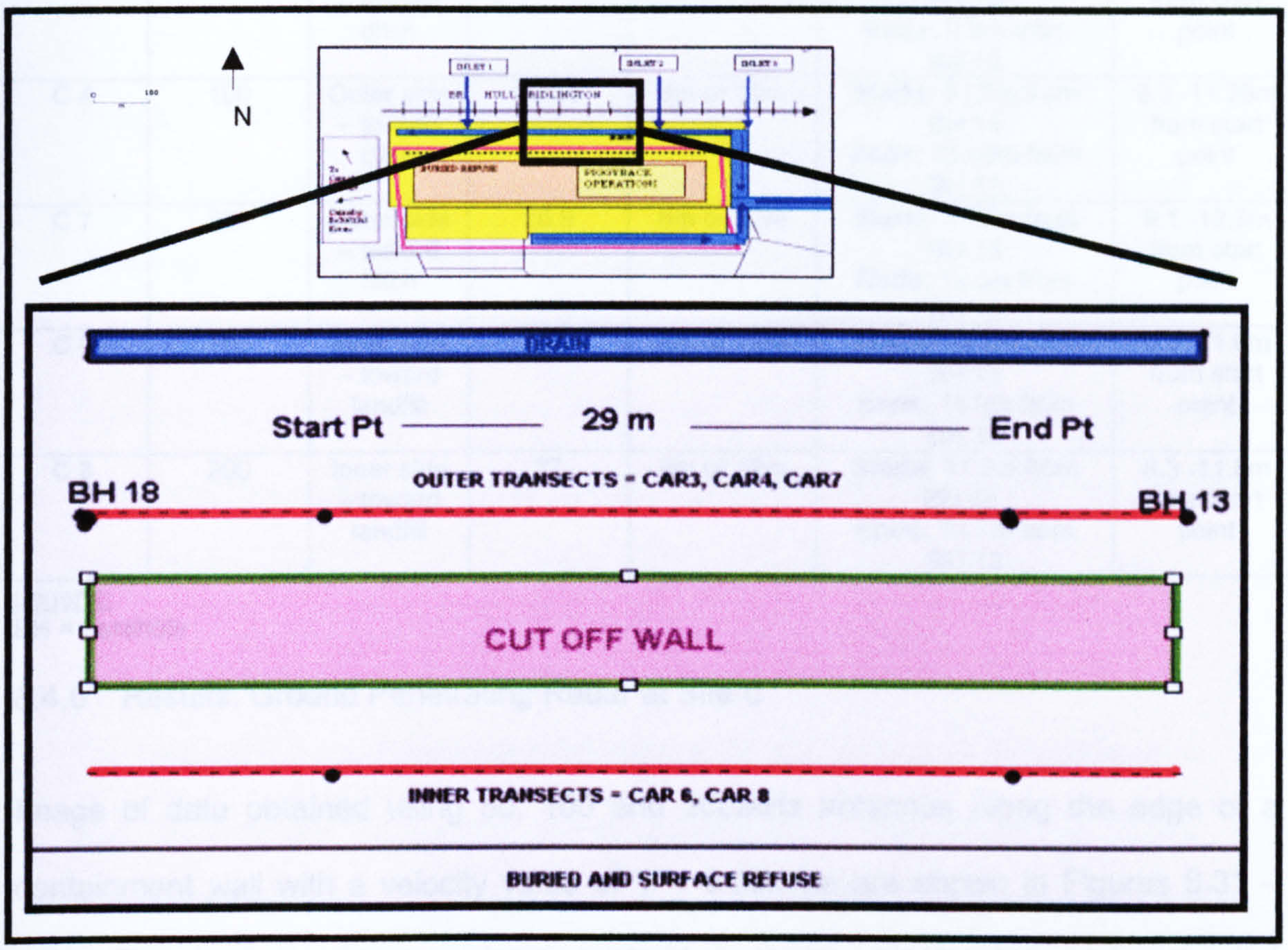


Table 8.13 Information about GPR survey lines collected at Site C

Transect Name	Antenna MHz	Location along Cut -Off Wall	Length of Transect (m)	Interpolated Transect Depth (m)	Location of Transect	Hot Spot
C 3	50	Outer side – toward ditch	27.8	8m or 12m	**Starts:** 11.5m from BH 18 **Ends:** 9.9m after BH 13	10.8m from start point
C 4	100	Outer side – toward ditch	20.55	8m or 12m	**Starts:** 11.5m from BH 18 **Ends:** 18.65m from BH 13	9.3 -11.25m from start point
C 7	200	Outer side – toward ditch	26.9	8m or 12m	**Starts:** 11.5m from BH 18 **Ends:** 12.3m from BH 13	9.1 -12.3m from start point
C 6	100	Inner side – toward landfill	26.5	8m or 12m	**Starts:** 11.5m from BH 18 **Ends:** 14.2m from BH 13	9.0 -11.0m from start point
C 8	200	Inner side – toward landfill	27	8m or 12m	**Starts:** 11.5m from BH 18. **Ends:** 13.7m from BH 13	8.3 -11.8m from start point

Legend:
BH = borehole

8.4.6 Results: Ground Penetrating Radar at Site C

Image of data obtained using 50, 100 and 200MHz antennas along the edge of a containment wall with a velocity value of V = 0.06m/ns are shown in Figures 8.31 – 8.34. The images show a sharp contrast between unsaturated and saturated subsurface features along the wall where leachate seepage occurred.

1) Results: Geologic Features

   GPR data enabled the identification of geological features at a depth of about 7m. A comparison of images, e.g. Figure 8.31, suggests a feature (approximately 1m thick) that extends along the transect length which is likely to be the sandy layer at 7m, indicated in Figure 8.30(b) as the yellow zone.

2) Results: Containment Cut-Off Wall Leakage

   A general unsaturated environment across both sides of the containment wall was identified, with the exception of a saturated zone at 9 through 11.5m from the transect starting points on both sides of the containment wall in the region between boreholes 18 and 13, e.g. Figure 8.32. Site conditions were dry during the investigation. By comparing images on both sides of the wall the investigation was able to confirm that a zone of saturation was present at the same point on both sides of the containment cut-off wall, e.g. Figure 8.33 for the inner side of the wall and Figure 8.34 for the outer side of the wall. The results using different antennae are compared in that there is a saturated zone on both sides of the wall. This feature is likely to be either a weak point in the bentonite clay containment wall or a higher permeability sand-gravel lens that was dissected when the wall was constructed. Both possibilities point to a leak in the containment wall, explaining the link between increased contaminant concentrations in nearby boreholes, e.g. Figure 8.30(d). Site records taken during the construction of the containment cut-off wall show that a weakness was noticed during construction, further confirming the conclusions.

3) Results: Hydrogeological Features

The images provided geological information that is critical in determining the next step of leachate containment and remediation. As discussed above, two conceptual models were possible. This data were used to construct two hydrogeological scenarios in the site's groundwater flow model. The first scenario assumed that a sand lens was located at about 7m from the surface. The second scenario assumed that there was a change in hydraulic conductivity and flow conditions on both sides of the containment wall, simulating the saturated weak zone inferred from the investigation's subsurface image. Both scenarios were tested, varying hydraulic conductivity values to verify whether the feature was a sand lens or whether it was a weak point in the wall. More GPR investigations are needed to determine which of the two conceptual models is correct. However, the information that was inferred from this investigation provided valuable hydrogeological information, which was helpful during model construction when constructing the different model scenarios and when distributing hydrogeological parameters within the mode domain.

Figure 8.31: Inner and outer wall images using a 100 MHz antenna showing a sand layer or gravel cell at about 7 m from the surface (see Figure 8.30(b))

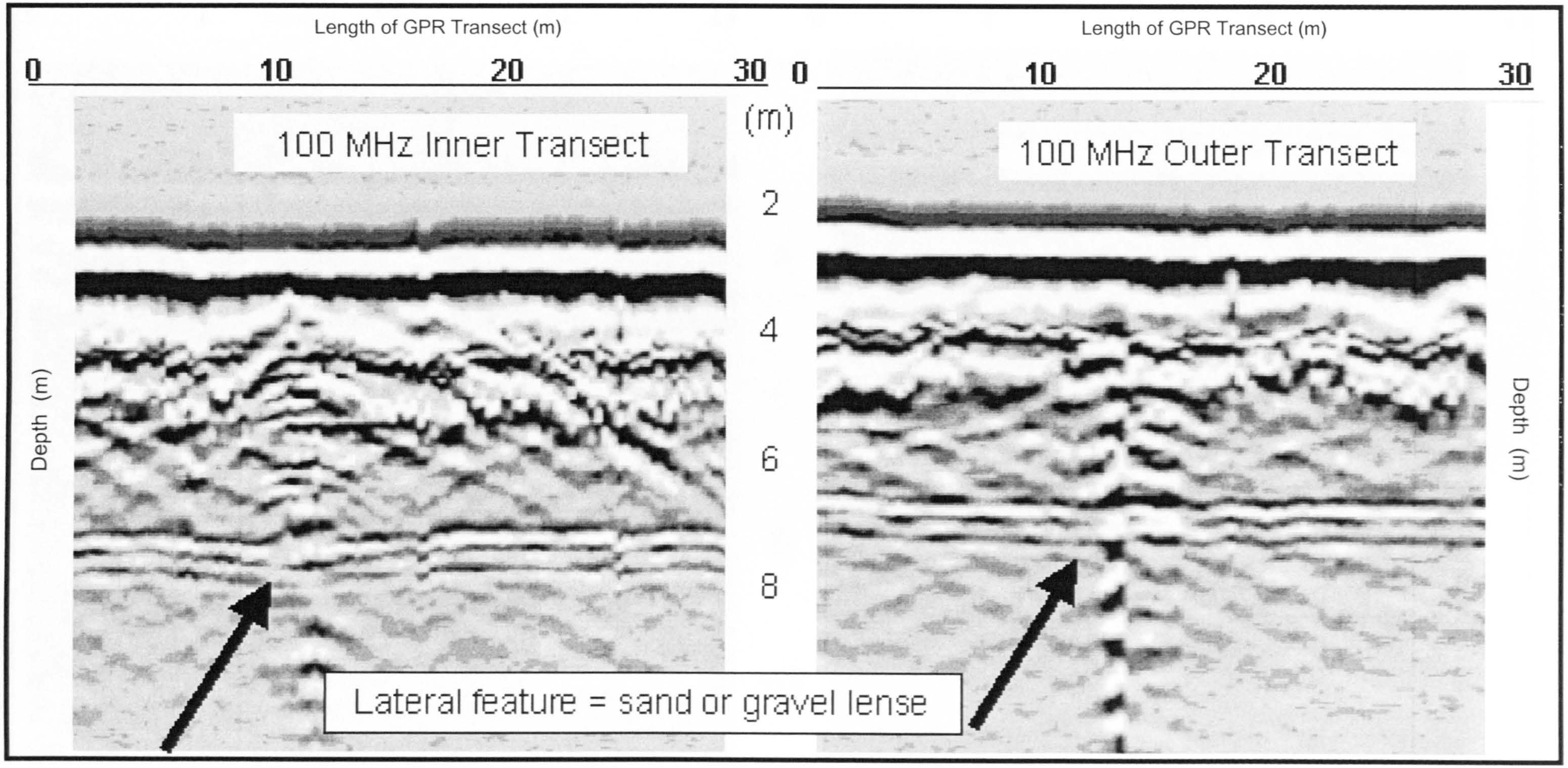


Figure 8.32: Comparison of the inner and outer 100 MHz transects showing a feature at the same location on both sides of the containment cut-off wall, suggesting a saturated zone or change in geology

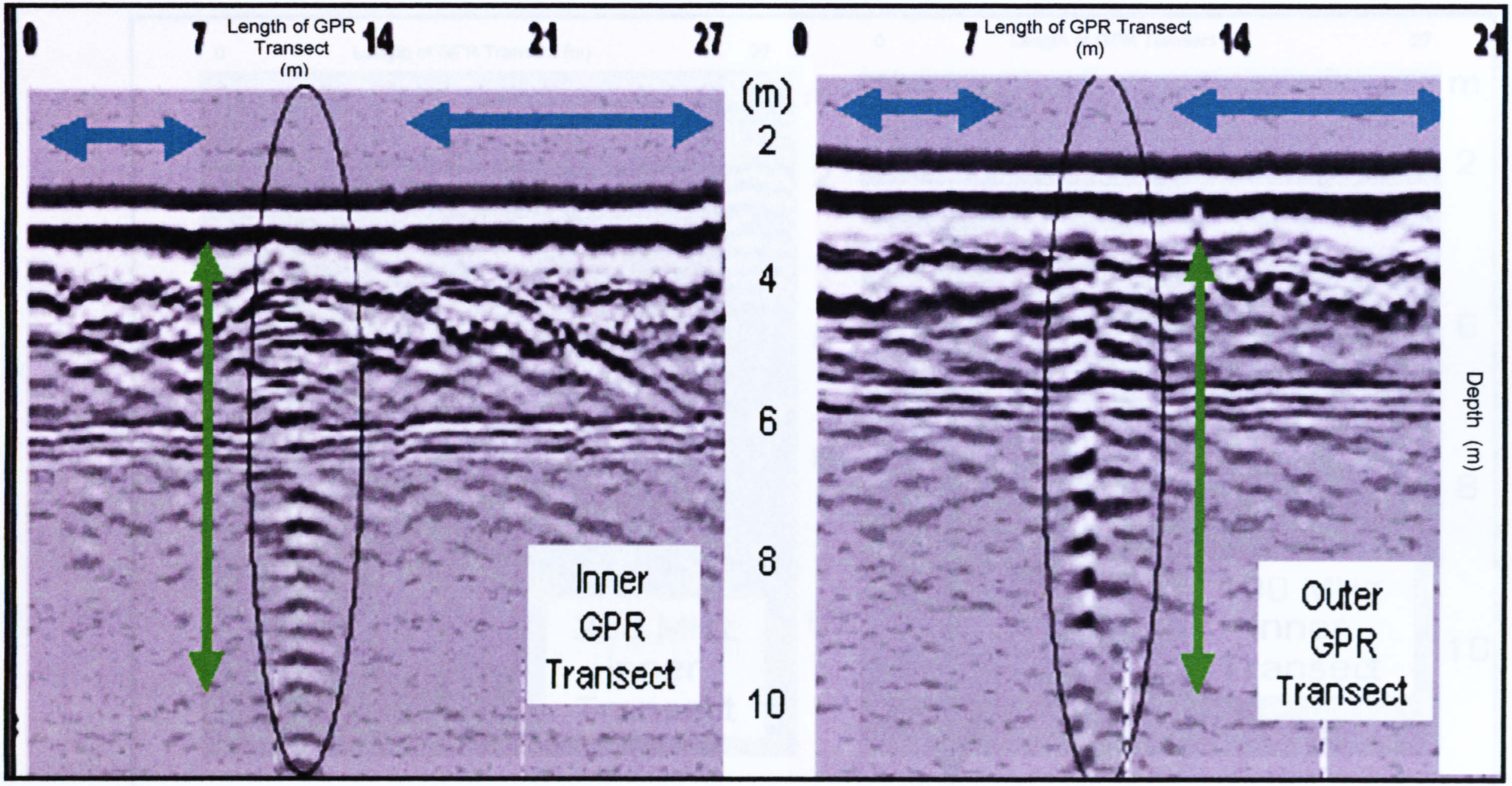


Figure 8.33: Comparison of the inner 100 and 200 MHz transects showing a feature at the same location on the inner side of the containment cut-off wall, suggesting a saturated zone or change in geology

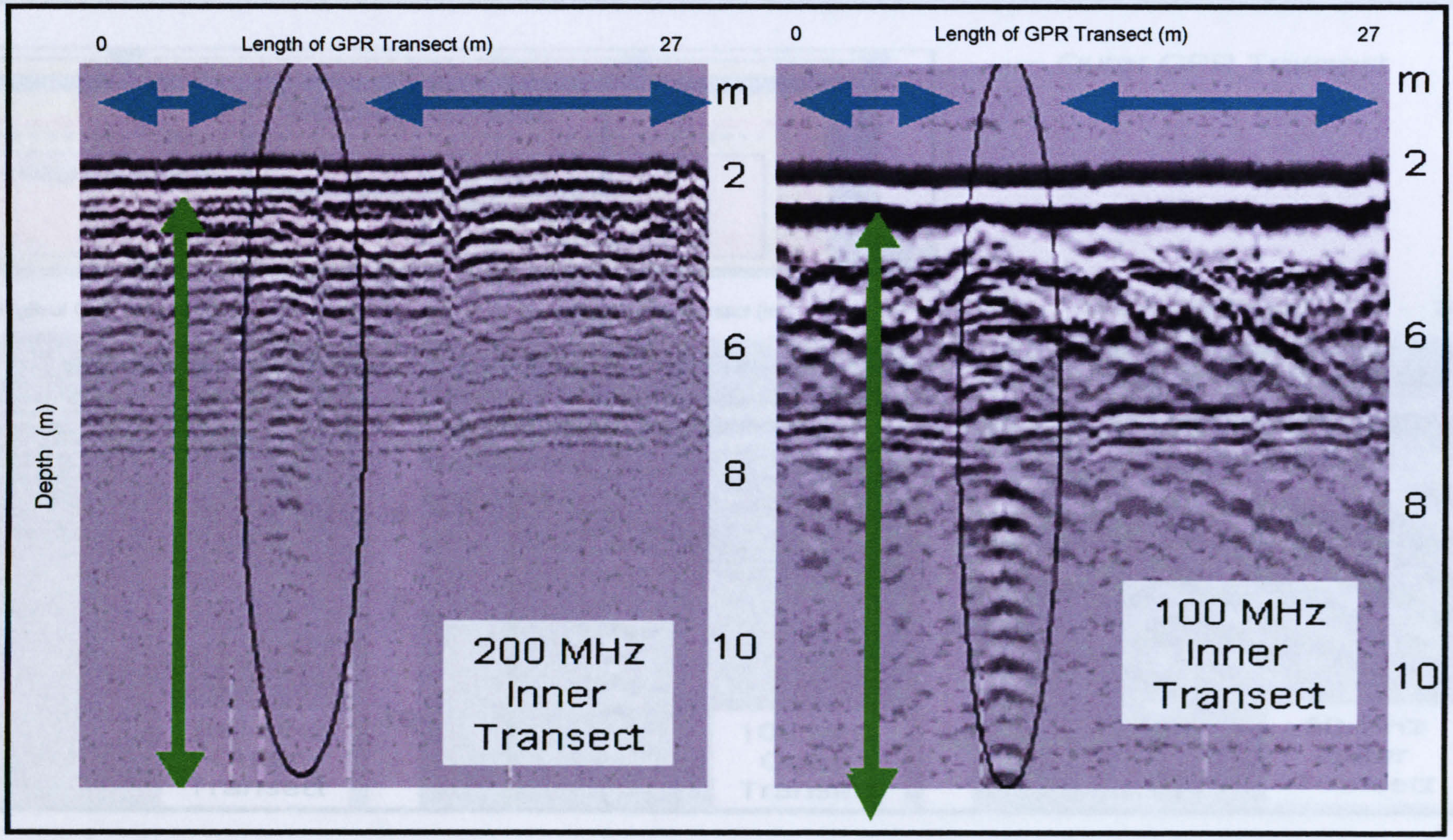
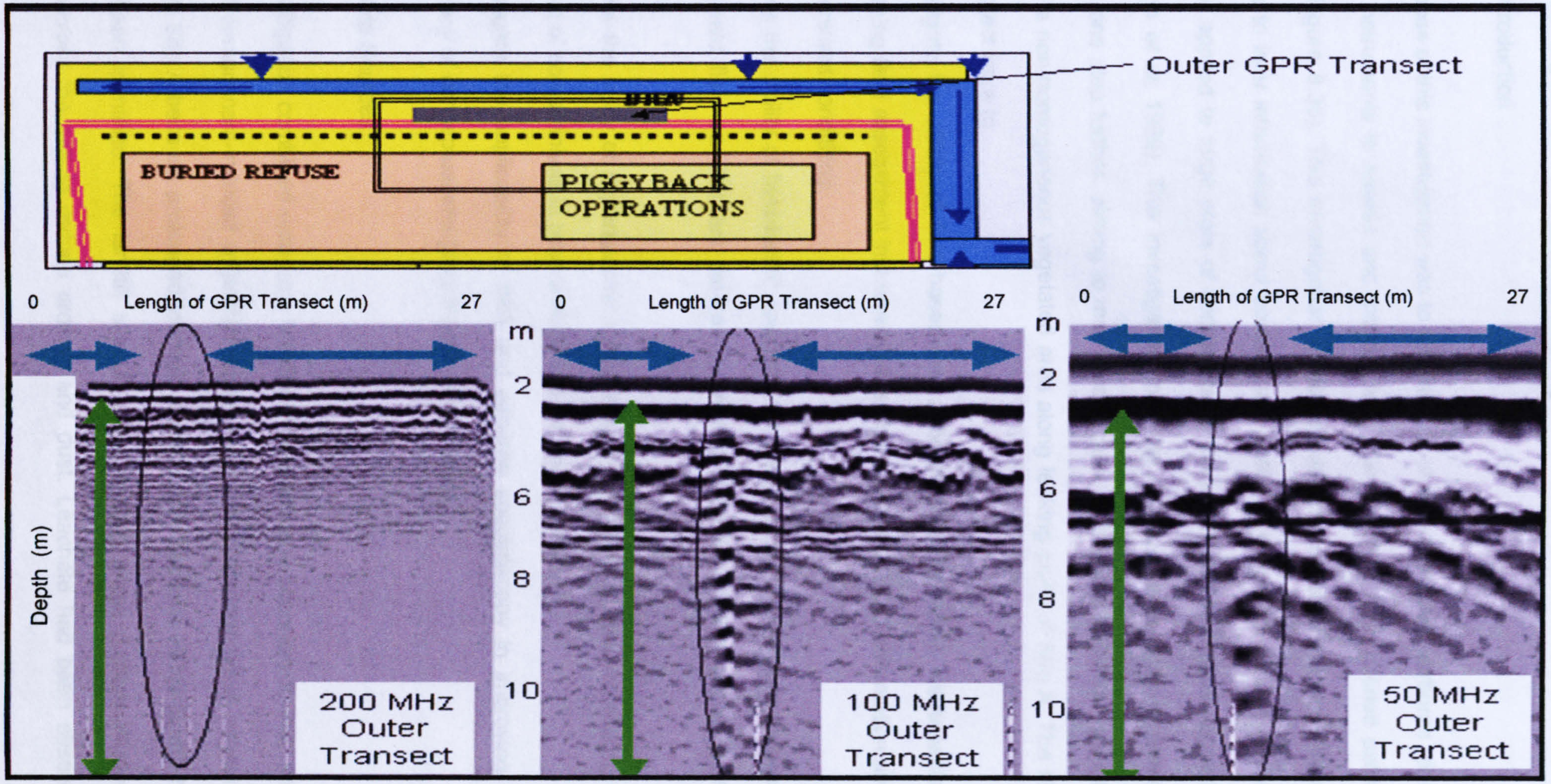


Figure 8.34: Comparison of the outer 50, 100 and 200 MHz transects showing a feature at the same location on the outer side of the containment cut-off wall, suggesting a saturated zone or change in geology



8.5 Investigation 3: Remote Sensing

8.5.1 Introduction

The purpose of this investigation was to test the use of field-based and airborne remote sensing instruments to assess and monitor leachate migration from unlined parts of Site A (Figure 8.35). This investigation combined data from hand-held and airborne scanners to infer information about contaminant paths of migration. Similar research has been applied to large plots of contaminated land with homogeneous vegetation (e.g. Jago *et al*, 1999). This investigation took the procedures outlined in previous research one step further, aiming to map small-scales of leachate-induced vegetation stress at a non-homogeneous vegetated area along leaking parts of Site A. The aims of the project were to:

1) Investigate the ability of field-based and airborne spectrographic instruments in becoming field assessment instruments that can provide detailed information about contaminant conditions
2) Assess the ability of field-based spectroradiometers to become industry-accepted hand-held field assessment devices to monitor contaminant-induced vegetation stress
3) Assess the utility of hyperspectral CASI imagery in mapping and monitoring the spread of contaminants in near-surface soil on landfill sites
4) Investigate the applicability of field and airborne spectroscopy in improving the accuracy of conceptual and groundwater flow models.

8.5.2 Data Sources

This investigation combined data from hand-held and airborne scanners to map small scales of contaminant-induced vegetation stress over a non-homogeneous vegetated surface at Site A because prolonged leachate escape from sections of the landfill had already been identified. The landfill site under study had grass (*Lolium Perenne*) planted across the site to reduce erosion and dust. Leachate had been observed escaping continuously from cells 5, 10 and 11 since 1997 with a marked increase noted after rainfall. Data from historical maps, monitoring data, hydrogeological reports, company records, and aerial photographs were compiled into a GIS to provide background information about conditions at Study Site A, e.g. Figures 8.36. An analysis of aerial photographs from 1992 through to 1999, all acquired during the summer, combined with preliminary field investigations identified that patches of stressed and dying vegetation growing up to 10m from the edges of cells 5, 10 and 11 had developed in recent years. The restricted area of this landfill, the non-natural source of the vegetation (giving an almost homogeneous vegetation cover), the constant soil and drainage conditions allows patches of unhealthy vegetation to be used as indicators of vegetation stress caused by leachate contamination of the soil with a high degree of confidence. Landfill gas was not considered to be a cause of this vegetation stress since most of the vegetation was often in the pathway of leaking surface leachate (e.g. Figure 8.37(e-f).

8.5.3 Arrangement of Survey Lines

The photographs in Figures 8.37(a)-(c), taken at heights varying from 500 – 1000m, showed that from 1992 to 1999 patches of vegetation developed adjacent to the unlined cells 5 and 10. Site records dating back to 1996 indicate that leachate seepage and surface water often accumulated in these small marshy regions after rainfall, e.g. Figure 8.37(d). Grass growing up to 100m from the edges of cells 5 and 10 was found to have patches of defoliated, discoloured and healthy-looking vegetation, e.g. Figure 8.37(e). After discussion with the managers of the landfill site, analysis of aerial photographs and preliminary field investigations, the section adjacent to cells 2 and 3 was selected as being the most likely area representative of vegetation unaffected by the leachate contamination of the soil, shown as area H in Figure 8.35. The section adjacent to cells 5, 10 and 11 was selected as being the most likely area representative of areas affected by leachate, shown in the close-up section of Figure 8.35. Five field spectroscopy survey lines were arranged parallel to the edges of the two representative sections separated by approximately 1.5m, Figure 8.35. Three survey transects sampling chlorophyll and heavy metal concentrations were carried out away from the landfill edge at cell 5 and 10 for a distance of 10m, shown as Transects A, B and C in Figure 1. One survey line sampling the heavy metal concentration of the soil and the grass was carried out away from the landfill edge at cell 5 for a distance of 30m while a second survey line sampling the heavy metal concentration of the surface water was carried out for a distance of 50m, both shown by the S, W and G titled survey line in Figure 8.35. Chlorophyll-a concentrations in the foliar samples were determined using acetone extraction and analysis by spectrophotometry.

Field spectral surveys along all the planned transects, with corresponding vegetation samples, were acquired. 143 sets of spectra were collected every 5m, e.g. Figure 8.35 and 8.37(f), using a GER 3700 spectroradiometer in April 1999 and a GER 1500 spectroradiometer in April 2000. A limited number of point samples were collected due to severe restrictions on sampling time caused by limited instrument availability and poor field and weather conditions.

Figure 8.35 Site A maps showing Spectroscopic survey lines: Transects 1 to 5 are the Spectroscopy survey lines; transect A, B & C are the chlorophyll survey lines; S, W & G survey line is the soil, water and grass heavy metal concentration survey line; areas labelled H were areas of unstressed grass

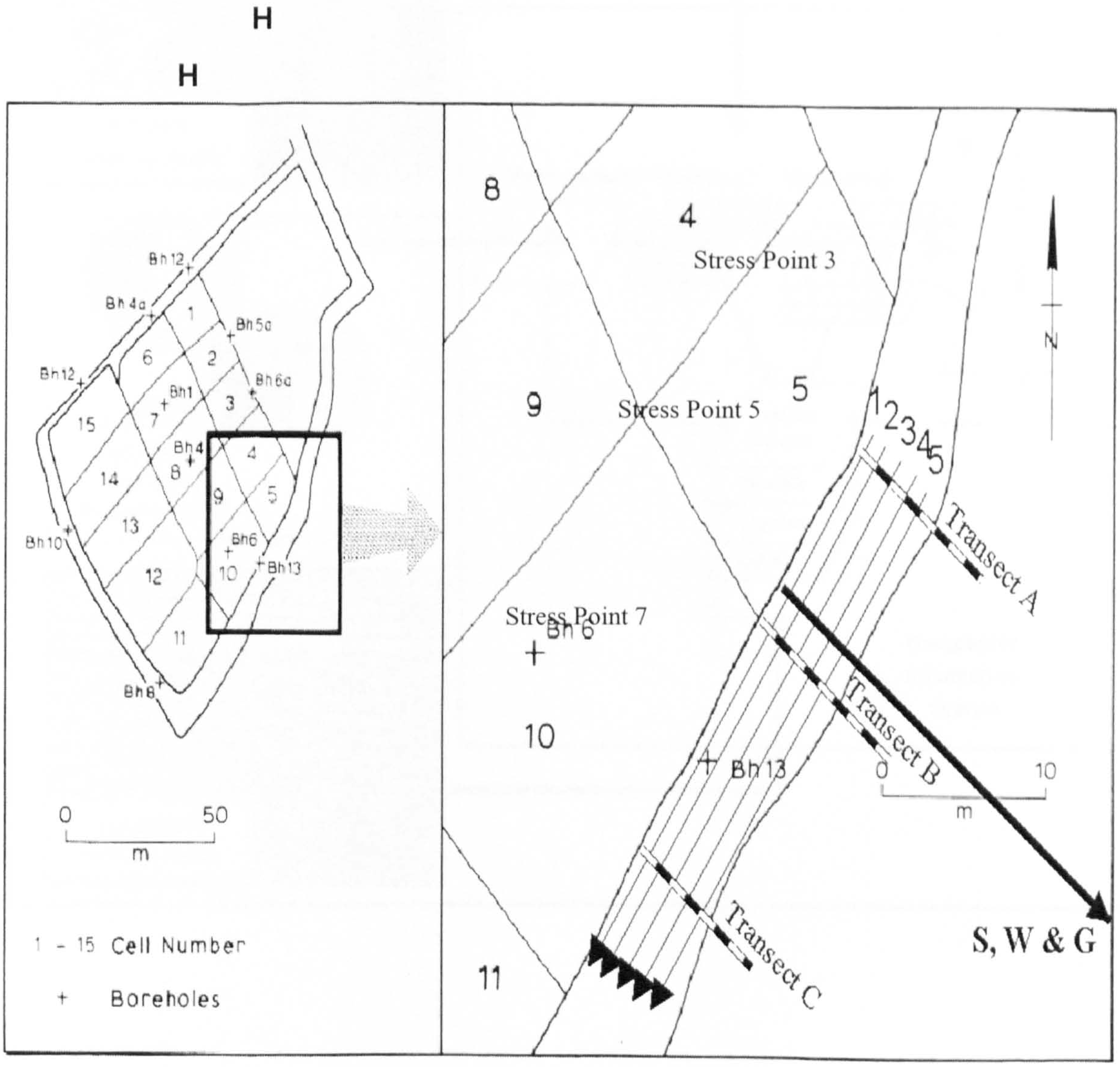


Figure 8.36 Data from field-based and airborne remote sensing instruments, aerial photographs, hydrological data, and other data sets were compiled into a GIS to provide background information about Site A conditions and to construct conceptual models of site conditions

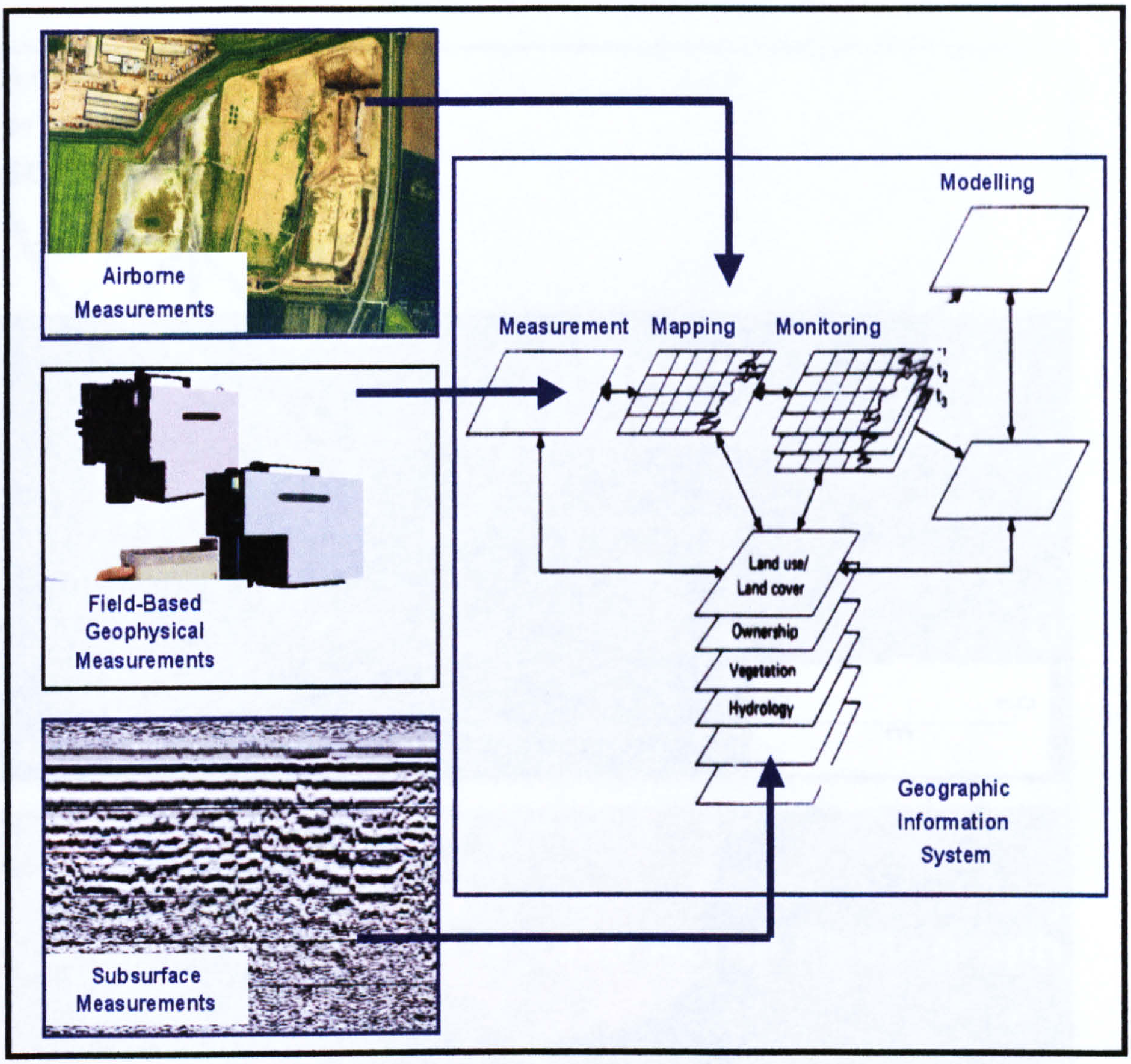


Figure 8.37(a) Aerial photographs of leachate and vegetation change at edge of cells 10 & 5; Top Image: The 1992 conditions show no indication of vegetation stress or leachate accumulation; Bottom Image: The 1999 conditions show vegetation stress (defoliation, discoloured vegetation) as well as leachate accumulation (The arrow shows areas of leachate accumulation that were not present in 1992)

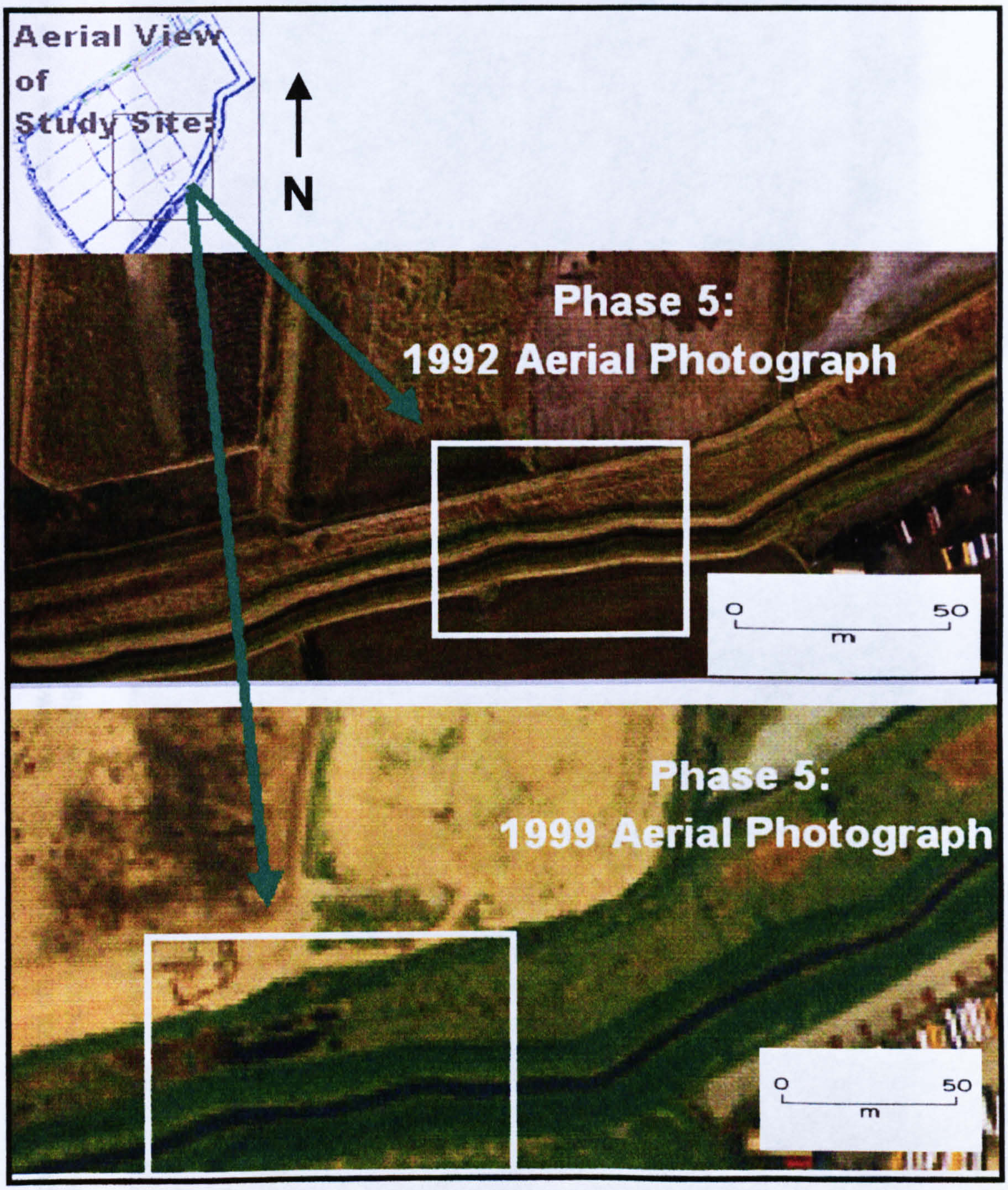


Figure 8.37(b): Enlarged 1992 aerial photographs of Site A showing no evidence of leachate and vegetation change in the study area adjacent to cells 10 and 5 (Photo taken by Wastewise, 1992)

Figure 8.37(c): Enlarged 1999 aerial photograph at Site A showing evidence of leachate presence and vegetation change close to the study area adjacent to cells 10 and 5 (Photo taken by NERC, 1999)

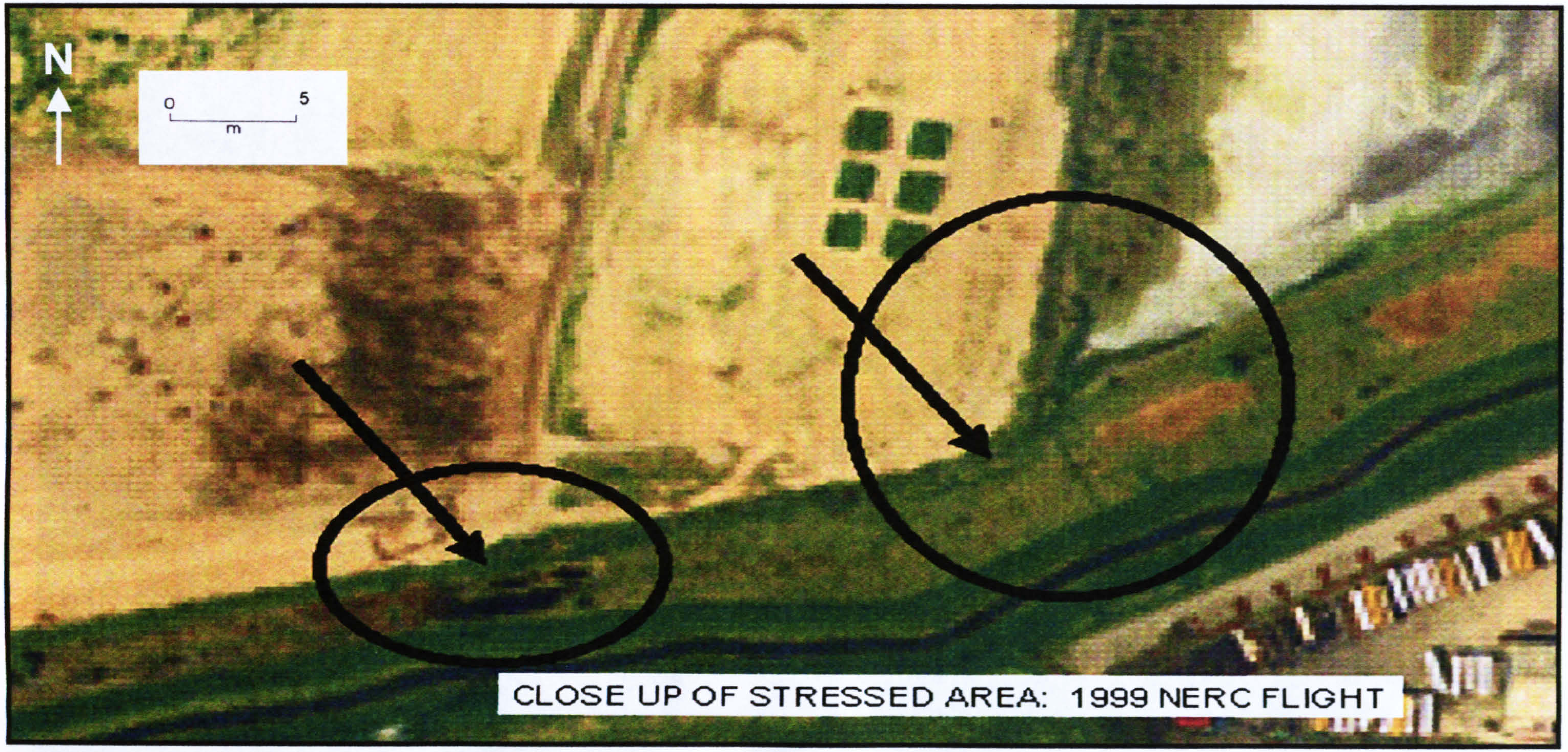


Figure 8.37(d): Surface water accumulations mixed with leachate seepage after precipitation located along unlined edges of the landfill (Photographed by T. Splajt, 2000)

Figure 8.37(e): Different levels of stressed vegetation located 10-20 m from the landfill edge showing (1) leachate accumulating on surface, (2) stressed grass that is discoloured, (3) stressed grass that retains a lush green colour all year around and (4) grass that does not show visible signs of stress (Photographed by T. Splajt, 2000)

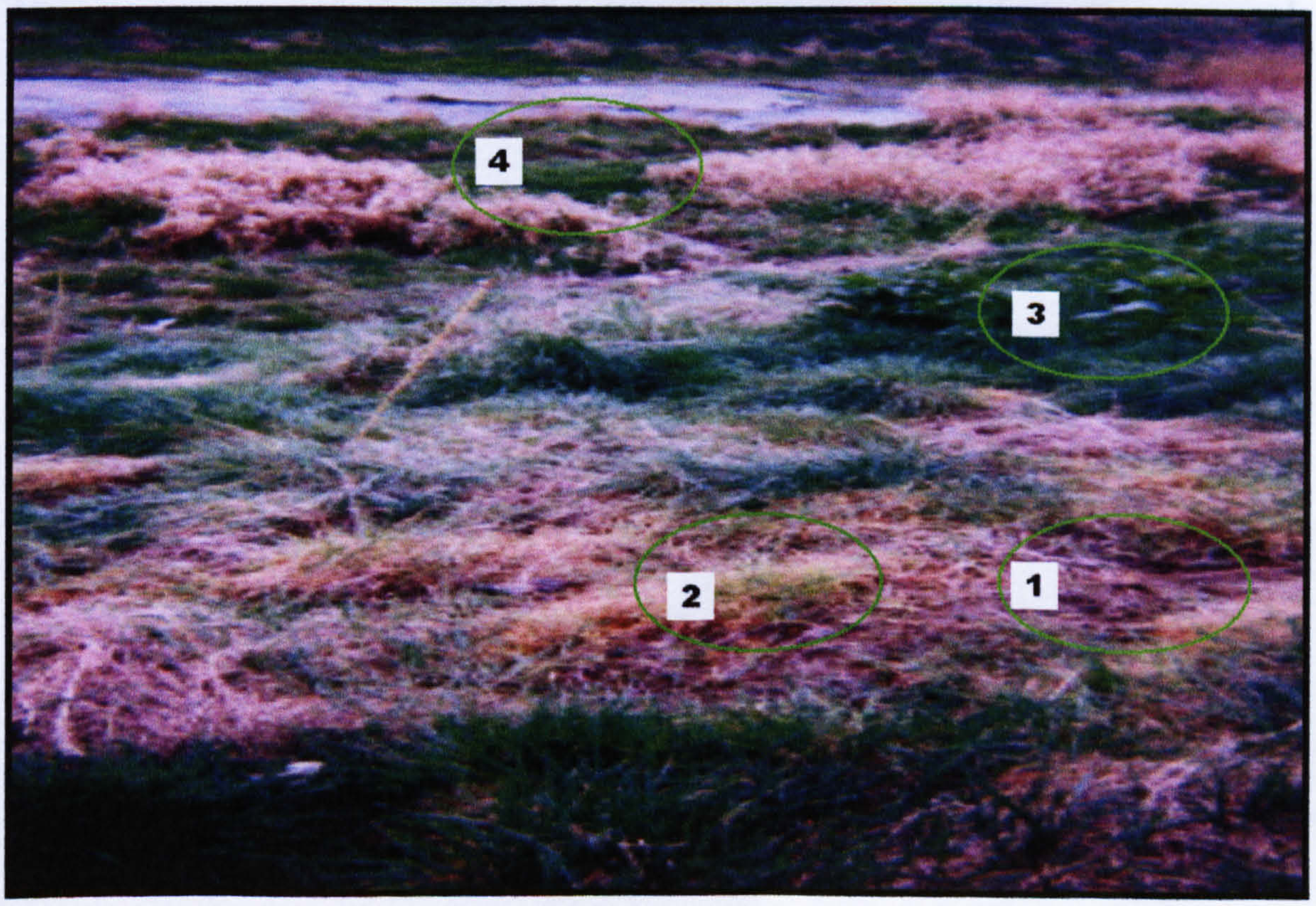
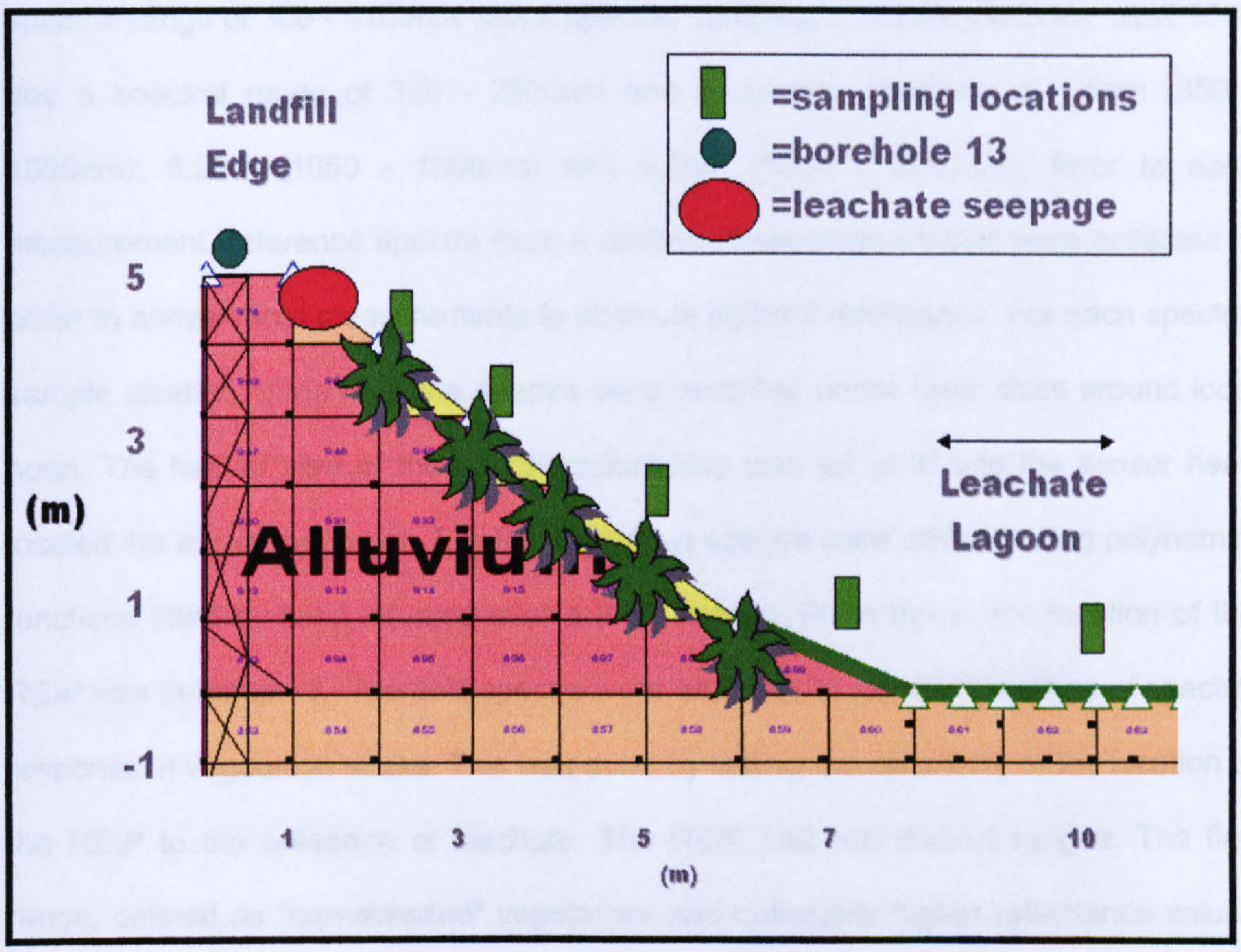


Figure 8.37(f): Cross-section image showing sampling locations for stressed points 3, 5 and 7 shown in Figure 8.35 where three types of measurements were taken: (i) hand-held spectral reflection measurements, (ii) heavy metal concentrations in vegetation, and (iii) chlorophyll concentrations in vegetation. Samples of each measurement were taken at 1m, 3m, 5m, 7m & 10m from the leachate outbreak points

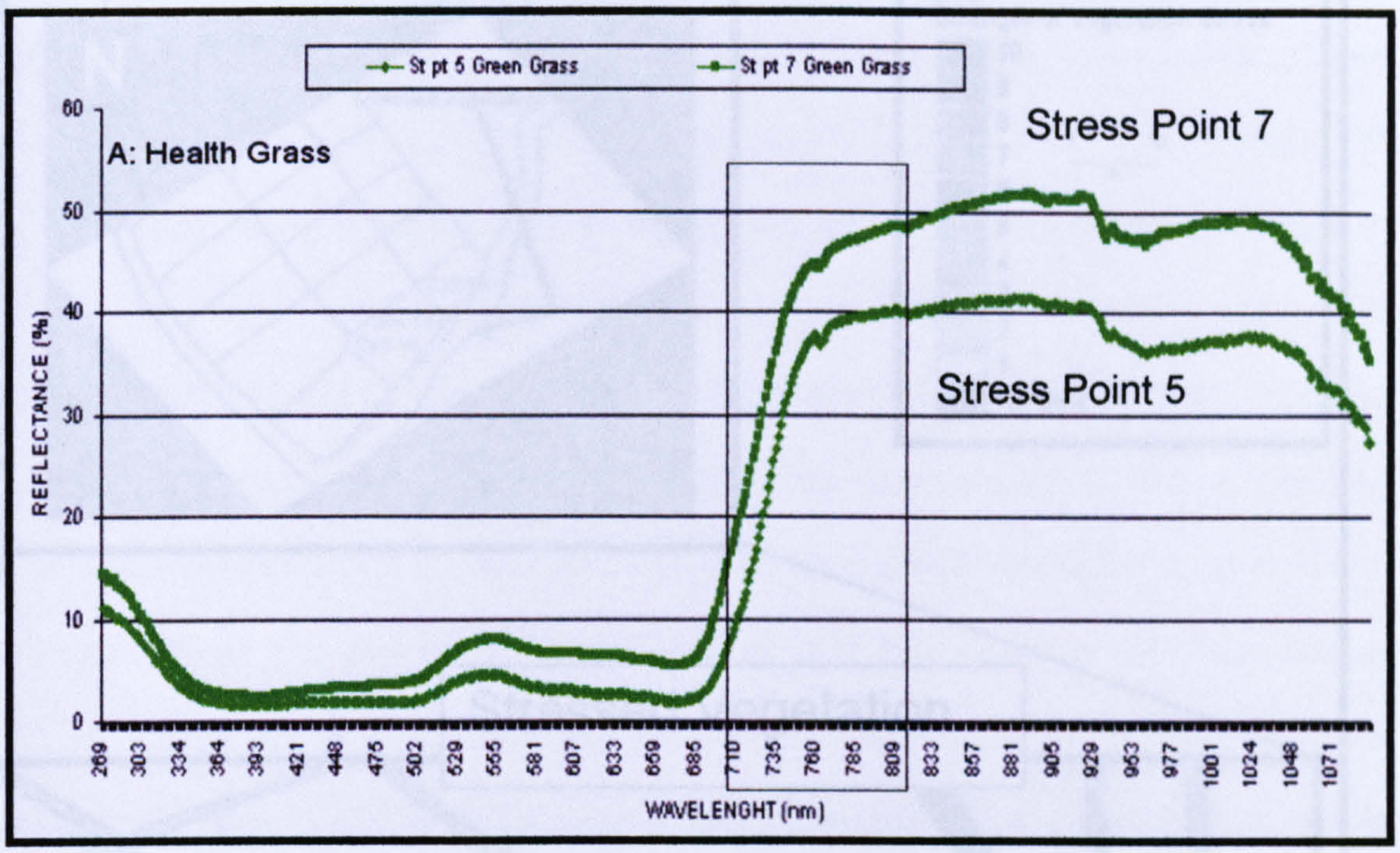


8.5.4 Analysis of Field Spectra

Two field spectroradiometers, the Geophysical and Environmental Research Corporation (GER) 1500 and 3700, were used in this study. The GER 1500 has a spectral range of 300 - 1100nm with a spectral sampling of 1.5nm while the GER 3700 has a spectral range of 350 - 2500nm and a spectral sampling of 1.5nm (350 - 1050nm); 6.2nm (1050 - 1900nm) and 9.5nm (1900 - 2500nm). Prior to each measurement, reference spectra from a calibrated spectralon tablet were collected in order to convert final measurements to absolute percent reflectance. For each spectral sample location, three replicate spectra were recorded under clear skies around local noon. The field of view of the spectroradiometers was set at 8° and the sensor head located 1m above the target. The first derivative spectra were derived using polynomial functions fitted by least squares over a 6nm interval. From these, the location of the REIP was determined. The field spectra were analysed to identify the range of spectral response in vegetation stress. This was done by testing the sensitivity of the location of the REIP to the presence of leachate. The REIP had two distinct ranges. The first range, defined as "non-stressed" vegetation, had noticeably higher reflectance values that were above 800nm, having REIP positions located near 730nm, e.g. Figure 8.38(a). The second range, defined as "stressed" vegetation, had generally lower reflectance values that were below 800nm, having REIP positions located near 705nm, e.g. Figure 8.38. A site map showing the degree of vegetation stress derived from these REIP ranges is shown in Figure 8.39 showing that vegetation stress was highest along the landfill edges near the leaking parts of cells 5 and 10.

Figure 8.38: Spectral reflectance plots using field-based measurements for (a) healthy grass (REIP ranged from 720 – 785 nm); (b) dying discoloured grass at Stress Points 3, 5 and 7 (REIP ranged from 700-740 nm) in which the REIP ranges decreased with increasing stress

(a) The spectral reflectance for healthy grass: 720-785 nm



(b) The spectral reflectance for dying (discoloured) grass: 700-740 nm

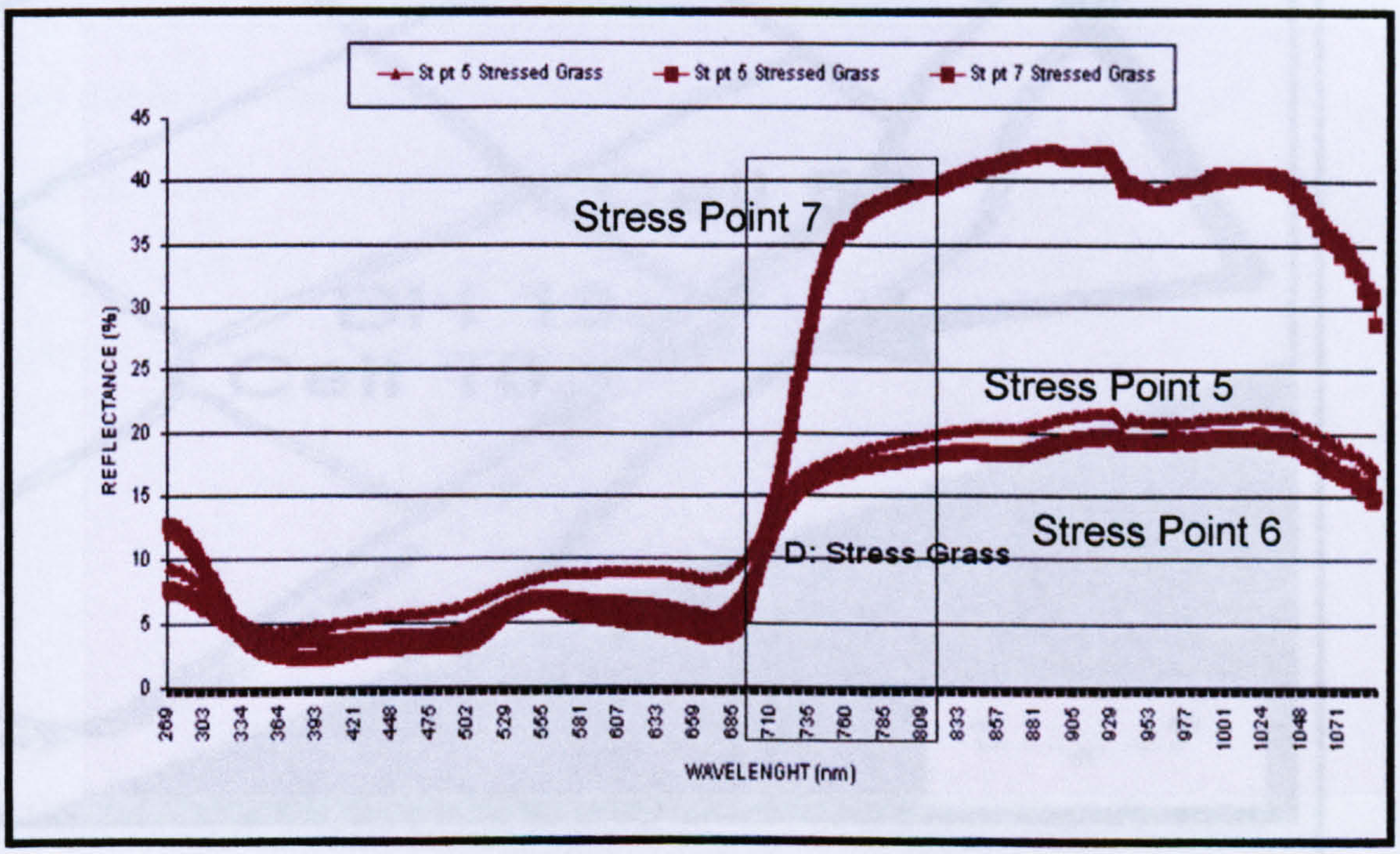
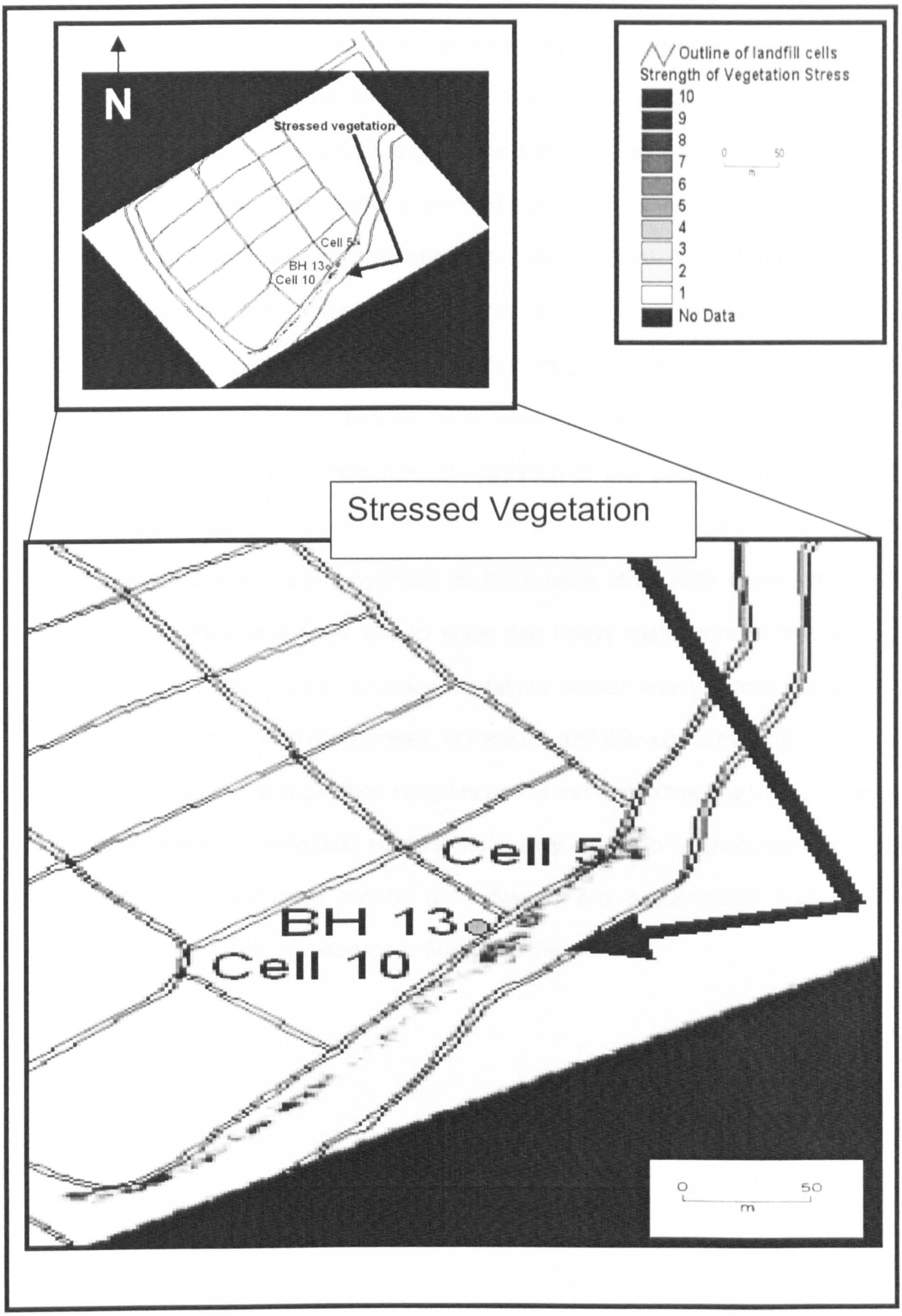


Figure 8.39: A site map showing the level of vegetation stress derived using REIP ranges collected across the site showing that vegetation stress was highest along landfill-leaking edges at cells 5 and 10 (Legend ranges from 10 = highest vegetation stress to 1 =no vegetation stress)



8.5.5 Integrating Data from Field Spectra, Chlorophyll Concentration and Measured Levels of Contamination in Vegetation and Soil around Site A

(a) Measuring Contamination in Soil, Vegetation, and Surface Water

A detailed site assessment was conducted sampling soil, vegetation, surface water and leachate up to 500m from the landfill's edge. Seven types of field measurements were collected and compiled into a GIS (e.g. Figure 8.36): (1) heavy metal concentrations in soil; (2) heavy metal concentrations in vegetation; (3) heavy metal concentrations in leachate; (4) heavy metal concentrations in surface water along landfill edges; (5) vegetation sampled to be tested for chlorophyll concentrations; (6) field-based spectral reflectance of vegetation; and (7) GPS points to map sampled points. The aim was to find a link between off-site stressed vegetation and landfill leachate. Heavy metal concentrations in soil and vegetation downstream of the landfill were measured to establish the extent of leachate migration off site. Ammonia was not an effective indicator of off-site leachate movement as there were other local contributors of this contamination. Figures 8.40 (a and b) show that heavy metal concentrations in soil, vegetation and surface water samples contained similar heavy metals, of which all three had significantly high manganese, chromium, and titanium in the soils, chromium, titanium and lead in the vegetation samples and chromium, manganese and vanadium in the surface water. These data suggest that heavy metals in leachate are transported in surface runoff and near-surface groundwater, are accumulated soils and are absorbed by vegetation that was up to 50m from the site.

Figure 8.40(a): Heavy metal concentrations in soil and grass from transect S, W & G (location shown in Figure 8.35)

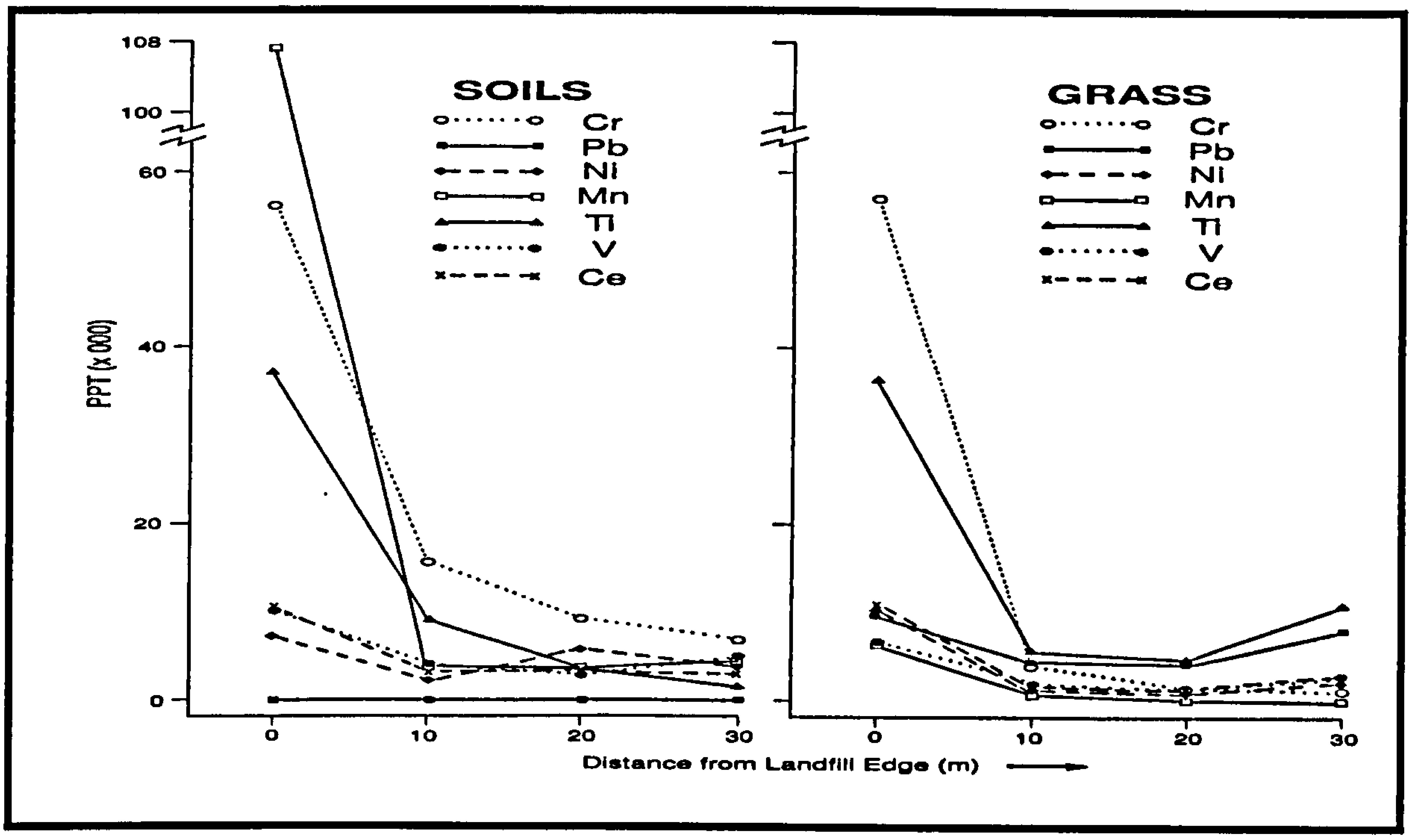
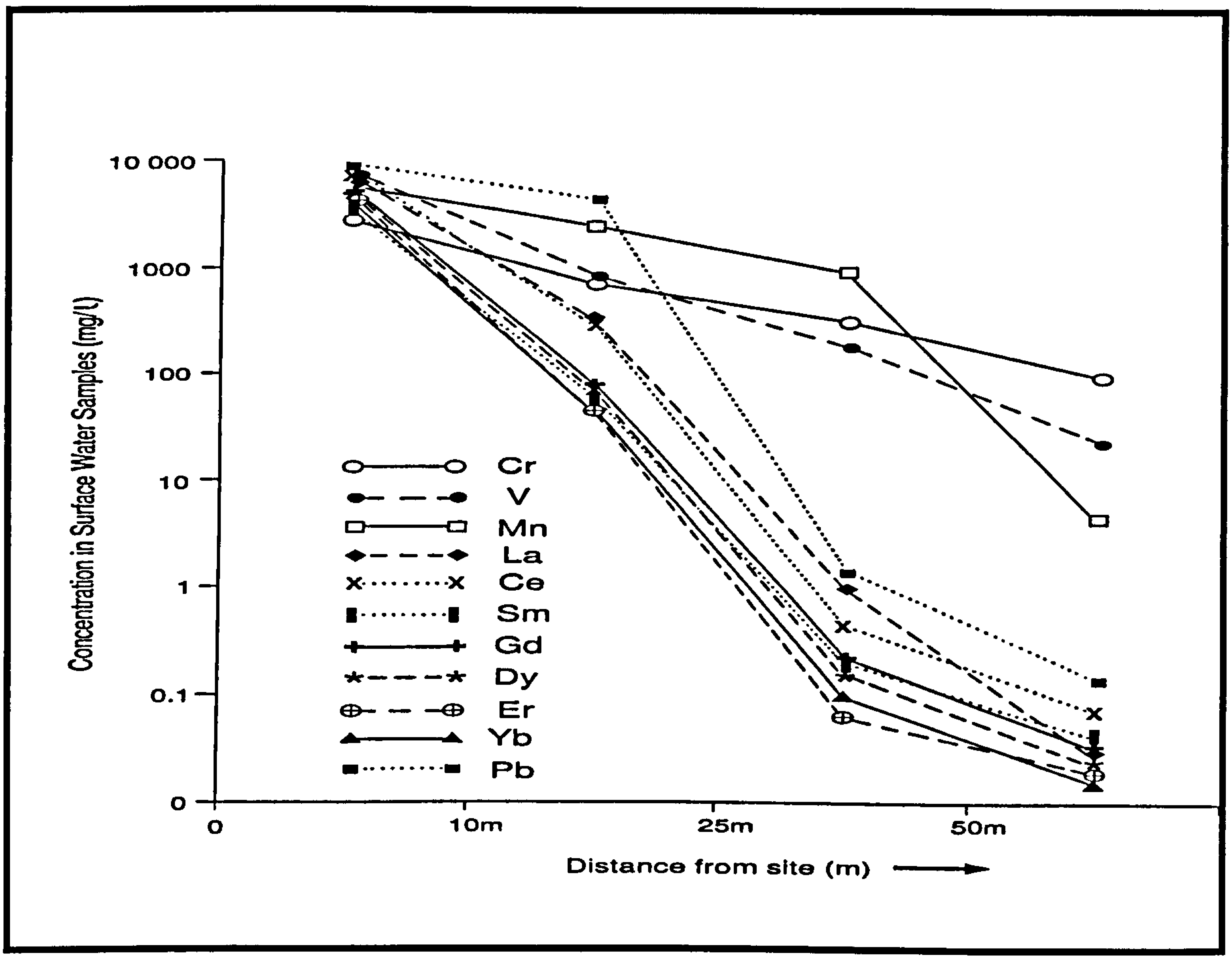


Figure 8.40(b): Heavy metal concentrations in surface water from transect S, W, and G (location shown in Figure 8.35)



(b) Chlorophyll Concentrations in Vegetation

The chlorophyll concentrations extracted from vegetation samples taken along the landfill edge at cells 5 and 10 additionally confirm leachate-induced vegetation stress, e.g. Figure 8.44. 51 foliar samples were collected and analysed for chlorophyll concentration in August 1999, April 2000 and August 2000. Table 8.14(a) summarises variability showing that the minimum and maximum values of chlorophyll concentration are significantly different in samples taken at high and low contaminated locations. The sample locations and their chlorophyll concentrations were compared with chlorophyll concentrations measured in healthy grass and with heavy metal concentrations in soils and vegetation sampled at the same locations. The samples showed three distinct areas of high, medium and low contaminant concentrations based on ammonia and heavy metal concentrations. Table 8.14(b) cross references the three categories of contaminated land (high, medium and low) with chlorophyll concentrations showing that areas of highest contaminants had the lowest chlorophyll concentrations and areas with lower contaminants had higher chlorophyll concentrations. This inverse correlation is also presented in Figures 8.41 and 8.42. The three categories of high, medium and low contaminated land were assumed because samples that had low contaminant concentrations also had higher chlorophyll concentrations that were similar in value to that of healthy grass, e.g. healthy grass had chlorophyll concentrations ranging from 832 – 979μg/g, grass which had low contaminant concentrations also had similar ranges (814.35 – 931.47μg/g).

Table 8.14(a) Summary of the variability of chlorophyll found in vegetation samples, showing that the minimum and maximum values of chlorophyll concentration have a significant difference. Samples with low chlorophyll concentrations were in areas with high contaminant concentrations

	Chlorophyll Concentration μg/g
Number of Samples	51
Mean Concentration	520.1
Standard Deviation	242.33
Minimum Chlorophyll Concentration	33.37
Maximum Chlorophyll Concentration	931.47

Table 8.14(b): Three distinct categories of high, medium and low contaminant concentrations based on ammonia and heavy metal concentrations are cross-referenced with chlorophyll concentrations showing an inverse correlation. Areas with high contaminant concentrations had low chlorophyll concentrations while areas with lower contaminant concentrations had higher chlorophyll concentrations (The data below identified three of eight stress points)

Contaminant Levels at each Plot	Stress Point 3 μg/g	Stress Point 5 μg/g	Stress Point 7 μg/g
Highly Contaminant Levels	540.67	150.62	33.38
Medium Contaminant Levels	654.4	483.93	606.09
Low Contaminant Levels	814.35	844.66	931.47

Figure 8.41: Chlorophyll concentrations graphed according to levels of contaminant concentrations showing those areas of high contamination have lower chlorophyll concentrations (based on Table 8.14(b))

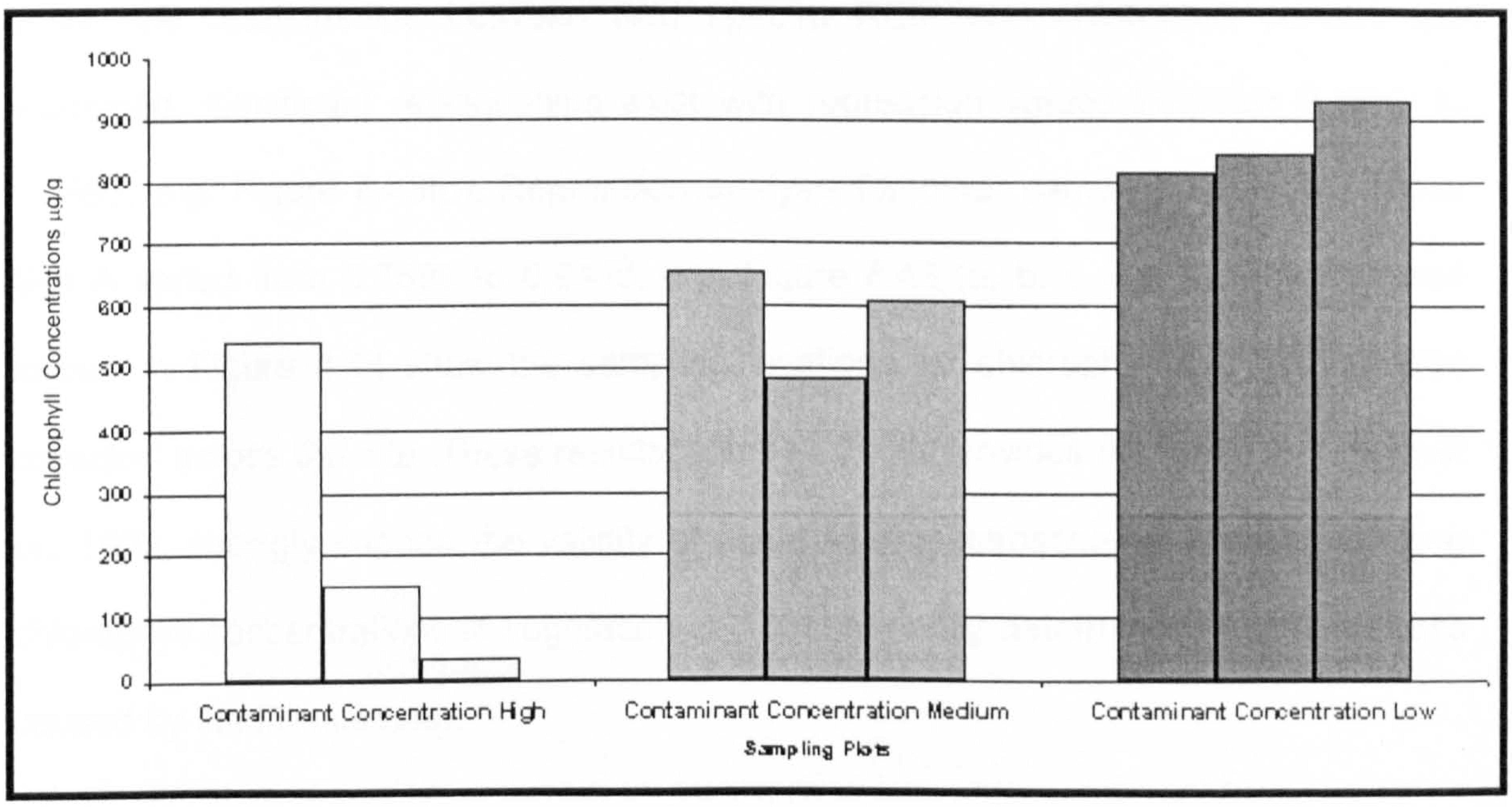
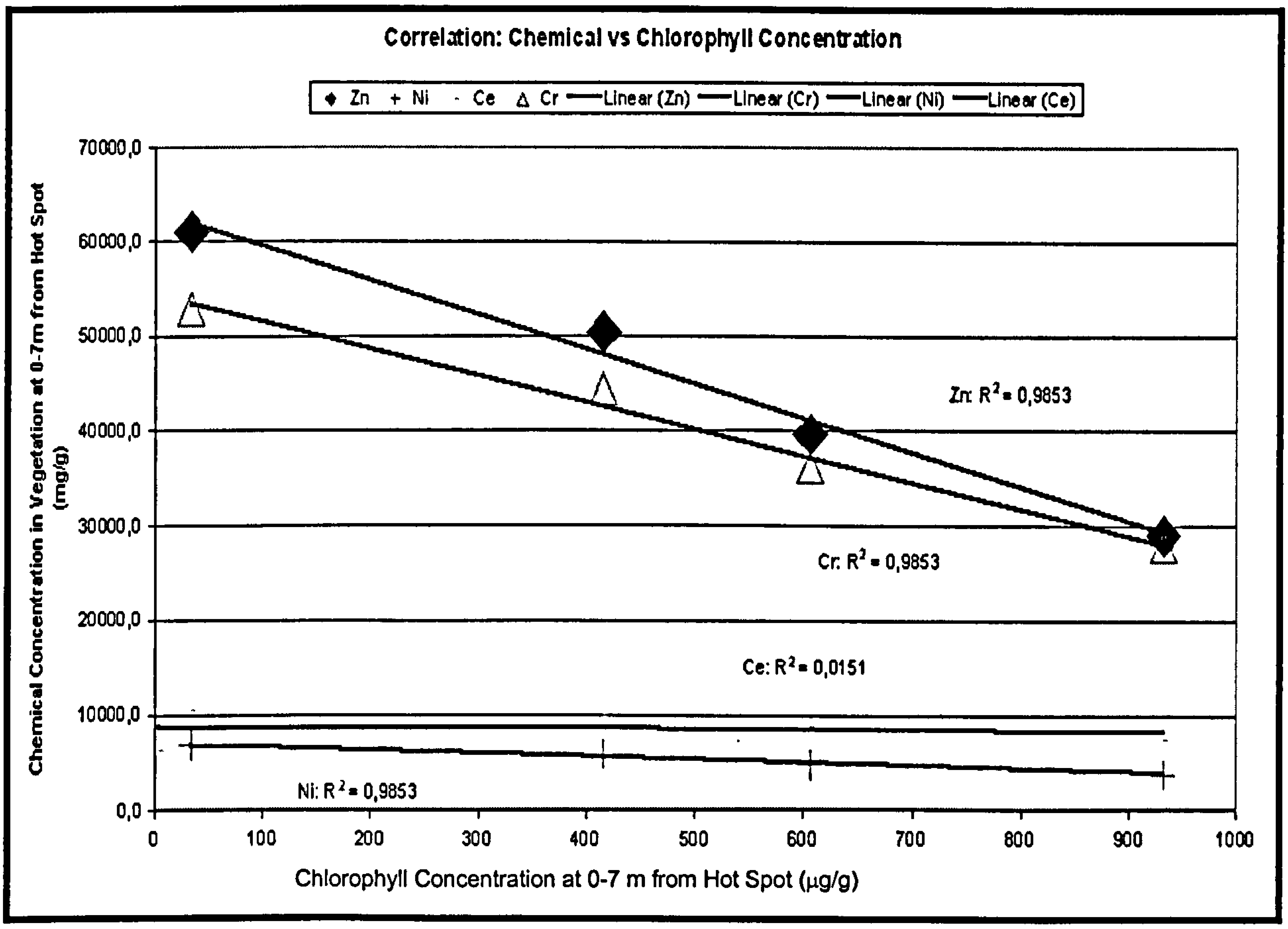


Figure 8.42: Comparison of vegetation samples taken at Stress Points 5 and 7 showing inverse correlation trends – as heavy metal concentrations decreased, chlorophyll concentrations increased



(c) Integrating Field Spectra and Chlorophyll Concentration

When the relationships between field spectra REIP and chlorophyll content are examined, significant relationships exist with regression values between 0.8353 to 0.8867, e.g. Figure 8.43(d). Regression analysis for other sampling locations across Site A varied from 0.7589 to 0.9446, e.g. Figure 8.43 (a, b, c and e). The coloured arrows in Figure 8.44 show the sampling locations for chlorophyll and field spectra collected across the site. These results, combined with previous research, e.g. Jago *et al.*, 1999, strongly support the validity of using field spectroscopy to identify stressed chlorophyll concentrations in vegetation and consequently determine vegetation stress caused by landfill leachate.

Figure 8.43: The relationship between REIP and chlorophyll concentration across Site A, showing positive correlation at all sampling locations

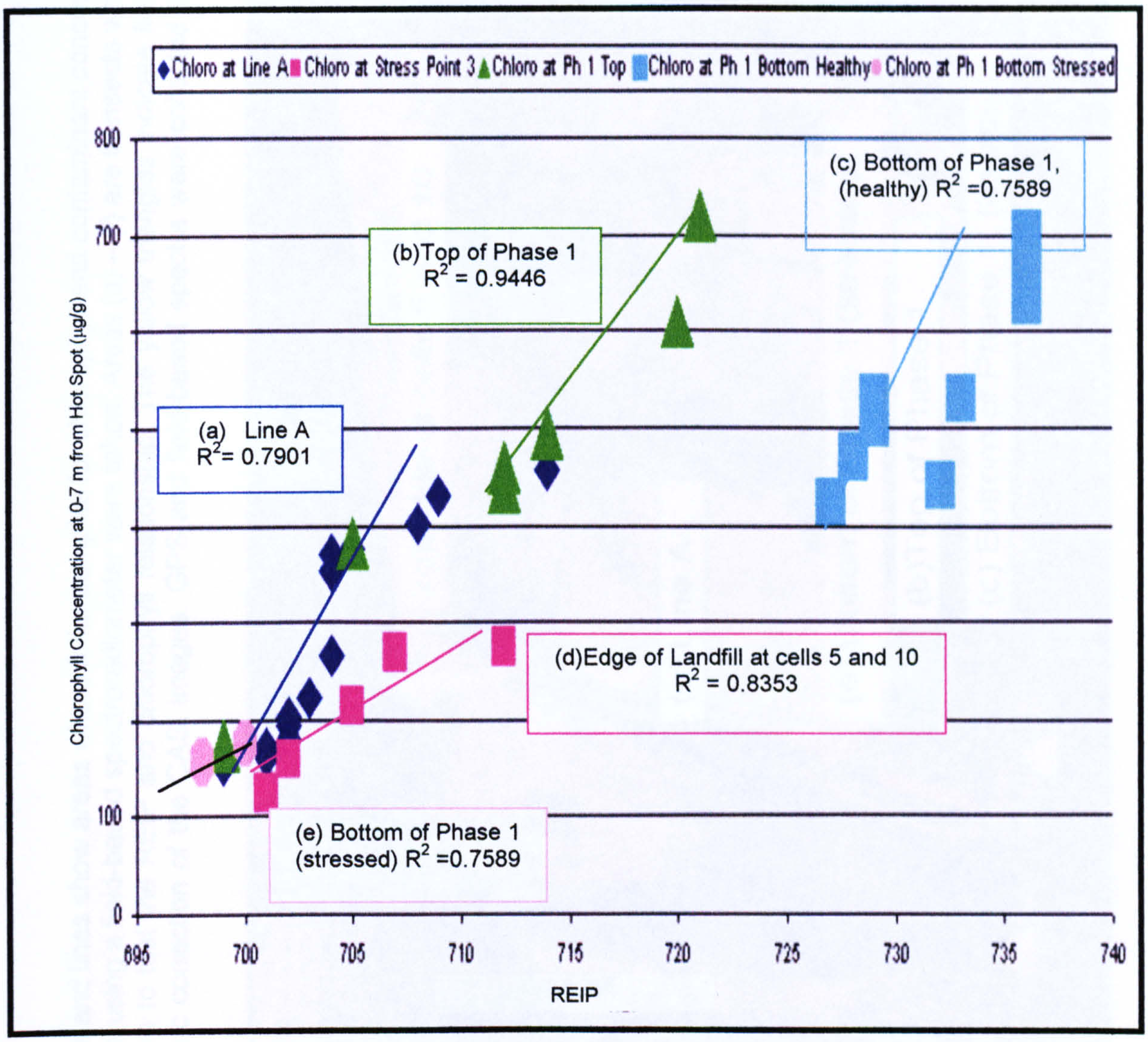
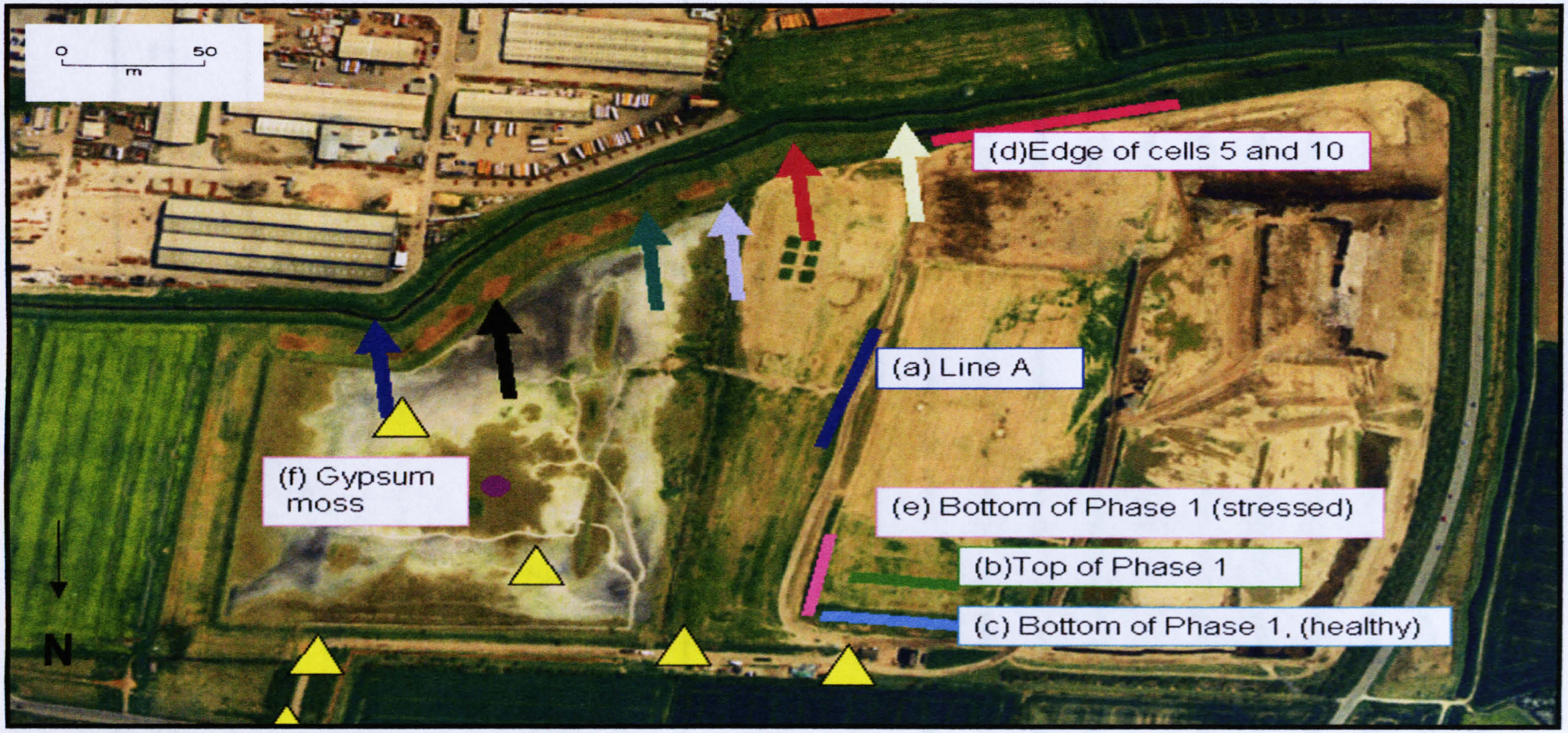


Figure 8.44 1999 CASI image of Site A: coloured arrows and lines show areas where foliar samples for chlorophyll and contaminant concentration, soil quality samples and spectral reflectance using a field-based spectroradiometer were taken. Areas (a) – (f) are transects where field spectra and vegetation samples were taken to test the REIP and chlorophyll relationship. The yellow triangles indicate field-based ground targets that were used for geometric correction of the CASI images. GPS and field-based spectra were collected at all the labelled areas on this image



8.5.6 Airborne Spectral Data

The objective was to assess the sensitivity of airborne spectroscopy in identifying vegetation health using red and infrared variations, the REIP in the CASI data. Two sets of CASI imagery were acquired for Site A on April 9, 1999 and September 6, 1999. The yellow triangles in Figure 8.44 show the locations of the targets used for geometric correction of the image. The sensitivity of CASI data for identifying vegetation health and differentiating 'stressed' from 'unstressed' vegetation was assessed in an initial analysis. Field spectra were calibrated to the CASI bandwidths, e.g. 13, 48 and 72-band setting, Figure 8.46. These spectral profiles show that the 48 and 72-band have sufficient spectral resolution to differentiate between the "stressed" and "non-stressed" vegetation, but this was not the case for the 13-band CASI image. However, significant increases in the slope (the 1st derivative) value at CASI band 7 (738 - 743nm) were noted in the "stressed" and "non-stressed" vegetation, e.g. Figure 8.45.

Figure 8.45: The first derivative of reflectance calculated for healthy and stressed vegetation in band 7 showing lower reflection in the 1st derivative for stressed vegetation

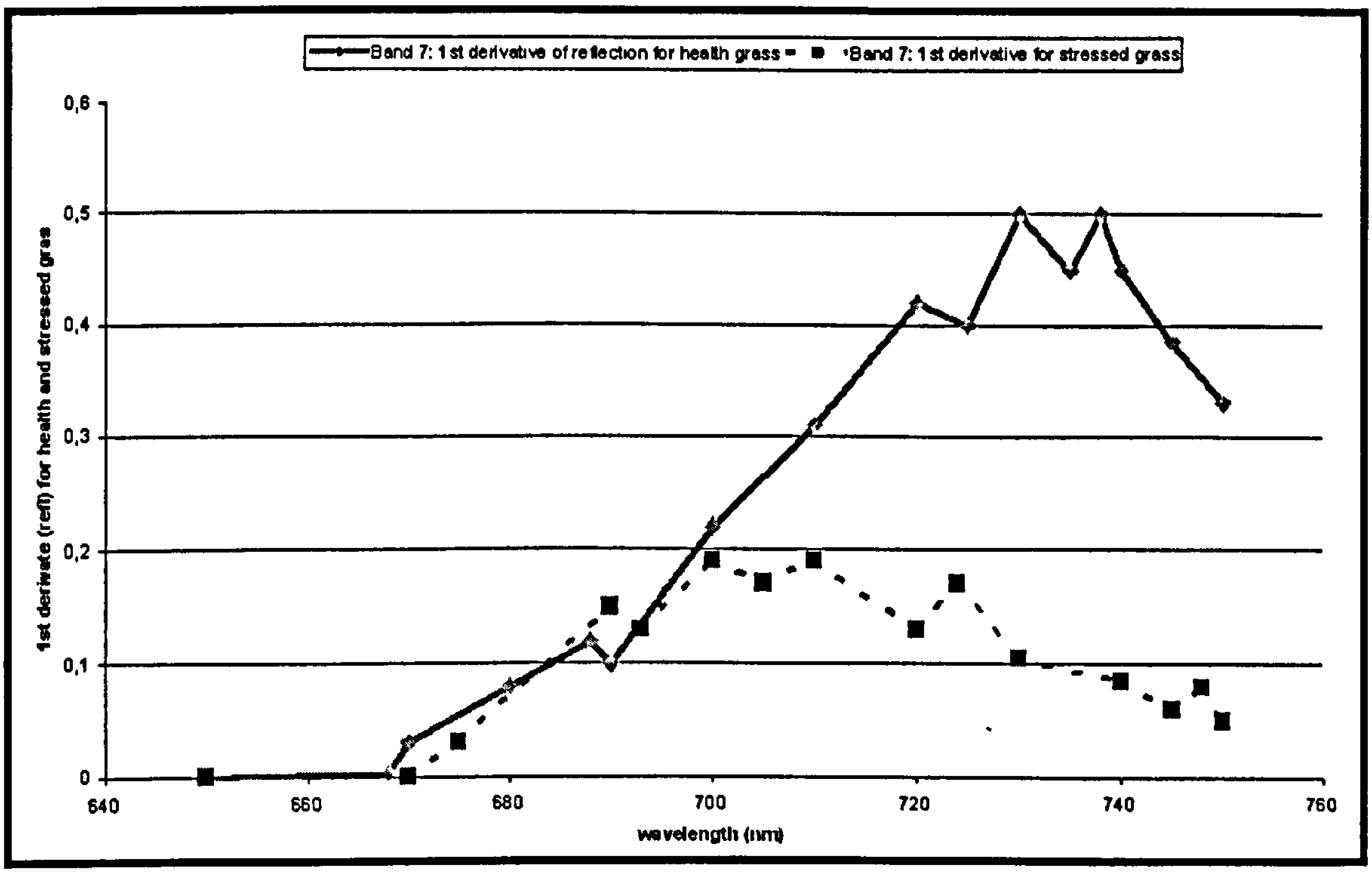
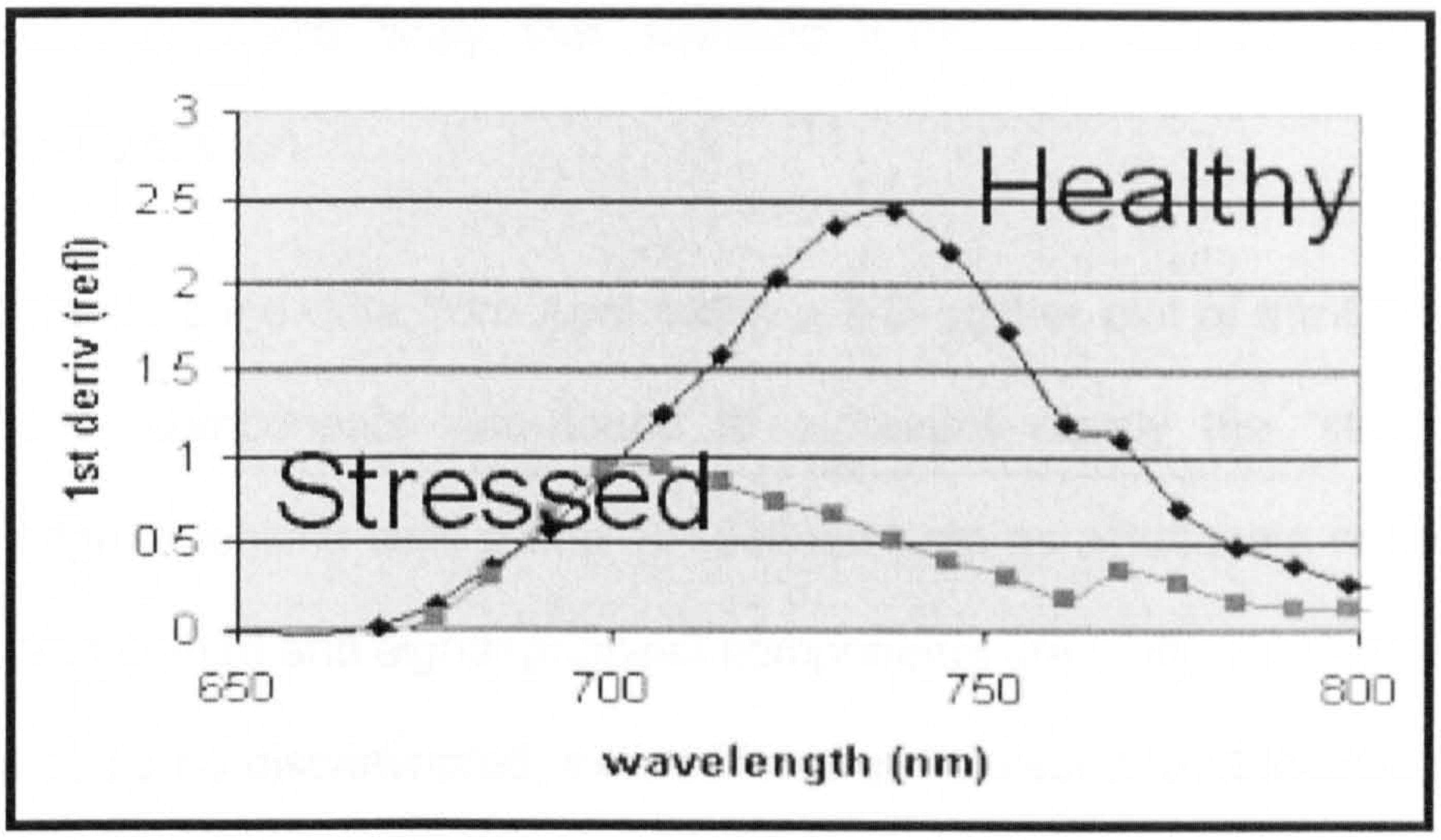
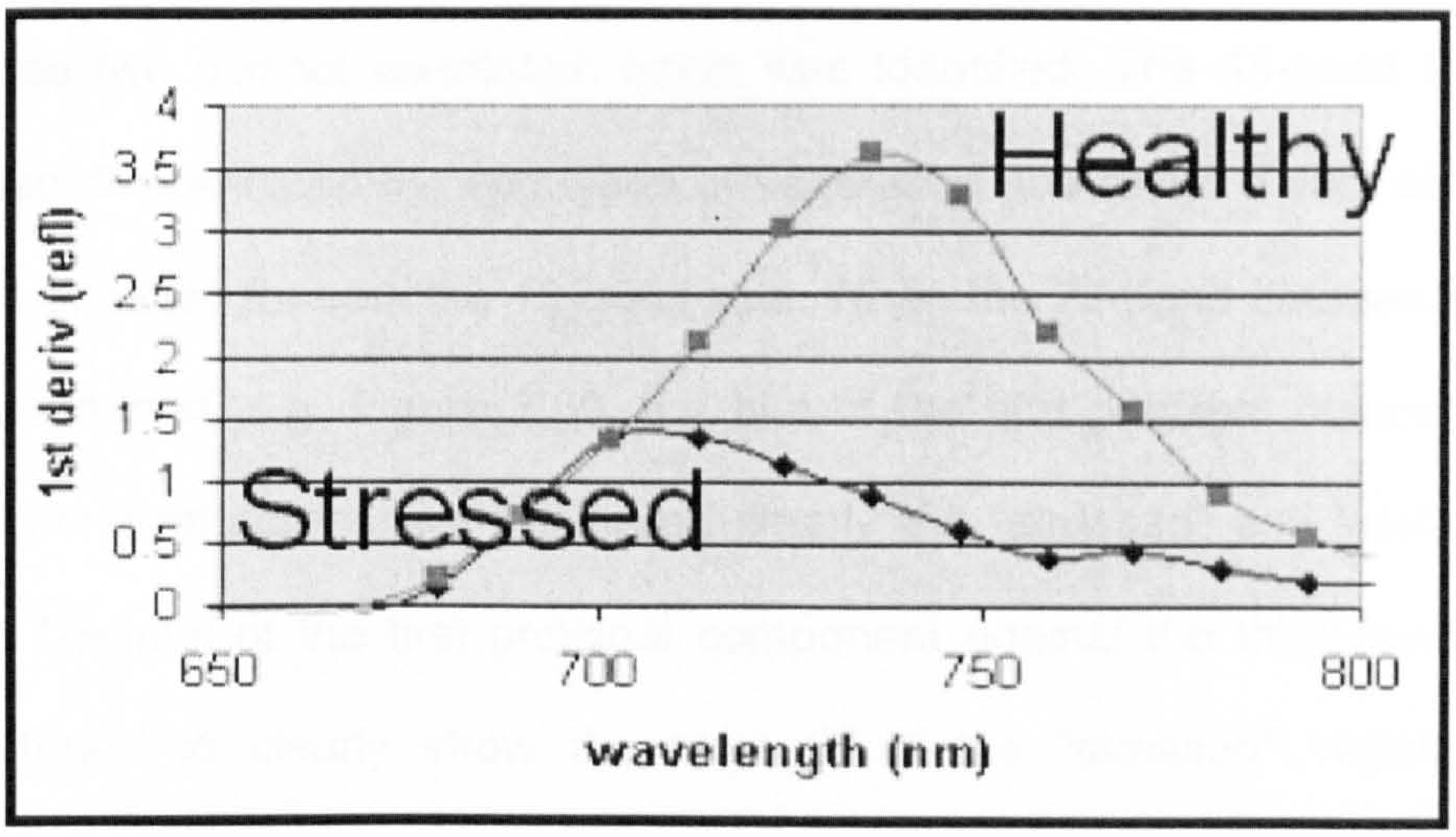


Figure 8.46: The first derivative of reflectance calculated for healthy and stressed vegetation using CASI bandset 48 and 72 showing that there is a distinct difference between healthy and stressed vegetation

(a) Band 48: 1st derivative of reflection for healthy and stressed vegetation.



(b) Band 72: 1st derivative of reflection for healthy and stressed vegetation.



An analysis of the CASI raw radiance data was used to identify whether any spatial patterns representing the "stressed" vegetation were identifiable at different stages in the vegetation growing cycle and to determine the sensitivity of the number of bands in the detection of the "stressed" vegetation. This was done using Minimum Noise Fraction (MNF) transform (Green *et al*, 1988) which produces a set of principal component images ordered in terms of decreasing signal quality. The results indicate that there is some spectral sensitivity associated with the "stressed vegetation", e.g. Figure 8.47. The analysis of bandset sensitivity to identify "healthy" and "stressed" vegetation was successful, Figure 8.48. CASI bands 13, 48 and 72 were all able to differentiate stressed vegetation. As the analysis was conducted at different points in the growing season, it seems likely that stressed vegetation can be identified throughout the growing season.

In the analysis of the 13-band data from April 1999, a 2-D scatter plot of the fifth and the seventh principal components was found to represent clearly the "stressed" vegetation at the edge of leaking cells 5 and 10 (defined here as a 'leachate wetland', Figures 8.49). When the third and eighth principal components are plotted, a distinctive type of vegetation could be discriminated, defined here as "stressed" and located in a very narrow band (10m wide) along the edges of cells 5, 10 and 11. When a similar analysis of 13-band CASI data from September 1999 was carried out, a very similar distribution of these two distinct vegetation types was identified. The 48-band CASI data from April also differentiated the two types of vegetation and found a very similar distribution to that identified for both the 13-band sets. When the 72-band data set from September was examined, e.g. Figure 8.50, the plot of the fifth principal component against the tenth principal component separated clearly the "stressed" and "wetland" vegetation types. The plot of the first principal component against the third principal component was found to clearly show the strength of the "stressed" vegetation decreasing from cell 11 towards cell 5 and was to show a very limited amount of mixing between the two vegetation types occurring immediately downstream from cell 5, e.g. Figure 8.50.

Figure 8.47: 1999 CASI image of Site A showing band 13 data using Minimum Noise Fraction transform analysis. The “stressed” vegetation is indicated in red while the coloured arrows are points of vegetation stress. The coloured lines are field spectrometer transects (The image is not very clear due to turbulence during flight imaging)

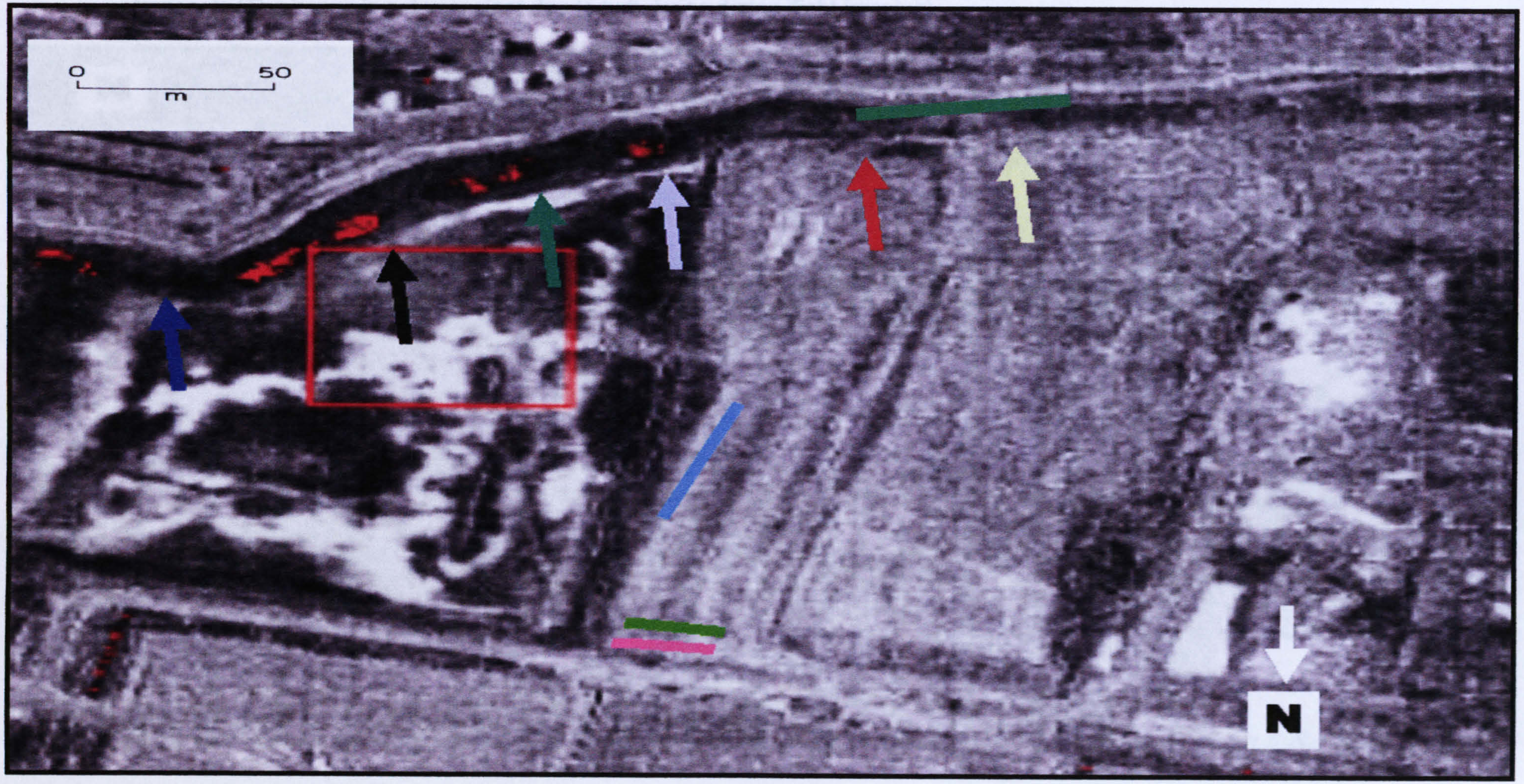


Figure 8.48: 1999 CASI image of Site A showing band 72 (poorer quality image than Figure 8.47) using Minimum Noise Fraction analysis. “Stressed” vegetation is indicated in red. The coloured arrows are points of vegetation stress. The coloured lines are field spectrometer transects (The image is not very clear due to turbulence during flight imaging)

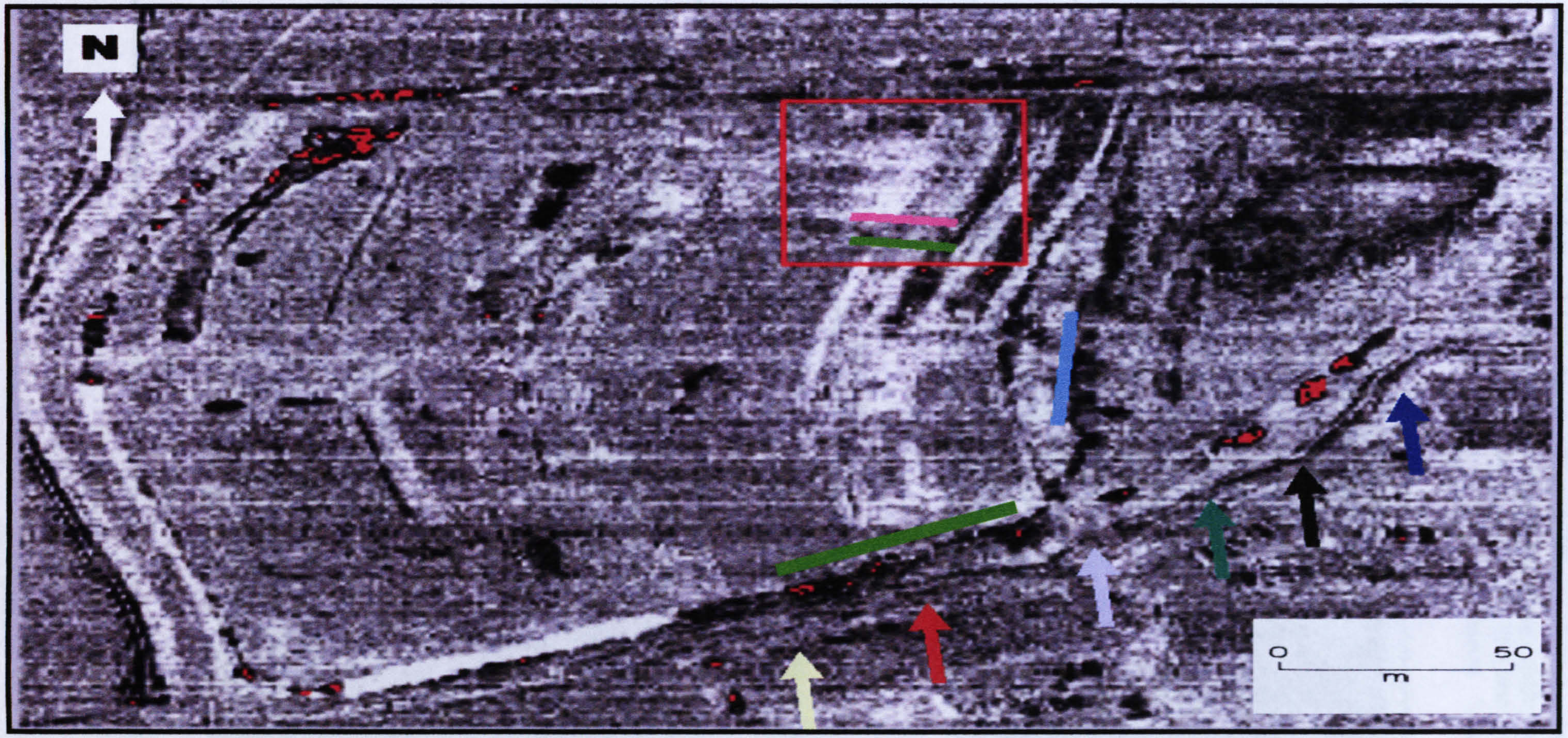


Figure 8.49: April 1999 CASI data set from band 13 using Minimum Noise Fraction analysis that produces image sets with decreasing signal quality, clearly showing "stressed" vegetation. A 2-D scatter plot of the fifth and seventh principal components was found to clearly represent "stressed" vegetation

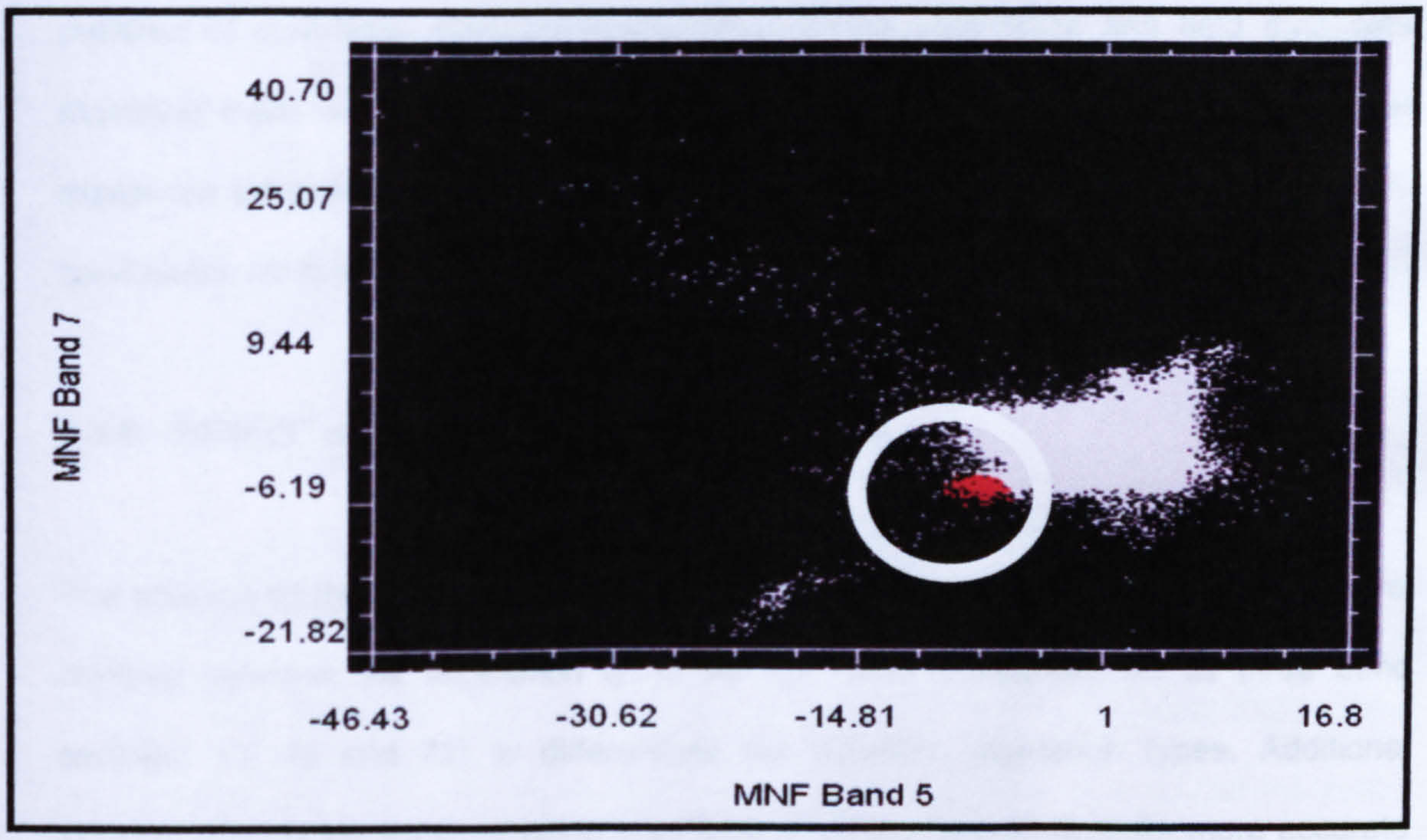
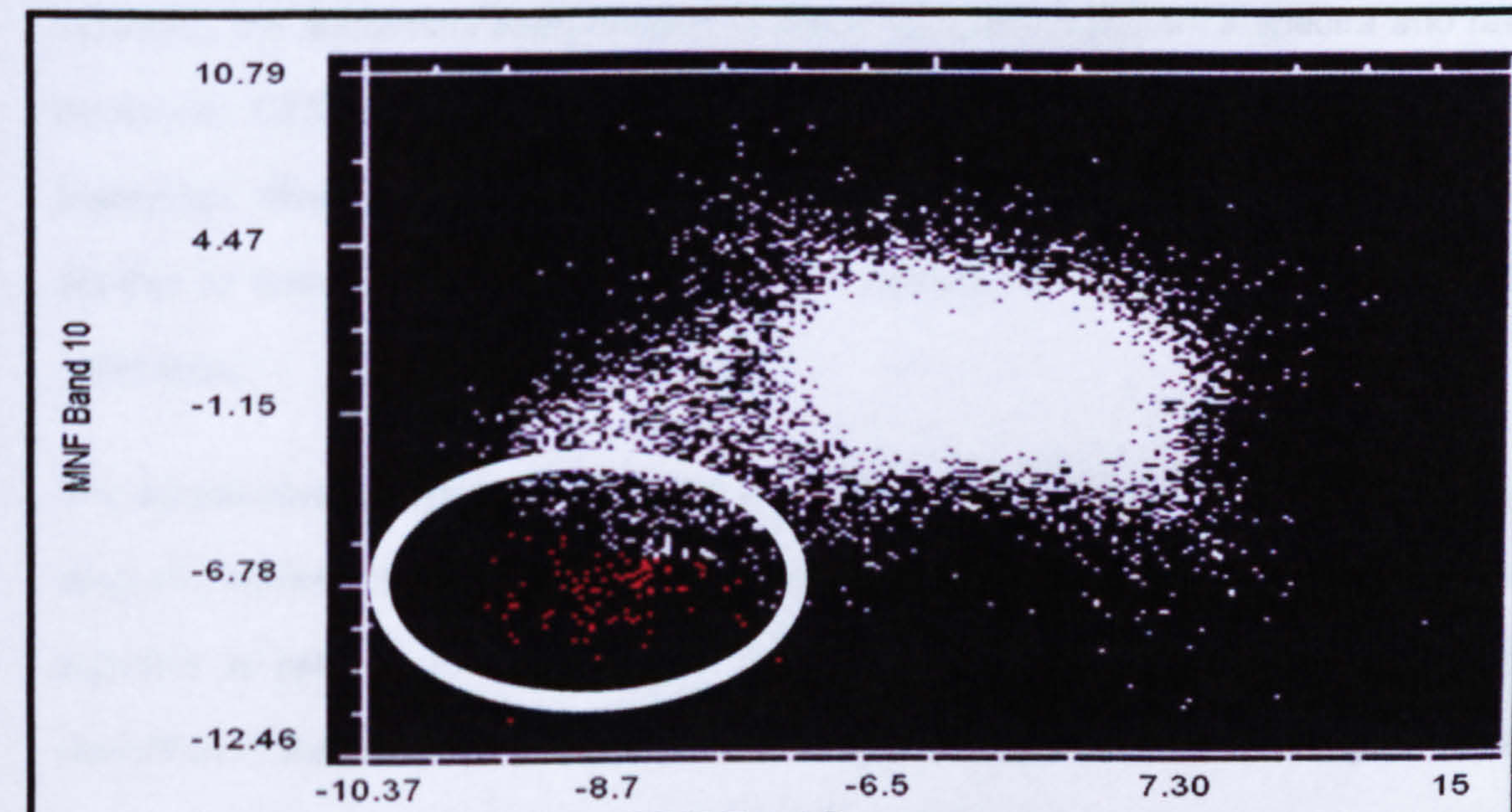


Figure 8.50: September 1999 data producing a 2-D scatter plot of the fifth and tenth principal components was found to clearly represent "stressed" vegetation using band 72



8.5.7 Spectral Sensitivity of Estimating Vegetation Health

To assess the sensitivity of airborne spectroscopy the objective was to search for patterns of correlation between REIP measurements for CASI and field data sets, matching them with physical parameters from field-based observations. Stress was measured using the NDVI parameter and field spectra were convolved to the CASI bandwidths for the 13, 48 and 72-band settings.

8.5.8 REIP (1st derivative) analysis of the CASI data

The analysis of the raw (radiance) CASI data showed that there was enough spectral contrast between the vegetation types for the CASI instrument (at all three band settings: 13, 48 and 72) to differentiate the different vegetation types. Additional spectral information acquired using the 72-band setting also allowed for more accurate differentiation of vegetation types both spatially and spectrally.

However, the successful interpretation of distributed field reflectance spectra and raw (radiance) CASI data requires prior detailed knowledge of site-specific contaminant dispersion. When such information is not available, a quantitative approach must be applied to both field and airborne spectra to enable the identification of "stressed" vegetation.

The quantitative analysis of CASI data processed to produce a first derivative image data set, showed the presence of small patches of anonymously "healthy" vegetation adjacent to cells 5, 10 and 11 while the "wetland" vegetation in the same region (identified in the raw CASI analysis) gave a very low response. Further analyses were conducted using the band 41 1st derivative image (703 - 712nm) and the 72-band CASI flight line from September. A distribution of very "stressed" vegetation was identified along the edges of cells 5, 10 and 11. The REIP at band 44 (727 - 731nm) clearly delineated the distribution of the "healthy" vegetation. Analysing the first derivative

CASI spectra for the distinct areas around cells 5, 10 and 11, clearly shows that there is quite a variation in the shape of the first derivative spectra, e.g. Figures 8.51 and 8.52. The pixels with the highest digital number (DN) in band 41 have broad, equi-dimensional plots with relatively low derivative values in band 44 whereas those pixels with a high digital number in band 44 and low digital number in band 41 have much narrower peaks with much higher derivative values in band 44, e.g. Figure 8.53. When the distribution of the "stressed" and "non-stressed" vegetation is analysed there is some overlap, especially along the edges of cells 5, 10 and 11. However, downstream the "stressed" vegetation cuts out abruptly.

When the spectral plots from the 72-band CASI 1st derivative images are analysed and compared to the ground survey results, the stressed and healthy vegetation were located close to one another at locations closer to the landfill, approximately 50cm apart. Since the individual pixels from the CASI images have a spatial resolution ranging between 0.25m^2 and 1m^2 they could not be composed purely of "stressed" vegetation but of a variable mixture of the "stressed" and "non-stressed" vegetation. This blurred both the CASI derived spectra and the CASI derived 1st derivative (REIP) profile. To analyse the effect of these "stressed" and "non-stressed" variations in one pixel on the overall spectra, the field spectra of "non-stressed" and "stressed" were combined with CASI spectra at different proportions. The first derivative of these synthetic spectra was then calculated, e.g. Figure 8.54.

Figure 8.51: Analysis of "stressed" and "non-stressed" vegetation showing that landfill leachate migrating off site caused "stressed" vegetation along the landfill edges (Legend: the strength of stressed vegetation from the 1st derivative of CASI band 41, 1st derivative in which 10 represents the highest level of stress while 1 represents the lowest level of stress)

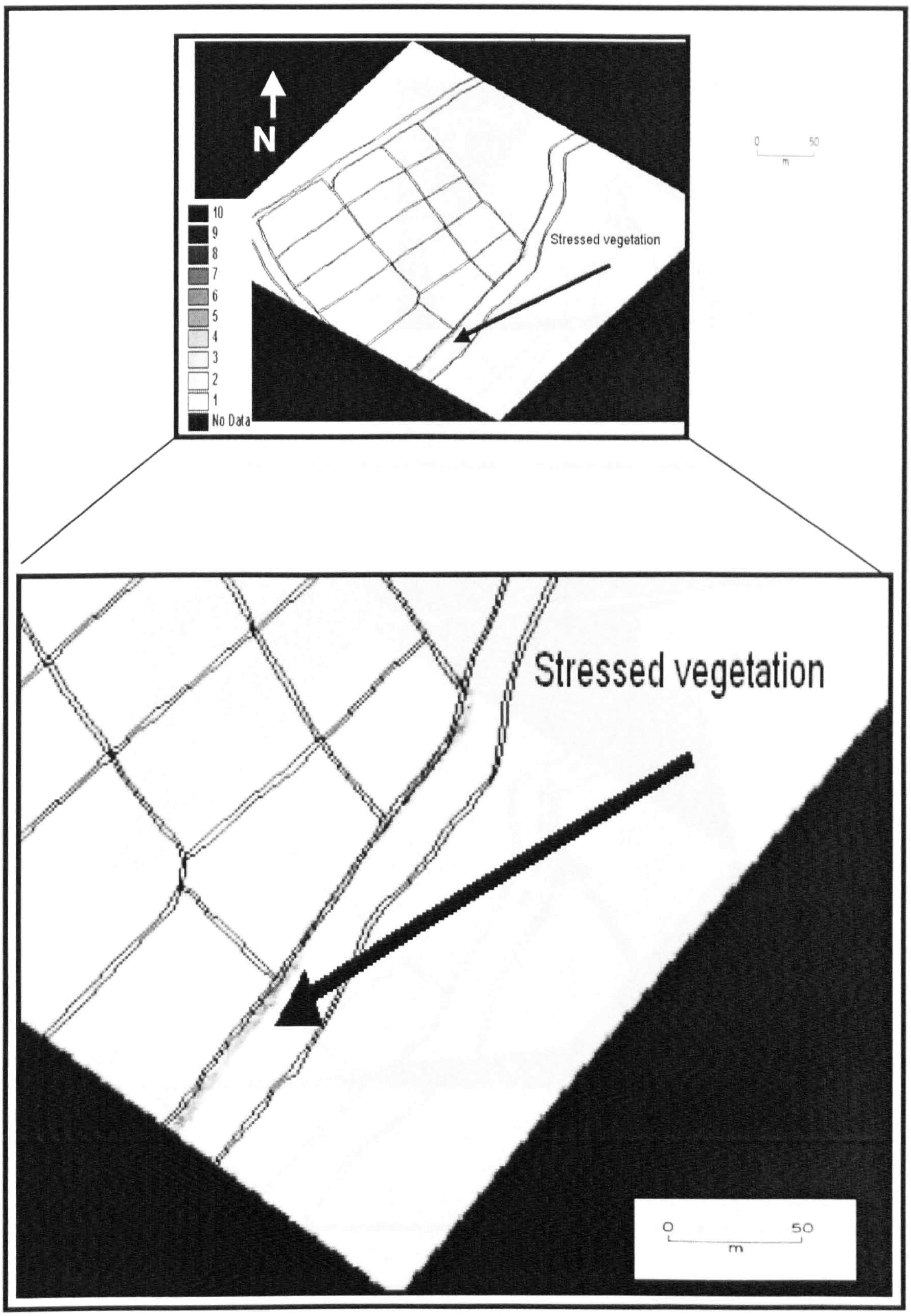


Figure 8.52: Analysing "stressed" and "non-stressed" vegetation showing that landfill-leaching caused "stressed" vegetation downstream from the site (Legend: the strength of stressed 'wetland' vegetation from CASI band 44 in which 10 represents the highest level while 1 represents the lowest level of stress)

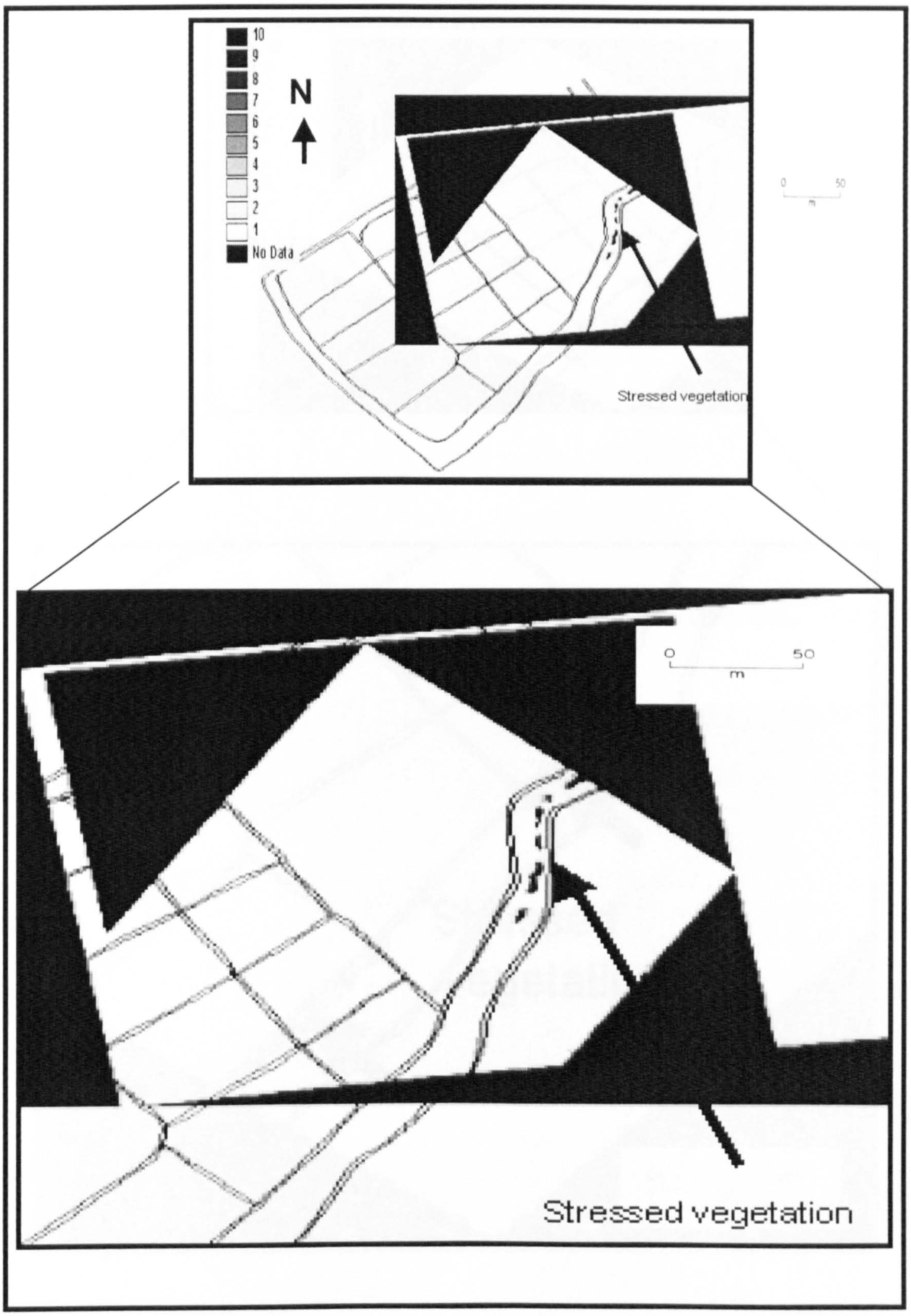


Figure 8.53: Analysing “stressed” and “non-stressed” vegetation clearly shows that landfill leachate migrating off site causes “stressed” vegetation along the landfill edges (Legend: the strength of stressed vegetation from Mixed Noise Filter analysis from CASI band 44 in which 10 represents the highest level while 1 represents the lowest level of vegetation stress)

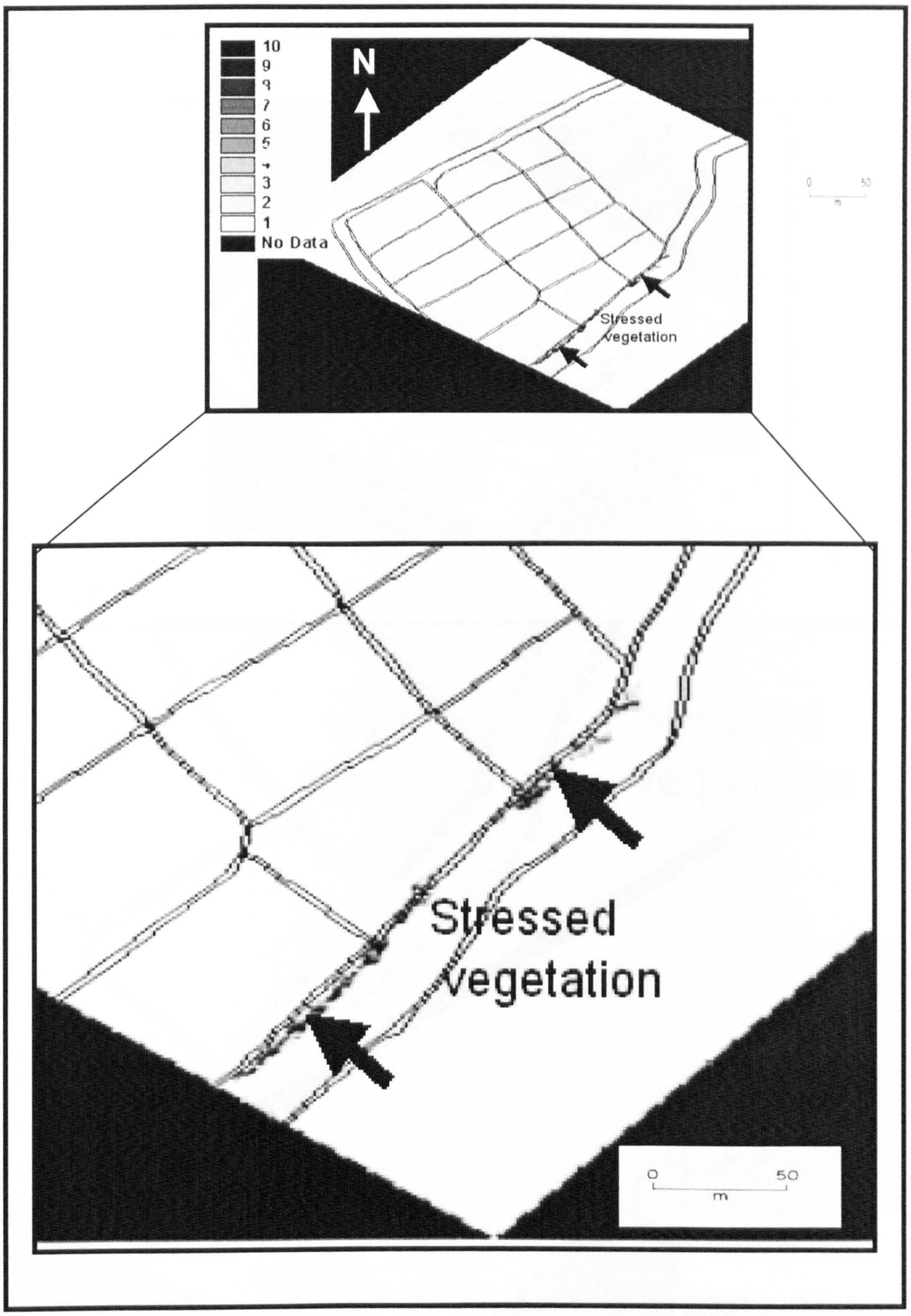
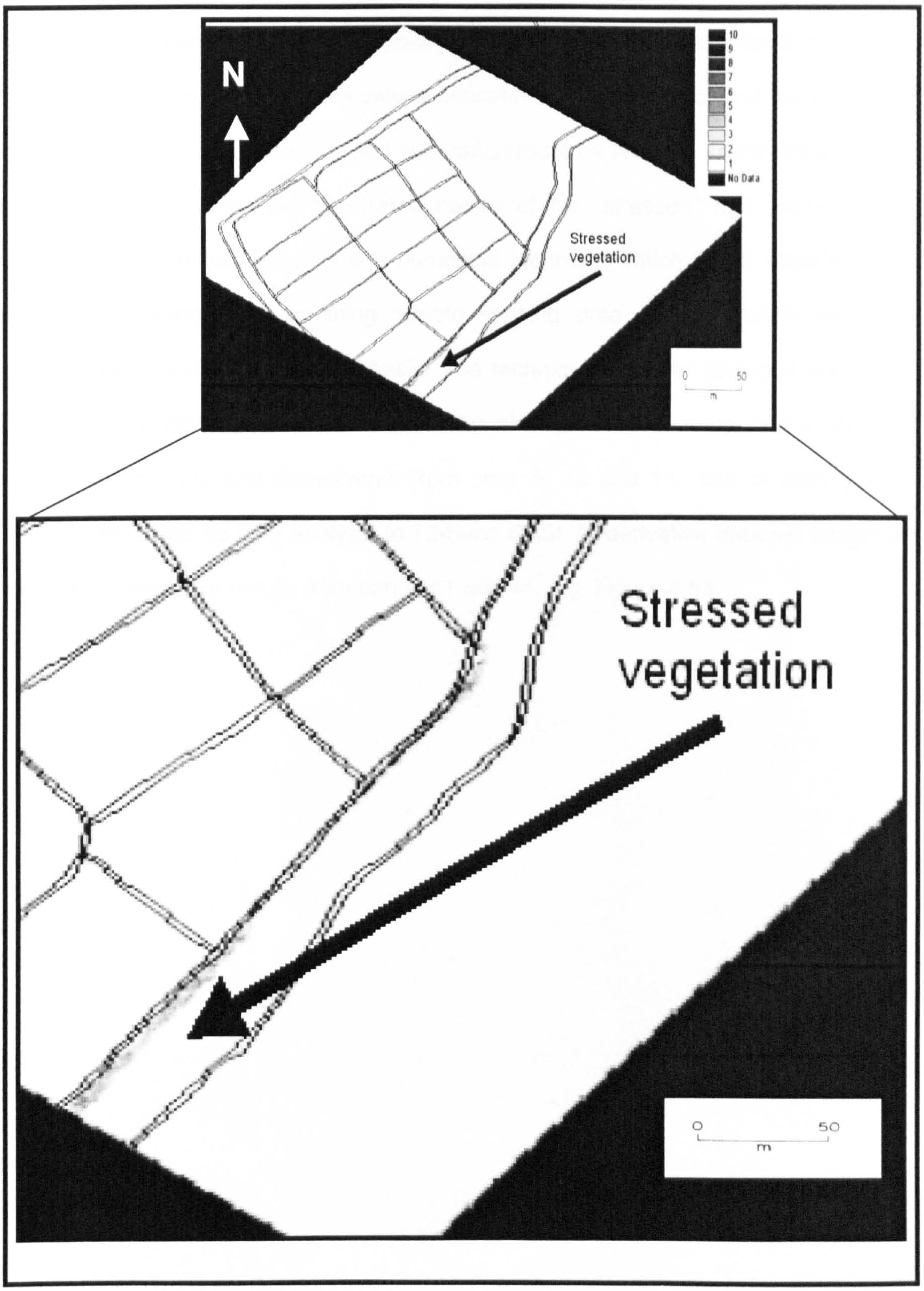


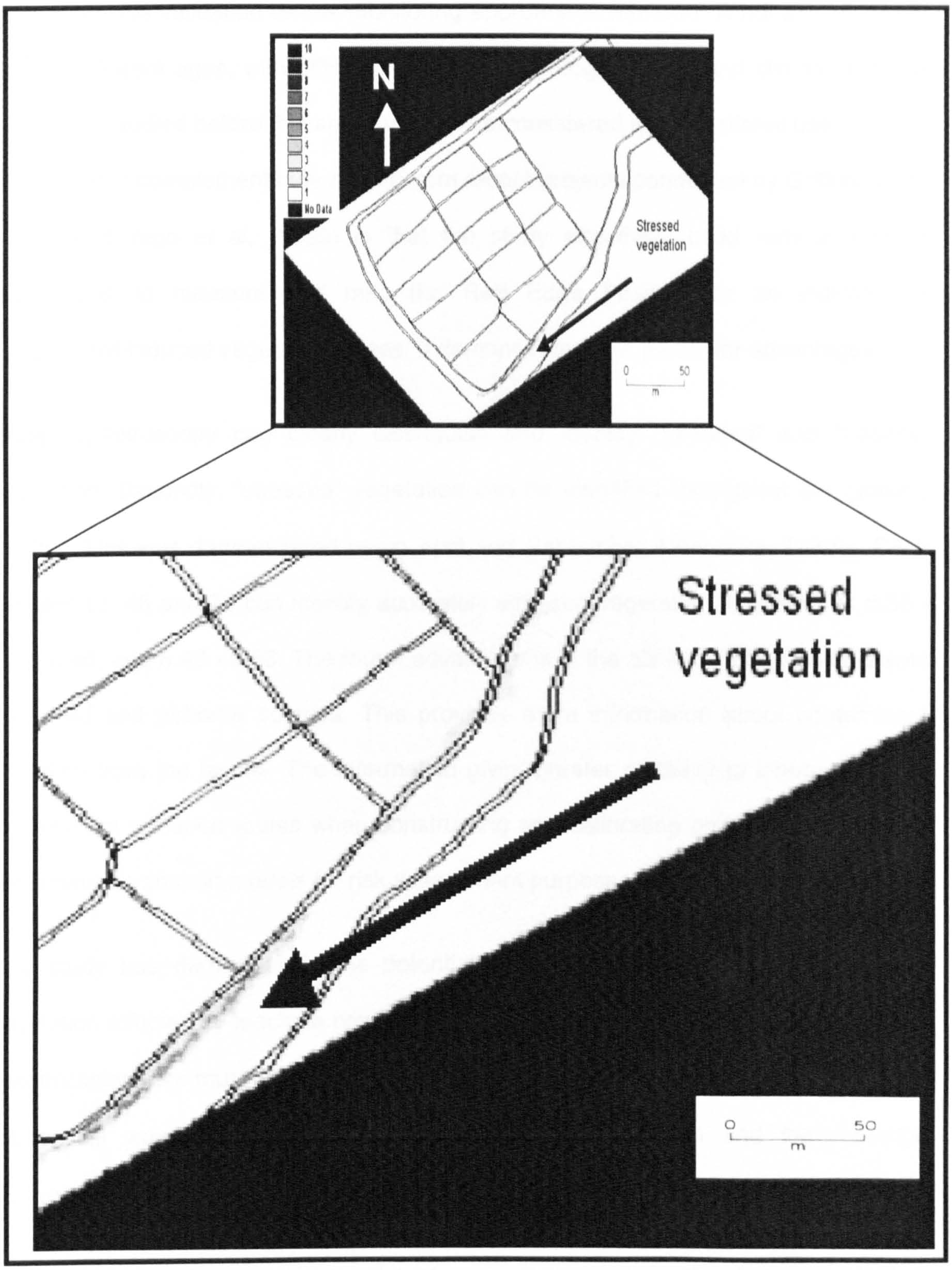
Figure 8.54: Field spectra of 'stressed' and 'non-stressed' vegetation were combined with CASI spectra at different proportions showing that landfill-leaching caused "stressed" vegetation off site (Legend: the strength of stressed vegetation using the 1st derivative of CASI band 72 in which 10 represents the highest level and 1 represents the lowest level of vegetation stress)



8.5.9 Semi-Quantitative Analysis of Vegetation Distribution

Many of the pixels had a small amount of very intensely stressed vegetation that was surrounded by healthy vegetation. The spectral response will not differentiate the small amounts of stressed vegetation in such pixels. In order to assess the degree of stressed vegetation located in each pixel, three points must be determined: the profile; the intensity of the first derivative values between band 39 and 45; and the proportion of the pixel made up by the stressed vegetation must be known. There are a number of methods to "un-mix" the spectral signature of the "stressed" and "non-stressed" vegetation. Match filter analysis is a numerical technique which is not classified as a quantitative method for examining remote sensing data as this usually relates to concentration data on a per pixel basis. The technique however provided successful results with a clear distribution of both the strength and quantity of the stressed vegetation adjacent and downstream from cells 5, 10 and 11. This is displayed by applying the matched filter analysis to 72-band CASI 1st derivative data set integrated with the 1st derivative results from bands 41 and 44, e.g. Figure 8.55.

Figure 8.55: The results of the match filter analysis on the CASI band 72 integrated with 1st derivative results from bands 41 and 45 showing that the distribution of "stressed" vegetation adjacent to cells 5, 10 & 11 (Legend: the strength of stressed vegetation in which 10 is the highest level while 1 is the lowest level of stress)



8.5.10 CASI and Contaminated Land: Summary Discussion

This study has presented results from only one study site. A much more comprehensive validation of this monitoring approach is required. A number of landfill sites of different ages, with different geological, hydrogeological and climatic settings need to be studied before this approach can be considered for operational use.

The research complements the results from recent projects conducted by Griffiths *et al*, (1996) and Jago *et al*, (1999) in that the study effectively used remote sensing applications to measure and map the Red Edge Position as an indicator of contaminant-induced vegetation stress. It demonstrates four particular advantages.

Firstly, spectroscopy can clearly distinguish and identify "stressed" and "healthy" vegetation. Secondly, "stressed" vegetation can be identified throughout the growing season. This was demonstrated using April and September 1999 data. Thirdly, CASI bandset 13, 48 and 72 can identify accurately stressed vegetation, e.g. Figures 8.38 - 8.39, 8.45, and 8.49 - 8.56. The fourth advantage is in the ability to integrate data sets from field and airborne sources. This provides more information about contaminant migration from the landfill. The information gives greater certainty to interpretation of contaminant transport routes when constructing and calibrating groundwater flow and contaminant transport models for risk assessment purposes, e.g. Figure 7.8.

This study has demonstrated the potential of reflectance spectroscopy to identify vegetation affected by leachate contaminated soil at a range of spatial resolutions. The spectroradiometer must have contiguous bands at sufficient spectral resolution over the critical wave range that measures chlorophyll absorption and the red-edge (between 650 and 750nm) to achieve this.

Hand-held or field-based spectroradiometers could be used as site assessment tools if airborne images and data sets became more readily available in industry. Re-designing hand-held or field-based spectroradiometers (if they were cost-effective and fieldfriendly when compared to other field assessment methods), could confirm the presence of vegetation stress and validate airborne data. Preliminary and repeat surveys using airborne- or satellite-based spectroradiometers could provide first-pass observations of large areas which could be followed up using ground-based spectroradiometers which could further focus field sampling surveys on the most affected areas in real time.

8.6 Summary of the Utility of New Innovative Methods

The increasing awareness of the environmental impact of contamination from landfill sites is drawing improvement in the accuracy of assessing site conditions and evaluating the risks posed by the site. Such assessments require detailed surveys that provide temporal and spatial information about hydrogeological conditions and contaminant concentrations. The difficulty is finding a method that improves accuracy without increasing costs unnecessarily.

This chapter has presented three new approaches that can enhance the site assessment. Investigation 1 (in Section 8.2) successfully tested Objective 2, applying kriging and using existing sample points at two study sites. The next step was to identify the optimal number of sample points needed to provide representative groundwater samples. The investigation results show that kriging can successfully simulate field conditions if an adequate number of initial samples are available and if regional or local conditions are adequately represented in the available data sets. The investigation underlined the role that regional and small-scale hydrogeological factors can have in both assessing and simulating site conditions. Both factors are heterogeneous, often unknown and overlooked or under-represented in field samples. Overall, kriging of groundwater levels proved to be an effective tool with which to improve the effectiveness of existing or future sample distributions, and confirm both regional and small-scale hydrogeological site conditions.

The second innovative approach implemented in investigation 2, was the application of Ground Penetrating Radar. The investigation (Section 8.3) tested Objective 1, the effectiveness of new techniques in providing valuable data that could be used to validate and improve the accuracy of site assessing findings. It demonstrates that a GPR can measure the depth of geological features that may be influencing contaminant migration from the landfill as well as the status of existing containment infrastructure. For old and unknown parts of a landfill, a GPR can measure the depth of buried waste and landfill cap thickness. The GPR also proved useful in investigating geophysical and hydrogeological theories about small-scale and unvalidated site assumptions. At Study Site A, the GPR effectively identified and mapped sand lenses originally thought to be present at the edge of cells 10 and 5. At Study Site C the GPR confirmed that both sides of the containment cut-off wall has a vertical feature with differing saturation levels, indicating either a weak spot in the wall or a sand-gravel lens which the wall cuts through. Both GPR applications proved effective in confirming assumptions about the small-scale near-surface heterogeneity around problem areas at both landfill sites. In addition, the data collected provided automatic images of subsurface conditions, adding conceptual certainty to the understanding of geophysical conditions around the sites.

The third investigation successfully used reflectance spectra to identify contaminated areas at landfill Site A. This investigation also tested Objective 1, demonstrating the ability of field and airborne spectral instruments to measure changes in the spectral response of vegetation on landfill surfaces. These changes can be used to infer the presence of landfill leachate, and paths of leachate migration.

These results suggest that field-based spectroscopy could be used to give immediate information on the presence and intensity of contaminant migration paths. This technique has obvious advantages over existing techniques as the immediate results could assist in targeting areas for higher spectral resolution water and biochemical sampling.

CHAPTER 9: MODELLING ANALYSIS - RESULTS

9.1. Introduction to modelling

9.1.1 Introduction

This chapter looks at groundwater flow and contaminant transport models that are frequently used to estimate risk at landfill sites. Constructing such models is similar to assembling a puzzle. The aim is to get as many pieces of information as possible in order to build a full picture. It would be ideal if all the pieces could be obtained but in the case of landfill sites, every additional piece of information increases the cost of site assessment. The modeller must therefore determine how much field data is needed to construct a valid model, keeping costs and time constraints in mind. Investigations 4, 5 and 6 attempt an evaluation of the influence of field data on model construction with two objectives:

- To test the influence of field data on model performance
- To test the influence of modelling assumptions and modelling practices during model construction and calibration.

It is generally difficult to evaluate the influence of field data on models because quantifying the impact of transferring field conditions into a model is affected by (a) landfill sites being site-specific, (b) model parameters also being model-specific and (c) modeller bias. These investigations will show that field data available for model construction can influence the modeller's understanding of site conditions, affecting the modelling practices and assumptions made during model construction, as well as the model's performance when simulating groundwater and contaminant transport.

Three study sites were used to construct groundwater flow and contaminant transport models for each site. The data collected during the site assessment, along with data collected and inferred from the kriging, GPR and remote sensing applications, discussed in Chapter 8, sections 8.3 – 8.5 were used to construct four models. Two of the models were full landfill-scale ones, labelled large-scale models, while two others focused upon leachate leaking areas of Sites A and C, labelled small-scale models.

Once the models were calibrated and validated, a three-part sensitivity analysis was conducted, see Table 9.1 – investigations 4, 5 and 6. The three investigations used Visual MODFLOW v.2.8.2 and MT3D (versions MS and 1.5) to construct four groundwater flow and contaminant transport models for Study Sites A, B and C. Each base-model, outlined in Table 9.1, was set up according to the site-specific conditions.

Table 9.1: The three-part sensitivity analyses conducted to test the influence of field data and modelling practices on groundwater flow and contaminant transport model performance

Inv. #	Investigation Description	Site A Small-Scale	Site A Large-Scale	Site B Large-Scale	Site C Small-Scale
4	Test the influence of field data sets on model performance	✓	✓	✓	✓
5	Test the influence of increasing grid size on contaminant concentrations	✓	X	✓	✓
6	Test the influence of increasing hydraulic conductivity values on contaminant concentrations	✓	✓	✓	✓

- Investigation 4 tested the impact of field data derived from the site assessment evaluating model reactions when additional sets of field data were used to infer parameters
- Investigation 5 conducted a sensitivity analysis testing changes in grid size aiming to evaluate the sensitivity of this parameter on contaminant transport
- Investigation 6 constructed several model scenarios using minimum, average and maximum site-specific values of hydraulic conductivity. The aim was to test the sensitivity of the hydraulic conductivity parameter on model-estimated contaminated transport.

9.1.2 Model Assumptions

The base-models for groundwater flow (Visual MODFLOW v.2.8.2) were calibrated under both steady state and transient conditions with data sets that stretched over 365-day periods to account for seasonal variations. Contaminant transport in MT3D was modelled using advection and dispersion equations. A thorough sensitivity analysis of model parameters was conducted to ensure they represent site conditions. Ammonia was assumed to be an effective indicator of leachate presence in all the models. They therefore used field-based ammonia concentrations for each investigation. The initial concentrations used are listed in Table 9.2 and were based on the average annual ammonia concentration from monthly leachate samples at each study site. Visual MODFLOW and MT3D parameters were kept constant running all three investigations under steady state conditions, and changing only the spatially distributed parameters applicable to each investigation. The contaminant transport time (in days) is given in Table 9.2. Variation was due to computing limitations in MT3D, determined by the grid size, the number of model layers, and contaminant transport processed defined in each model (Zheng, 1990; Zheng *et al*, 2000). In each investigation, model sensitivity was evaluated by comparing ammonia concentrations (also referred to in this chapter as 'contaminant concentrations') in two ways. Firstly, by comparing contaminant concentrations away from the landfill. Secondly, by comparing changes in contaminant concentrations through time (in days). The aim of modelling contaminants was not to quantify whether ammonia concentrations were above, below or within accepted levels, but to observe model behaviour and identify whether it would significantly alter contaminant transport patterns inferred in each model.

Table 9.2: Description of initial concentrations and contaminant transport times used in Visual MODFLOW v.2.8.2 and MT3D models

	Inv. #	Site A Large-Scale Model	Site A Small-Scale Model	Site B Large-Scale Model	Site C Small-Scale Model
Initial Contaminant Concentrations In Model	4 5 6	a) 30mg/L unlined parts of model b) 100mg/L lined parts of model	30mg/L unlined parts of model	65mg/L landfill area	300mg/L landfill area
# of Simulated Days in Each Model	4 5 6	365 days	365 days	18250 days	5000 days

9.1.3 Evaluating Model Behaviour

In order to evaluate model behaviour, five questions were applied to the model results in each investigation:

(1) Did increasing amounts of field data, grid size or hydraulic conductivity influence contaminant transport in the models?

(2) Did the hydrogeological scenarios constructed for each site influence results?

(3) Did contaminant concentrations increase away from the site when data sets, grid size or hydraulic conductivity were increased?

(4) Did contaminant concentrations increase through time when data sets, grid size or hydraulic conductivity were increased?

These questions assisted in classifying contaminant and groundwater flow behaviour in which all three investigations showed that: (1) concentrations away from landfill were altered. By increasing data availability, grid size or hydraulic conductivity value contaminant concentrations varied, producing higher contaminant concentrations away from the landfill; (2) contaminant concentrations changed through time. By increasing the amount of data used in each model, different scenarios of grid size or hydraulic conductivity values were defined, altering contaminant concentrations through time; (3) hydrogeological assumptions influence model results. By producing different hydrogeological scenarios of each site, the inferred contaminant plume shape changed regardless of the grid size, conductivity value or amount of data used. This behaviour therefore confirmed three outcomes when detailed data sets are used during model construction. Table 9.3 cross-references the investigation results with the outcomes:

Outcome 1 Increasing the amount of field data available for model construction will influence the conceptual model of site conditions.

Outcome 2 Increasing the amount of field data available for model construction will influence the modeller's understanding of site conditions, therefore affecting how they define grid size, hydraulic conductivity or other model parameters that influence groundwater flow and contaminant transport.

Outcome 3 Increasing the amount of field data available for model construction will alter the contaminant concentration being modelled and is likely to increase the accuracy of such models, under the condition that model construction, calibration and validation are conducted by following good modelling practices.

The following sections will discuss each investigation, focusing on evaluating model behaviour.

9.2 Investigation 4: Field Data Used in Model Construction

9.2.1 Objective

The aim of investigation 4 was to test the sensitivity of model results on different data sets, derived from different field assessment methods. Four site models were used:

- The large-scale model of Site A
- The large-scale model of Site B and
- The small-scale model of Site A
- The small-scale model Site C.

Table 9.3 Cross-referencing investigation results with the evidence that they provide for each of the three outcomes (Outcomes 1-3 are described in section 9.1.3)

Investigation and Models Produced	Outcome 1	Outcome 2	Outcome 3
Investigation 4: Tested the availability of field data and field assumptions in model construction			
Study Site A Large-Scale	✓		✓
Study Site A Small-Scale	✓		✓
Study Site B Large-Scale	✓		✓
Study Site C Small-Scale	✓		✓
Investigation 5: Tested the sensitivity of grid size			
Study Site A Small-Scale		✓	✓
Study Site B Large-Scale		✓	✓
Study Site C Small-Scale		✓	✓
Investigation 6: Tested the sensitivity of hydraulic conductivity			
Study Site A Large-Scale		✓	✓
Study Site A Small-Scale		✓	✓
Study Site B Large-Scale		✓	✓
Study Site C Small-Scale		✓	✓

9.2.2 Description of Models Used

The investigation took the findings of preliminary and detailed studies at each landfill site and produced several hydrogeological scenarios, in which the amount and extent of data used was increased in successive scenarios. The amount of data varied according to the amount of historical information and assessment instruments used at each study site, Table 9.4. The large-scale models (Site A and Site B) both assumed two hydrogeological scenarios. For Site A, scenario 1 it was assumed that landfill cells were not important contributors to off-site leachate migration, while the scenario 2 model assumed that they were, e.g. Figure 9.1. For Site B, scenario 1 assumed that there were low regional groundwater levels, while scenario 2 assumed high regional groundwater levels, e.g. Figure 9.2. Both models tested three data sets, e.g. Table 9.3. In the small-scale models (Sites A – Figure 9.3 and Site C – Figure 9.4), one hydrogeological scenario was assumed, using three data sets, e.g. Table 9.4.

Table 9.4: The type of data used to produce each modelling scenario

	# of Hydrogeological Scenarios	Boreholes and Piezometer data sets	GPR and regional hydrological data	Remote sensing data sets
Site A (LG)	2			
Scenario A		✓		
Scenario B		✓	✓	
Scenario E		✓	✓	✓
Site B (LG)	2			
Scenario A		✓		
Scenario B		✓	✓	
Scenario C		✓	✓	✓
Site A: (SM)	1			
Scenario 1		✓		
Scenario 2		✓	✓	
Scenario 3		✓	✓	✓
Site C (SM)	1			
Scenario 1		✓		
Scenario 2		✓	✓	
Scenario 3		✓	✓	✓

<u>Legend:</u>
GPR = Ground Penetrating Radar
= Type of data used in each scenario
LG = Large-scale Model
SM = Small-scale Model

Figure 9.1 Two hydrogeological models of Site A with field data incrementally added

Location in Model		Scenario A , B and E Conductivity Values
Landfill	(pink)	Kxyz=0.0001 m/s
Gypsum Layer	(white)	Kxyz=$1 e^{-5}$ m/s
Glacial Till	(dark blue)	Kxyz=$1 e^{-7}$ m/s
Sand Gravel Lens	(orange)	Kxyz=0.0099 m/s
Clay Landfill Liner	(light blue)	Kxyz=$1 e^{-10}$ m/s
Sand-Till and Landfill Liner	(green)	Kxyz=$1 e^{-5}$ m/s
Circled Area		Changes in initial conditions

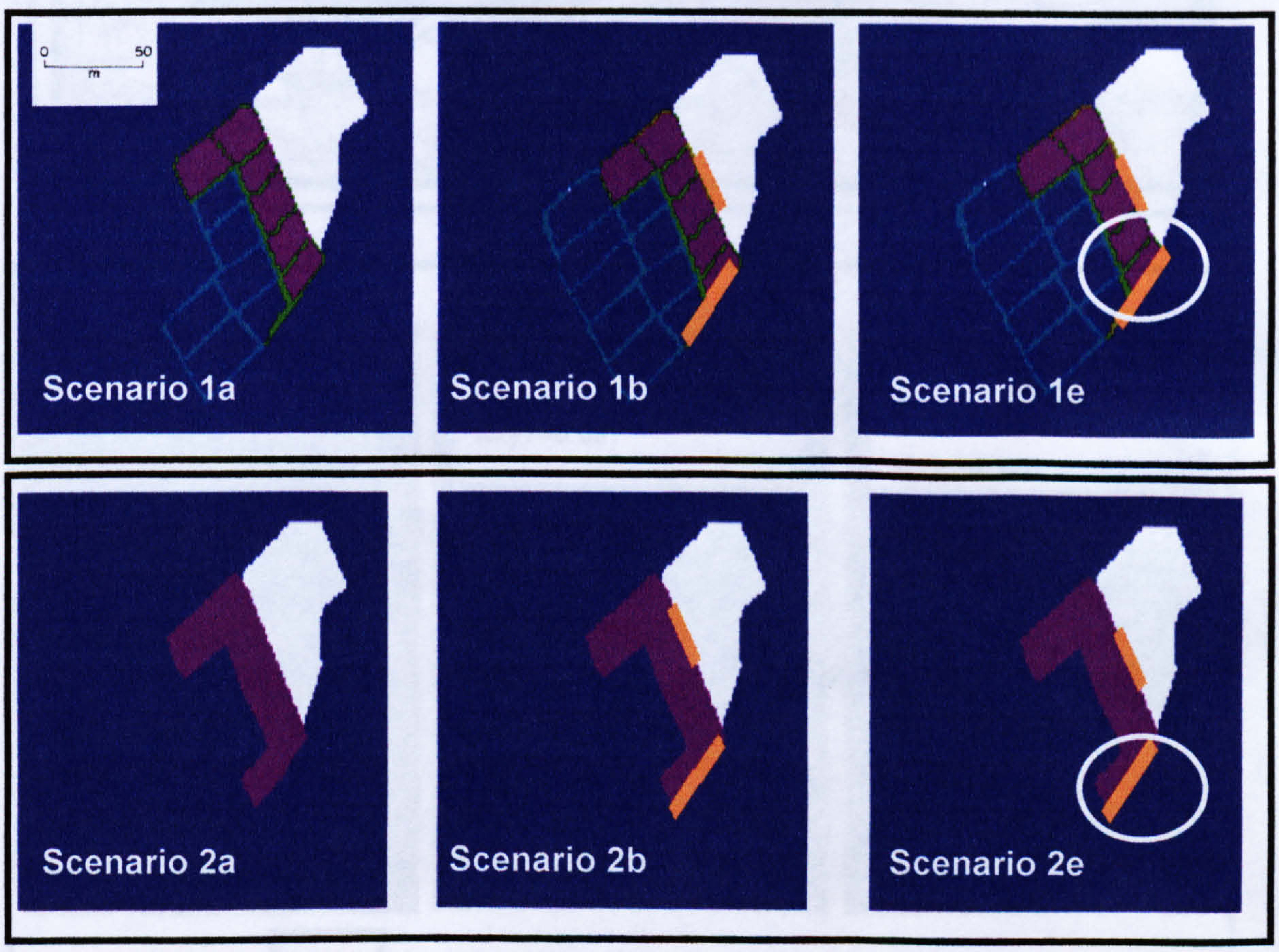
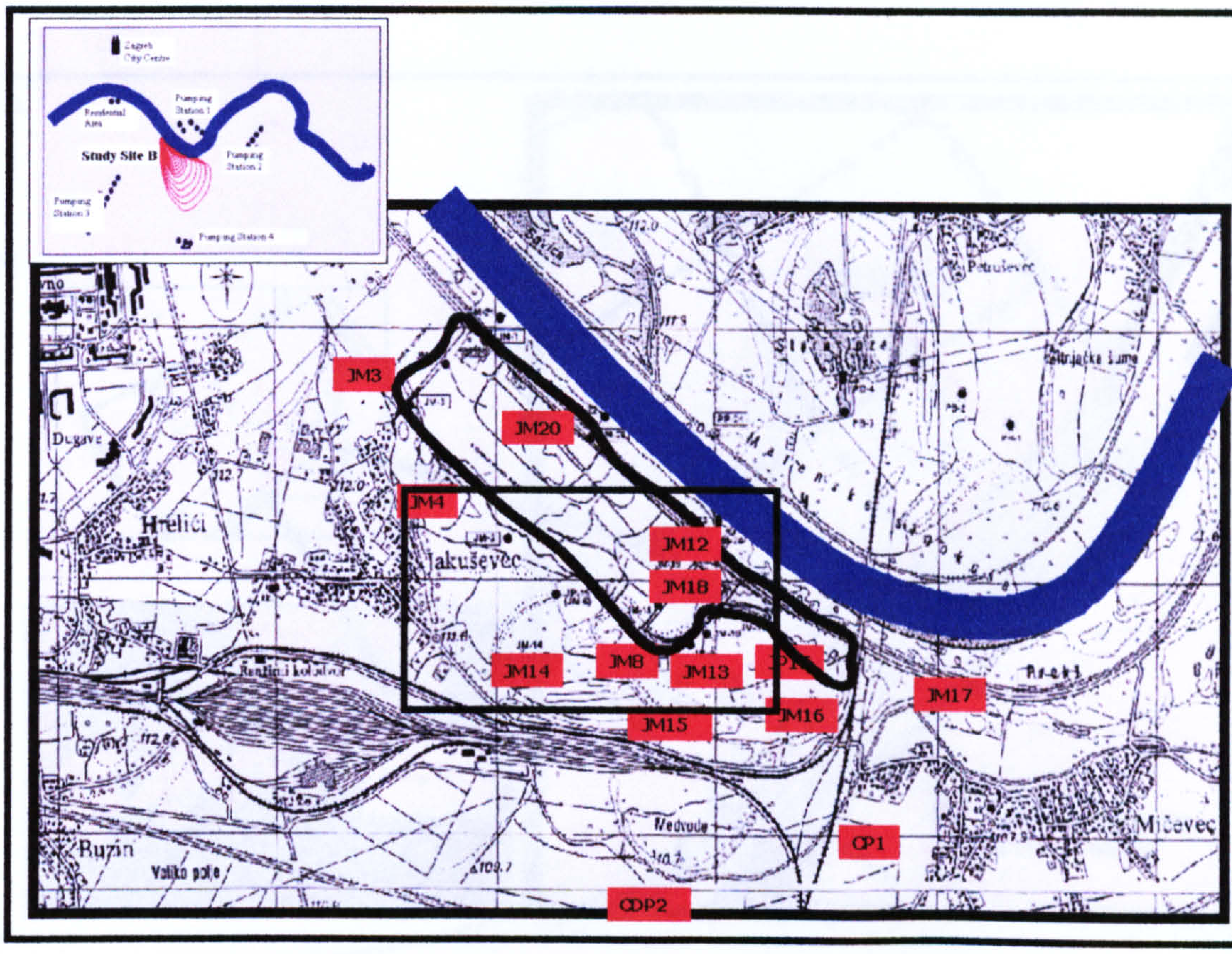
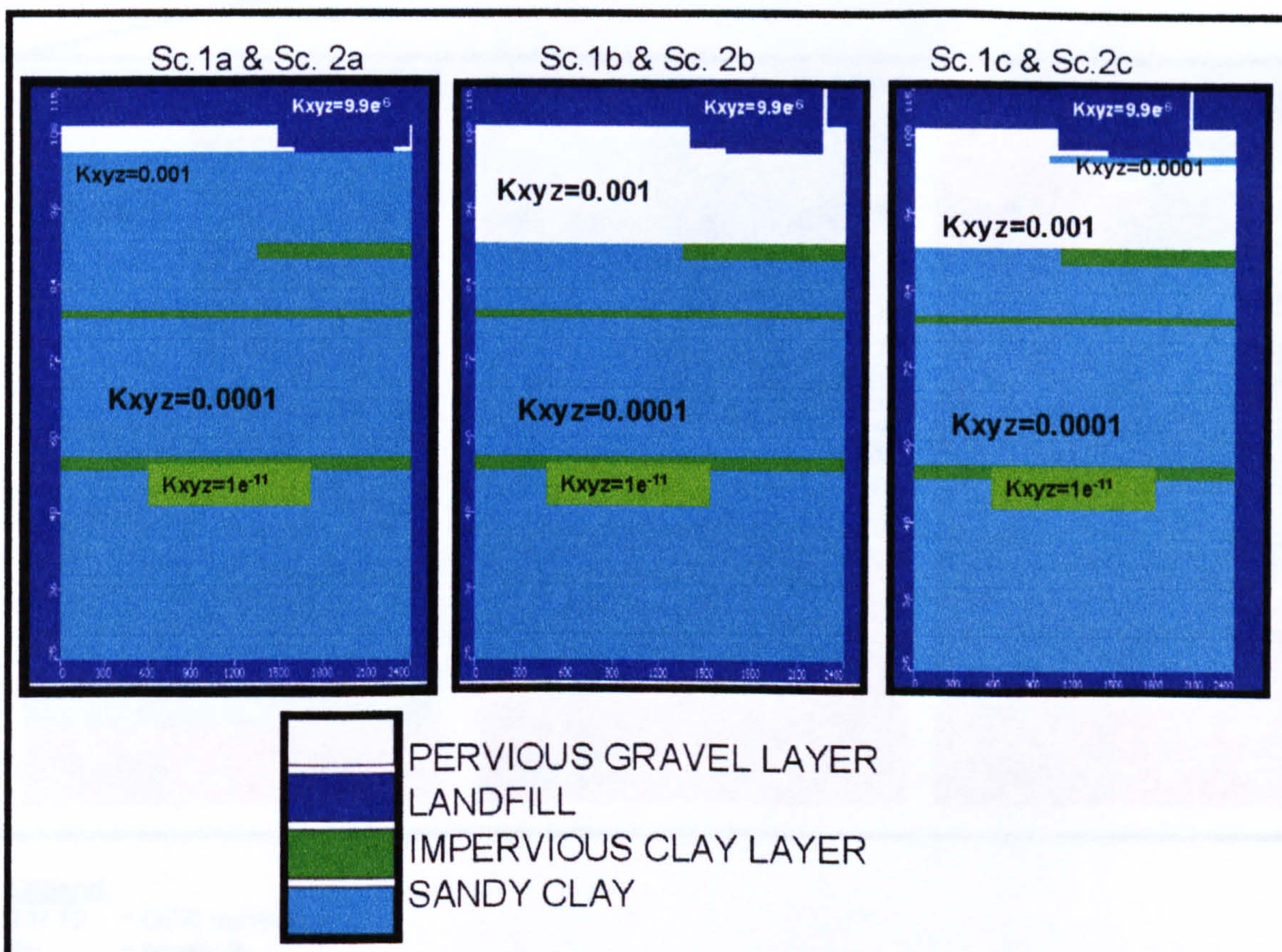


Figure 9.2 Site B cross-sections of scenarios A, B and C





Legend:

Hydraulic conductivity values are in m/s

Scenario 1= Low regional groundwater Levels

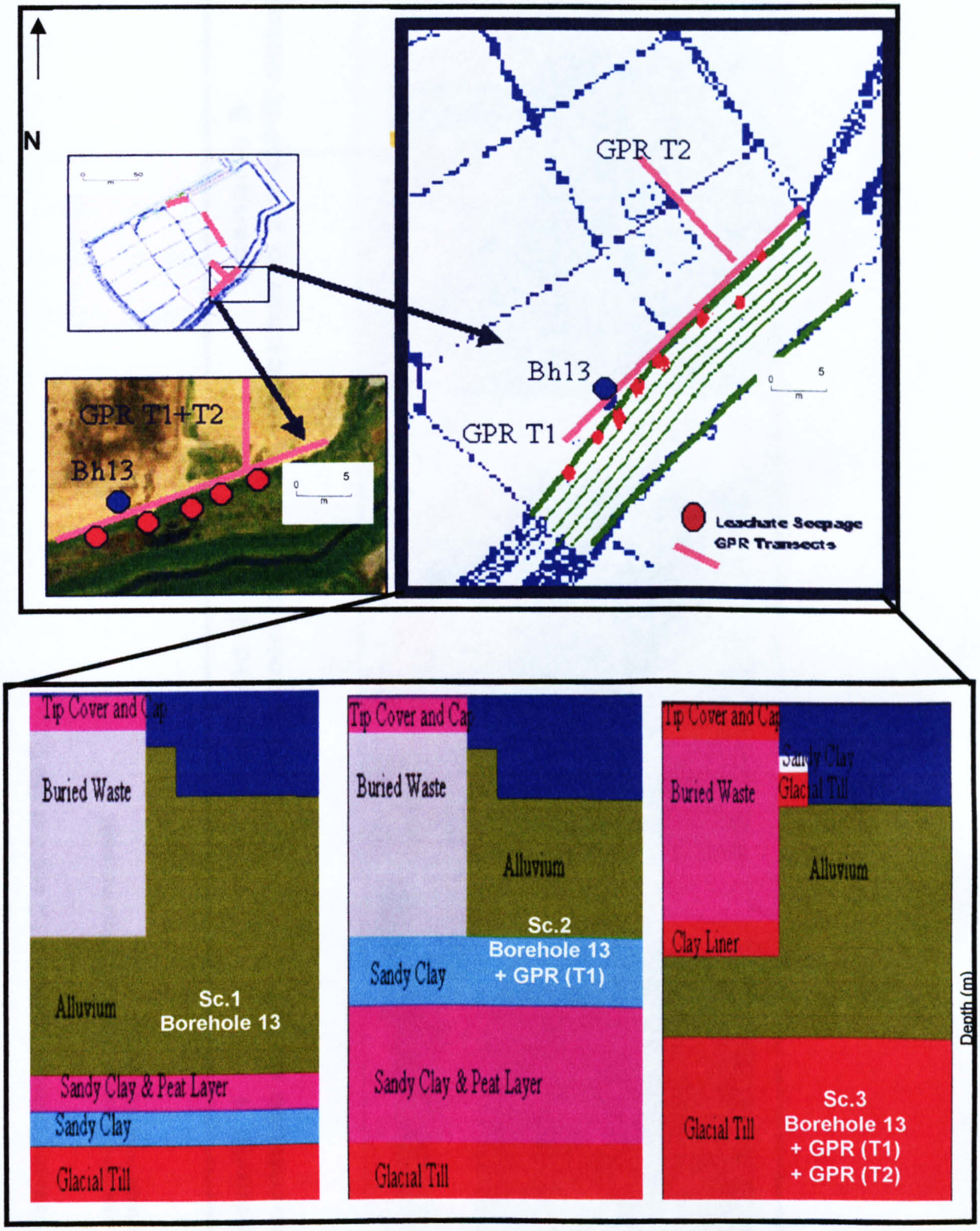
Scenario 2 = High regional groundwater levels

JM20 = piezometer points

= Sava River

= landfill edge

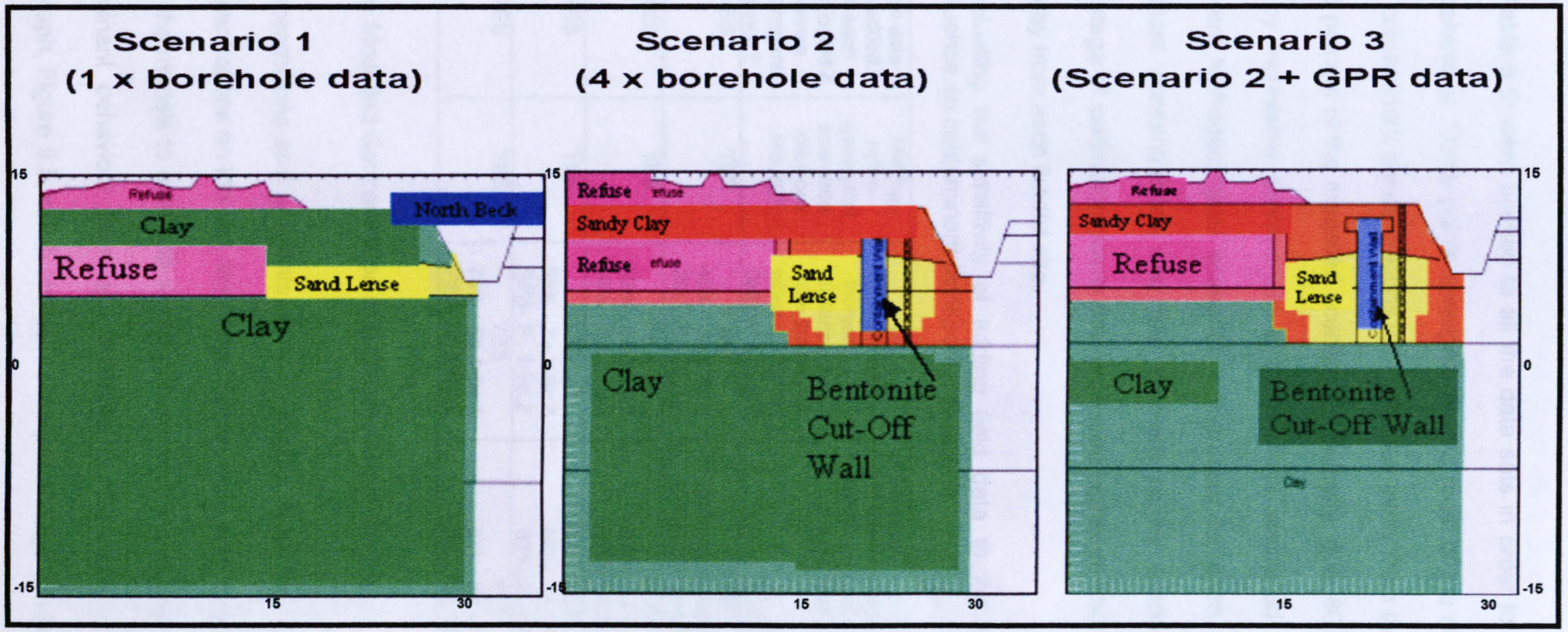
Figure 9.3 Cross-section illustration of the small-scale model scenarios of Site A



Legend:
T1/ T2 = GPR transect
Bh = borehole
X axis in the cross-section maps = distance in metres away from the landfill
Y axis in the cross-section maps = depth in metres

Figure 9.4 Cross-section models of scenarios 1,2 and 3 of Site C

Legend:
X axis in the cross-section maps = distance in metres away from the landfill
Y axis in the cross-section maps = depth in metres



9.2.3 Model Behaviour

Five questions (Table 9.5) were applied to all the data sets in order to infer common trends in model behaviour. Three points emerged. Firstly, most of the models reacted by having higher contaminant concentrations with distance away from the landfill. This was noticed in 75 percent of the modelled scenarios. Secondly, over 80 percent of the models reacted by increasing contaminant concentrations simulated through time. Thirdly, the different hydrogeological scenarios constructed for each site produced different contaminant concentrations and had different plume shapes. Figure 9.5 graphs the percentage of calibration points that produced higher or lower contaminant concentrations away from each landfill site.

Table 9.5 Evaluating the sensitivity of adding field data to the model and its influence on contaminant transport simulations

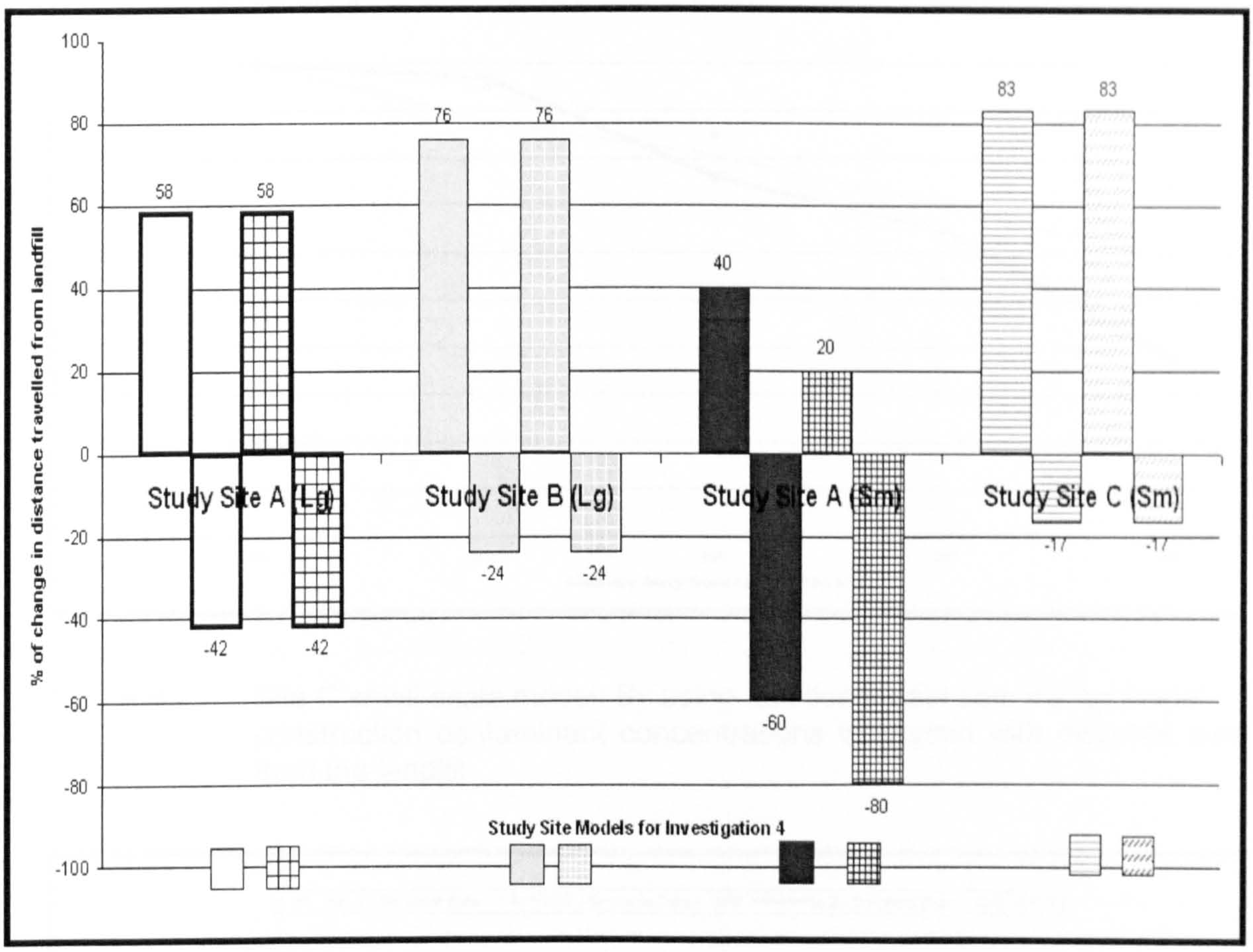
Study Site	When data was added with each scenario, did it influence contaminant transport?	Did the hydro-geological scenarios influence results?	Did concentrations increase with distance away from the landfill when additional data was used?	Did contaminant concentrations increase through time when additional data was used in model construction?
Small-scale model of Site A	YES	YES	NO 40% - Sc.1-Sc.3 20% -Sc.1-Sc.2	No 100% decrease SC.1-Sc.3 100% decrease SC.1-Sc.2
Small-scale model of Site C	YES	YES	YES 83% - Sc.1-Sc.3 83% - Sc.1-Sc.2	YES 100% Sc.1-Sc.3 100% Sc.1-Sc.2
Large-scale model of Site A	YES	YES	YES 58% - Sc.1-Sc.3 58%- Sc.1-Sc.2	YES 66%- Sc.1-Sc.3 33%-Sc.1-Sc.2
Large-scale model of Site B	YES	YES	YES 76% - Sc.1-Sc.3 76%- Sc.1-Sc.2	YES 83% -Sc.1-Sc.3 83%- Sc.1-Sc.2

9.2.4 Comparing Modelled Concentration with Distance from the Site

Contaminant concentrations and gradients were compared in all four of the models. Contaminant concentrations on the last day of calculation were compared at calibration points in each of the models to establish whether there were increasing or decreasing trends in contaminant behaviour from one scenario to the next. The data were compiled into a graph, Figure 9.5, which compared the percentage of calibration points in each model in which contaminant concentrations increased or were influenced when data were added. The graph shows that most of the calibration points in all the models were significantly altered when additional field data were used to infer model parameters. On the last day of simulation, contaminant concentrations increased in three out of four models when additional data was used (Site A-Lg., Site B and Site C).

The small-scale models showed sensitivity to field data in that the difference between contaminant concentrations in each scenario increased with distance from the landfill. The Site A small-scale model experienced lower concentrations, e.g. Figure 9.6, while the Site C model produced higher concentrations away from the landfill, Figure 9.7. When comparing the contaminant gradients in these two models, the Site C model (Figure 9.8(b)) shows that the extent of the contaminant is the same in all three scenarios however the internal distribution of patterns in the different data scenarios produced differing results. This was not as evident in the Site A model (Figure 9.8(a) indicating that the influence of data sets on contaminant transport could be site-specific as well as model-specific. The large-scale models also showed sensitivity to adding data sets. They (Site A and B) produced 58 percent and 76 percent higher contaminant concentrations away from the site, Figure 9.9(a). In the Site B model, scenarios B and C produced identical results meaning that the models were sensitive to the additional information in scenario B, but not in scenario C, e.g. Figure 9.9(b).

Figure 9.5 Comparing contaminant concentrations at calibration points in each model showing that contaminant concentrations increased in three out of four models when additional data was used (Site A-Lg., Site B and Site C)



Legend:

Sm. = small-scale model, Lg. = large-scale model

- Positive values = % of calibration points in the model that produced higher contaminant concentrations (mg/l of N) when additional data was used in model construction
- Negative values = % of calibration point that produced lower contaminant concentrations (mg/l of N) when additional data was used in model construction

Figure 9.6: Site A small-scale model: By adding data, the contaminant concentrations generally decreased with distance away from the site

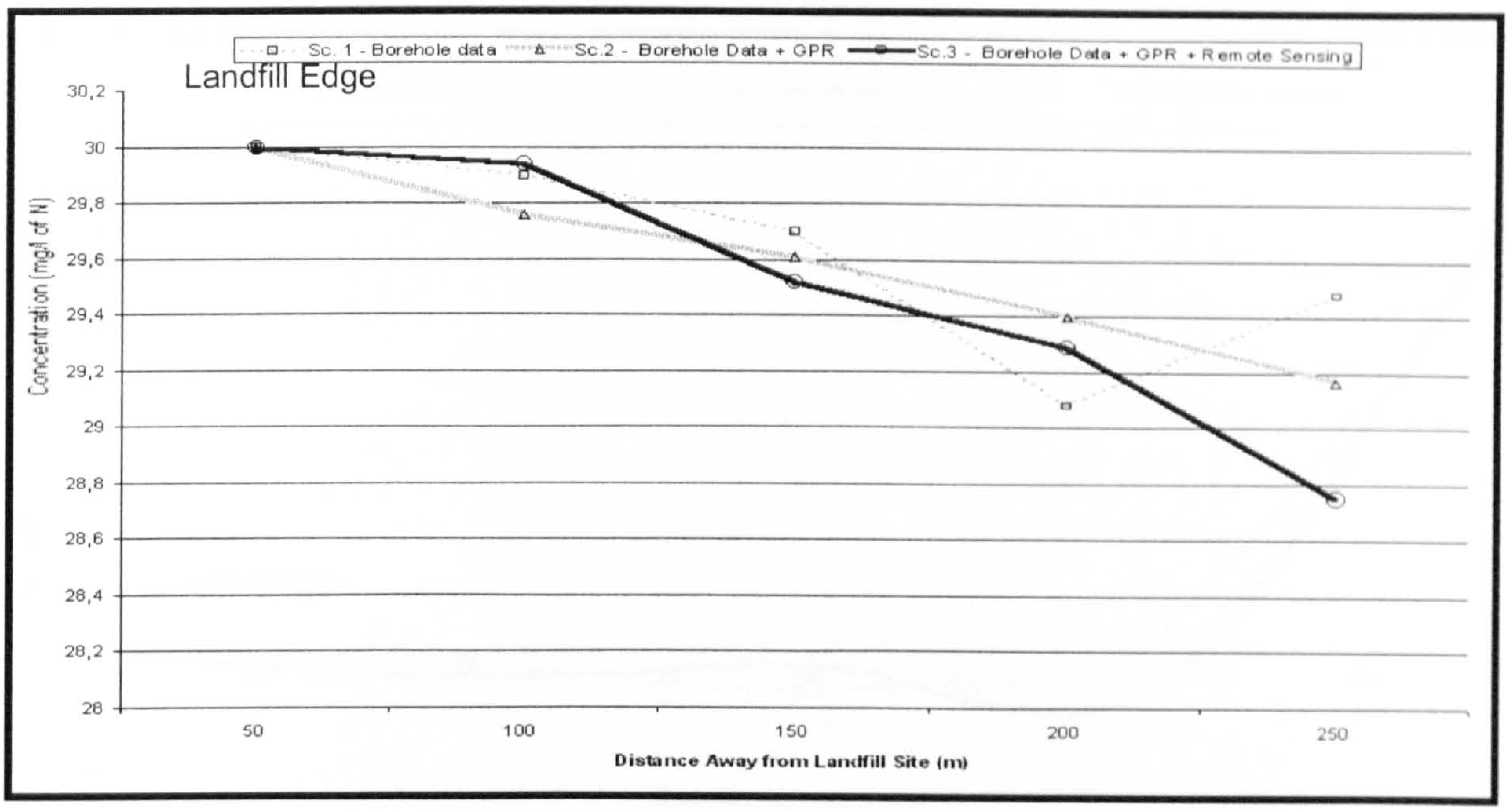


Figure 9.7 Site C small-scale model: By using additional data sets during model construction contaminant concentrations increased with distance away from the landfill

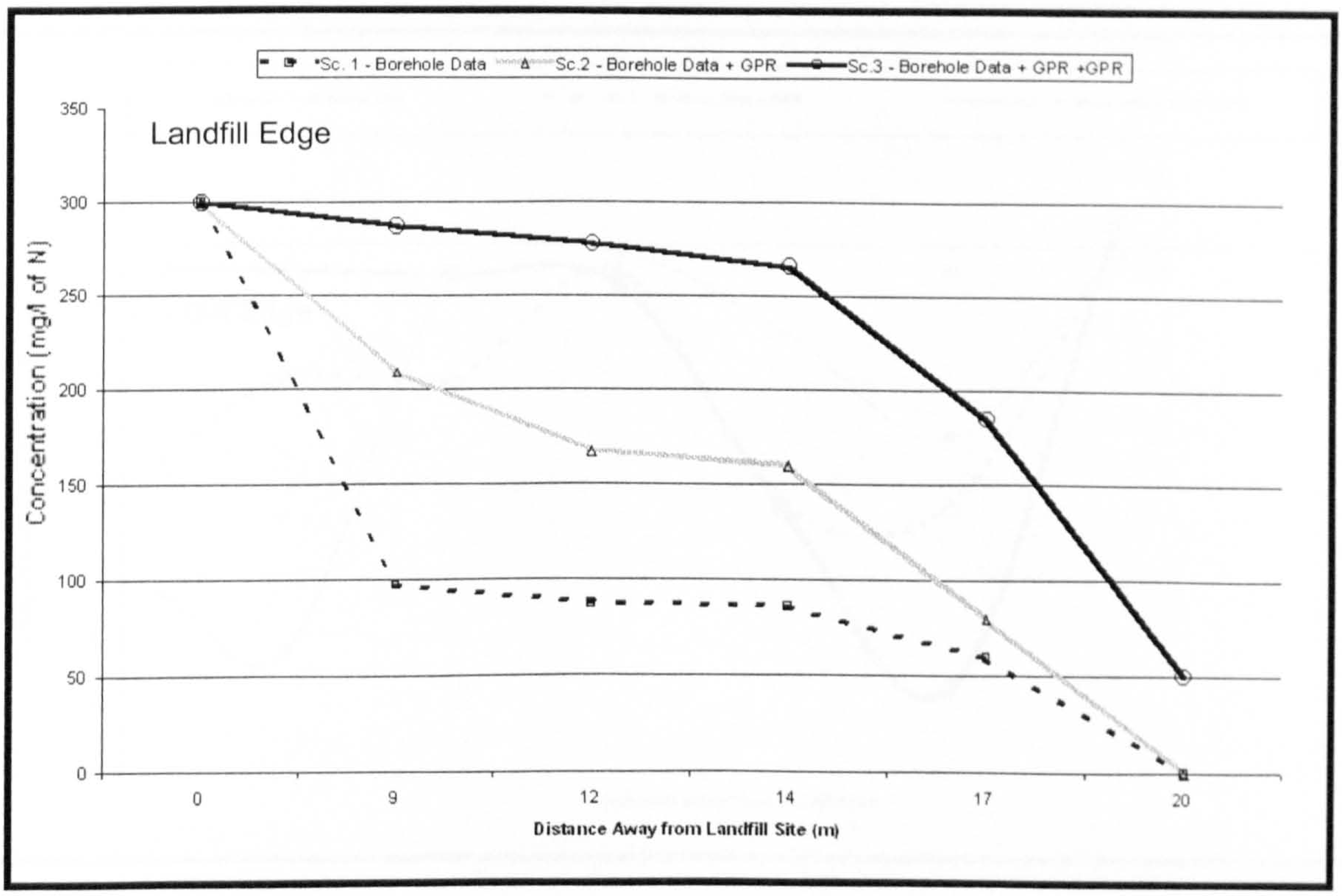


Figure 9.8(a) Site A small-scale model: comparing contaminant gradients showing that there was a slight difference in the scenario that used the largest data set at about 150 m from the site

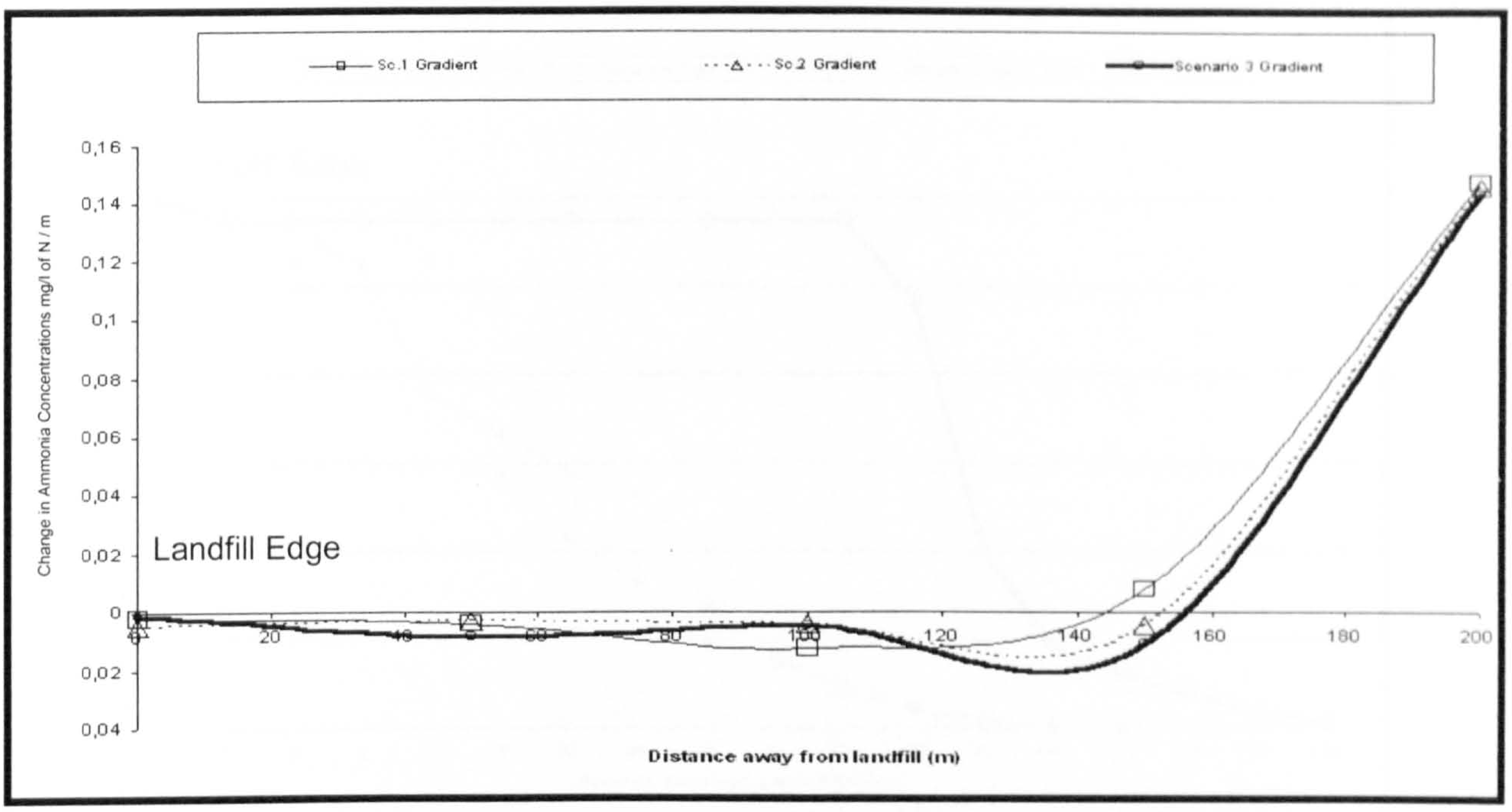


Figure 9.8(b) Site C small-scale model: The extent of contamination is the same however the internal distribution of contaminant concentrations differed significantly when additional data was used in model construction

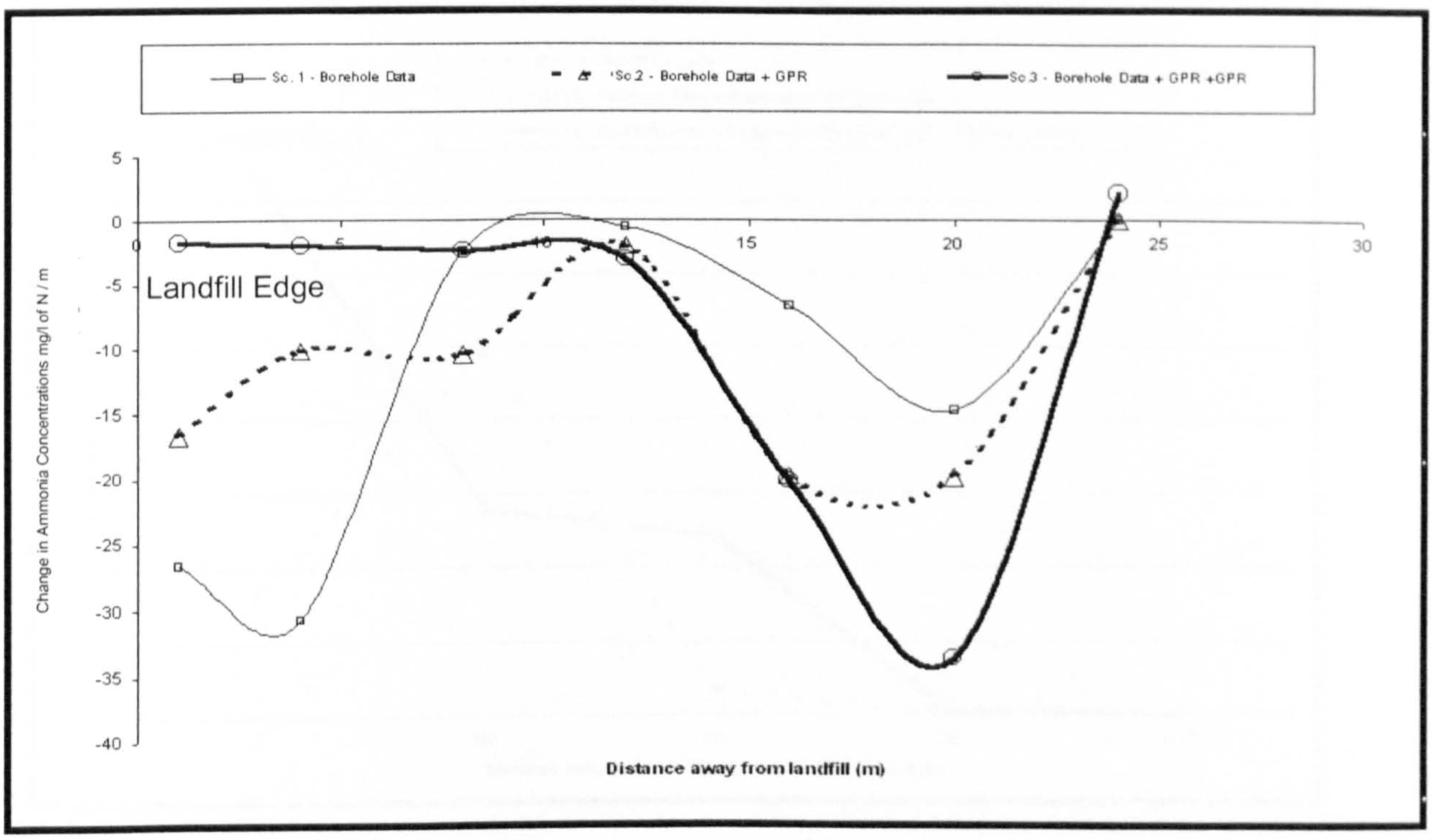


Figure 9.9(a) Site A large-scale model: Contaminant concentrations increased in scenarios 2 and 3 when additional data was used

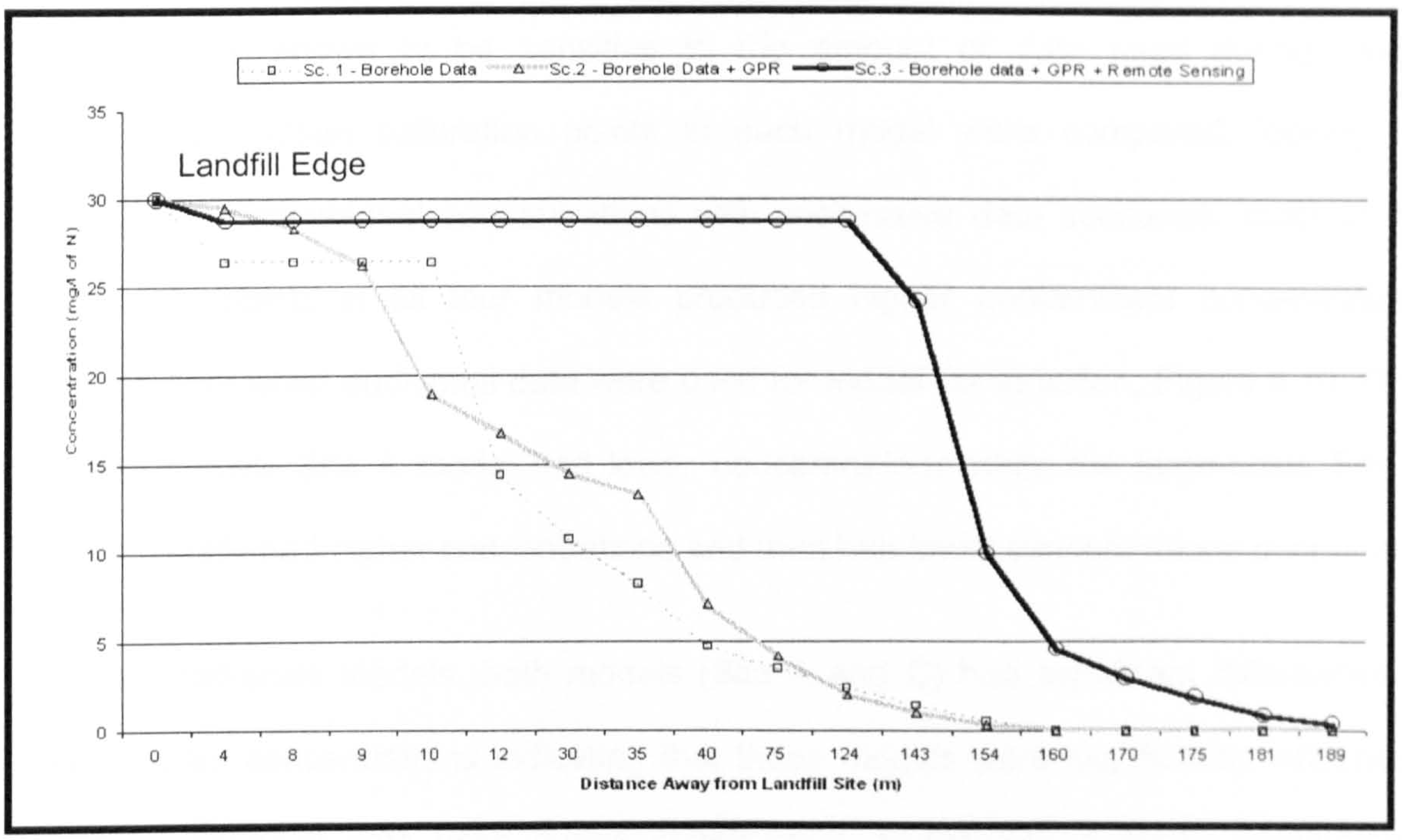
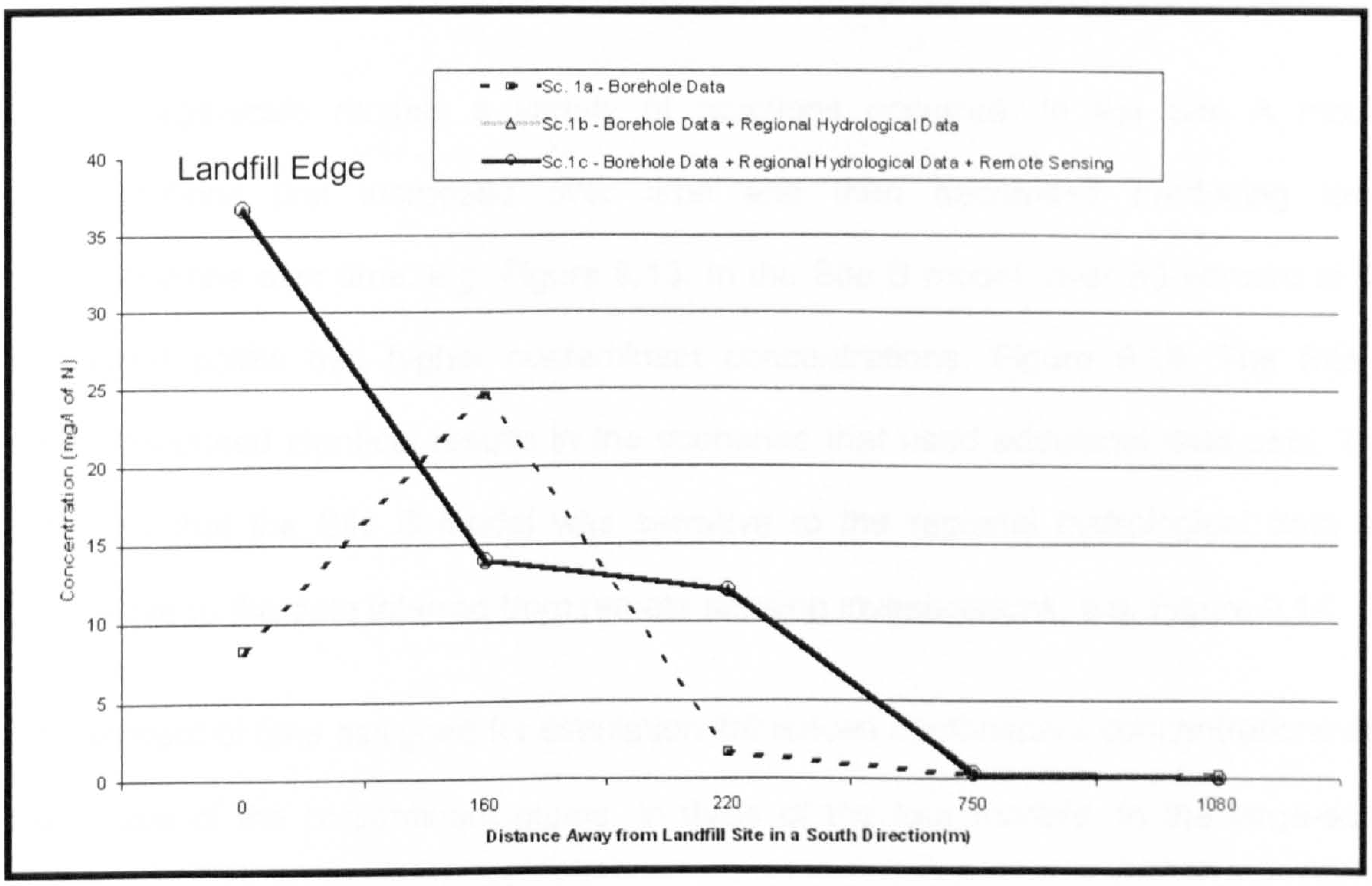


Figure 9.9(b) Site B large-scale model: When additional data was used in model construction, scenarios B and C produced identical results and contaminants were higher with distance away from the landfill



9.2.5 Comparing Contaminant Concentrations through Time

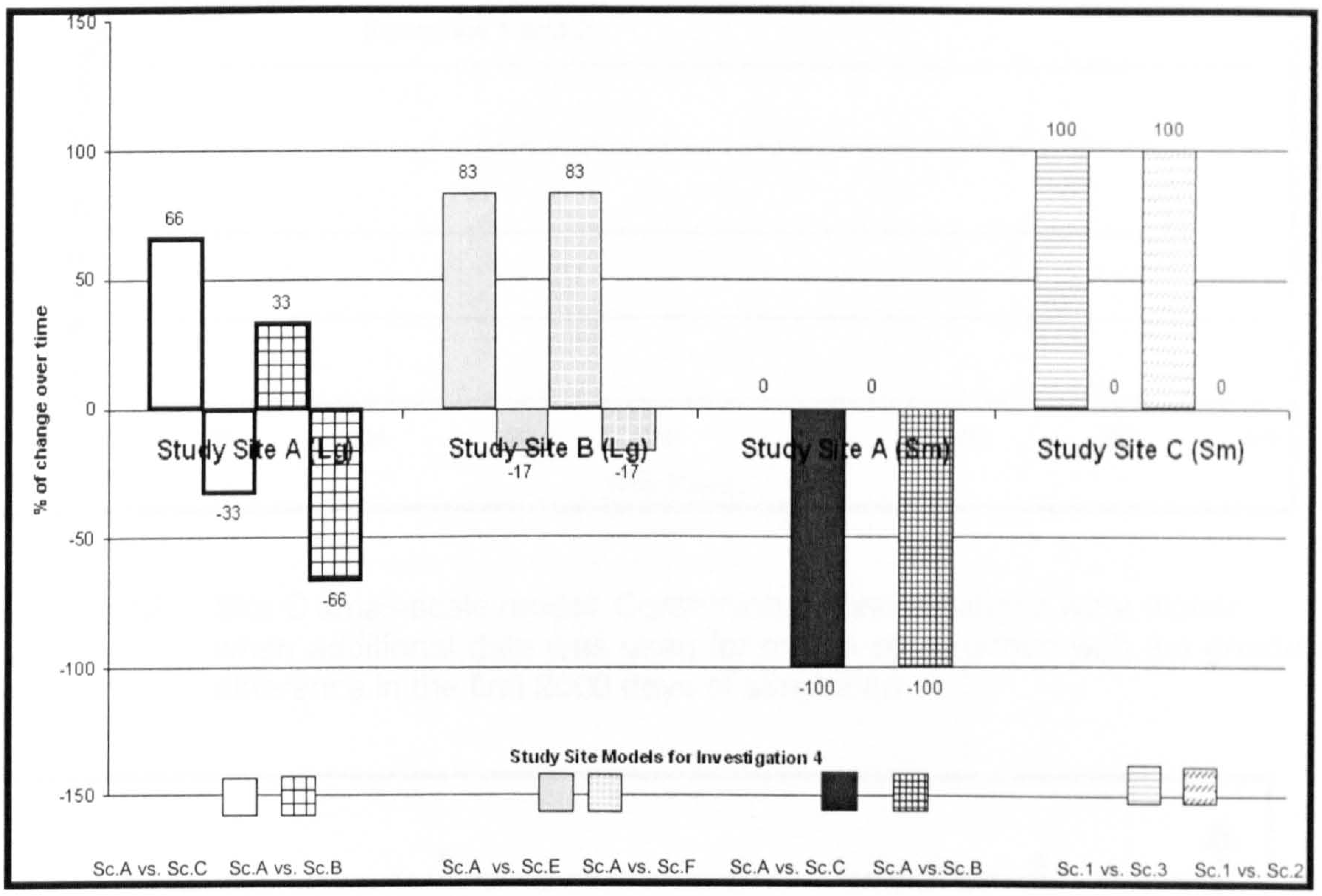
Contaminant concentrations were compared through time in each model. All the models were shown to be sensitive to the amount of data used during model construction. When calibration points in each model were compared, looking for changes in contaminant concentrations with successive data scenarios, most of the calibration points in all four models produced higher contaminant concentrations through time when additional data were used for model construction, Figure 9.10. Only the small-scale Site A model had lower concentrations while the large-scale Site A model initially had higher concentrations and then had lower concentrations over time.

In the small-scale models, both models (Site A and C) had significant differences in contaminant concentrations indicating that these models were significantly influenced by the amount of data used during construction. In the Site A model contaminant concentrations were lower, e.g. Figure 9.11, while in the Site C model, concentrations were higher when additional data were used for model construction, Figure 9.12.

In the large-scale models a variety of reactions occurred. In the Site A model, concentrations first increased over time and then decreased producing lower concentrations over time, e.g. Figure 9.13. In the Site B model, over 80 percent of the measured points had higher contaminant concentrations, Figure 9.10. The Site B model produced identical results in the scenarios that used additional data sets. This indicated that the Site B model was sensitive to the regional hydrological data but insensitive to the data inferred from remote sensing investigations, e.g. Figure 9.14.

The amount of time assigned for simulation influenced contaminant concentrations and the shape of the contaminant plume, in three of the four models. In the large-scale model of Site A, the largest difference in contaminant migration patterns occurred in the first 100 days after which contaminant concentrations levelled off, e.g. Figures 9.13. In the Site A small-scale model the greatest difference occurred in the first 150 days. In the Site C model, the concentrations levelled off after 2000 days, remaining constant after this time. Only the Site B model showed independent behaviour throughout the model in which contaminant concentrations varied over time according to the location of the calibration point and according to the scenario being modelled.

Figure 9.10 Five out of eight modelled scenarios produced higher contaminant concentrations through time when additional data was used for model construction



Legend:

- Positive values = % of calibration points in the model that produced higher contaminant concentrations on the last day of simulation
- Negative values = % of calibration point that produced lower contaminant concentrations on the last day of simulation

Figure 9.11 Site A small-scale model: In all three scenarios the contaminant concentrations differed most in the first 150 days of simulation

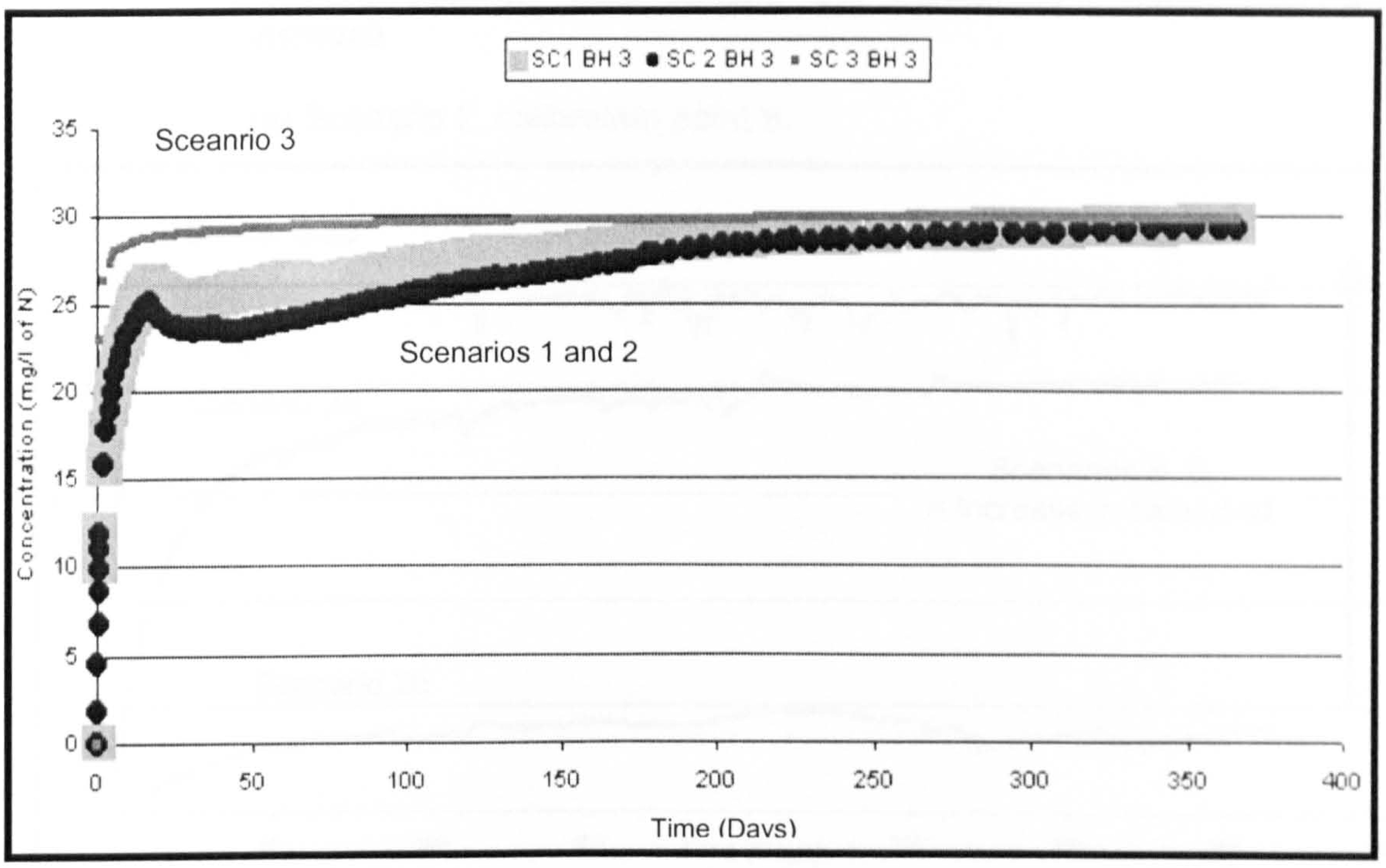


Figure 9.12 Site C small-scale model: Contaminant concentrations were higher when additional data was used for model construction with the greatest difference in the first 2000 days of simulation

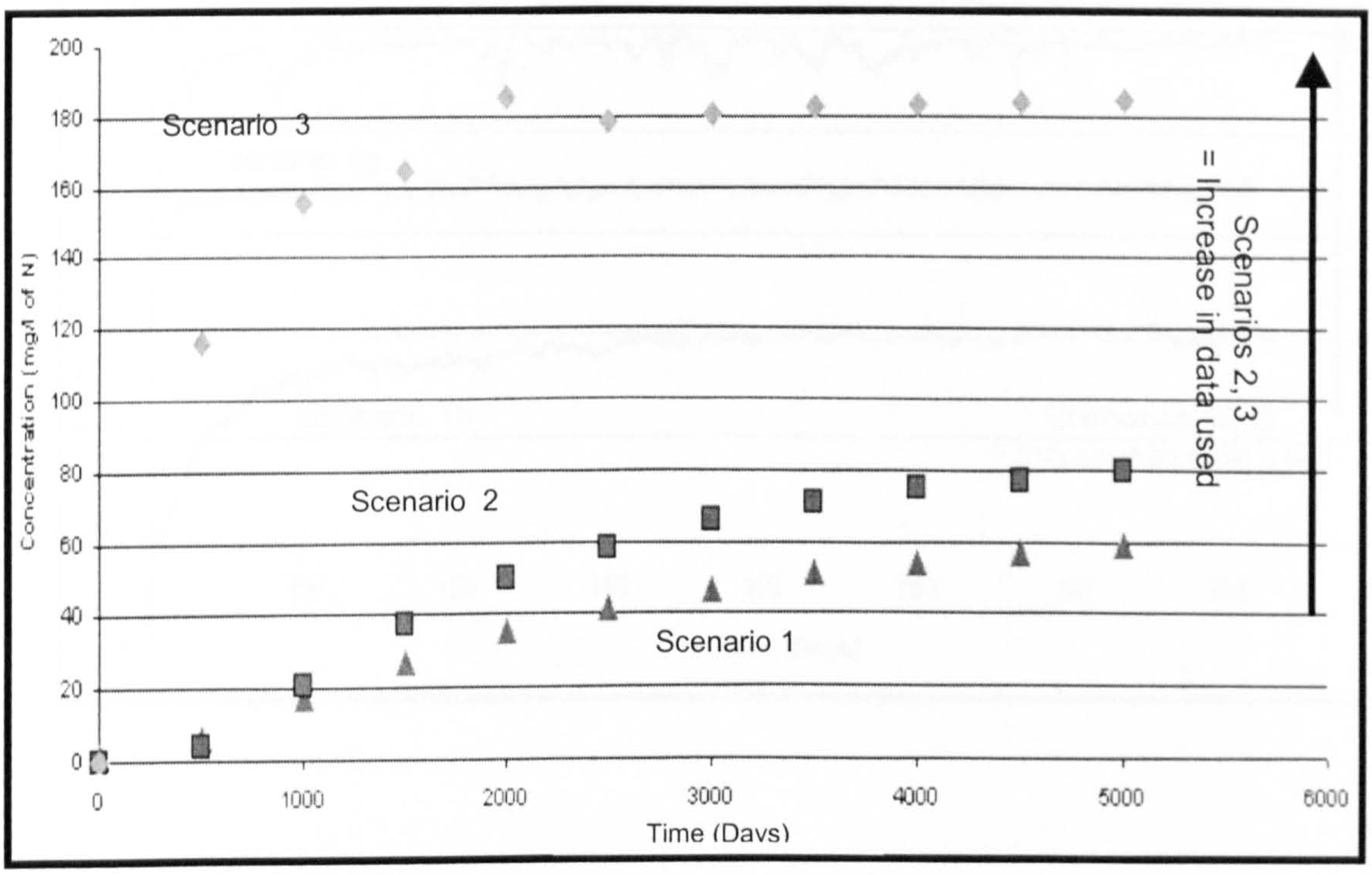
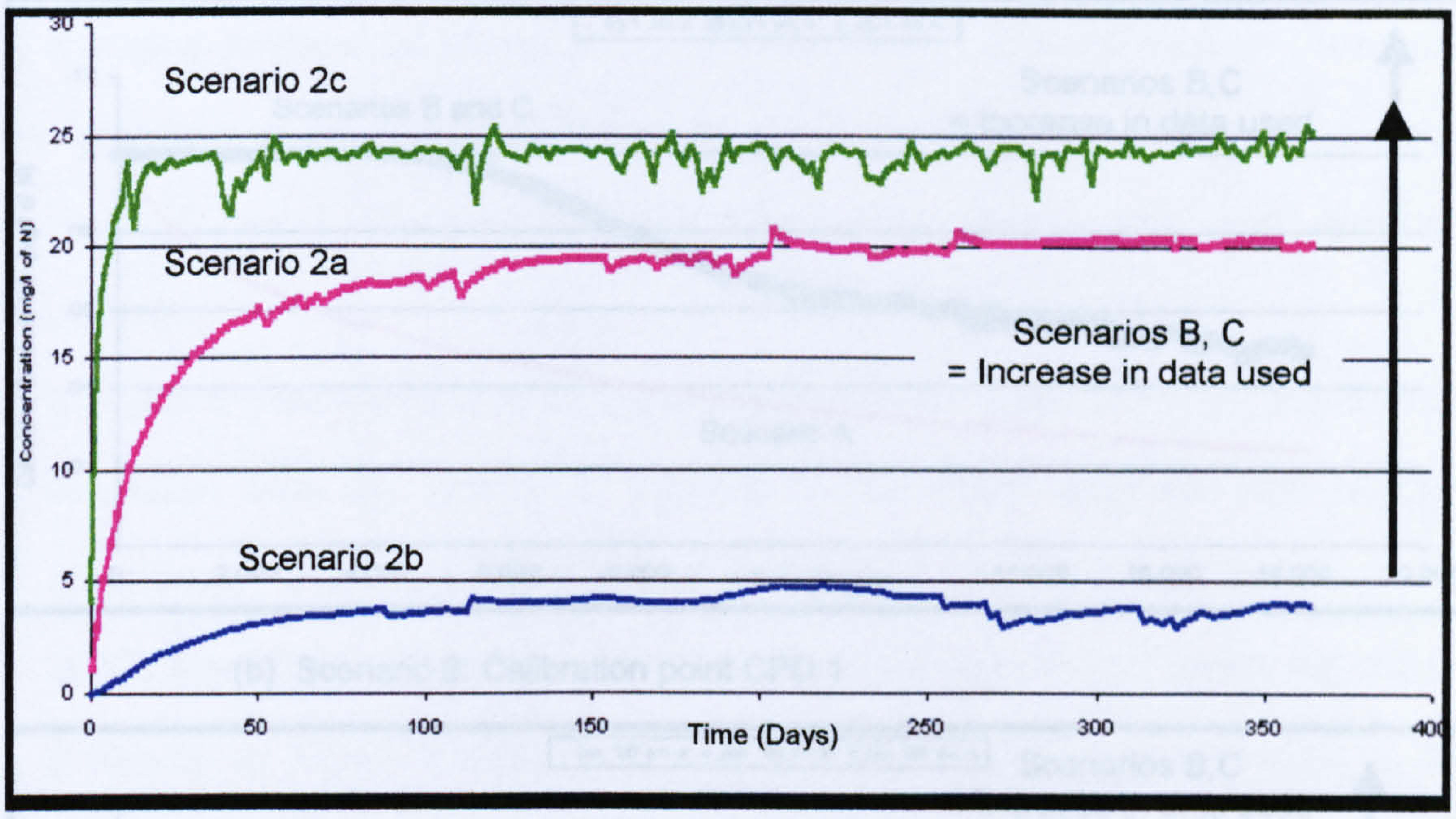


Figure 9.13 Site A large-scale model: The GPR data (Scenario 1b and 2b) caused contaminant concentrations to decrease while adding the GPR and Remote Sensing data caused the contaminant concentrations to increase

(a) Scenario 2, calibration point 6.



(b) Scenario 1, calibration point 13.

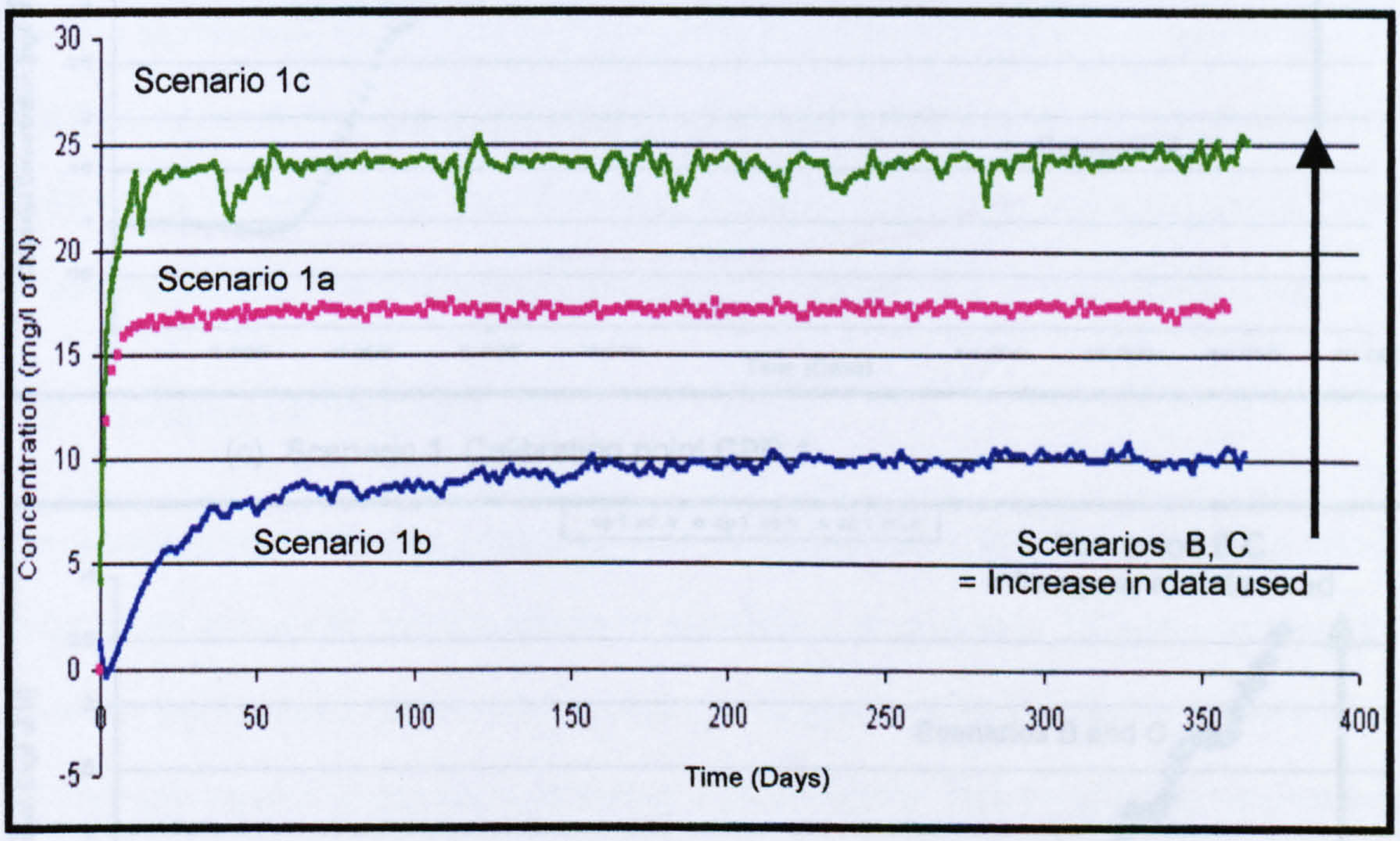
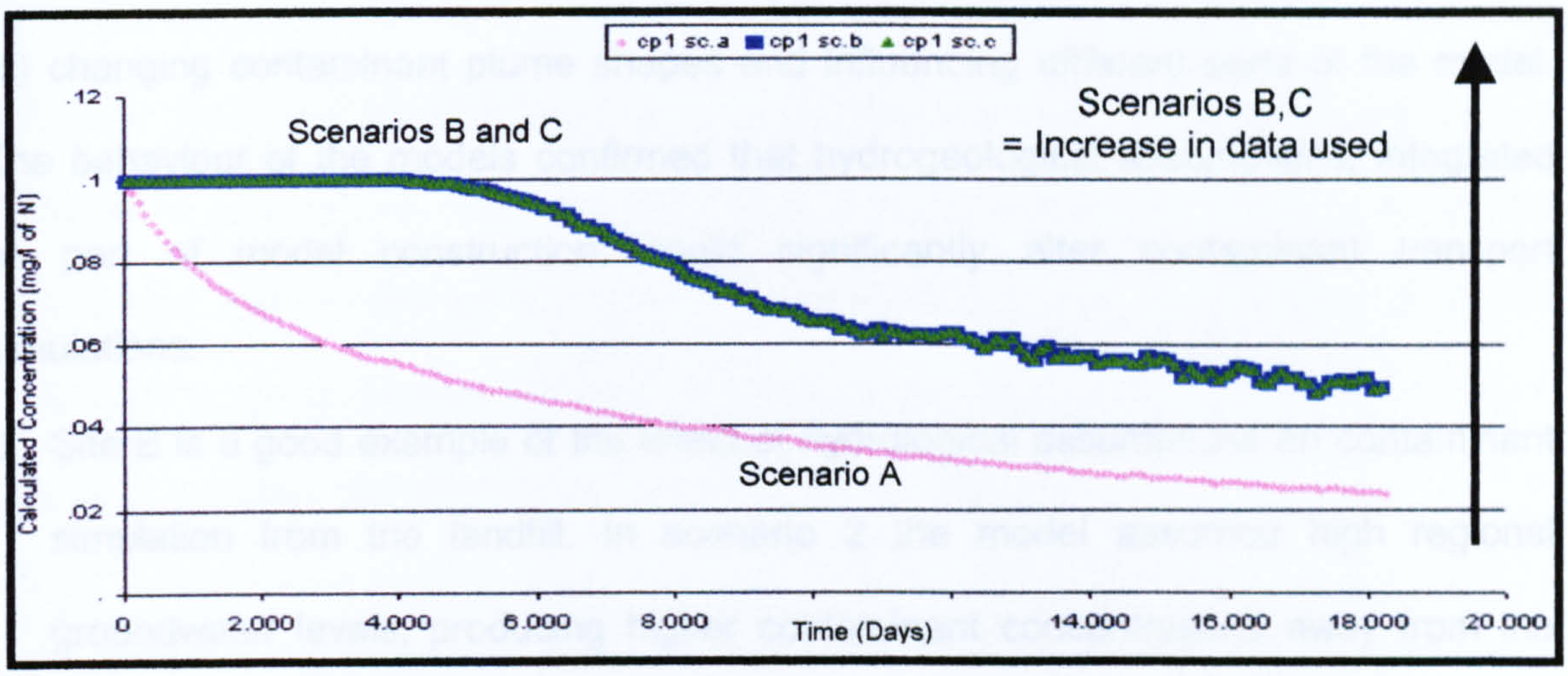
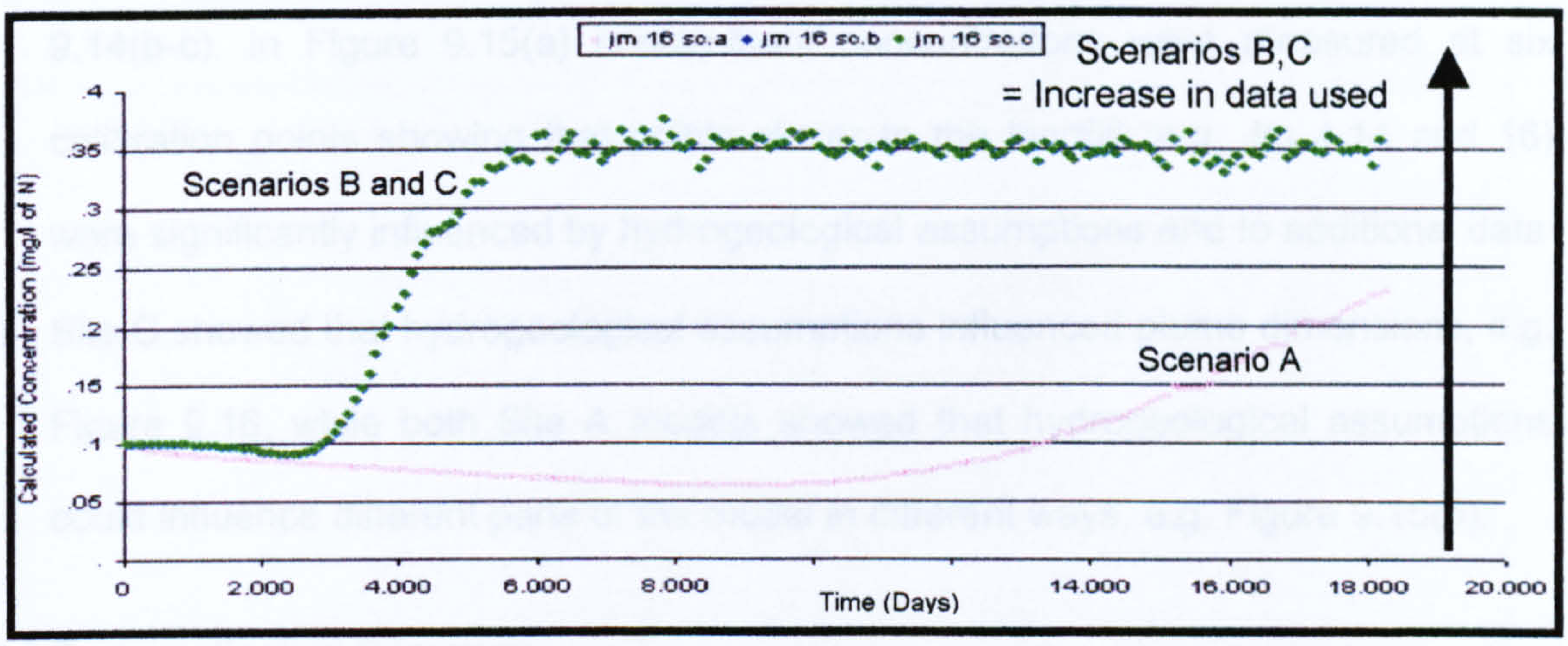


Figure 9.14 Site B large-scale model: Scenarios B and C (which used additional data sets) produced identical results. Most of the calibration points produced higher contaminant concentrations through time when additional data was used for model construction

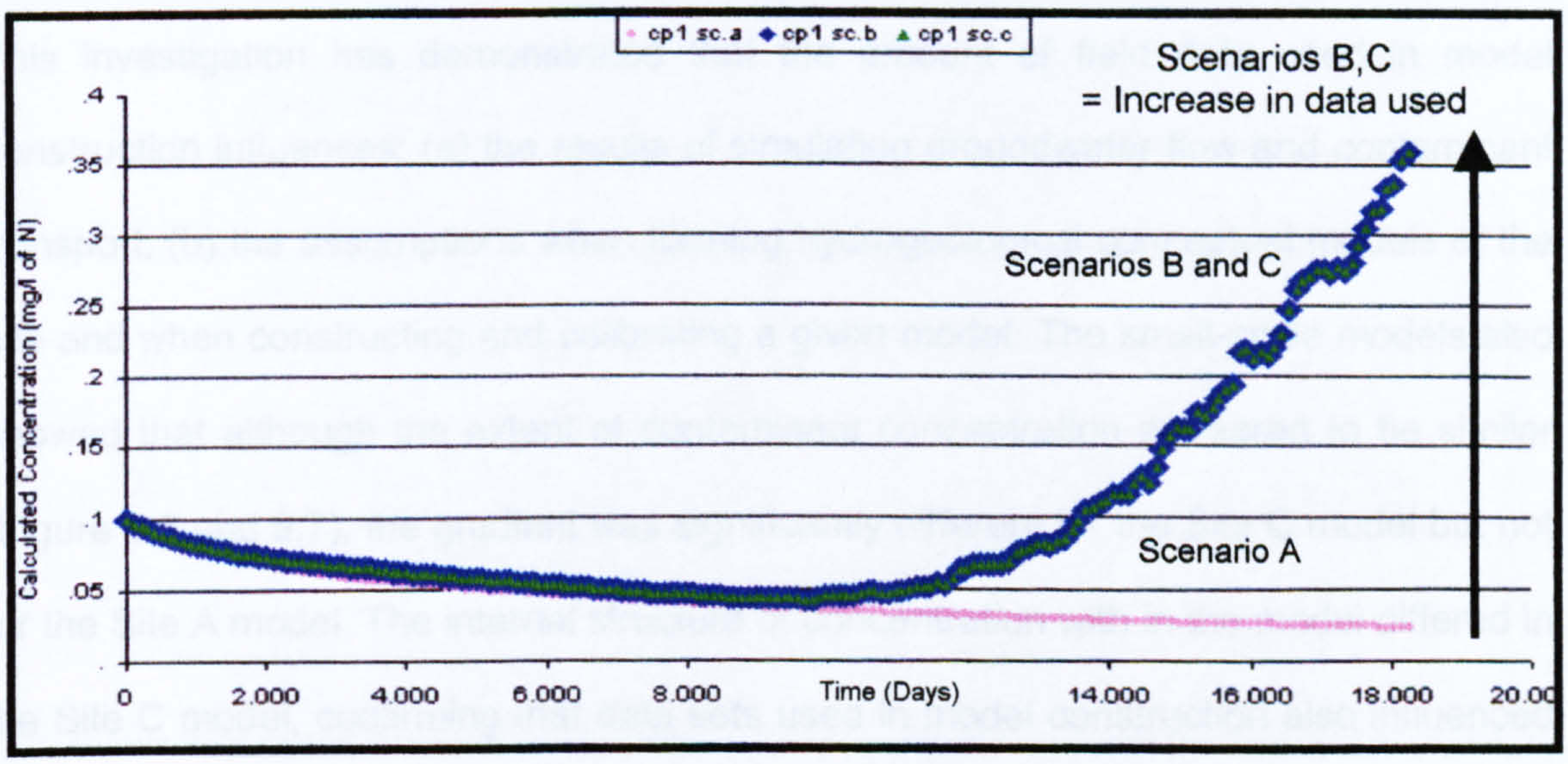
(a) Scenario 1: Calibration point Jm 16



(b) Scenario 2: Calibration point CPD 1



(c) Scenario 1: Calibration point CPD 1



9.2.6 Hydrogeological Assumptions

The different hydrogeological scenarios used in each model showed significantly different contaminant concentrations (a) with distance from the landfill, (b) through time, (c) changing contaminant plume shapes and influencing different parts of the model. The behaviour of the models confirmed that hydrogeological assumptions, integrated as part of model construction, could significantly alter contaminant transport simulations.

- Site B is a good example of the effect of hydrological assumptions on contaminant simulation from the landfill. In scenario 2 the model assumed high regional groundwater levels, producing higher contaminant concentrations away from the landfill than those in scenario 1, which assumed low groundwater levels, e.g. Figure 9.14(b-c). In Figure 9.15(a) contaminant concentrations were measured at six calibration points showing that points closer to the landfill (e.g. Jm 4,14 and 16) were significantly influenced by hydrogeological assumptions and to additional data
- Site C showed that hydrogeological assumptions influenced plume dimensions, e.g. Figure 9.16, while both Site A models showed that hydrogeological assumptions could influence different parts of the model in different ways, e.g. Figure 9.15(b).

9.2.7 Investigation Summary

This investigation has demonstrated that the amount of field data used in model construction influences: (a) the results of simulating groundwater flow and contaminant transport; (b) the assumptions when forming hydrogeological conceptual models of the site and when constructing and calibrating a given model. The small-scale models also showed that although the extent of contaminant concentration appeared to be similar (Figure 9.6 and 9.7), the gradient was significantly different for the Site C model but not for the Site A model. The internal structure of concentration with in the model differed in the Site C model, confirming that data sets used in model construction also influenced internal model calculations, explaining why Figure 9.16 produced differing plume shapes. In order to strengthen the findings of this investigation, further modelling needs to be conducted in which the same types of data are used for each site model. For example, this investigation used borehole data, GPR and remote sensing data to construct models of Site A and borehole and GPR data for Site C. The conclusions would be strengthened if the models were constructed using the same types of field methods, e.g. borehole data and GPR data from all three sites. Further modelling should also standardise the contaminant concentrations and simulation time. In the investigation, these parameters were defined by model-specific conditions making it difficult to compare directly results from different models.

Figure 9.15(a): Site B: Points closer to the landfill (e.g. Jm 4,14 and 16) were significantly influenced by hydrogeological assumptions and additional data

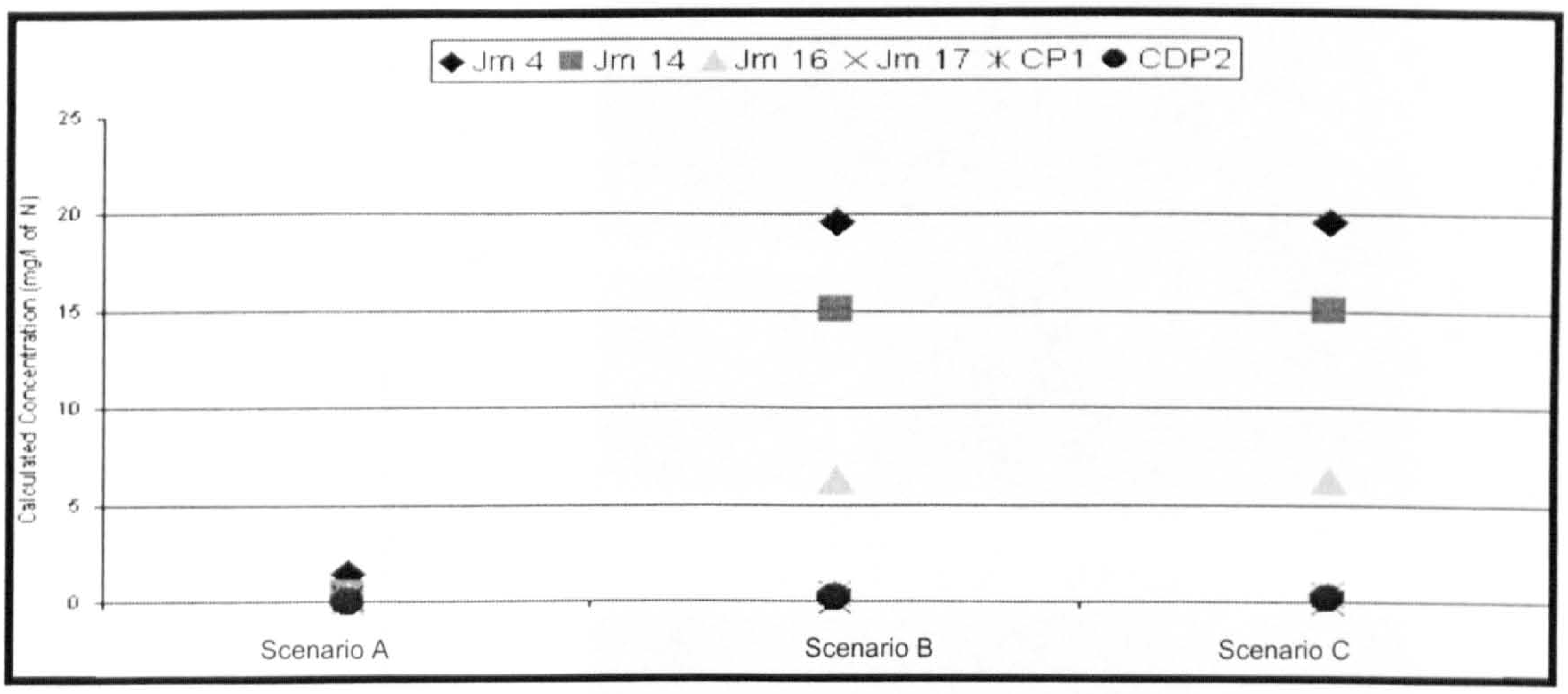


Figure 9.15(b) Site A small-scale model: Contaminant concentration at three points in three hydrogeological scenarios show that Bh 1 and Bh 2 had quite different contaminant concentrations while Bh 3 had similar results

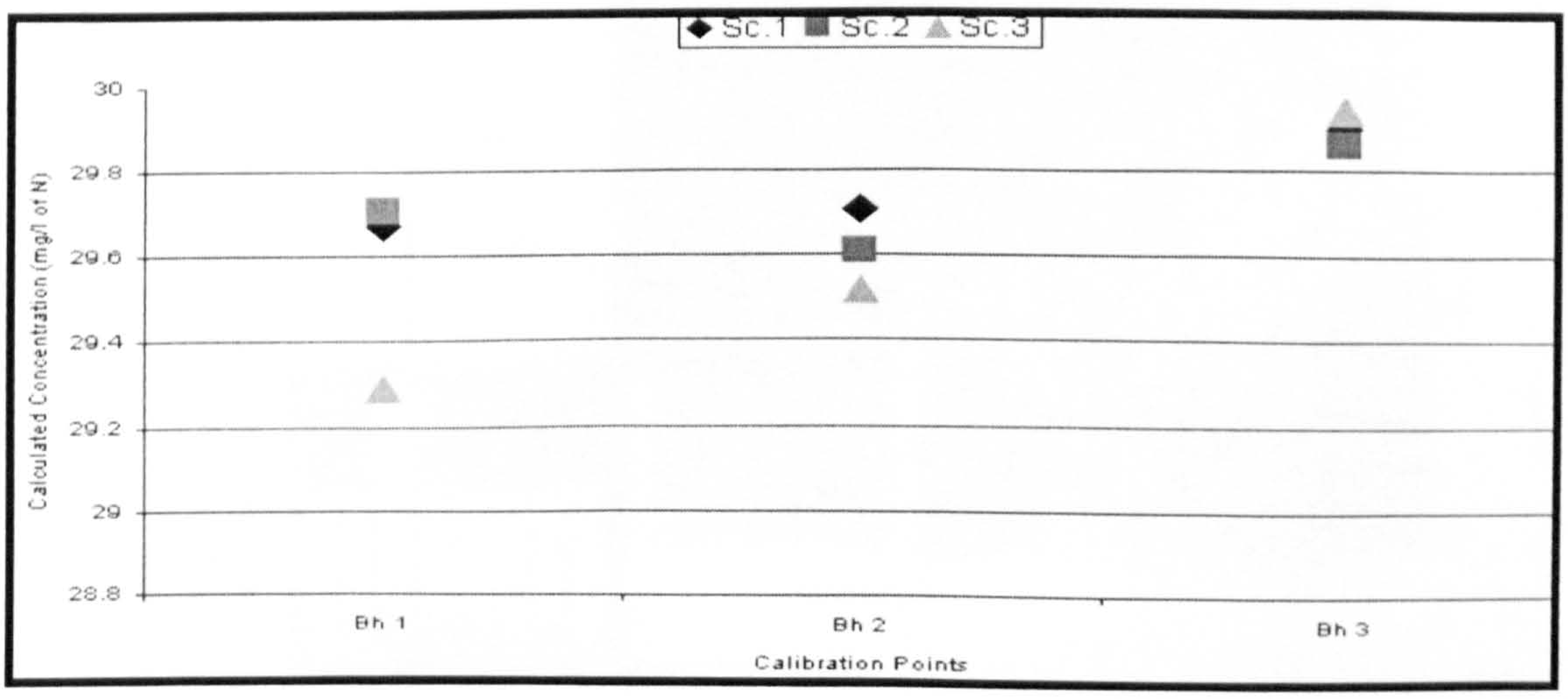


Figure 9.16 Cross-section contaminant plumes calculated on day 5000 near borehole 7 in the Site C model showing that hydrogeological assumptions that were used to construct each scenario produced different plumes shapes and dimensions

Legend:

_______ = Edge of contaminant plume

X axis in the cross-section maps = distance in metres away from the landfill

Y axis in the cross-section maps = depth of subsurface in metres

9.3 Investigation 5: Grid Density Variations

9.3.1 Objective

The aim of Investigation 5 was to test the sensitivity of models to grid sizes. Previous studies have been conducted to test the sensitivity of grid size on model results, e.g. Matanga, (1996). However, in this investigation the focus is on evaluating the influence of modelling assumptions and practices when inferring parameters such as grid size in leachate estimation models. In this context, the results can confirm the importance of grid size as a parameter important in contaminant migration but will also confirm that the modeller and field data have a significant influence on assumptions during construction. In context of the site assessment, this could significantly change leachate simulations. Three site models were used:

- the small-scale model of Site A
- the large-scale model of Site B and
- the small-scale model Site C.

Each model produced three scenarios with varying grid densities. These are listed in Table 9.6. A large-scale model of Site A was also constructed, however the model could not be calibrated to field data. As a result, these results are not included. Grid spacing varied in the models, depending on three factors. Firstly, upon the size of the area being modelled. The small-scale models used grid sizes of 1x1 m^2 or 2x2 m^2 while the landfill scale Site B model used larger grids (25-100^2 m). Secondly, the initial modelling objective was to incrementally increase grid size to measure the influence of small grid changes that could be considered as unimportant modelling decisions made by the modeller during model construction. The investigations initially planned to test the influence of using very small incremental changes (of 0.10 cm for the small-scale models and 10 m increases for the Site B model), however Visual MODFLOW was not able to converge many of these scenarios. This could be due to computing limitations in the MT3D contaminant transport model, which calculates migration from the centre of one grid node to the centre of the next grid node. Zheng (1990) explains that computing errors may occur if the grid density and number of model layers are too numerous for MT3D. Sensitivity analyses were conducted to identify which grid sizes MT3D could successfully compute. As a result, the small-scale models used grids with increases of $0.5m^2$ and $1.0m^2$ while the Site B model increased using $25m^2$ and $50m^2$. The results were investigated by comparing differences in contaminant (a) migration distances from the landfill and (b) the change in contaminant concentrations through time.

Table 9.6 Grid sizes assigned to each study site model in Investigation 1 (Scenario = Sc.)

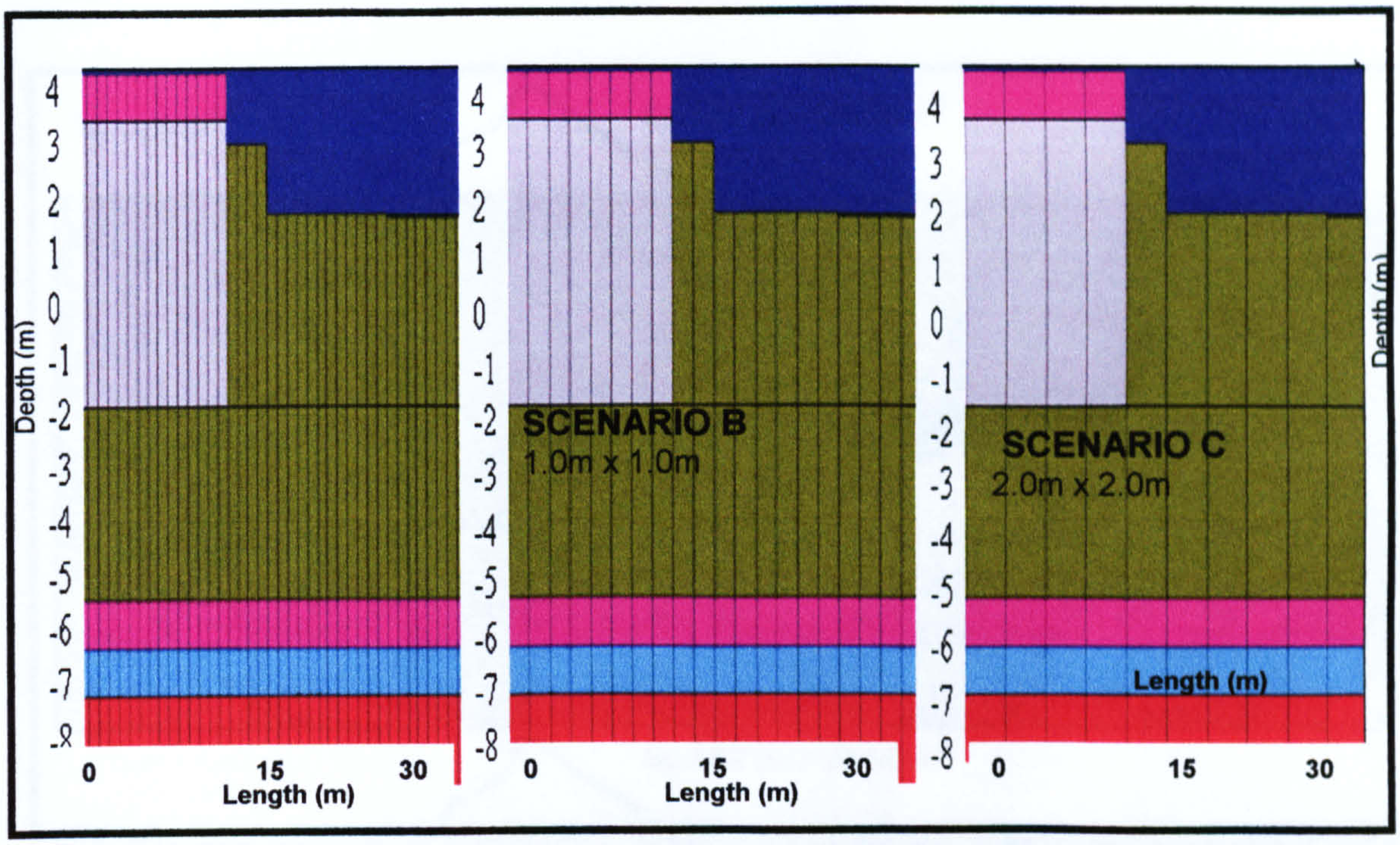
Study Site	Scale of Model	Grid Size Sc.1	Grid Size Sc.2	Grid Size Sc.3	No. of Calibration Points
Site A	Small-Scale	$0.50m^2$	$1m^2$	$2m^2$	4
Site C	Small-Scale	$0.5m^2$	$1m^2$	$2m^2$	4
Site B	Large-Scale	$25m^2$	$50m^2$	$100m^2$	14

9.3.2 Description of Models Used

The three site models in Table 9.6 were also presented in Section 9.2, which tested data sets. Each site had several hydrogeological models, e.g. Figure 9.17 – 9.19, which were used to test the sensitivity of each model to grid size changes.

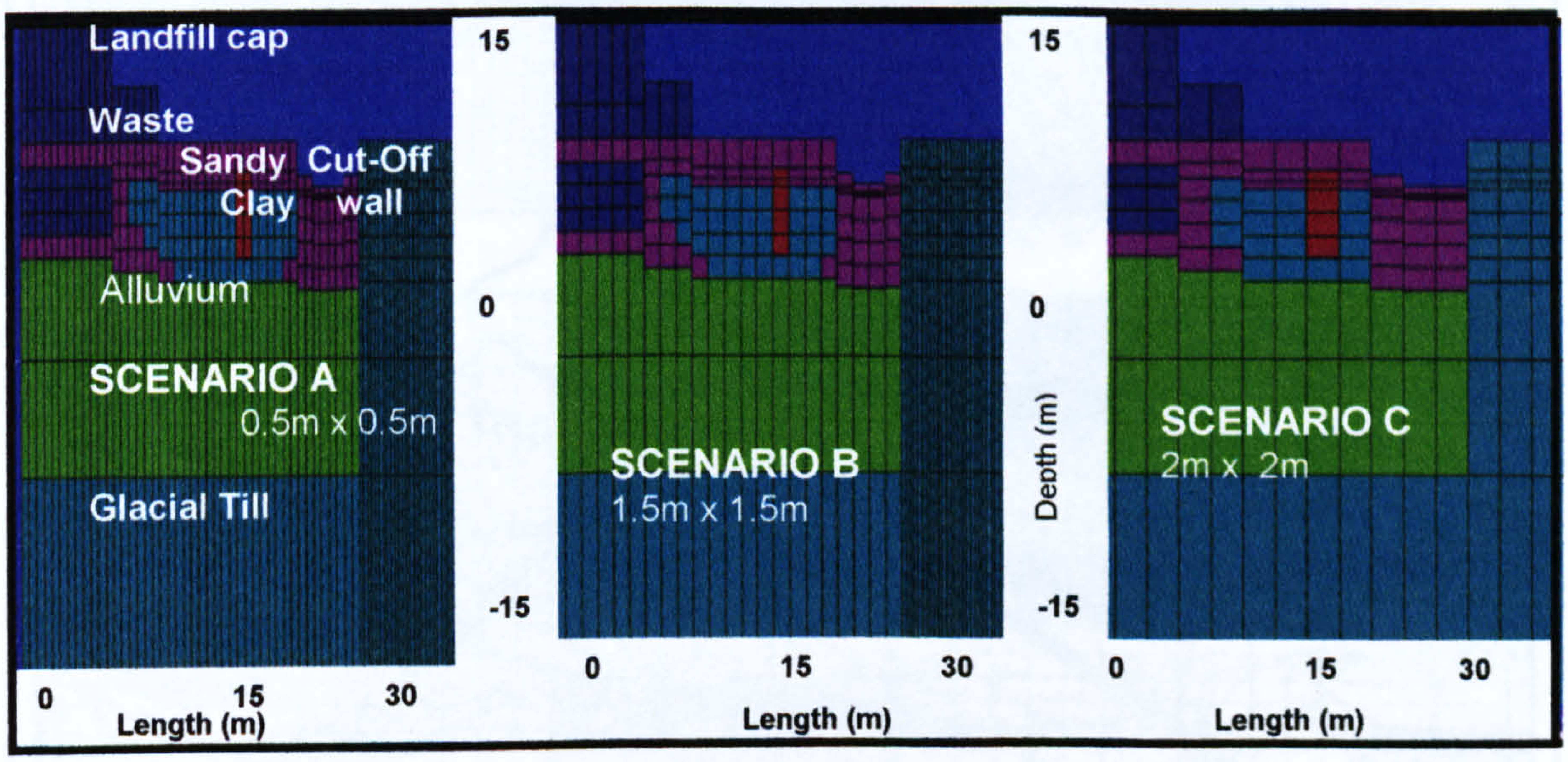
- Site A small-scale model: Two hydrogeological scenarios of the small-scale Site A model were used to test variations in grid size. Scenario A had a grid size of $0.5m^2$, scenario B $1m^2$ and scenario C $2m^2$ (e.g. Figure 9.17)
- Site C small-scale model: Three hydrogeological scenarios of Site C were used to test variations in grid size in the Site C model. Scenario A used a grid size of $0.5m^2$, scenario B used $1.5m^2$ and scenario C used $2\ m^2$ (e.g. Figure 9.18)
- Site B large-scale model: Two hydrogeological scenarios of Site B were used to test the influence of grid sizes on modelled results. The scenario used grid sizes of $25m^2$, $50m^2$, and $100m^2$ (e.g. Figure 9.19).

Figure 9.17: Site A small-scale model: Cross-section view of scenarios A, B and C models at Site A showing grid sizes of 0.50m², 1m² and 2m²



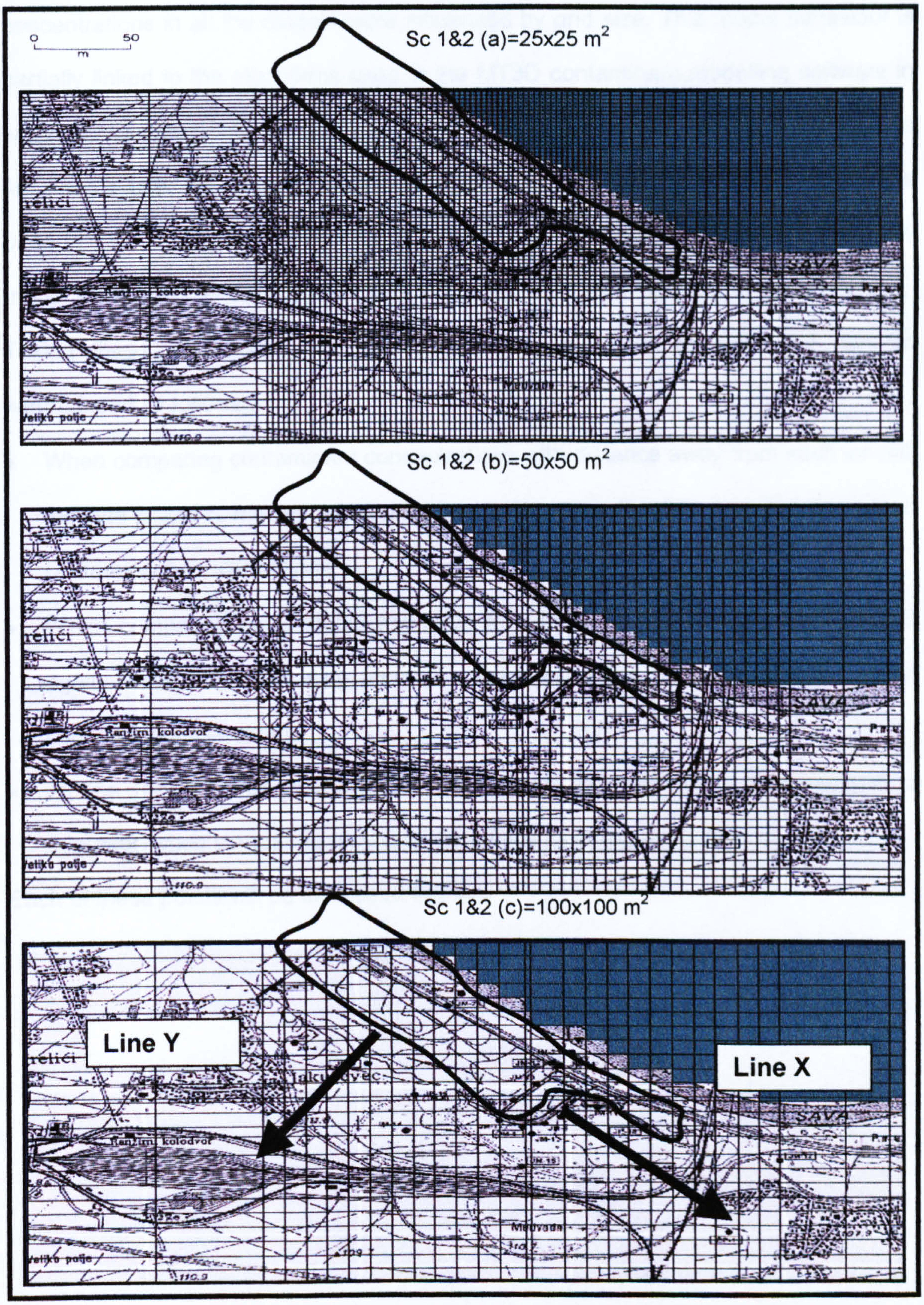
Legend:
Sc. 1 & 2 (a) = Sc. 1 & 2 (b) = Sc. 1 & 2 (c) = X axis in the cross-section maps = distance in metres away from the landfill
Y axis in the cross-section maps = depth of subsurface in metres

Figure 9.18 Cross-section view of Site C models (scenarios A, B and C) showing grid sizes of 0.50m², 1.5m², and 2m²



Legend:
Sc. A = 0.5 x 0.5m grid; Sc. B = 1.5 x 1.5 m grid; Sc. C = 2.0 x 2.0 m grid
X axis in the cross-section maps = distance in metres away from the landfill
Y axis in the cross-section maps = depth of subsurface in metres

Figure 9.19: Plan view of grid sizes used in the Site B model, assuming two Scenarios: Sc. 1 had low groundwater levels while Sc. 2 had higher groundwater levels.



9.3.3 Investigation 5 Results

The investigation results are summarised in Table 9.7, which shows that contaminant concentrations in all the models were influenced by grid size. This model behaviour is partially linked to the algorithms used in the MT3D contaminant modelling software in which contaminant concentrations are calculated from one grid node to the next. The smaller the grid, the more complex the contaminant transport calculation. The models used in this investigation had different site-specific conditions, which determined the model scale and number of model layers. The findings were therefore expected to vary. Taking these two factors into account, the trends shown in Table 9.7 can be summarised as follows:

- When comparing contaminant concentrations with distance away from each landfill, the Site B and C models reacted by increasing while the Site A model decreased contaminant concentrations
- When comparing contaminant migration through time, the Site B and C models again reacted by increasing temporal fluxes while the Site A model decreased concentrations over time
- All the models produced different contaminant concentrations and patterns of migration when comparing results of the different hydrogeological scenarios.

Each of these points will be discussed further.

Table 9.7 Five questions that were used to evaluate model behaviour when grid size was increased

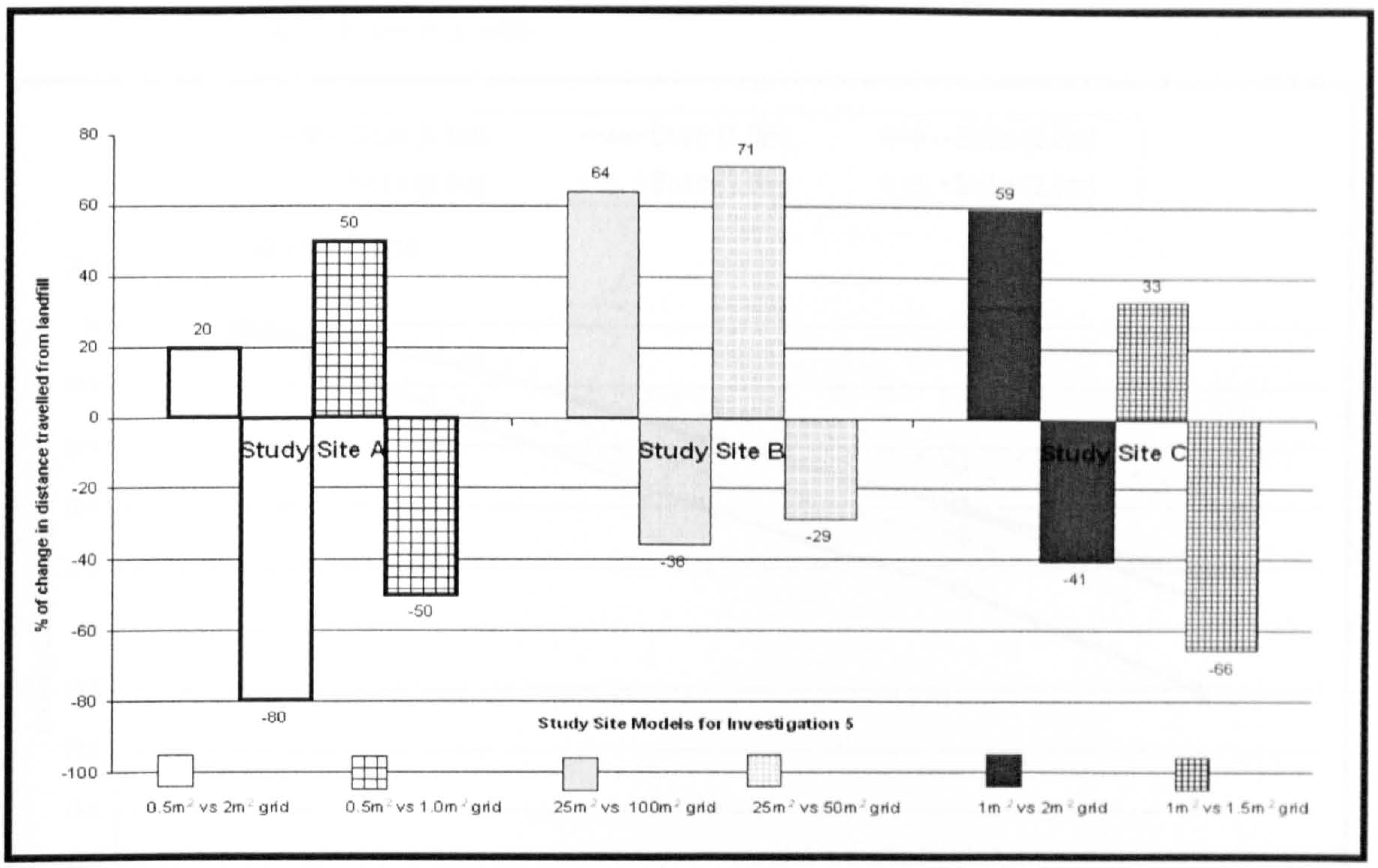
Study Site	Did increasing grid size influence contaminant transport in the models?	Did the hydro-geological scenarios influence results?	Did concentration concentrations increase away from the site when grid size was increased?	Did contaminant concentrations increase through time when grid size was increased?
Small-scale model of Site A	YES	YES	NO 20% - Sc.A-Sc.C 50% - Sc.A-Sc.B	N0 25% - Sc.A-Sc.C 0% - Sc.A-Sc.B
Small-scale model of Site C	YES	YES	YES 59% - Sc.A-Sc.C 33% - Sc.A-Sc.B	YES 80% - Sc.A-Sc.C 50% -Sc.A-Sc.B
Large-scale model of Site B	YES	YES	YES 64% - Sc.A-Sc.C 71% - Sc.A-Sc.B	YES 86% - Sc.A-Sc.C 71% -Sc.A-Sc.B

Legend:
- % = The number of calibration points in the model that produced higher contaminant concentrations away from the landfill

9.3.4 Comparing Modelled Concentrations with Distance from the Site

In context of the landfill risk assessment, the results showed two trends. Firstly, each model reacted to grid changes individually, e.g. Figure 9.20. This was shown in all three models. In the Site A model, contaminant concentrations first decreased then increased. In the Site C model, the concentrations first increased and then decreased. The Site B models had high contaminant concentrations which continued to increase when larger grids were used. The second trend showed that the scenarios which used smaller grid sizes tended to produce higher contaminant concentrations with distance away from the landfill, e.g. Table 9.8 and Figures 9.21 – 9.23. When comparing calibration points for sensitivity to grid size, over 50 percent produced higher contaminant concentrations when smaller grid sizes were used. Figure 9.21 comparing contaminant concentrations and gradient between the different scenarios showed that there were differences in both the concentration simulated (Figure 9.21(a)) and in the model's internal distribution of ammonia concentrations (Figure 9.21(b)). In the Site B model, (Figure 9.22) grid size influenced model simulation in different ways. The smallest grid size (25 x 25 m^2) produced the highest contaminant concentrations in the first 480 m while the larger scenario (50 x 50 m^2) produced higher concentrations and steeper gradients (e.g. Figure 9.22(a)). In the Site B large-scale model the 25 x 25 m^2 grid produced the highest contaminant gradients and concentrations while scenario 2b was highest for scenario 2 which assumed higher groundwater levels (Figure 9.22(b)). In the Site C small-scale models, the scenario using larger grid sizes (2 x 2 m^2 grids) produced the highest contaminant concentrations away from the landfill (Figure 9.23(a)). Contaminant concentrations were inverse when comparing the 1m^2 and 2m^2 grid - the 1m^2 model had high gradients in the first 10 m followed by low gradients after 20m while inverse behaviour occurred in the 2m^2 model, confirming the influence of grid size on contaminant simulation (Figure 9.23(b)).

Figure 9.20 When increasing grid size the percentage of calibration points that had higher concentrations away from the site varied



Legend:

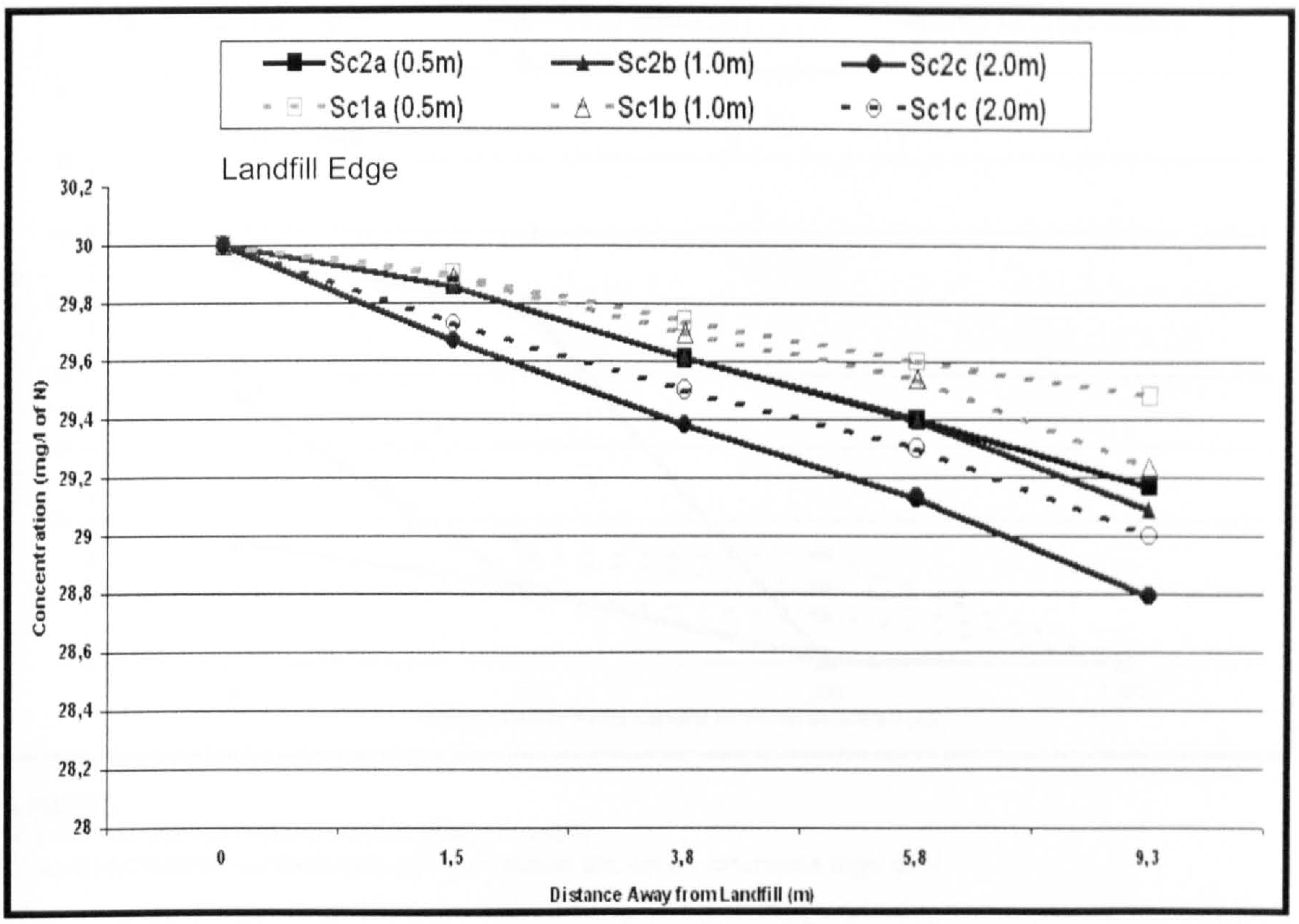
- Positive values = % of calibration points in the model that produced higher contaminant concentrations away from the landfill
- Negative values = % of calibration points that produced lower contaminant concentrations away from the landfill

Table 9.8 The smallest grid sizes gave the highest contaminant concentrations in Site A models while in the Site B and C models, the largest grid size produced the highest contaminant concentrations with distance from the landfill

		Hydrogeological Scenario 1 (mg/l of N)	**Hydrogeological Scenario 2** (mg/l of N)	**Hydrogeological Scenario 3** (mg/l of N)
SITE A	0.5 x 0.5m^2	**29.48**	**29.17**	N/A
	1 x 1m^2	29.23	29.07	N/A
	2 x 2m^2	29	28.79	N/A
SITE B	25 x 25m^2	0.0237	0.06	N/A
	50 x 50m^2	0.023	0.07	N/A
	100 x 100 m^2	**0.029**	**0.079**	N/A
SITE C	0.5 x 0.5m^2	0.1	12.3	0.1
	1.5 x 1.5m^2	9.5	14	14
	2 x 2m^2	**50**	**70**	**16.3**

Legend:
Contaminant concentrations = ammonia mg/l of N)

Figure 9.21(a) Site A small-scale model: The scenario using smaller grid sizes (scenarios 1a and 2a) produced the highest contaminant concentrations away from the site.



Legend:
X-axis represents distance away from landfill
Y-axis represents contaminant concentrations shown as ammonia in mg/l of N

Figure 9.21(b) Site A small-scale model: In scenario 2c, the internal distribution of ammonia concentrations differed indicating that the grid size influenced the contaminant simulations but not significantly

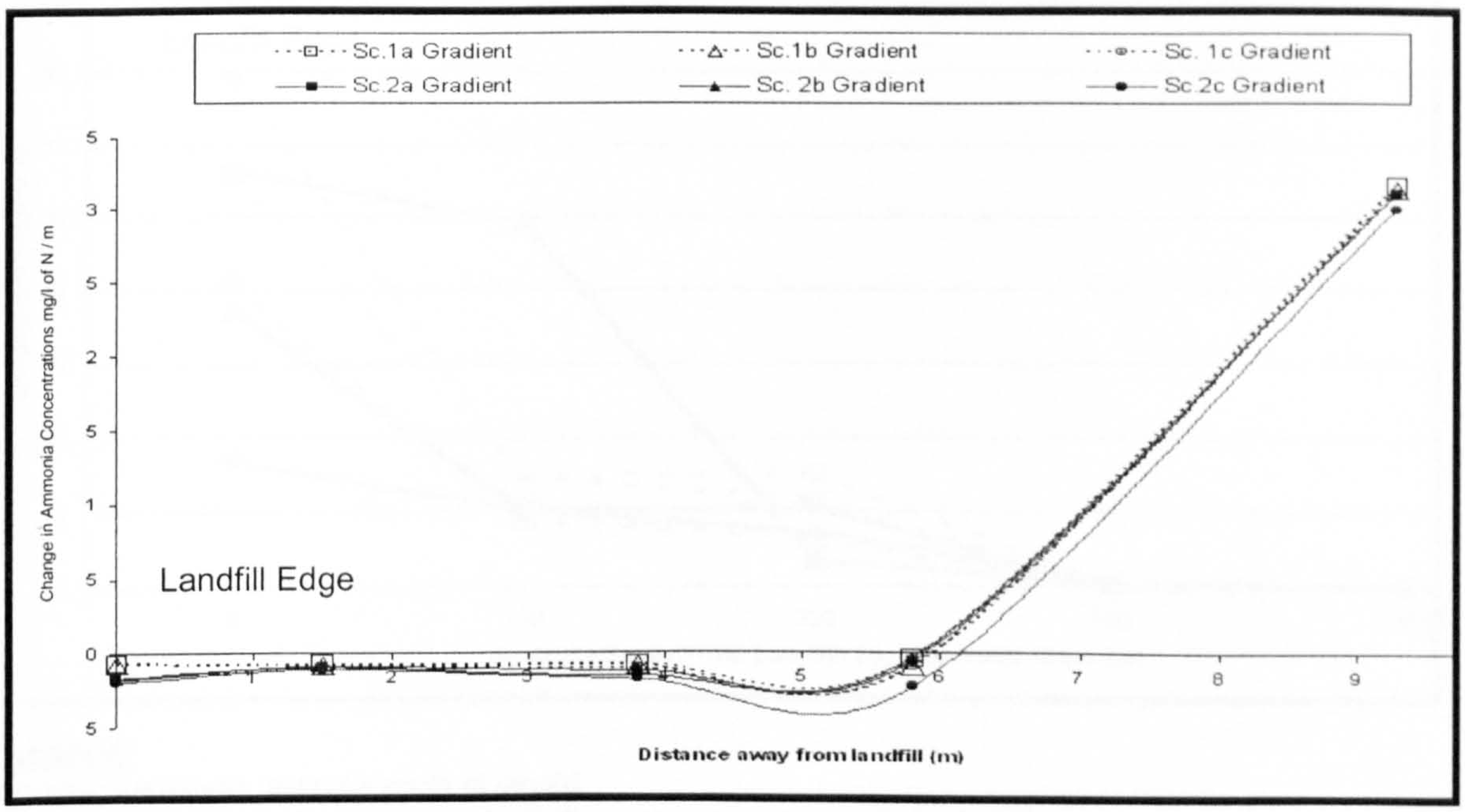
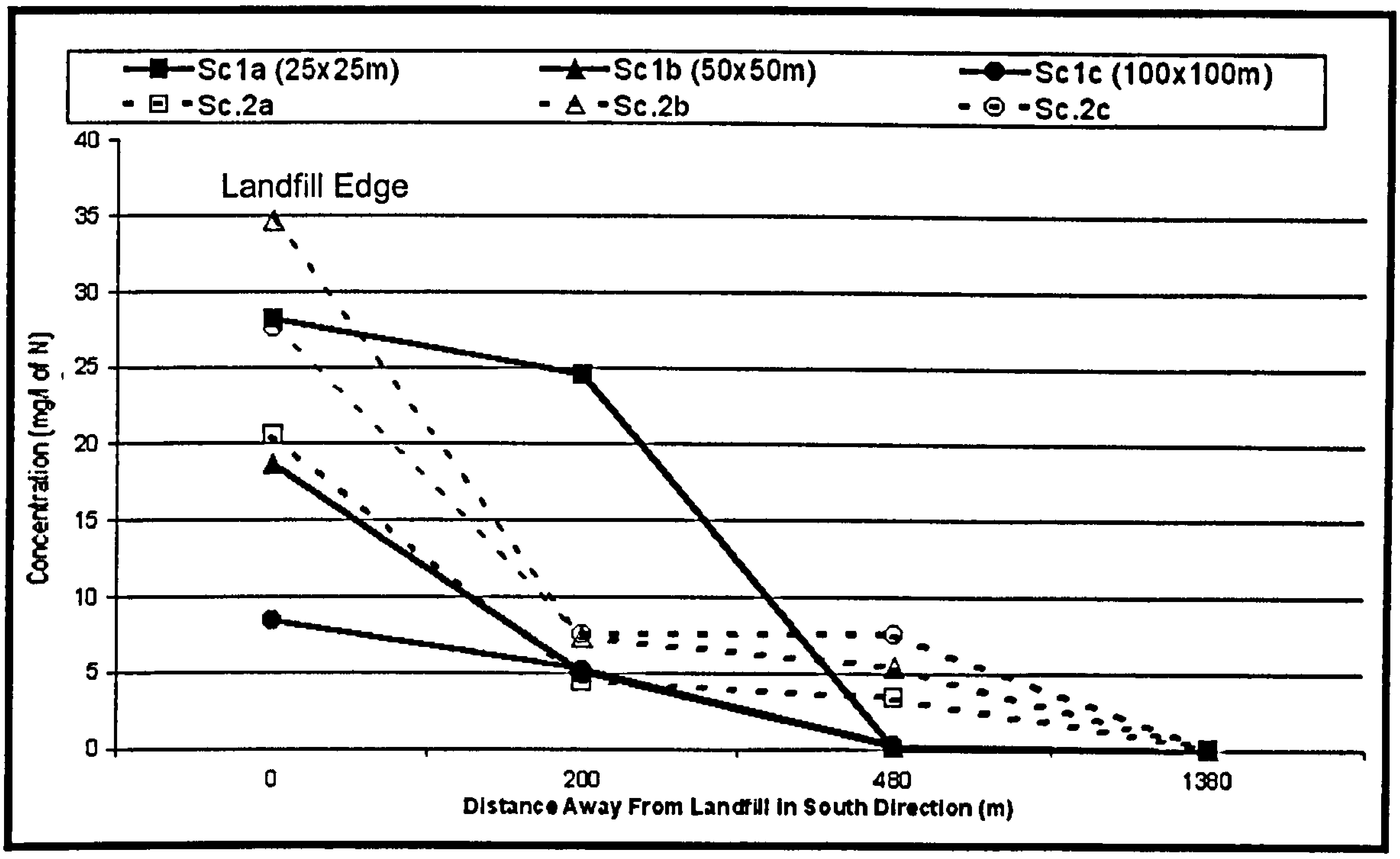
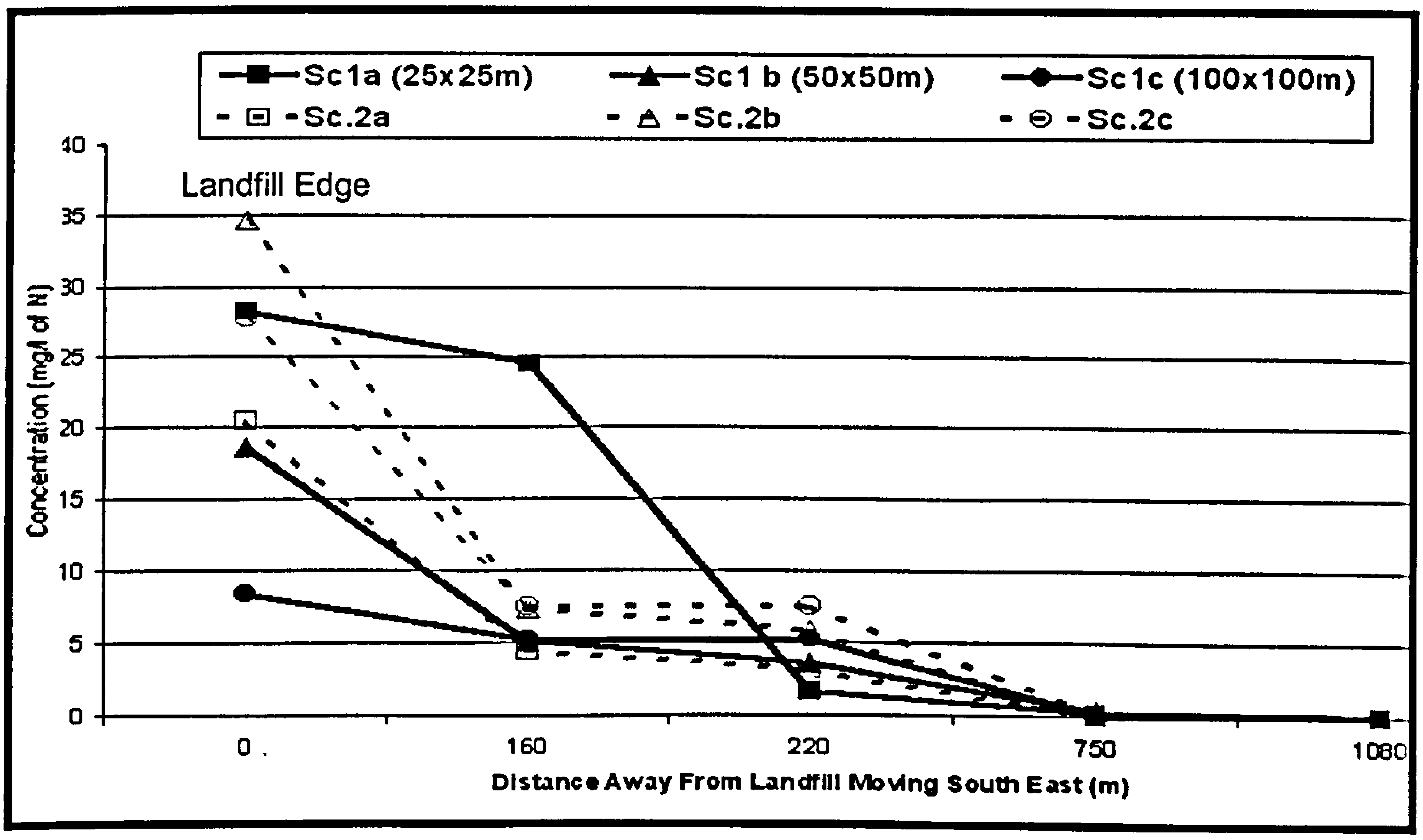


Figure 9.22(a) Site B: The smallest grid size (25 m^2) produced the highest contaminant concentrations while the larger scenario (50 m^2) produced higher concentrations and steeper gradients



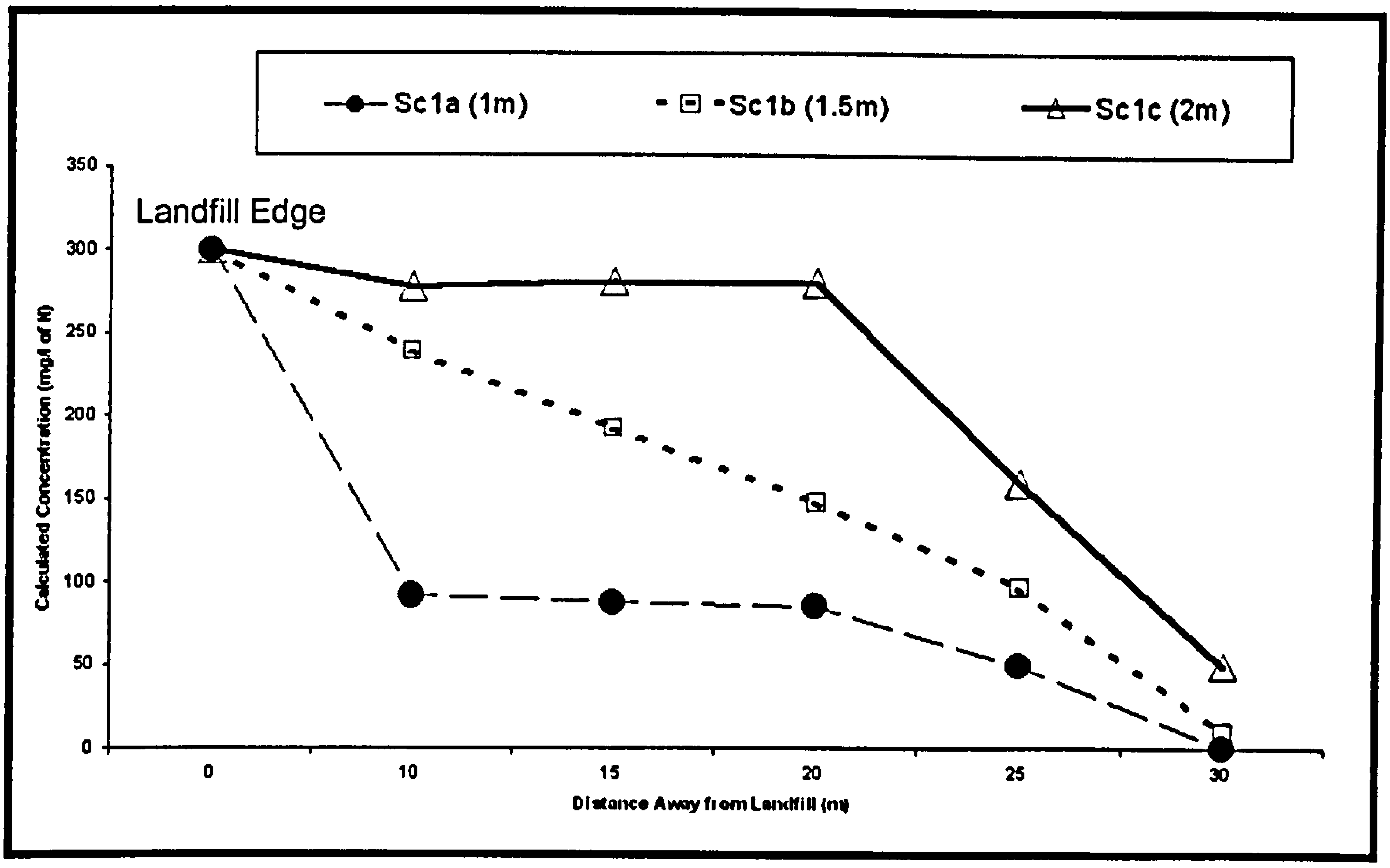
Legend:
X-axis represents distance south east of landfill
Y-axis represents contaminant concentrations shown as ammonia mg/l of N

Figure 9.22(b) Site B large-scale model: The 25m^2 grid produced the highest contaminant gradients and concentrations for scenario 1 while the 50m^2 grid was highest for scenario 2



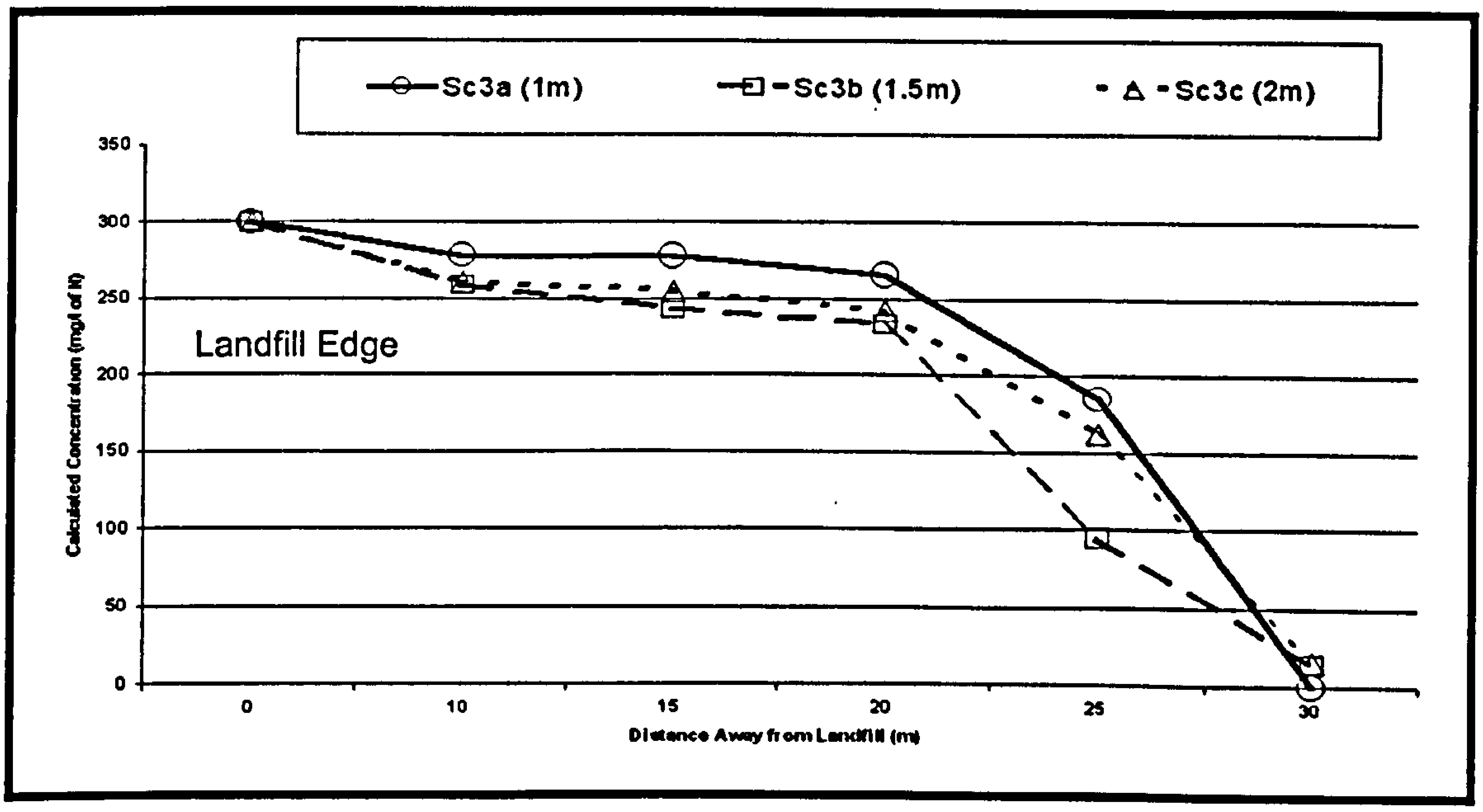
Legend:
X-axis represents distance south of landfill
Y-axis represents contaminant concentrations shown as ammonia mg/l of N

Figure 9.23(a) Site C small-scale model: The scenario using larger grid sizes (2 m^2) produced the highest contaminant concentrations away from the landfill.



Legend:
X-axis represents distance from landfill
Y-axis represents contaminant concentrations shown as ammonia mg/l of N

Figure 9.23(b) Site C small-scale model: The scenarios using larger grid sizes produced the highest contaminant concentration



Legend:
X-axis represents distance from landfill
Y-axis represents contaminant concentrations shown as ammonia mg/l of N

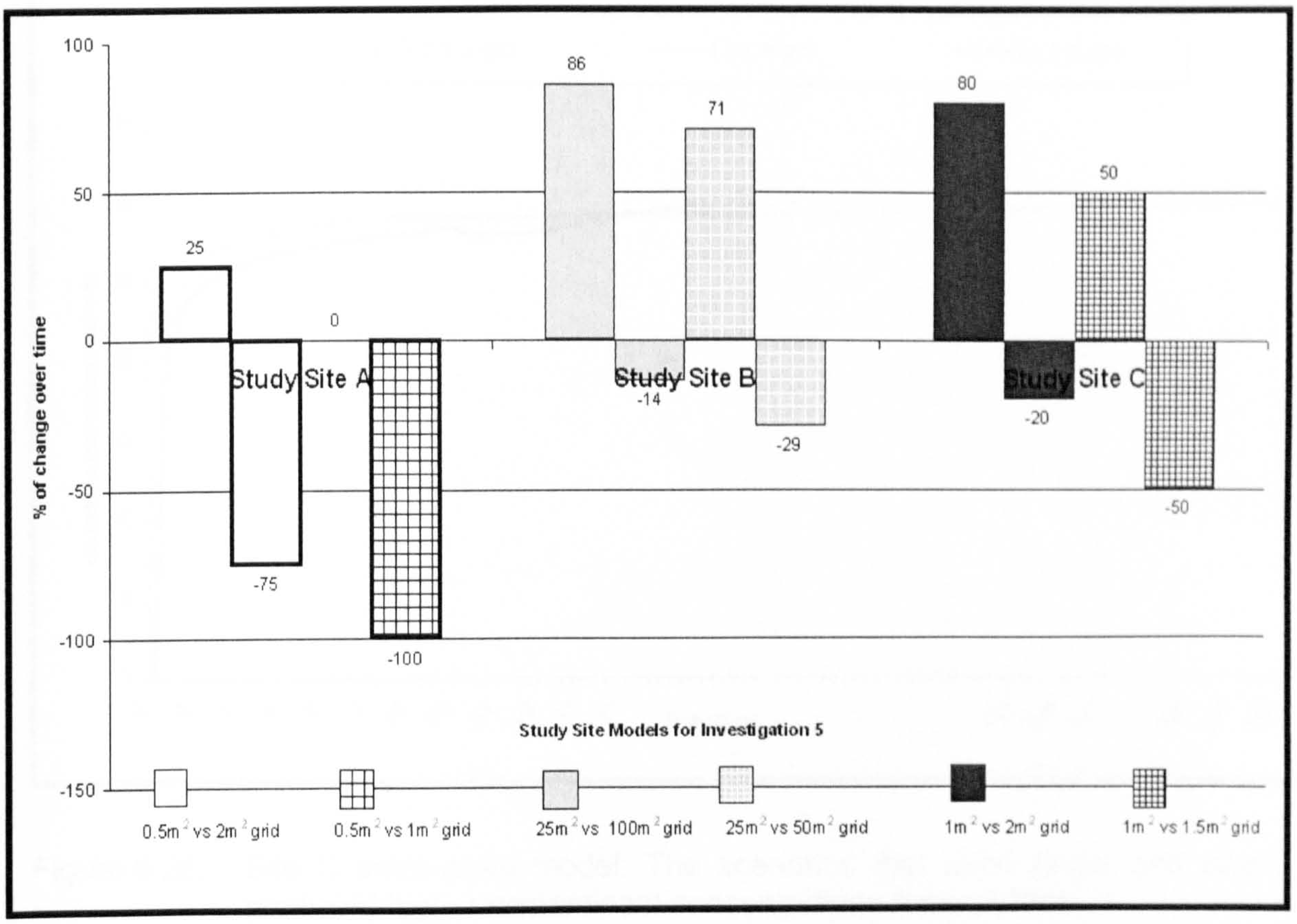
9.3.5 Comparing Contaminant Concentrations through Time

Contaminant concentrations were evaluated looking at each model's behaviour through time showing that all three site models were influenced by grid size. When grids were enlarged, the Site A model produced lower contaminant concentrations through time while the Site B and C models produced higher concentrations, e.g. Figures 9.24 - 9.30. The graph in Figure 9.24 compares contaminant concentrations at calibration points in each scenario on the last day of simulation, showing the percentage of points that had higher / lower concentration concentrations when grid sizes were increased.

Time was a factor that also influenced contaminant concentrations in the small-scale models (Site A and Site C). The first 20 days of simulation in the Site A model and the first 2000 days in the Site C model were critical periods before which the contaminant concentrations levelled off, Figure 9.25 and 9.26. In the Site A small-scale model the impact of grid size on contaminant concentrations was compared over time showing that the largest grid size (2 m^2) gave the lowest contaminant concentrations (Figure 9.25). In the Site C small-scale model, the scenarios that used larger grid sizes produced higher contaminant concentrations through time (e.g. Figure 9.26). The influence of grid size on model behaviour was also observed in the Site B model (Figure 9.27), in that calibration points closer to the landfill produced significantly different results when comparing scenarios, e.g. Figure 9.28, calibration points Jm4, Jm12 and Jm13.

In context of the landfill site assessment, these results indicate two trends. Firstly, by increasing grid size, temporal fluctuations in contaminant concentrations differed, depending on the grid size used. Secondly, time as a parameter of modelling influences patterns and concentrations of contaminant distribution.

Figure 9.24 Contaminant concentrations compared in each model showing the percentage of points that had higher and lower concentration concentrations when grid sizes were increased



Legend:

- Positive values = % of calibration points in the model that produced higher contaminant concentrations on the last day of simulation
- Negative values = % of calibration points that produced lower contaminant concentrations on the last day of simulation

Figure 9.25: Site A small-scale model: The largest grid size (2 m^2) gave the lowest contaminant concentrations through time

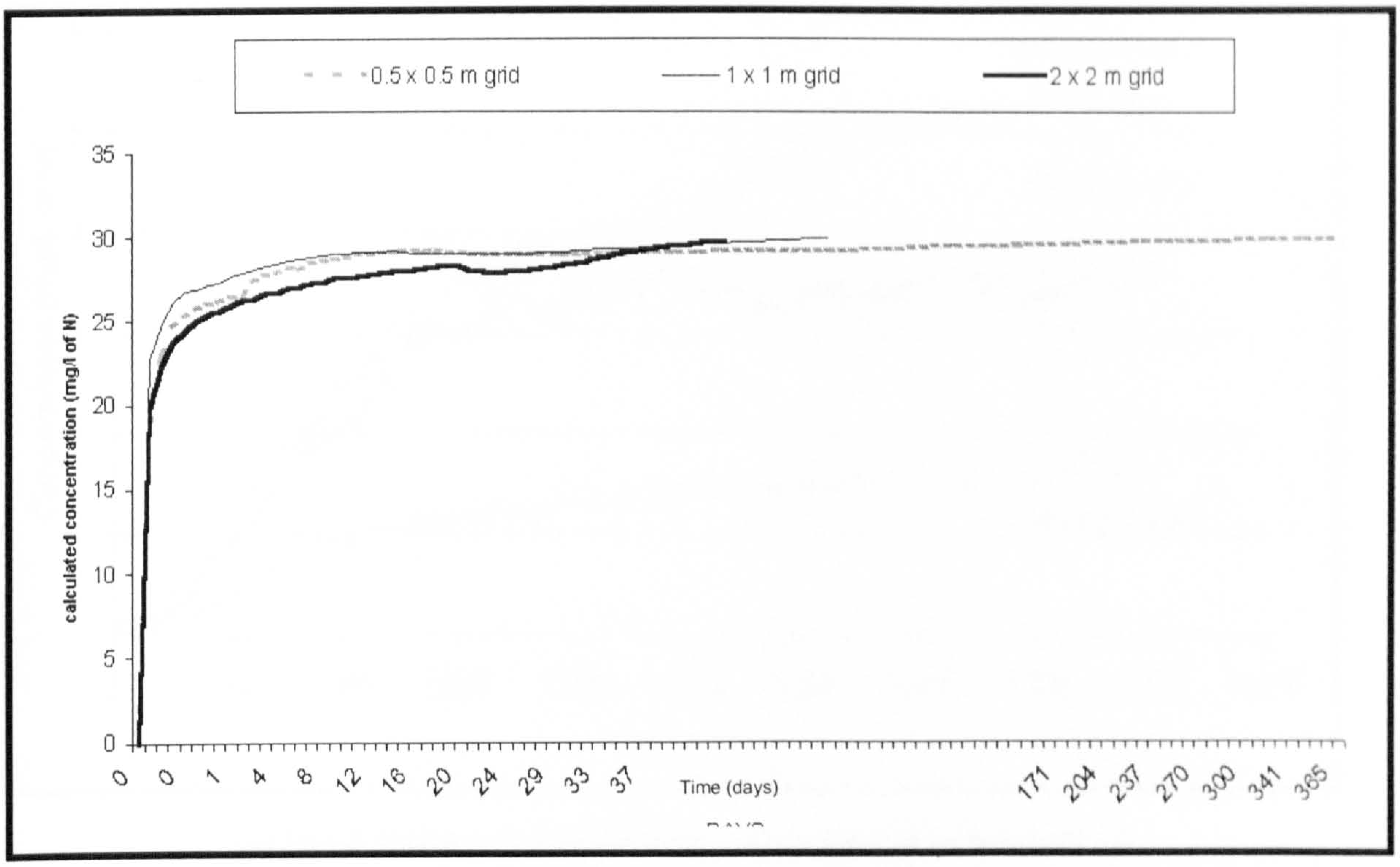


Figure 9.26: Site C small-scale model: The scenarios that used larger grid sizes, produced higher contaminant concentrations through time

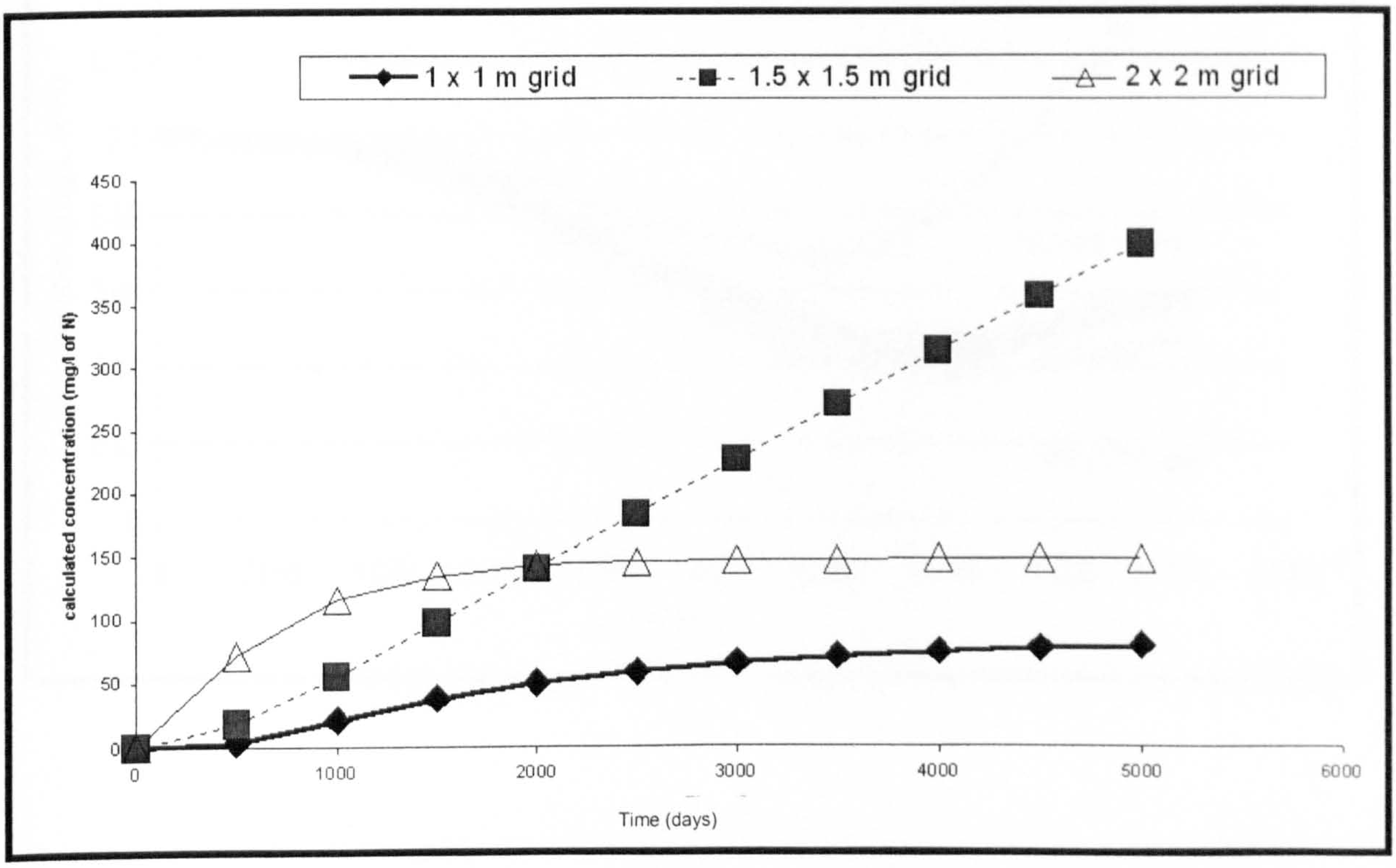
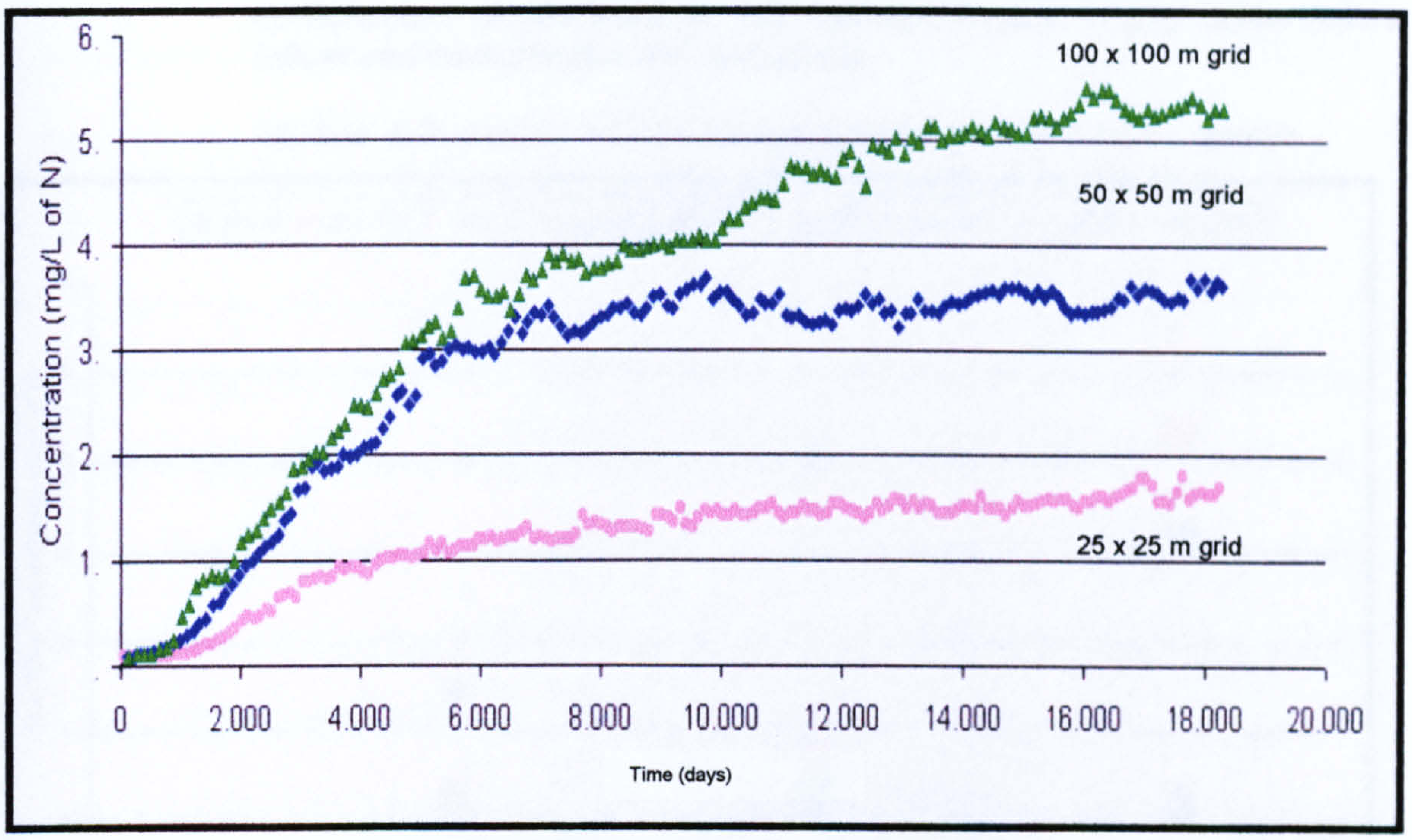


Figure 9.27 Site B large-scale model: The large grids produced higher contaminant concentrations and different migration patterns

(a) Site B scenario 1: Contaminant concentrations at borehole 13.



(b) Site B scenario 2: Contaminant concentrations at borehole 16.

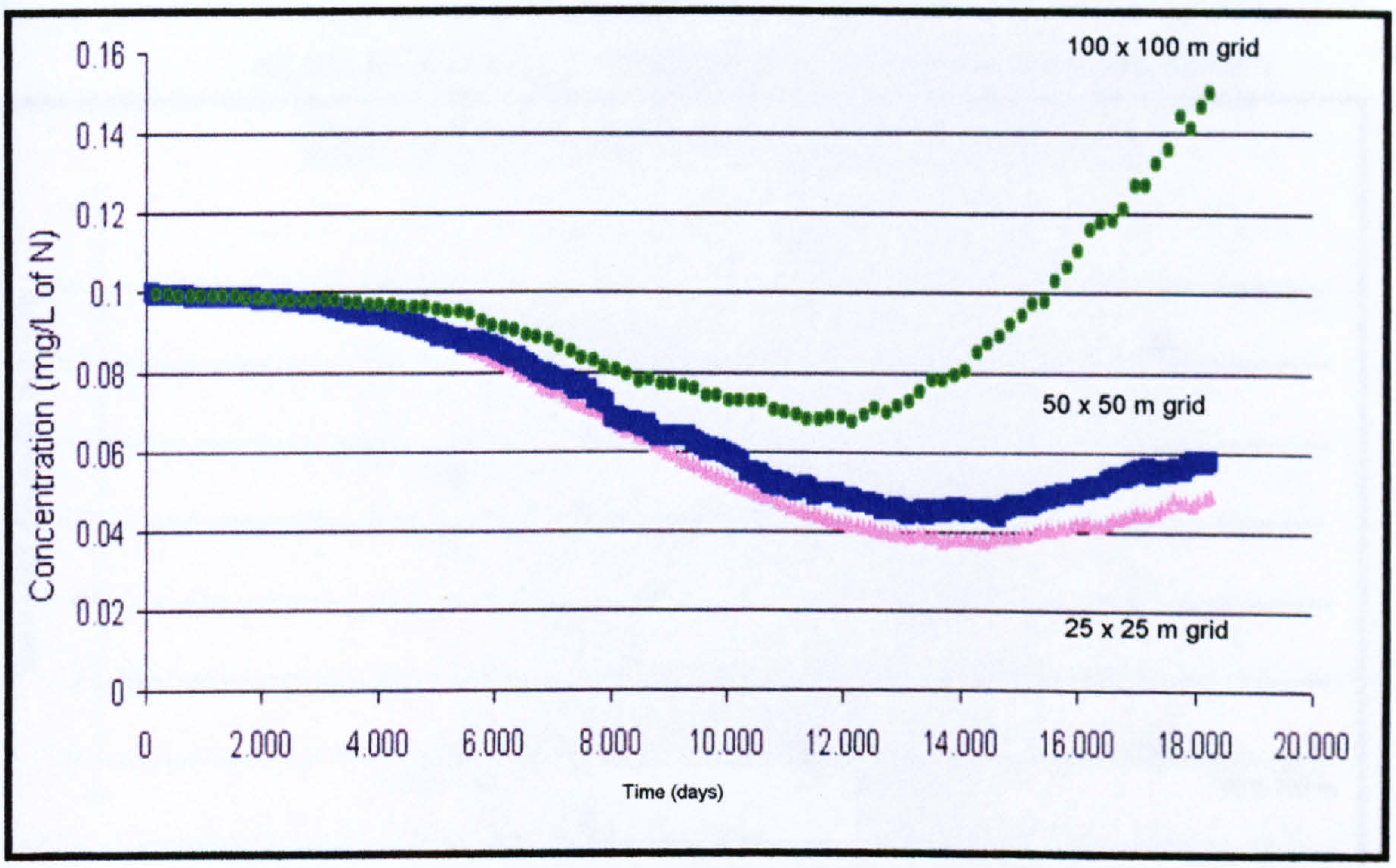
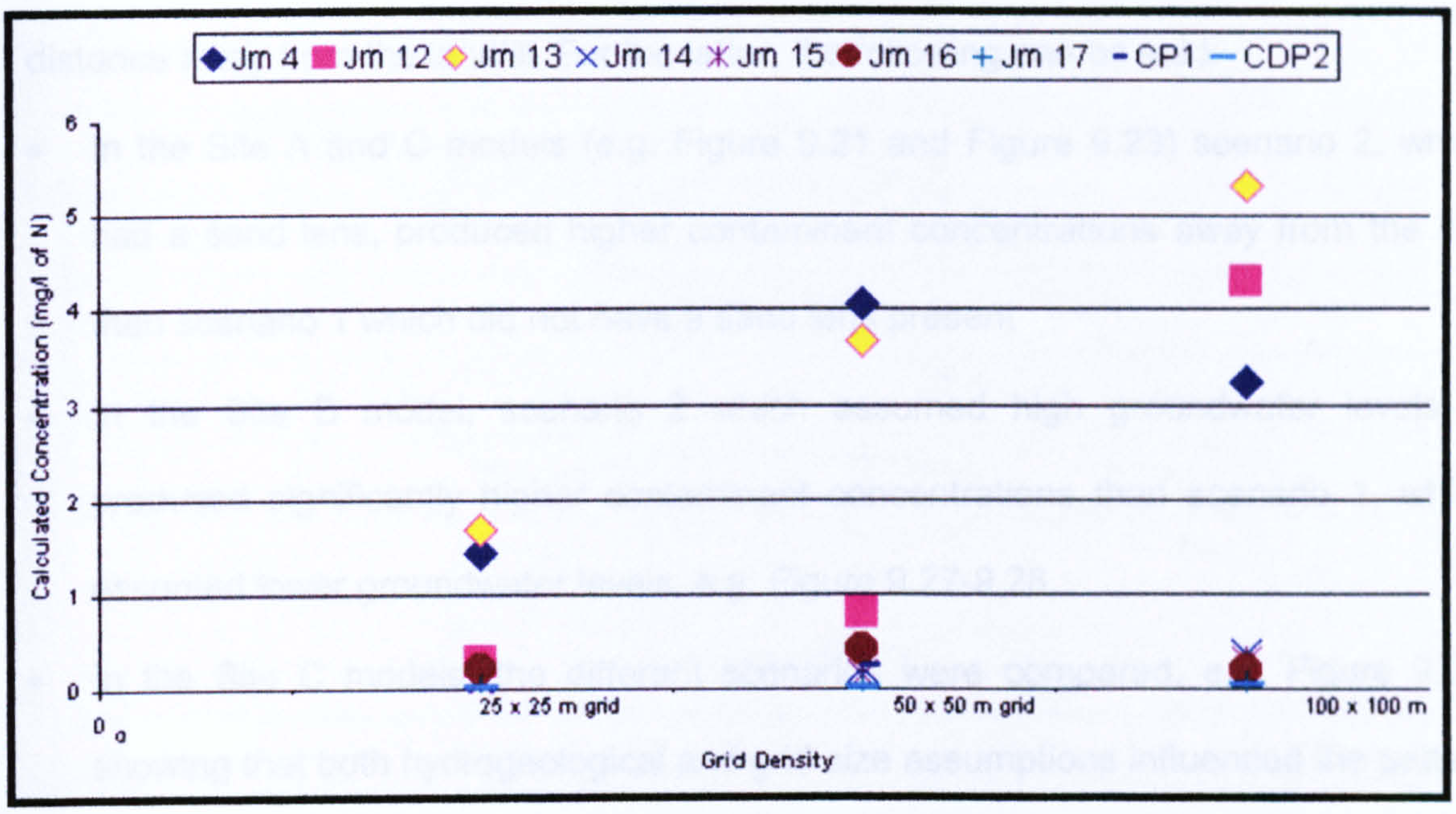
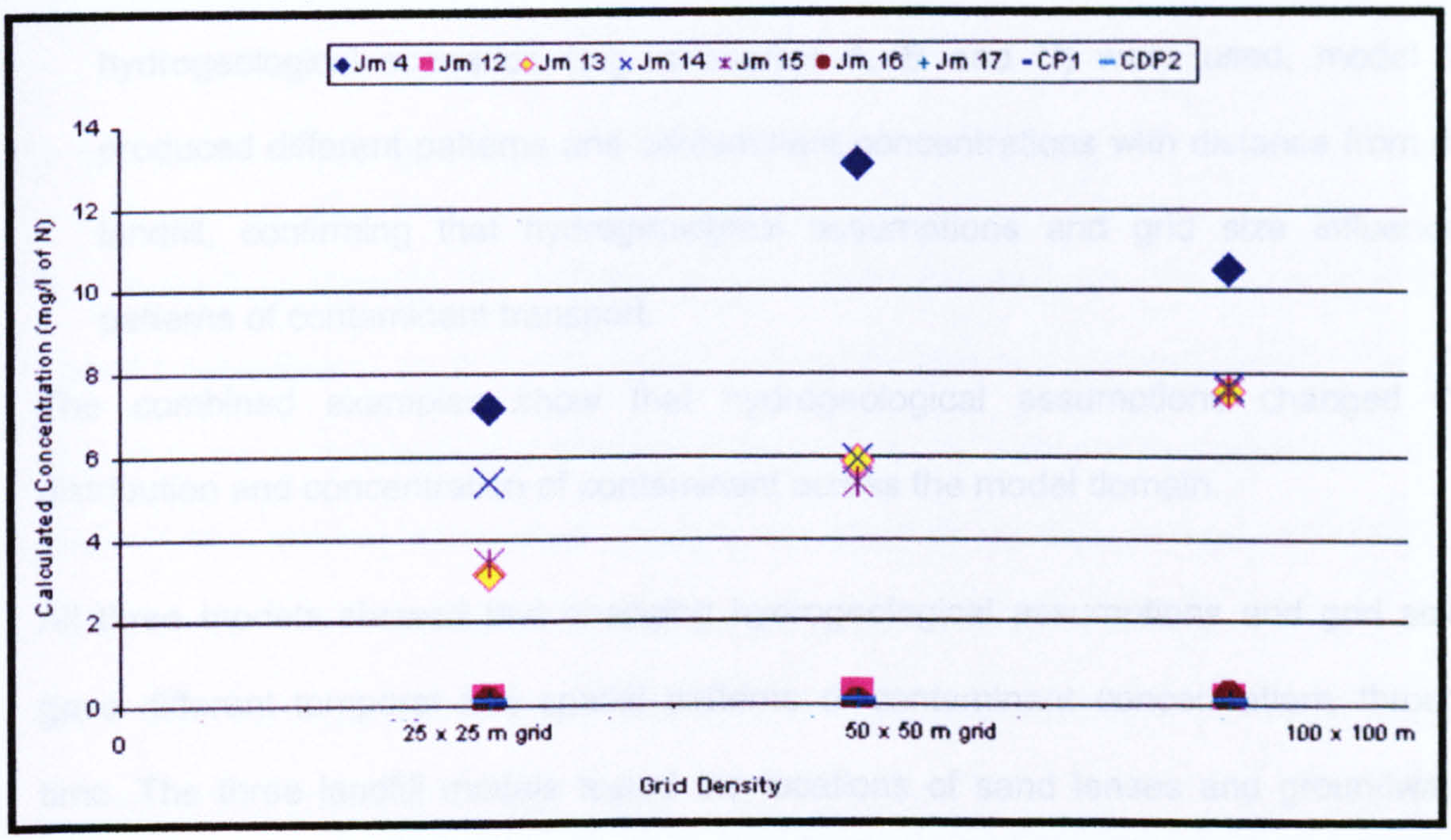


Figure 9.28 Site B large-scale model: Three trends were observed: (1) calibration points closer to the landfill were most sensitive to changes in contaminant concentrations; (2) increasing grid size gave higher contaminant concentrations; (3) hydrogeological model assumptions influenced contaminant concentrations

(a) Site B: Scenario 1 – Comparing grid sizes across the model domain



(b) Site B: Scenario 2 – Comparing grid sizes across the model domain



9.3.6 Hydrogeological Assumptions

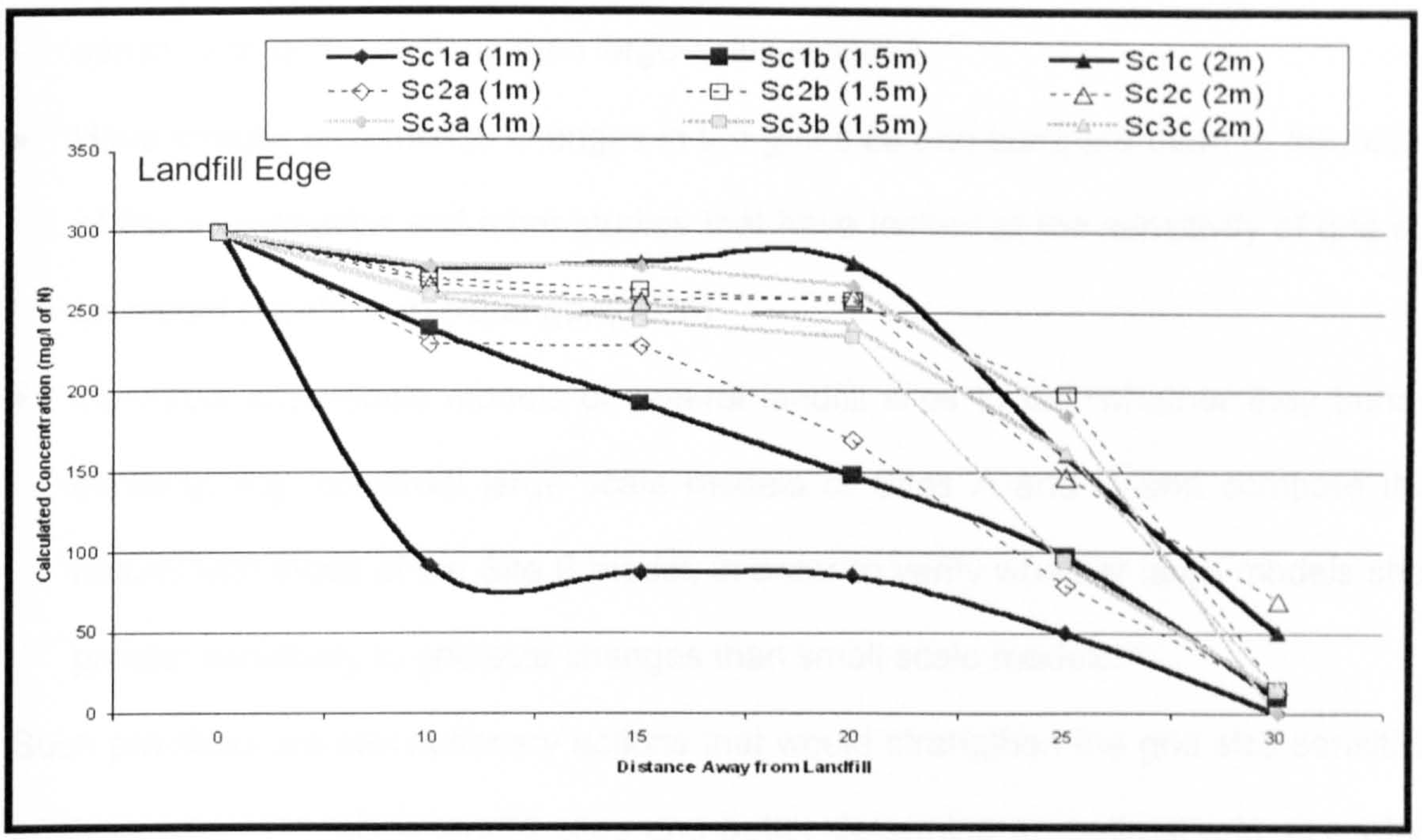
Model specific hydrological assumptions influenced contaminant concentrations and patterns of migration in all the models. This was observed when comparing both sets of model results, those compared through time and those comparing contaminants with distance away from the landfill. For the latter, the following can be said:

- In the Site A and C models (e.g. Figure 9.21 and Figure 9.23) scenario 2, which had a sand lens, produced higher contaminant concentrations away from the site than scenario 1 which did not have a sand lens present
- In the Site B model, scenario 2 which assumed high groundwater levels, it produced significantly higher contaminant concentrations than scenario 1, which assumed lower groundwater levels, e.g. Figure 9.27-9.28
- In the Site C models, the different scenarios were compared, e.g. Figure 9.29, showing that both hydrogeological and grid size assumptions influenced the pattern and concentration of contaminant migration. When three different grid sizes and hydrogeological scenarios (e.g. scenarios A, B and C) were used, model 'A' produced different patterns and contaminant concentrations with distance from the landfill, confirming that hydrogeological assumptions and grid size influenced patterns of contaminant transport.

The combined examples show that hydrogeological assumptions changed the distribution and concentration of contaminant across the model domain.

All three models showed that changing hydrogeological assumptions and grid sizes gave different temporal and spatial patterns of contaminant concentrations through time. The three landfill models tested the locations of sand lenses and groundwater levels. These model parameters are directly derived from field data collected during the site assessment. In context of the risk assessment, the differing model results show that hydrogeological information derived from the site assessment significantly influenced contaminant concentrations simulated in each landfill model.

Figure 9.29 Site C small-scale model: Three different grid sizes and hydrogeological scenarios (e.g. scenarios A, B and C) produced different contaminant patterns and concentrations, confirming that hydrogeological assumptions and grid size influenced patterns of contaminant transport



Legend:
Scenario a = 1 x 1 m grid
Scenario b = 1.5 x 1.5 m grid
Scenario c = 2 x 2 m grid

9.3.7 Grid Analysis Summary

It is difficult to quantify the findings of grid sensitivity analysis due to: (a) the site-specific conditions in each landfill model; and (b) the different grid sizes tested. Despite this, all three models behaved similarly showing that grid size and hydrogeological data derived from the site assessment need to be carefully considered when constructing model scenarios. Model behaviour can be condensed into three trends:

(a) The contaminant concentration changed with distance away from the site when grid size was increased

(b) Contaminant patterns of migration and concentrations were altered through time when grid size was increased

(c) Hydrogeological assumptions in each of the landfill models influenced contaminant concentrations.

When evaluating the sensitivity of grid size in future modelling of landfill sites, good modelling practice should be to:

- Quantify the influence of model scale (whether small-scale models are more sensitive to grid variations than large-scale models)
- Have smaller incremental changes in the grid size and compare them to the results of this investigation and other studies that have looked at the sensitivity of grid size on model results, e.g. Matanga, (1996)
- Construct large-scale models of several landfill sites to test whether they behave similarly, e.g. construct large scale models of Sites A and C and compare their results with those of the Site B model, in order to verify whether large models show greater sensitivity to grid size changes than small scale models.

Such practices are precautionary actions that would strengthen the grid size sensitivity analysis when modelling landfill sites and better determine an optimal grid size which can influence the direction and gradient of contaminant migration with landfill models simulating groundwater flow and contaminant transport.

9.4 Investigation 6: Hydraulic Conductivity Analysis

9.4.1 Objective

The purpose of this investigation was to test the sensitivity of hydraulic conductivity values assigned during model construction and calibration. This parameter influences the direction and velocity of groundwater flow as well as contaminant transport. Its value can be either derived from soil samples or measured directly in the field or laboratory. It can also be derived from regional or local information about hydrogeological and soil conditions. Several studies have tested the sensitivity of this parameter on groundwater flow models and on pesticide distribution, (e.g. Sudicky, 1986; Hanor, 1993; McDougall *et al*, 1996; Gburek and Folmar, 1999). This investigation focuses on groundwater flow and contaminant transport models that are used for risk estimation purposes. As a result the experiments focused on the impact that field data and the modeller could have upon contaminant transport simulations.

9.4.2 Models Used and Modelling Assumptions

The sensitivity of hydraulic conductivity was tested using four models initially presented in Section 9.1, e.g. Figures 9.1 – 9.4. Several sets of hydraulic conductivity values were tested for each site, e.g. Table 9.9. The first scenario used site-specific conductivity values derived from field data collected during the site assessment. The other scenarios used conductivity values taken from previous studies, regional data and model calibration. This experimental design was used because small-scale hydraulic conductivity data about landfill conditions is difficult to collect and is therefore often missing. The experimental design therefore reflected conditions that are often found when constructing groundwater and contaminant transport models of landfill sites. The availability of conductivity data varied in each model. In the large-scale Site A model, there were two sets of data and in the remaining models there were three sets of data.

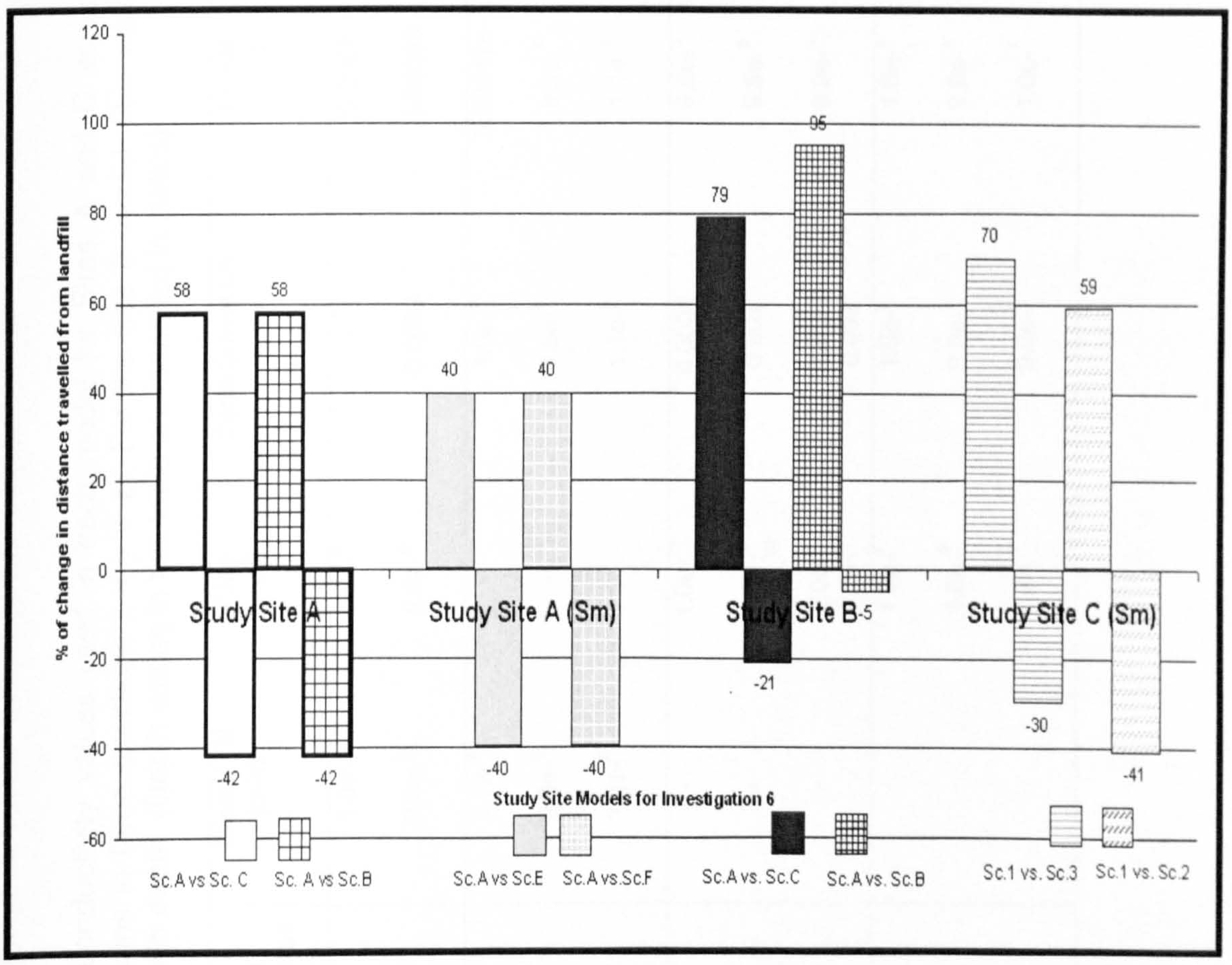
Assuming isotropic conditions is common practice in groundwater flow modelling, (e.g. Zheng *et al,* 2000). This is due to three reasons: (1) it is difficult to measure hydraulic conductivity in three dimensions (x, y and z); (2) it keeps modelled conditions simple and homogeneous; (3) this type of data is often not available at site-specific scales needed for model construction. However, the implications of such assumptions may significantly affect the results if field conditions differ from model assumptions. The Site A and C models assumed isotropic conditions for hydraulic conductivity. For Study Site B, extensive hydrological studies had been conducted in which there was evidence of flow similarity in the x and y directions, but changing with depth, e.g. Table 9.9. The investigation expected this parameter to influence model results. However, in context of the risk assessment, the modelling objective was to transfer field measurements (hydraulic conductivity measurements) into a model and then evaluate the influence of different assumptions (different hydraulic conductivity values) on contaminant simulations. The aim was therefore to isolate the influence of the field data and modelling assumptions.

The models reacted to increasing conductivity values, in the following way: (1) contaminant concentrations changed and generally increased with distance away from the landfill; (2) the migration patterns and contaminant concentrations were altered through time, and (3) contaminant concentrations reacted to differences in hydrogeological assumptions in each model.

9.4.3 Comparing Modelled Concentrations with Distance from the Site

The results showed that three of the four models significantly increased contaminant concentrations away from the landfill when hydraulic conductivity values were increased, e.g. Figures 9.30 – 9.33. Only the small-scale model of Site A reacted differently in which 20 percent of the measured points remained unchanged while the remaining 80 percent varied with higher and lower contaminant concentrations, e.g. Figure 9.30. This reaction was significant because it indicated that the model was sensitive to the first set of conductivity value changes but was insensitive to the second set. It also indicated that hydraulic conductivity values would influence model results, depending upon the hydrogeological assumptions in the model. This was identified in Figure 9.3.2 in which some scenarios in the Site A small-scale model caused contaminant concentrations to increase slightly (sc.1) however the models were not sensitive to conductivity values used in scenario 1F while scenario 2F was sensitive, producing lowered concentrations. Figure 9.33 shows that by increasing hydraulic conductivity values, contaminant concentrations increased.

Figure 9.30 Comparing model behaviour when hydraulic conductivity values are increased showing that the percentages of calibration points that had higher or lower concentration concentrations away from the site



Legend:

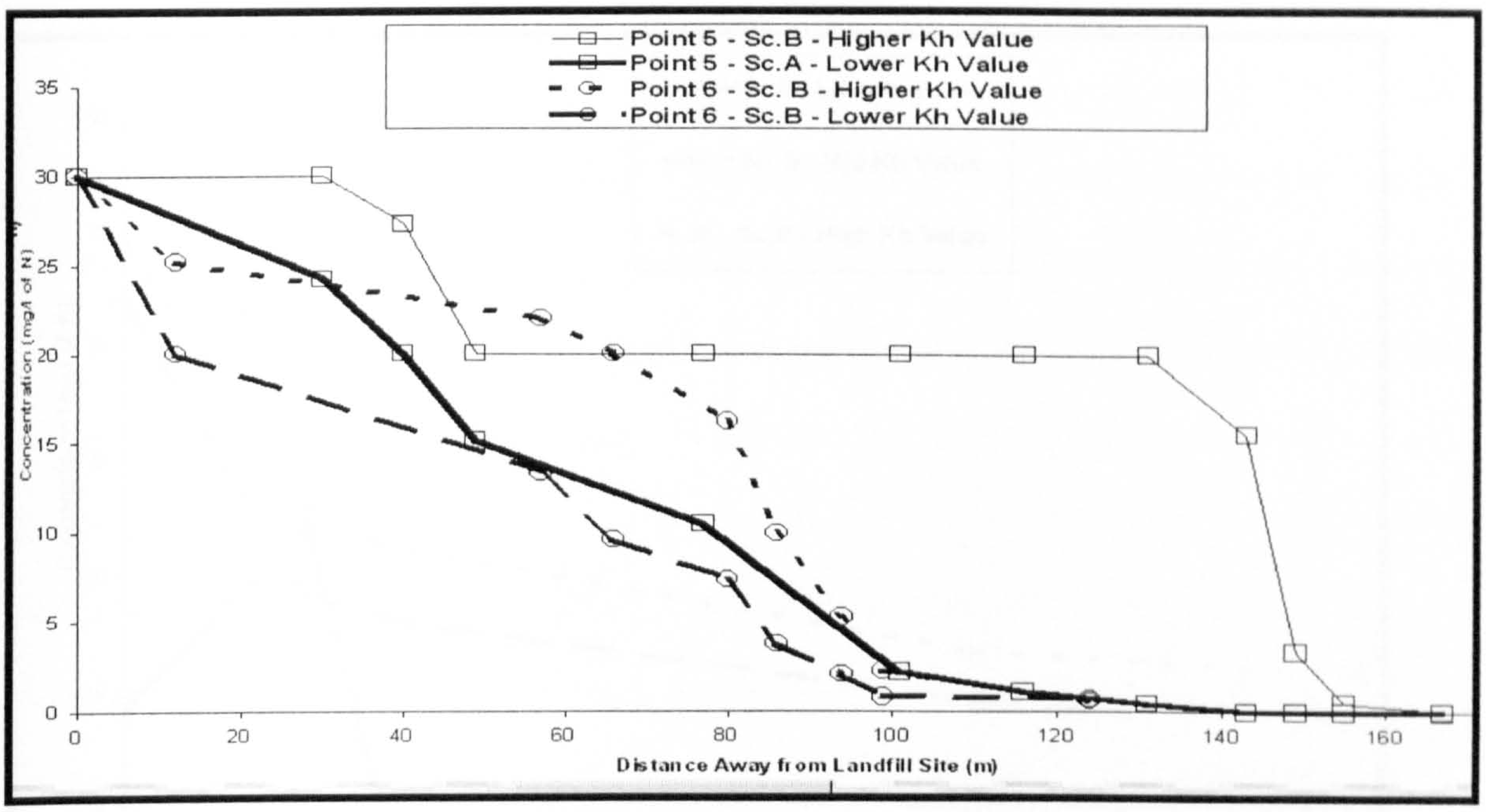
- Positive values = % of calibration points in the model that produced higher contaminant concentration concentrations away from the landfill
- Negative values = % of calibration point that produced lower contaminant concentration concentrations away from the landfill

Table 9.9: Hydraulic conductivity values used in each model for Sites A and C assumed isotropic conditions (equal hydraulic conductivity concentrations in three directions x = y = z). For the Site B model, flow was considered equal in the x and y directions, changing vertically with depth (Depth values in this table are identified in *Italics*)

Study Site Model	Kh Data Set that was tested	Glacial Till (m/s)	Clay (m/s)	Sand-Gravel Lens (m/s)	Waste (m/s)	Cut-Off Wall & Lined Landfill (m/s)	Alluvium (m/s)
Site A	**Sc.1**	$1.0e^{-7}$	$1.0e^{-8}$	-	0.0001	$1.0e^{-10}$	$1.0e^{-7}$
Large-Scale	**Sc.2**	$9.9e^{-7}$	$9.9e^{-8}$	0.0099	0.00099	$9.9e^{-10}$	$9.9e^{-7}$
Site A	**Sc.1**	$1.0e^{-8}$	-	$1.0e^{-5}$	$0.00017e^{-4}$	-	$1.0e^{-7}$
Small-Scale	**Sc.2**	$9.9e^{-8}$	-	$9.9e^{-5}$	$9.9e^{-4}$	-	$9.9e^{-7}$
	Sc.3	$1.1e^{-8}$	-	$1.1e^{-5}$	$1.1e^{-4}$	-	$1.1e^{-7}$
Site B	**Sc.1**	-	$1.0e^{-11}$	0.0001	$9.9e^{-6}$	-	0.001
Large-Scale	**Sc.2**	-	$1.0e^{-8}$ *$6.7e^{-10}$*	0.001	$9.9e^{-5}$	-	0.00049 *$6.6e^{-6}$*
	Sc.3	-	$1.0e^{-8}$ *$6.7e^{-9}$*	0.00256 ***0.008***	$9.9e^{-6}$	-	0.00099
Site C	**Sc.1**	-	$1.0e^{-9}$	$1.0e^{-5}$	$1.0e^{-5}$	$1.0e^{-12}$	-
Small-Scale	**Sc.2**	-	$1.0e^{-9}$	$9.9e^{-5}$	$9.9e^{-5}$	$1.0e^{-12}$	-
	Sc.3	-	$2.0e^{-9}$	$9.9e^{-5}$	$1.0e^{-7}$	$1.0e^{-12}$	-

Legend:
Kh = Hydraulic Conductivity
Sc. = Scenario

Figure 9.31 Site A large-scale model: Contaminant concentrations away from the landfill showing that the higher conductivity scenarios produced higher contaminant concentrations



Legend:

Bh = Borehole   Kh = Hydraulic Conductivity

Point 5 = Scenario 2   Point 6 = Scenario 3

Figure 9.32 Site A small-scale model: Contaminant concentrations away from the landfill showing that contaminant concentration varied. The models were insensitive to conductivity values used in scenario 1F but sensitive to scenario 2F

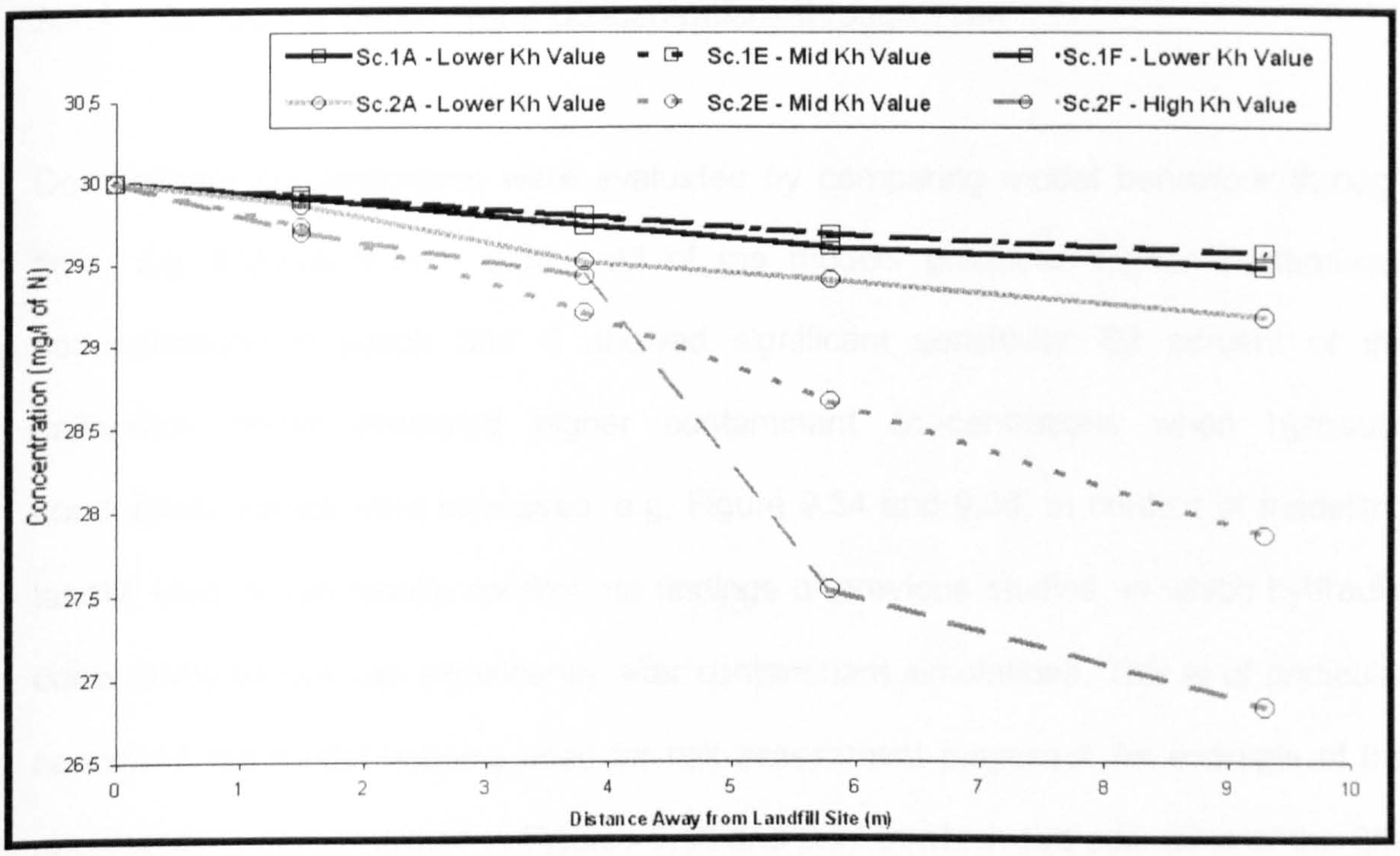
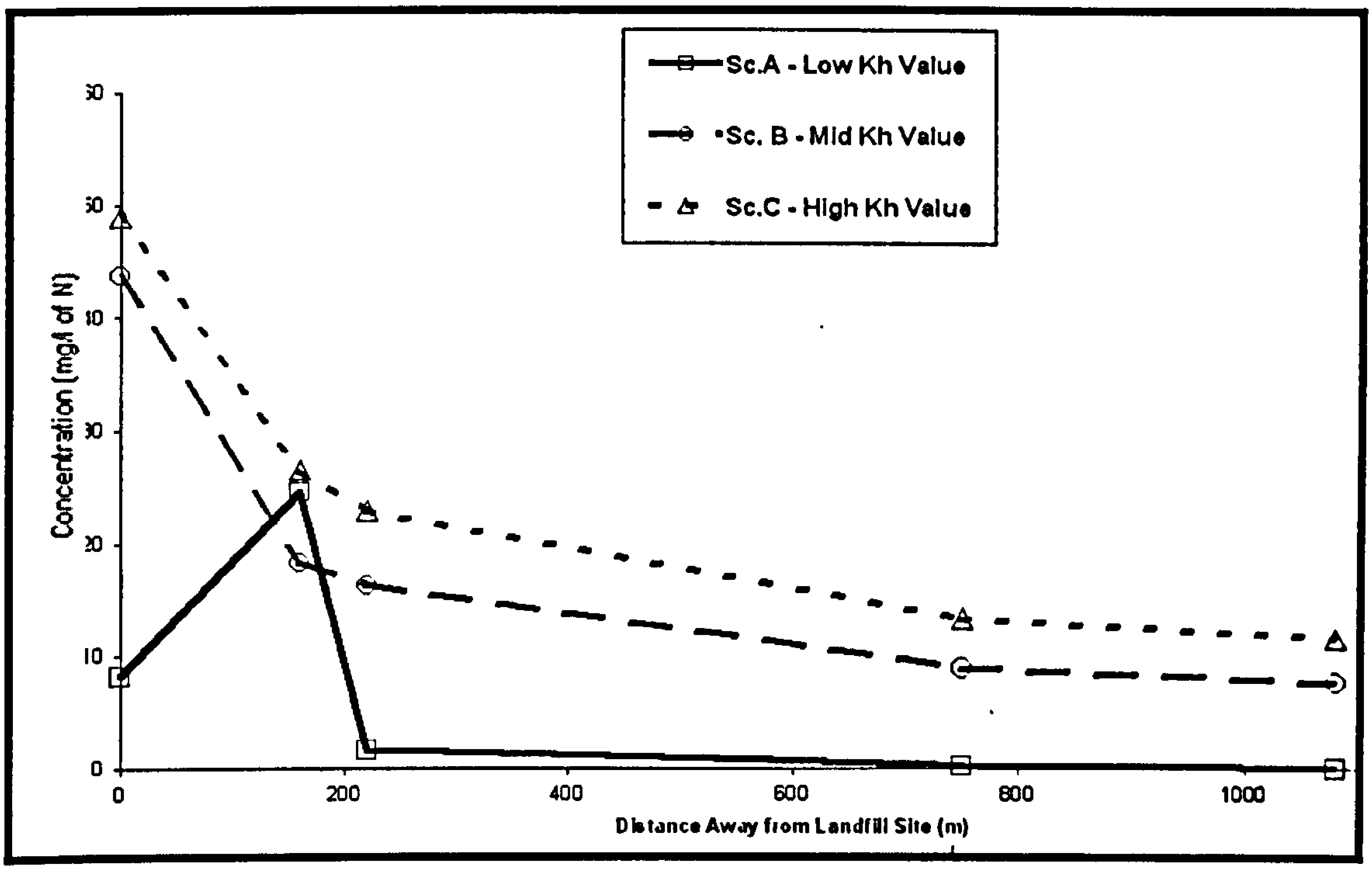


Figure 9.33 Site B large-scale model: by increasing hydraulic conductivity values, contaminant concentrations increased away from the landfill

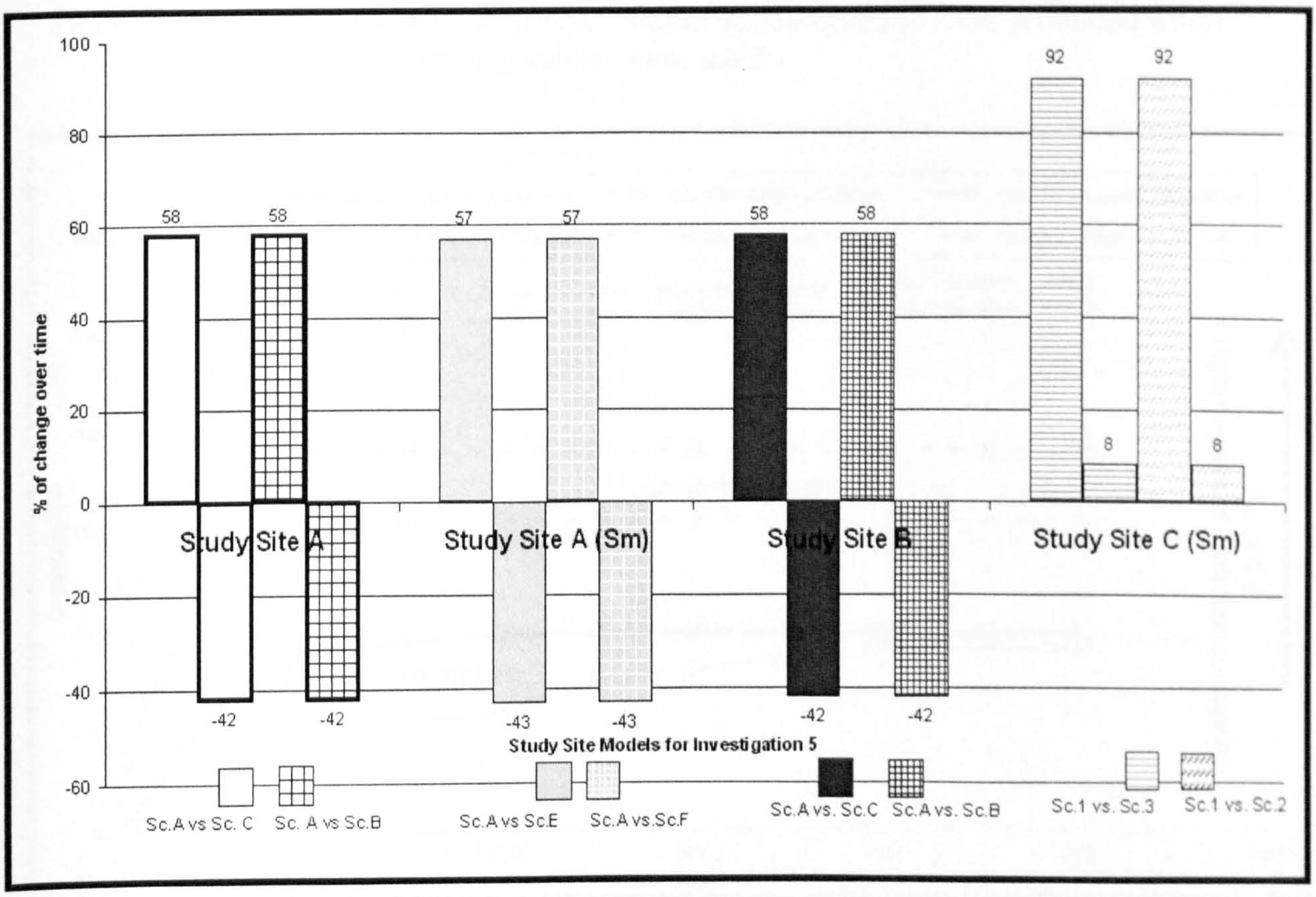


Legend:
Sc. = Scenario
Kh = Hydraulic Conductivity

9.4.4 Comparing Contaminant Concentrations through Time

Contaminant concentrations were evaluated by comparing model behaviour through time, e.g. Figures 9.34 – 9.38. All of the models produced higher contaminant concentrations in which Site C showed significant sensitivity: 92 percent of the calibration points produced higher contaminant concentrations when hydraulic conductivity values were increased, e.g. Figure 9.34 and 9.36. In context of modelling landfill sites, these results confirm the findings of previous studies, in which hydraulic conductivity values can significantly alter contaminant simulations. This is of particular concern if the model is being used for risk assessment purposes. An example of the uncertainty is demonstrated in Figures 9.35 and 9.37 in which two scenarios of the Site A and Site B models are presented. The models used hydraulic conductivity ranges that reflected the minimum and maximum values found or measured during the site assessment. Each of the models reacted differently to the conductivity values, altering contaminant patterns of migration and producing higher contaminant concentrations. If these models were used for risk assessment purposes to estimate whether leachate migration away from the landfill would increase with time, the modelling results would significantly differ. In the case of the Site B model (Figure 9.37), the risk assessment would have quite varied results for stakeholders to consider since the potential dates by which contaminant transport was expected to increase varied from day 1000 through to day 3000 for scenario 1 and from day 4000 through to day 10000 in scenario 2. The variability in modelled results points to the importance of the hydraulic conductivity parameter and also points to the importance of having effective field data with which to form sound hydrogeological assumptions about a site's conditions.

Figure 9.34 Comparing model behaviour through time when increasing values of hydraulic conductivity



Legend:

- Positive values = % of calibration points in the model that produced higher contaminant concentrations on the last day of simulation
- Negative values = % of calibration point that produced lower contaminant concentrations on the last day of simulation

Figure 9.35 Site A small-scale model: Increasing conductivity values produced higher contaminant concentrations in scenario 1

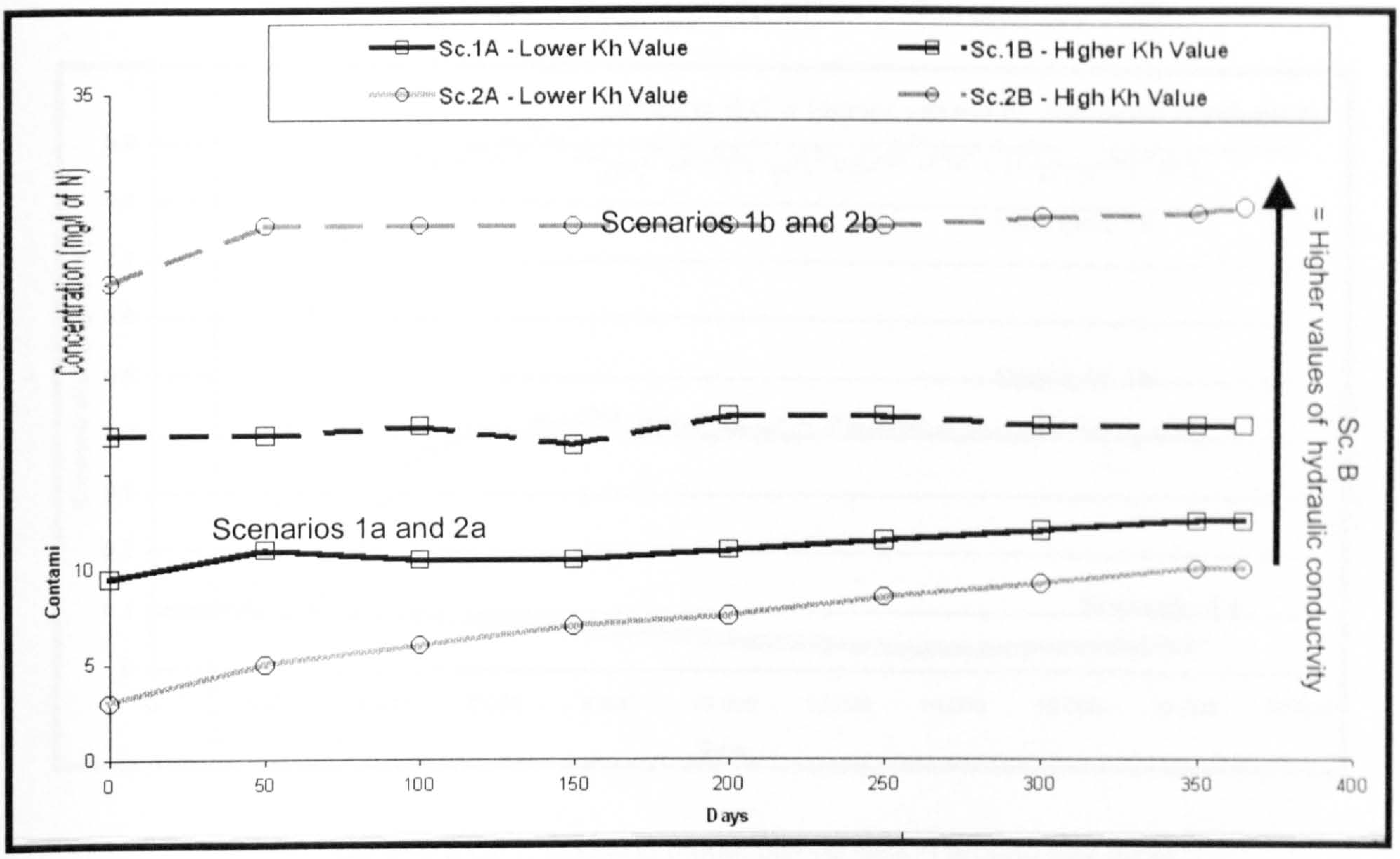


Figure 9.36 Site C small-scale model: Higher concentrations were produced when higher conductivity values were used

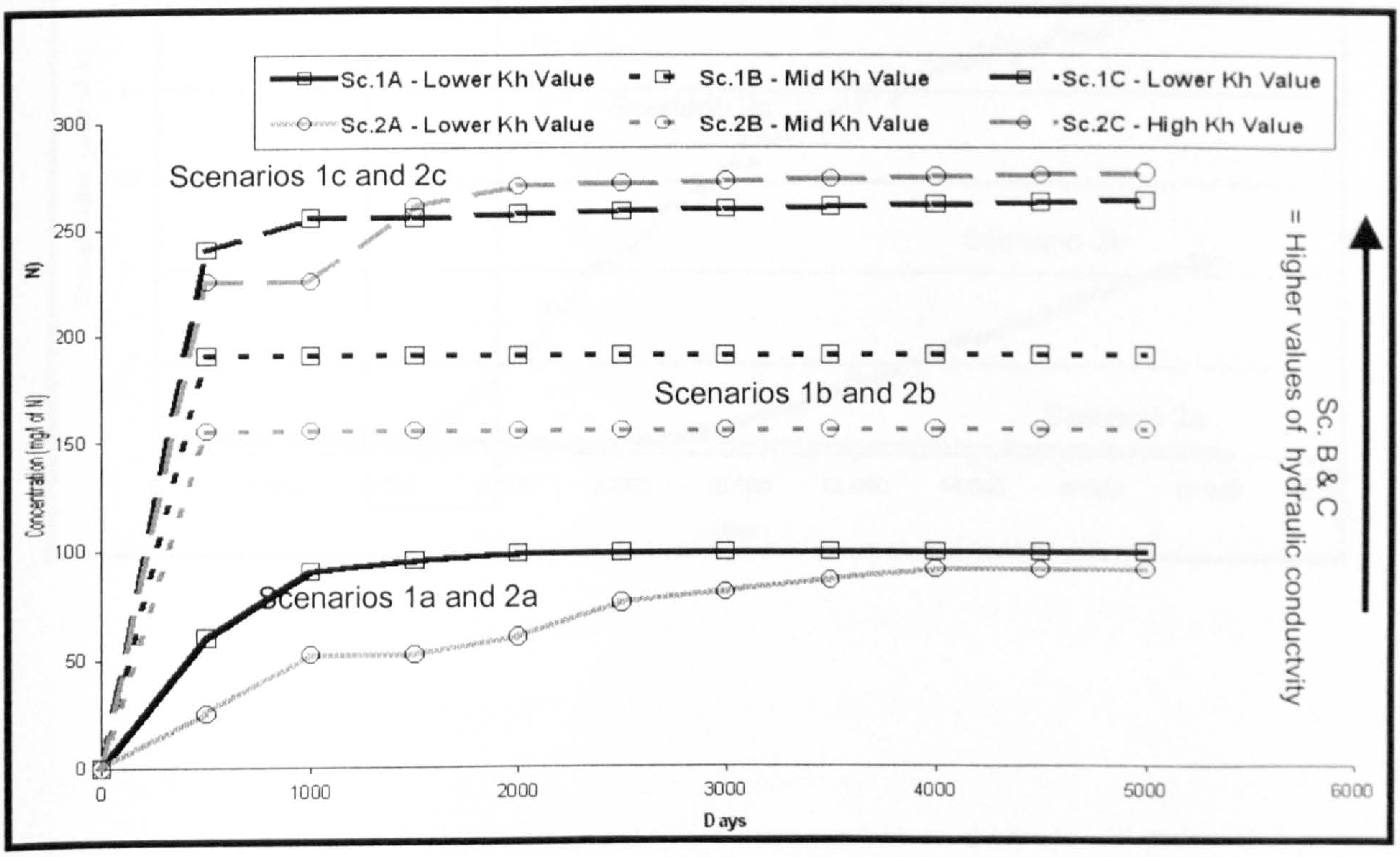
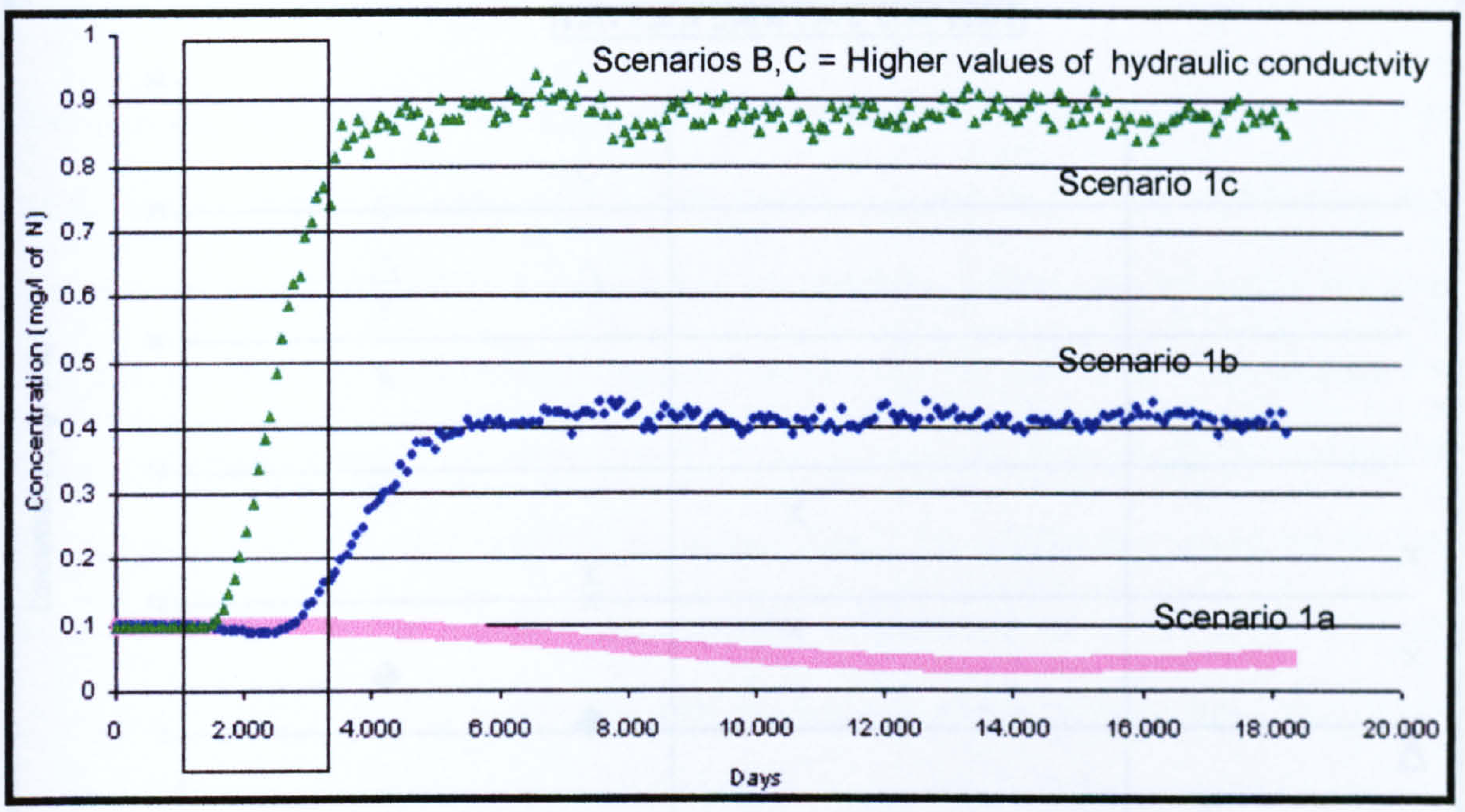


Figure 9.37 Site B large-scale model: Contaminant concentrations through time increased rapidly at some point in time in scenarios that used higher hydraulic conductivity values

(a) Scenario 1: Jm16 showing model differentiation after day 1000.



(b) Scenario 2 CP1: Showing model differentiation around day 5000.

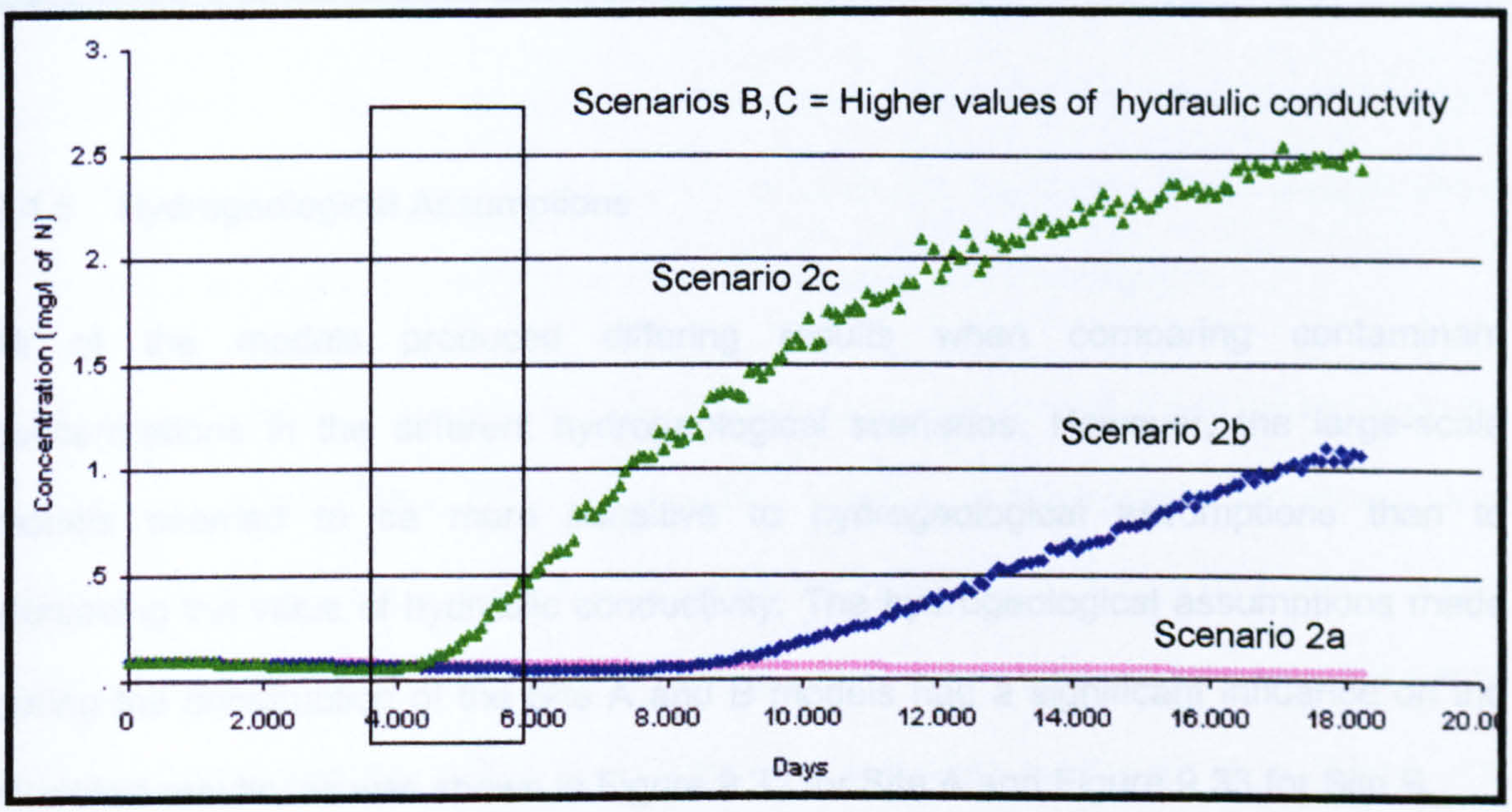
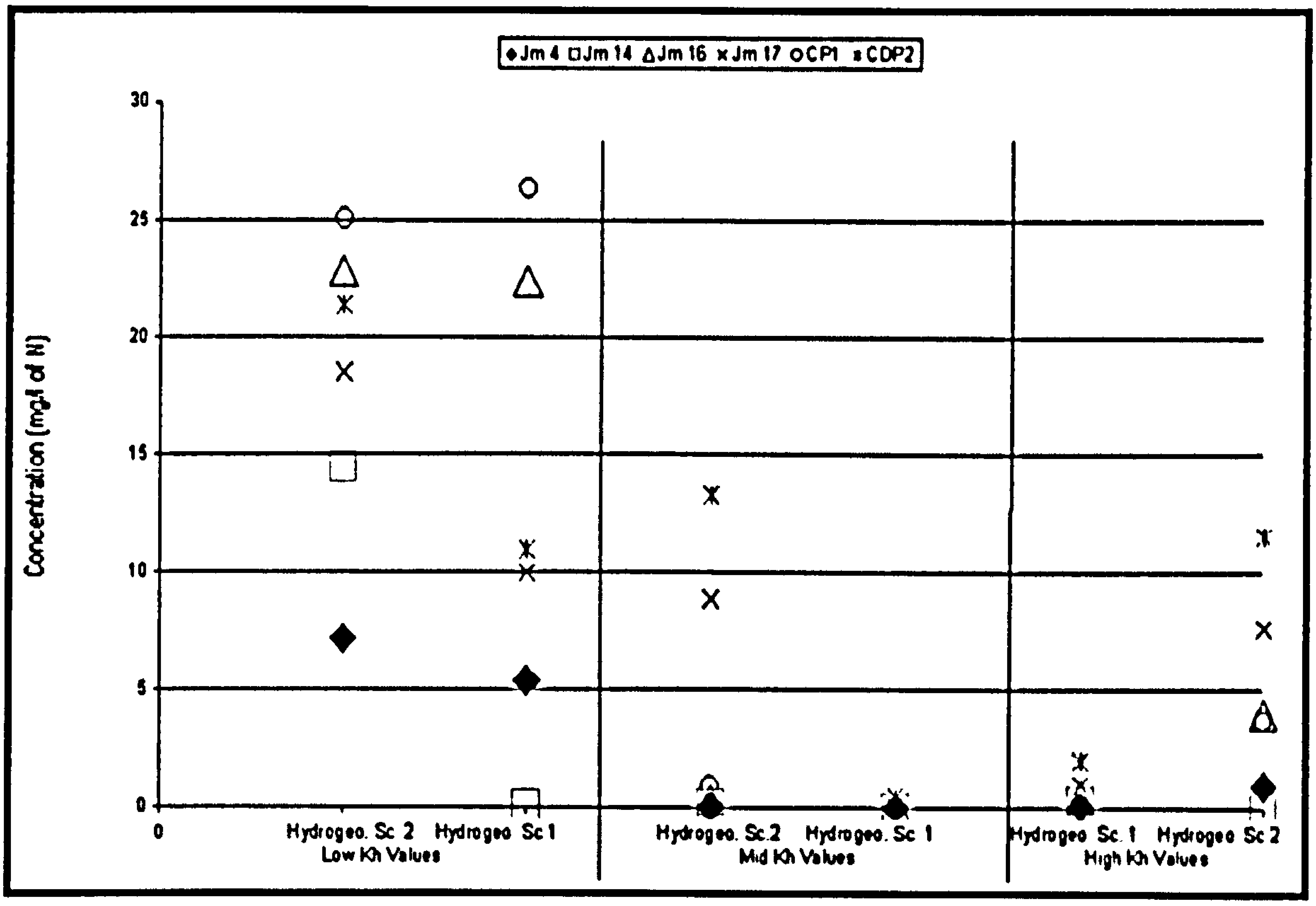


Figure 9.38 The Site B models assumed higher regional groundwater levels and produced higher contaminant concentrations, then scenario 1, which assumed lower regional groundwater levels



9.4.5 Hydrogeological Assumptions

All of the models produced differing results when comparing contaminant concentrations in the different hydrogeological scenarios. However, the large-scale models seemed to be more sensitive to hydrogeological assumptions than to increasing the value of hydraulic conductivity. The hydrogeological assumptions made during the construction of the Site A and B models had a significant influence on the modelled results, as was shown in Figure 9.32 for Site A and Figure 9.33 for Site B. The Site B model is a good example of this, in that scenario 1 which assumed lower groundwater levels produced lower contaminant concentrations while scenario 2 which assumed higher levels, produced higher contaminant concentration over time, e.g. Figure 9.38. In the small-scale models, the different hydrogeological scenarios did have some impact on contaminant concentrations, e.g. Figure 9.35 for Site A, however hydraulic conductivity had a greater effect, altering contaminant concentrations through time.

9.4.6 Investigation Summary

The models that were tested used different hydraulic conductivity values and had incremental ranges. Despite these differences, model behaviour confirmed that hydraulic conductivity values are sensitive parameters, which can influence the rate and concentration of contaminant migration away from the site as well as change the pattern of concentration migration through time. In order to strengthen the findings of this investigation, further modelling analyses should be conducted, focusing on two areas. The first is to model using smaller increments of increasing hydraulic conductivity values, comparing results with the findings of this investigation. This would provide validation of the impact of increasing conductivity values when constructing models. The second area of focus should be to standardise the amount of time used in model simulations so that all four models simulated contaminant concentrations over the same period of time. This would provide more insight into the influence of time as a parameter to consider when simulating contaminant transport. Such investigations would reinforce the trends that have been identified in this investigation.

In context of the site assessment, the results presented in this investigation indicate that by increasing hydraulic conductivity values, contaminant simulation was significantly influenced, which may influence the accuracy of a risk estimation model. Many of the models produced higher contaminant concentrations, however there was some variation indicating that the relationship between increasing conductivity and contaminant flow concentrations was not always similar. The different hydrogeological scenarios that were constructed for each site also significantly influenced patterns of contaminant migration, regardless of the hydraulic conductivity value used. Given that all the models showed significant sensitivity to both conductivity and hydrogeological assumptions, model construction should pay close attention when interpreting these data into a model. Inaccurate assumptions could have implications on the accuracy of risk estimation.

9.5 Investigations 3 - 6: Modelling Summary

The second part of this chapter focused on groundwater flow and contaminant transport models, testing Objectives 3, 4 and 5 which evaluated the influence of field data during model construction on (a) model performance and (b) modelling practices and assumptions. In context of the risk assessment, the three investigations assessed the value of additional data collected during the site assessment. Investigation 4 (Field Data Used in Model Construction and Calibration) used several data sets to construct landfill models. The results showed that by increasing the amount of field data available for model construction, the conceptual and hydrogeological conditions within each model were also changed. This inherently altered contaminant concentrations and plume shapes. It is difficult to determine which of these data sets produced the most accurate simulations since further field data are needed to validate the assumptions of each model. Instead, what is important to note is that the additional data improved the conceptual understanding of site conditions, which is a critical piece of information needed during model construction. Investigation 4 therefore confirmed that additional data will influence contaminant transport results but it did not confirm whether the models themselves were more accurate. Investigations 5 and 6 took the same models and tested Objectives 4 and 5, the impact of changing grid size and hydraulic conductivity values, evaluating whether limited knowledge of site conditions when inferring these model parameters, could also influence the assumptions made during model construction. Here again, the investigations confirmed that such assumptions had a significant effect on contaminant concentrations and patterns of migration through time. The sensitivity to data sets and modelling assumptions reinforces the role of the modeller during model construction. In future modelling of landfill sites, good practice should be to ensure that the modeller:

(a) has a solid understanding of hydrogeology

(b) has a sound understanding of the groundwater modelling software being used

(c) adheres to good modelling practices and

(d) is able to determine the type and quality of field data needed for model construction.

The three investigations were initially motivated by studies conducted on pesticide transport models conducted by Bergstrom and Jarvis (1994); Diekkrugger *ot al* (1995); and van Clooster *et al* (2000). The main differences between these studies and investigations 4 - 6 is that investigations 4 - 6 used a 3-D groundwater flow and advection-dispersion model to simulate landfill leachate migration, using one modeller, one modelling software, and several sets of data. In contrast, the listed publications used several different pesticide simulation models, several modellers, and only one set of data for model construction. In order to validate the findings of this research, further modelling analyses could build upon these finding in which the landfill models should have similar: (a) model scales; (b) time frames for simulating contaminant transport; and (c) initial contaminant concentrations. In concluding, the investigations were not able to determine whether field data would significantly improve the accuracy of contaminant simulation. Instead, the investigations confirmed that uncertainties inherited from missing field conditions could have a profound impact on the assumptions and decisions made during model construction and on groundwater and contaminant transport simulations.

The evidence presented in this research once again brings attention to the importance of field data in both field assessments, modelling predictions and risk assessments. Many authors have called for caution when using field data to use groundwater and contaminant transport models for risk assessment purposes including Beven (1991); Bergstrom and Jarvis (1994); Diekkrugger *et al* (1995); Doherty (1999) and van Clooster *et al* (2000). An early example of such warnings is Anderson (1979) who stated:

'...Contaminant transport models are currently operating yet the realisation of their full potential must await the resolution of problems in acquiring field data and theoretically defining the complex interaction of groundwater flow and contaminant transport...' (Anderson, 1979)

It is interesting to note that almost 25 years has passed and the same message is again being emphasised through the findings of this research. This arguably indicates that despite significant technological developments that have influenced and improved methods, equipment and tools used when assessing and modelling contaminated site conditions (e.g. insitu digital measurement equipment, GIS, and user-friendly risk estimation models), little has changed since Anderson's 1979 statement. This is confirmed by Zheng (2000) who again states that risk-based groundwater flow and contaminant transport models have developed faster then the science needed to understand diverse hydrogeological transport processes and the methods available to measure these contaminant transport processes. In the case of contaminated land assessments, field techniques have remained at one and two dimensions, focused mainly upon point-based sampling scales. On the contrary, risk estimation models (such as groundwater flow and contaminant transport models) have expanded to simulate two and three-dimensional subsurface conditions. Several documents have been developed in the last decade to provide invaluable guidance, setting site assessors and modellers on the right track when determining the types and amounts of field data needed for risk assessment modelling of groundwater and contaminated transport. Examples include:

- Department of Environment (1994-1995) which outline sampling strategies for contaminated land and landfill assessments;
- ASTM (1994-1998) which provide guidelines for building contaminant transport and groundwater flow models used in risk assessment applications;
- CAMASE *cited*, (2000) which provide step-by-step instructions on constructing and calibrating contaminant transport models;

- McMahon *et al,* (2001(a-c)) which provide guidance on selecting appropriate mathematical models for subsurface contaminant transport as well as guidance in constructing conceptual models, selecting appropriate parameter values and analysing the effectiveness of these models;
- DEFRA and Environment Agency (2002(a-c)) which provide several helpful guidelines and lists of potential contaminants for the assessment of land, contaminants in soils and lists of toxicological data and intake values for humans.

This chapter has tested Objectives 3, 4 and 5 which focused upon testing the influence of field data, modelling practises and data assumptions on model outcomes. The results of this research have perhaps not produced new results in context of the findings from the modelling analyses. Instead, the empirical evidence presented in this chapter is a very strong confirmation of the value of effective field data, emphasising the value of collecting extensive and site-specific geophysical information at contaminated landfills and other similar types of contaminated sites.

CHAPTER 10: DISCUSSION

10.1 Chapter Introduction

This chapter assesses the results presented in Investigations 1-6, in terms of the five research objectives. It provides an overall review of: (a) how the investigations link back to the research objectives, (b) the value added by using each of the new methods, (c) the limitations faced with each new field method that was tested, and (d) how these different new methods can improve landfill site assessments.

The overall research objectives stated in Chapter 1 have been met to a large extent. The field assessment methods that have been evaluated all provided important additional information that could improve risk-based landfill assessments. It has been possible to formulate recommendations based on each of the five research objectives. The recommendations made at the end of this chapter aim to improve the accuracy of landfill site assessment and risk-based modelling of leachate migration, building on existing standards, models, and organisations involved in these areas in the UK. They are intended for government authorities, and professional associations dealing with landfill management.

10.2 Linking the Research Results to Project Objectives

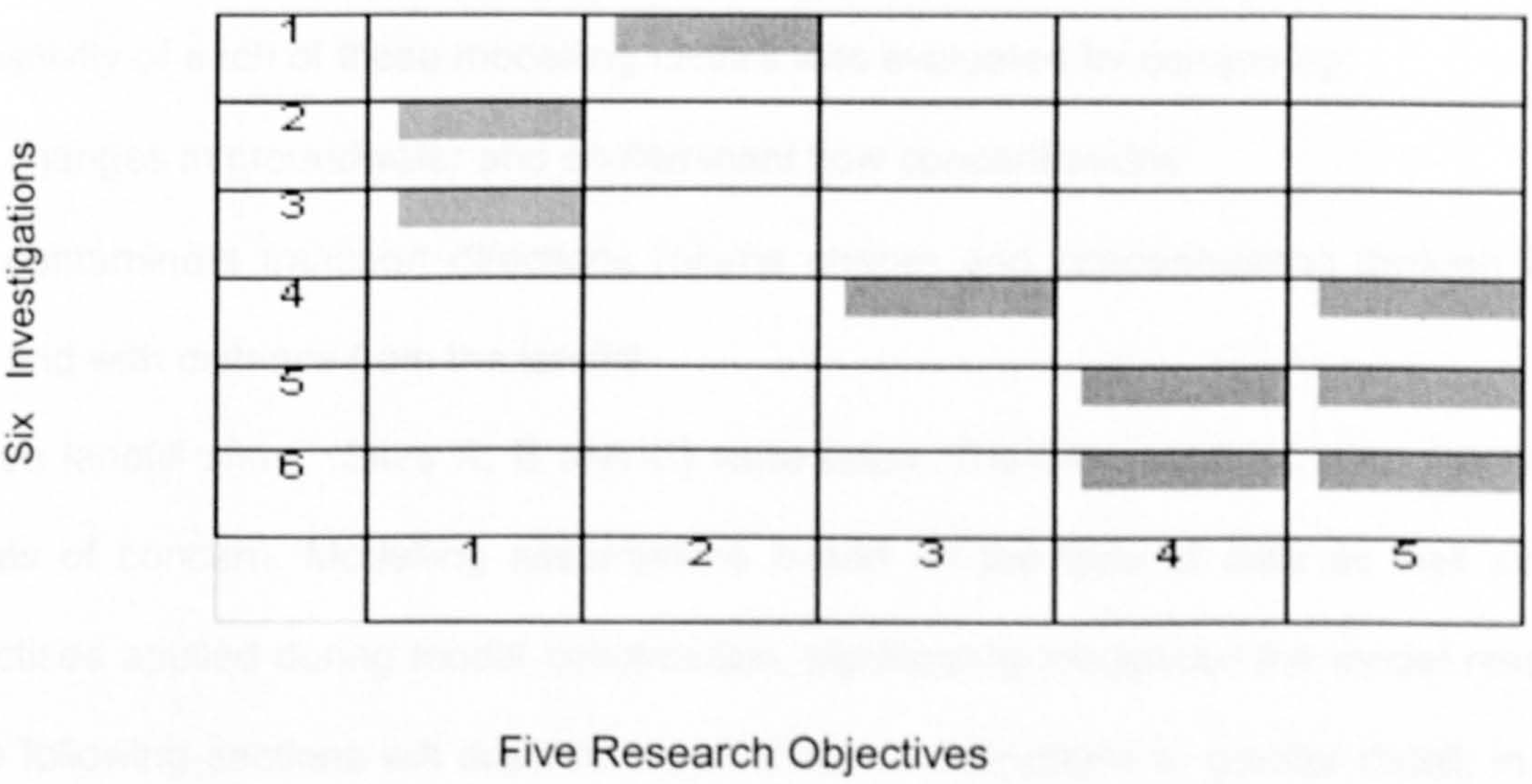
Six investigations were undertaken and presented in chapters 8 and 9, focusing upon site assessment methods and modelling practises that influence risk estimations used when evaluating the level of risk posed by contaminating landfill sites. The research objectives outlined in Chapter 1 include:

1. Evaluating whether innovative field assessment methods can provide new insights into site-specific, subsurface and landfill structure,
2. Evaluating whether geostatistical models can be used to improve the sampling pattern and evaluate trends in data sets collected for heterogeneous and contaminating landfill sites,

3. Testing the influence of data that is used during groundwater flow and contaminant transport model construction, especially when these models are used to assess risks posed by a landfill,
4. Testing the influence of modelling practises that may effect leachate simulations in groundwater flow and contaminant transport models that are used in landfill risk assessment applications,
5. Evaluating the influence of assumptions made on groundwater flow and contaminant transport models, when these models are used for landfill risk assessment purposes. The assumptions can be related to the model used, the data available during model construction, the modeller's level of experience or comprehension of site-specific conditions.

Each of the six investigations presented in Chapters 8 and 9 tested one or two of the research objectives, as outlined in Table 10.1.

Table 10.1 Cross-referencing the five research objectives with the six investigations presented in Chapters 8 and 9



Investigations 1-3, which tested the effectiveness of kriging, GPR and remote sensing instruments present interesting findings that demonstrate the utility of each of the methods employed in characterising landfill conditions. The utility of each method is reviewed on the basis of:

- How it links back to the research objectives
- Its contribution in the context of the overall landfill site assessment (relating back to objectives 1 and 2),
- How it can be applied and integrated with other methods to improve both the robustness of a site assessment, and modelling accuracy (relating back to objectives 1 and 3), and
- Limitations and cost-effectiveness of each method identifying aspects requiring further scientific research.

All three applications demonstrated that if used under appropriate conditions, the inherent uncertainties that are often found in the site assessment could be reduced. The modelling investigations (investigations 4-6) were focused upon the data used, the assumptions made and the modelling practises applied during model construction. The sensitivity of each of these modelling factors was evaluated by comparing:

- changes in groundwater and contaminant flow concentrations
- contaminant transport directions (plume shape) and concentrations through time and with distance from the landfill.

Three landfill sites (Sites A, B and C) were used. The investigations have identified areas of concern. Modelling assumptions based on the type of data as well as on practises applied during model construction, significantly influenced the model results. The following sections will describe each of the investigations in greater detail, in the context of the research objectives and the contribution each can made to landfill site assessment.

10.2.1 Kriging

The results from the kriging investigations demonstrated that additional insight into the field conditions required for site assessment could be resolved. The aim of the investigation was to test whether kriging could optimise sampling locations at two landfill sites. The investigation is linked back to research objectives 2, which aimed to establish whether geostatistical modelling could provide the site assessor with a better understanding of heterogeneous site conditions and whether such tools could be used to optimise the search for new sampling locations.

Investigation 1 confirmed that kriging (as a geostatistical-modelling tool) can meet the objectives set out in research objective 2. It can verify existing or planned sampling locations, as well as provide insight into subsurface and hydrogeological conditions around a landfill site. The results confirm the effectiveness of this tool when using limited data sets coming from older and unlined landfill sites with poor landfilling records. The findings of investigation 1 demonstrate that kriging can make a major contribution to the understanding of data from heterogeneous landfill sites. It is particularly effective when:

- evaluating whether additional sampling points are needed at a landfill site
- designing a sampling pattern, e.g. placing new piezometer points around a site to measure groundwater levels
- planning a site assessment to identify areas that should be subject to a more detailed investigation
- evaluating areas of the site that are difficult to access, (e.g. rough terrain at a part of a landfill). If such areas have even one sample point (e.g. one borehole, with historical leachate data), kriging can provide additional insight into the variability of hydrological and contaminant fluctuations at such areas.

It is a cost-effective tool widely available as an independent modelling package or as an extension of a proprietary GIS software (e.g. ArcView GIS v.3.2). In the context of improving groundwater flow and contaminant transport models, kriging can be used to evaluate the spatial distribution of field data used in model construction. It can also be used to improve the representation of spatially distributed hydrogeological parameters in models. The main disadvantage of kriging is that to give reliable results it requires high levels of spatially distributed data to identify new sampling locations. However if aiming to investigate the level of heterogeneity at a given landfill site, then even data sets with very few samples can be evaluated using this tool. This was demonstrated in investigation 1, using the data sets for Site B. The investigation showed that although there was a limited number of groundwater level measurements, kriging could identify zones of higher and lower heterogeneity and also outline seasonal trends in groundwater flow.

The wider utility of kriging still needs to be evaluated using data sets from other types of contaminated sites, e.g. abandoned mines, land around oil refineries, power stations, gas works, petroleum stations and chemical work, accidental spillage on roads, industrial sites or ministry of defence sites. This would broaden its applicability as a standard part of contaminated land and landfill risk assessments. The Environment Agency has already recommended models for improving site and risk assessment. Examples include the CLEA and the LandSim models, which simulate contaminant migration and estimate risks posed by contaminated landfills. Government agencies should also consider suggesting geostatistical models such as kriging, especially those models that can be integrated with existing GIS and contaminated land software.

10.2.2 Ground Penetrating Radar

The GPR investigations (investigation 2) carried out in this study delineated near-surface contamination and located subsurface hydrogeologic features in landfills with complex geology. The aim of the investigation was to test whether GPR could be used to delineate subsurface landfill features at two landfill sites. The hydrogeological features of interest relate back to the site assessment – aiming to delineate groundwater levels, paths of leachate migration, sand lenses, depth of buried waste and containment wall integrity. The investigation links back to research objective 1 testing relatively new multi-spatial methods that have been applied in other fields of science or that have been tested on other types of contaminated sites. The purpose was to evaluate whether these new methods could be used to improve the accuracy of landfill site assessment.

The results of the GPR investigations show that the instrument can provide excellent results when applied at older and uneven parts of leaching landfill sites. The investigation met the research objectives set out in objective 1 in that the GPR-derived data that was collected at Sites A and C provided qualitative subsurface information about leachate levels, landfill edges, old and unlined landfill cells and hydrogeological conditions around each of the landfill sites. The accuracy of the site assessment conducted at Sites A and C was much improved when the GPR data was integrated with the existing field information. At Site A, it confirmed the presence of a sand lens that acted as a conduit for leachate migration off-site. It also provided depth estimates for buried waste in older parts of the landfill that were previously unknown. At Site C, it confirmed the likelihood of weak spots in the landfill's containment wall as well as verifying the presence of sand lenses. These findings therefore confirmed the utility of a number of ways in which GPR could be used to improve the understanding of small-scale and site-specific landfill risks particularly in terms of providing information about leachate and subsurface conditions at older landfill sites:

(a) landfill conditions - the location and thickness of subsurface features providing historical and geophysical information can assist in landfill management and planning, e.g. identifying buried waste and old access roads, measuring landfill cap thickness, identifying areas of containment wall leakage

(b) contaminant plume conditions – characterising near-surface contaminant conditions, e.g. leachate levels within a landfill cell and the dimensions of near-surface leachate plumes

(c) hydrogeologic conditions - the depth, location and thickness of hydrogeologic features, e.g. sand lenses along landfill edges, clay and gravel layers, local groundwater levels.

The cost-effectiveness of GPR compares well with other near-surface geophysical survey instruments. The cost of hiring a consultant to conduct a GPR survey or hiring a GPR instrument is similar to other comparable survey methods (e.g. seismic refraction, electrical resistivity, gravity, electromagnetic surveys). Selecting the most appropriate instrument is determined by the site-specific landfill conditions and on the survey objective. When compared with other survey methods, GPR is quick and easy to use and is effective in identifying hydrogeological and contaminant plume characteristics (e.g. leachate seepage, high water tables and shallow geology). The GPR can also provide valuable information for model construction. Examples include: (a) identifying subsurface areas of higher conductivity (e.g. locating sand-gravel lenses along the landfill edges that act as leachate migration pathways); (b) more accurately defining head and boundary conditions (e.g. using landfill cell leachate levels); and (c) defining subsurface landfill conditions that might influence groundwater and contaminant flow into the model (e.g. gravel pathways that were used during landfilling operations at unlined parts of a landfill but have been buried and serve as a pathway for off-site leachate migration).

Despite its successful application at two landfill sites in north eastern England, several operational difficulties were experienced due to the design of the instrument and due to the novelty of deployment on landfill sites. In order to address these limitations, future GPR research needs to be focused upon the following:

- Improving the design of a GPR, making it easier to use in rough terrain (e.g. pulse radar sensors that can be placed between boreholes to map heterogeneous contaminant plume extents between two or more points or adding wheels or a trolley to make the instrument mobile).

- Deriving GPR instrument parameter values for landfill conditions (e.g. dielectric constant, electrical conductivity, velocity and attenuation). These input parameters are needed to calibrate the instrument to site-specific, hydrogeological, and landfill conditions. Input values are known for many types of geological materials however applicable landfill values have not been well researched.
- Conducting GPR investigations at other contaminated sites in the UK (e.g. abandoned industrial sites, petroleum stations and chemical works, accidental spillage on roads, and buried dumps). Such applications have been successful in the United States however UK applications need to be investigated further due to the different hydrogeological environment. If the research is successful, the instrument could be suggested for wider use during the site assessment of such lands.

The results of the GPR investigations at Sites A and C suggest that, in landfill site assessments where contamination problems have been known to occur (e.g. leachate-leaking cells, unlined landfill edges, older buried parts of a landfill site, and leaking containment structures), GPR investigations will assist in resolving much more detailed near-surface information. In order to optimise the data retrieval, the surveyed area needs historical hydrogeologic information to calibrate and verify both the instrument parameters prior to data collection and the data collected. For optimum results, field conditions should be dry and the instrument should be set up and calibrated using direct observations of the geological conditions of the subsurface being surveyed.

10.2.3 Remote Sensing

The purpose of the remote sensing investigation was to test the use of field and airborne remote sensing instruments at landfill sites causing near-surface groundwater and soil contamination. The investigation took methods that were applied in previous studies in different environments (e.g. Jago *et al*, 1999) and successfully modified and applied these methods to a contaminating landfill site – Site A. The investigation demonstrated the effectiveness of these methods when used as part of a site assessment in landfill management applications. It links back to the first research objective which aimed to test new multi-spatial field methods that could provide new scales and types of field data. The results fully met research objective 1 in that they confirmed that measuring the 'red edge' of spectral reflection over landfill sites assisted in improving the understanding of landfill-related risks in that it identified patches of leachate accumulation and paths of leachate migration as well as areas off-site that have already been altered or damaged by leachate presence.

(a) Field-based Spectroscopy

Field-based spectroscopy can be used to infer paths of near-surface contamination using patterns of vegetation stress. When comparing the effectiveness of the field-based and airborne instruments, the field-based spectroradiometer provided a new type of data set, significantly different from the standard hydrogeological or biochemical data sets that are routinely collected during landfill site assessments. The instrument measured the spectral reflectance of different surfaces around Site A. It was found to be most effective when integrated with other field data such as that for groundwater, contaminant soil concentrations and aerial photographs. When these data sets were cross-referenced and integrated, paths of leachate migration could be delineated. The field-based method demonstrated distinct advantages over other field techniques, providing immediate results when measuring leachate-stressed conditions around a landfill site.

However two difficulties were experienced when conducting field spectroscopy surveys at Site A. The first was that only experienced research personnel would be able to carry out the field survey and the second was linked to the instrument's design. It is very sensitive and extremely cumbersome making deployment on landfill sites time consuming due to lengthy setting up periods and frequent instrument failures. These limitations severely limit the cost effectiveness of current field spectroscopy surveys. If field spectroradiometers were specifically re-designed to increase their ease of use and robustness then it would be an effective method.

(b) Airborne-based Spectroscopy

This research showed that airborne-based spectroscopy can resolve areas of stressed vegetation caused by leachate contamination. This method is most effective when calibrated using field spectra and integrated with other data sets (e.g. ground water, soil and vegetation quality). The utility of airborne spectroscopy as a cost-effective methodology is inhibited by three main factors. The availability of airborne spectroscopy data is more limited and is much more expensive (approximately 10 times) than standard aerial photography. Secondly the quality of airborne spectroscopic data is not guaranteed as external conditions such as wind speed (causing turbulence during data collection) and cloud cover (altering the spectral reflectance) will seriously effect the data quality. Thirdly, once the data is collected, highly qualified personnel are essential for data processing, data analysis and interpretation. When compared to aerial photography airborne spectroscopy is currently not a cost-effective methodology for assessing landfill contaminant conditions. If further research using airborne instruments could improve on the operational limitations then such data could be used to improve the risk assessment (e.g. mapping vegetation stress along the landfill edges and isolating pathways of leachate migration away from the site).

(c) An Integrated Remote Sensing and Field Data Approach

When field and airborne remote sensing data sets were integrated with other field information within GIS, maps of leachate migration could be produced. These maps were used to calibrate contaminant transport parameters in the Site A model. This approach can provide robust field information needed in the site assessment if:

- the site has evidence of leachate or landfill gas seepage
- large areas around or on the site are vegetated land
- field and airborne reflectance is collected at the same time (on the same day)
- field conditions during data collection are sunny, without strong winds or cloud cover
- GPS readings, along with vegetation and soil samples are taken at the same time as field-based measurements in order to infer levels of contaminated land
- qualified personnel and adequate software and computing capacities are available for data analysis and interpretation.

Integrated data of this type provides real-time information about the extent of a near-surface contaminant plume that is not available and cannot be inferred over regional scales, even when integrating several data sets derived from direct assessment methods. To strengthen the applicability of spectroscopy-integrated assessment, future research should focus upon:

- conducting studies that build on the findings of this research, further verifying whether spectral instruments can effectively map and monitor small-scale vegetation stress under heterogeneous vegetation conditions
- improving the quality of airborne spectral remote sensing data
- developing simpler airborne remote sensing instruments such as aerial videography, that are cost effective and can meet the required resolutions needed for field assessment (providing data resolution of less then 5m and a temporal resolution of one flight every couple of days) and

- developing direct field assessment and monitoring techniques that will record and transmit real-time measurements of groundwater levels and leachate quality.

The success of novel remote sensing methodologies was demonstrated by using field- and airborne-based spectrophotometers that measured the 'red edge' of spectral reflectance around Site A. The investigation confirmed research objective 1 in that 'red edge' remote sensing methods can identify the location and intensity of vegetation stress caused by leachate migration, significantly improving the value of data collected during the site assessment.

10.2.4 Groundwater Flow and Contaminant Transport Modelling

Three modelling investigations were undertaken as part of this research (presented in Chapter 9). They link back to research objectives 3,4, and 5, testing the influence of: data sets available during model construction, modeller assumptions and modelling practises, and how these factors influenced groundwater flow and contaminant transport simulations. The modelling results presented in this thesis met the research objectives in that approximately seventy percent of the tested models in investigations 4,5 and 6 produced higher contaminant concentrations when field measurements and resolutions of the field data used in model construction were increased. The modelling analyses undertaken is a direct verification of the fact that both the modeller and the field data used can significantly influence the outcome of landfill-estimated risk models which are used in landfill site assessments. Investigation 4 addressed research objective 4, focusing on the influence of field data used during model construction. The results show that by conducting a comprehensive site assessment (using both direct and indirect assessment methods), detailed information can be obtained about hydrogeological, landfill and contaminant plume conditions, avoiding many of the field assumptions shown in Table 10.2 (e.g. Assumptions A-G). Different conceptual model and hydrogeological conditions within each model scenario, Investigation 4 produced differing results – depending upon the type and the amount of field data available during model construction. The scenarios were non-unique in that they were based on different hydrogeological assumptions and produced different contaminant plume shapes yet all the scenarios were calibrated and validated using the same data sets. The results provided strong evidence of the fact that field data used in model construction influenced site assumptions that were inherently incorporated into the groundwater flow models. These influenced the way landfill risks were perceived (landfill risks in the models are shown in the form of ammonia concentration modelled through time and with distance from the landfill).

However, the modeller plays a critical role in compiling these data sets and integrating them into a model domain. Modelling investigations 5 and 6 show that there is a need to adhere to good modelling practices when constructing such models. The results show that close attention must be paid to site-specific and model-specific characteristics that can significantly influence model results. Site-specific characteristics include considering the landfill age and waste composition as these factors reflect the type of site-specific contaminant transport mechanisms that may be present (e.g. diffusion, advection etc.) as well as hydrogeological factors that influence groundwater, hydrological and contaminant transport conditions. Model-specific characteristics include a review of model parameters and their appropriate values, verifying whether they reflect site-specific flow conditions.

In general all three modelling investigations found that when more data are used in model construction, the predicted contaminant concentration and risk are higher. This was the case for a significant number of models. In only 30 percent of the case studies was this trend not found. It is therefore evident that the amount of data available will influence the way in which landfill risk is perceived.

Table 10.2 Five out of the eight field and modelling assumptions frequently causing model failure (listed in Table 5.1) were tested in Investigations 1 - 6.

	MODEL ASSUMPTION	FIELD CONDITIONS THAT ARE DIFFICULT TO MEASURE	CATEGORY CAUSING PREDICTIVE FAILURE	RESEARCH OBJECTI VE LINKED TO INVESTIGATIONS 1 - 6
A	Homogeneous layers	Geological layers are often heterogeneous	FIELD DATA CONCEPTUAL MODEL GMP	Objective 1 – New Field Assessment Methods Investigation 2: GPR Provided data about heterogeneous near-surface geological conditions around two landfill sites
B	Isotropic transport in Cells and zones	Transport is often anisotropic but is difficult to measure due to its distributed and heterogeneous nature	MODELLING CODE FIELD DATA CONCEPTUAL MODEL GMP	Objective 1 – New Field Assessment Methods Investigation 2: Use of GPR provided data about leachate flow directions and areas of higher and lower flux, which can be used to estimate areas in a model that have anisotropic transport which influences contaminant transport
C	No dispersion or diffusion occurs other then what is specified as the source	There may be other local sources of dispersion & diffusion	FIELD DATA GMP	Objective 1 – New Field Assessment Methods Investigation 3: Airborne CASI spectra integrated with other field data sets mapped paths of vegetation stress identifying that the landfill was the only local source of contaminant dispersion causing vegetation stress
D	Constant contaminant and hydrogeologic al flow properties	These properties vary depending upon climatic and groundwater flow conditions.	FIELD DATA CONCEPTUAL MODEL GMP	Objective 3, 4 and 5 – Data Used in Model Construction, Modelling Practises and Data Assumptions Investigations 1,2, 3 and 4: The varying groundwater flow conditions were inferred from data sets derived from kriging, GPR and remote sensing instruments. The integrated data sets provided a broader understanding of groundwater and contaminant flow directions. These data sets were integrated and tested in investigation 4
G	Constant hydraulic conductivity across model layers and zones	Hydrogeological and contaminant parameters are difficult to measure because the subsurface is variable and heterogeneous	FIELD DATA CONCEPTUAL MODEL GMP	Objectives 3 and 5 – Data Used in Model Construction and Data Assumptions made during model construction Investigation 2: GPR Provided data about heterogeneous near-surface geological conditions around two landfill sites. This data was used to assign hydraulic conductivity values across the models in investigations 4,5 and 6

Legend:
GMP = good modelling practises

10.3 Optimising Risk Assessment through Reduction of Uncertainty

10.3.1 Research Findings

Based on the results of this research, the following can be said: in order to improve the accuracy of a site assessment and of risk estimation models, there is a need to use both indirect and direct field assessment methods to investigate site-specific landfill conditions. However, several factors need to be considered when selecting the methods of assessment, these are:

(a) the nature of the contaminant plume

(b) site-specific hydrogeologic and geochemical landfill conditions

(c) the availability of landfill site background information

(d) the stakeholders and risks involved

(e) the future planned use for the site

(f) the budget available for site assessment

(g) the amount of time available for site assessment and risk assessment.

The methods tested in investigations 1-3 provided insight into detailed hydrogeological and contaminant flux conditions. All three methods are limited by the fact that they need a large amount of background data to establish good results due to the variability caused by site-specific landfill conditions. The remote sensing applications were also hindered by logistical restraints such as availability of field instruments, inclement weather and operational difficulties with the NERC aircraft.

Despite these limitations, kriging, GPR and remote sensing methods can provide worthwhile information that can assist in both conceptualising and modelling site conditions. This is shown in Table 10.2 where five out of the eight field and modelling assumptions that cause model failure (listed in Table 5.1) were addressed by the methods tested in investigations 1, 2 and 3. These innovative methods can improve modelling and site assessment assumptions related to (a) geological layers, (b) anisotropic transport, (c) local sources of contaminant diffusion and dispersion, (d) groundwater flow and contaminant transport properties and (g) hydraulic conductivity distribution across a model domain. Assumption (D) in Table 10.2 (representing contaminant and hydrogeological flow properties) requires a variety of field methods in order to understand site-specific conditions. This emphasises the need to evaluate whether a given field method is effective as a stand-alone method or as a method whose data set is integrated with other field data sets. Table 10.3 lists the different methods and their effectiveness in conceptualising site conditions and shows that they all provide information that compliments each other. The exception is airborne spectral images, which require field spectra for calibration and which provide the most comprehensive information when integrated with other data sets.

Table 10.3 Evaluating the effectiveness of using different site assessment methods

	1 Boreholes Groundwater Levels Contaminant Concentrations	**2** Kriging of borehole groundwater data	**3** GPR on Landfill	**4** Hand-held spectroradiometer on Landfill	**5** Airborne spectral images of Landfill	**6** Field and Airborne Spectra integrated in GIS
1	■	✓	✓	✓	X	✓
2	✓	■	✓	✓	X	✓
3	✓	✓	■	✓	X	✓
4	✓	✓	✓	■	X	✓
5	X	X	X	✓	■	✓
6	✓	✓	✓	✓	✓	■

Legend:

1 = Borehole and groundwater levels and contaminant concentrations;
2 = Kriging of borehole groundwater data;
3 = GPR applied at problem areas of landfill;
4 = Hand-held spectroradiometer applied at problem area of landfill
5 = Airborne spectral images collected over the entire landfill
6 = Field and airborne spectra integrated and mapped in GIS

✓ The method in the column provides effective site assessment information when integrated with the matching method in the top row

X The method in the column does not provide effective site assessment information when integrated with the matching method in the top row

■ Does not apply

10.3.2 Implications of Addressing Site Assessment Methods and Modelling on Reducing Risk Uncertainty

The results presented in investigations 1-6 have three implications that are linked to the selection of methods, models and assumptions for a landfill risk assessment. Firstly, as demonstrated in modelling investigation 4, the amount of representative field data available during the site assessment and during model construction will significantly influence how accurately landfill conditions are conceptualised and how well a risk-estimation model can simulate site conditions. The long-term implications of incorrectly conceptualising conditions are in most cases, inaccurate and/or inappropriate remedial decisions that will not moderate landfill-related risks in the long run. An example of this is Site C that had several site assessments conducted in the 1990's. However these studies did not calculate correctly the volume of leachate in each landfill cell and geophysical surveys of subsurface *strata* around the landfill were not conducted. The implications are shown in Figure 7.16 in which the leachate containment wall began leaking some five months after it was constructed.

The second implication is that the data collected during the site assessment will influence how the site is conceptualised, how site-specific risks are perceived and how the risk model is constructed. More importantly, it will define assumptions that both the site assessor and the modeller will use throughout the site, risk assessment and model-building process. These site assumptions are inherently added into assessment reports and risk estimation models but are not addressed as uncertainties or assumptions of the landfill risk assessment. Examples were shown in investigations 5 and 6 when setting grid size or hydraulic conductivity values. Both parameters can significantly skew contaminant gradients, depending on the modeller's assumptions when defining the parameter values. By not stating site-specific assumptions that are made during a site assessment and modelling, uncertainties about site conditions are potentially overlooked. This was evident when historical records were reviewed for

Sites A and C. Given that hydrogeological information about both sites was limited, the investigation relied on the findings of previous studies as a guideline to identify sampling locations and possible routes of leachate migration. However, the previous reports did not clarify how hydraulic conductivity values were determined (e.g. number of samples used to derive their values, method used, sampling distribution etc.) or how the two- and three-dimensional geological maps presented in the previous reports were developed (e.g. number of samples used to interpolate subsurface conditions, the assumptions made, the maximum estimated subsurface depth etc.). Initial errors were therefore compounded in later models.

The third implication is that the modeller's knowledge of contaminant hydrogeology, landfilling processes, good modelling practises and familiarity with software are factors which will influence the decisions made during model construction. These can produce different and over-predicting contaminant simulations. This was demonstrated in investigations 5 and 6 where higher leachate concentrations were produced when larger grid and higher conductivity values were used. The assessor's experience and training also significantly influences site assessment findings in that these issues will influence the types of field data that will be collected during the site assessment and how landfill characteristics are interpreted and constructed into a model.

10.3.3 Optimising the Reduction of Uncertainty from Site Assessment through the Use of Multiple Methods

In order to reduce the level of uncertainty from site assessments, multiple field methods should be applied, integrating the various data sets to construct a multi-dimensional conceptual model. Since the modelling investigations in this research showed that both field data and modelling assumptions affected contaminant simulation, several factors (listed in Section 10.3.1) should be considered, in order to improve the quality of data collected during the landfill site assessment. These factors (Figure 10.1) help to determine the type of data that needs to be collected in order to determine site-specific landfill risks (e.g. ammonia concentrations, groundwater levels, damage to near by crops or local streams etc.). They will also assist in identifying the appropriate type of field methods for the site assessment. A wide variety of field assessment methods are becoming increasing cost-effective and more readily available. One aspect that must be considered is the need for different scales of field data. What can be done depends on the amount of time and money available for the landfill assessment. However, by collecting groundwater samples, aerial photographs, vegetation and soil samples as well as historical records and some form of geophysical survey data, an integrated conceptual model of site-specific conditions can be constructed. As was demonstrated in investigations 1,2 and 3, innovative methods such as GPR, remote sensing instruments and kriging can be very successfully applied and integrated in GIS with directly measured data sets (e.g. piezometer measurements of groundwater levels and borehole leachate concentration measurements). Such data also provides invaluable qualitative and quantitative information about site conditions, valuable to the site assessor, the modeller and other stakeholders involved in the landfill risk assessment process.

However, there are several problems associated with an integrated approach. One common problem is that highly trained staff are needed for geophysical and remote sensing data collection and analysis, making such assessment methods more costly. In addition, airborne remote sensing data may not always provide good results due to cloud cover and wind direction during the data collection phase. With geophysical methods, care must be taken when interpreting 2-dimensional and 3-dimensional images constructed using GPR data and only trained professionals should interpret such data sets. When integrating the different data, care must be taken when up- and downscaling the data sets in order to fit them into an integrated GIS model. This proved to be a problem during investigation 3 in which groundwater, leachate, vegetation and soil quality data were used to build GIS-layer maps in which point-samples as well as airborne remote sensing data were up-scaled to represent contaminated conditions across the landfill. In the latter case, the reflectance measurements had pixels with spatial resolutions ranging between $0.25m^2$ and $1m^2$, which had to be up-scaled to fit the GIS-layer maps that had grid sizes of $6.75m^2$ in each direction. Although the investigation produced effective results, data integration was problematic and could only be achieved by a significant amount of post-processing.

In summary, four steps should be taken to reduce uncertainty in the site assessment (shown in Figure 10.1). These are to:

(a) Define the purpose of the site assessment and the site-specific landfill conditions

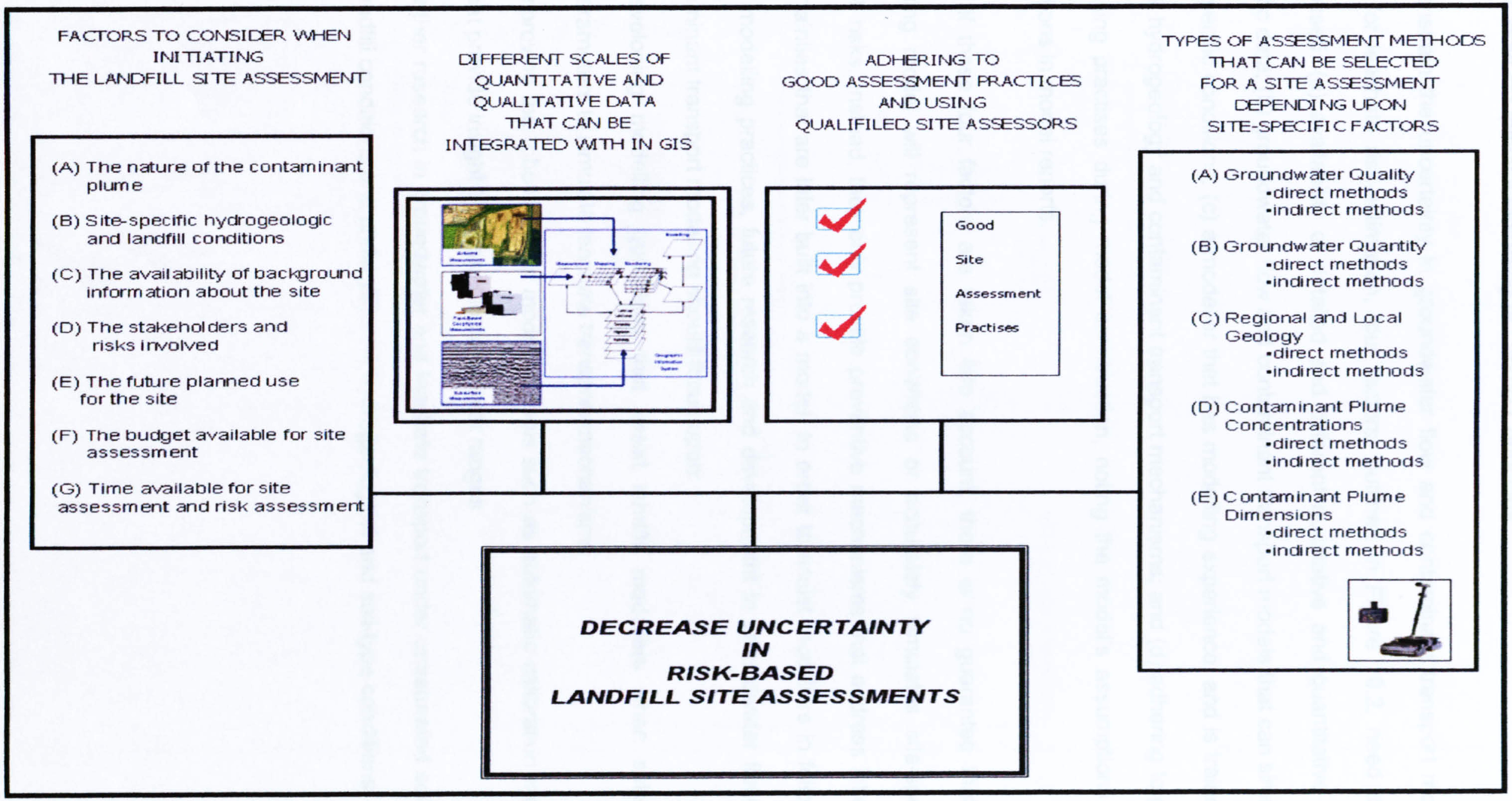
(b) Evaluate whether the methods used for site assessment are effective under site-specific conditions

(c) Use GIS as a tool for data management, and data integration

(d) Ensure that good site assessment practises are adhered to and that qualified professionals are used to conduct the site assessment.

Evaluating the assessment purpose and specific conditions will help to identify appropriate field assessment methods, keeping in mind that several types of data should be collected. By using GIS to integrate qualitative and quantitative data, site-specific landfill risks can be more clearly conceptualised. By verifying that qualified personnel conduct the site assessment, there will be a higher probability that best practises and professional codes of conduct will be respected.

Figure 10.1 Optimising reduction of uncertainty from the Site Assessment



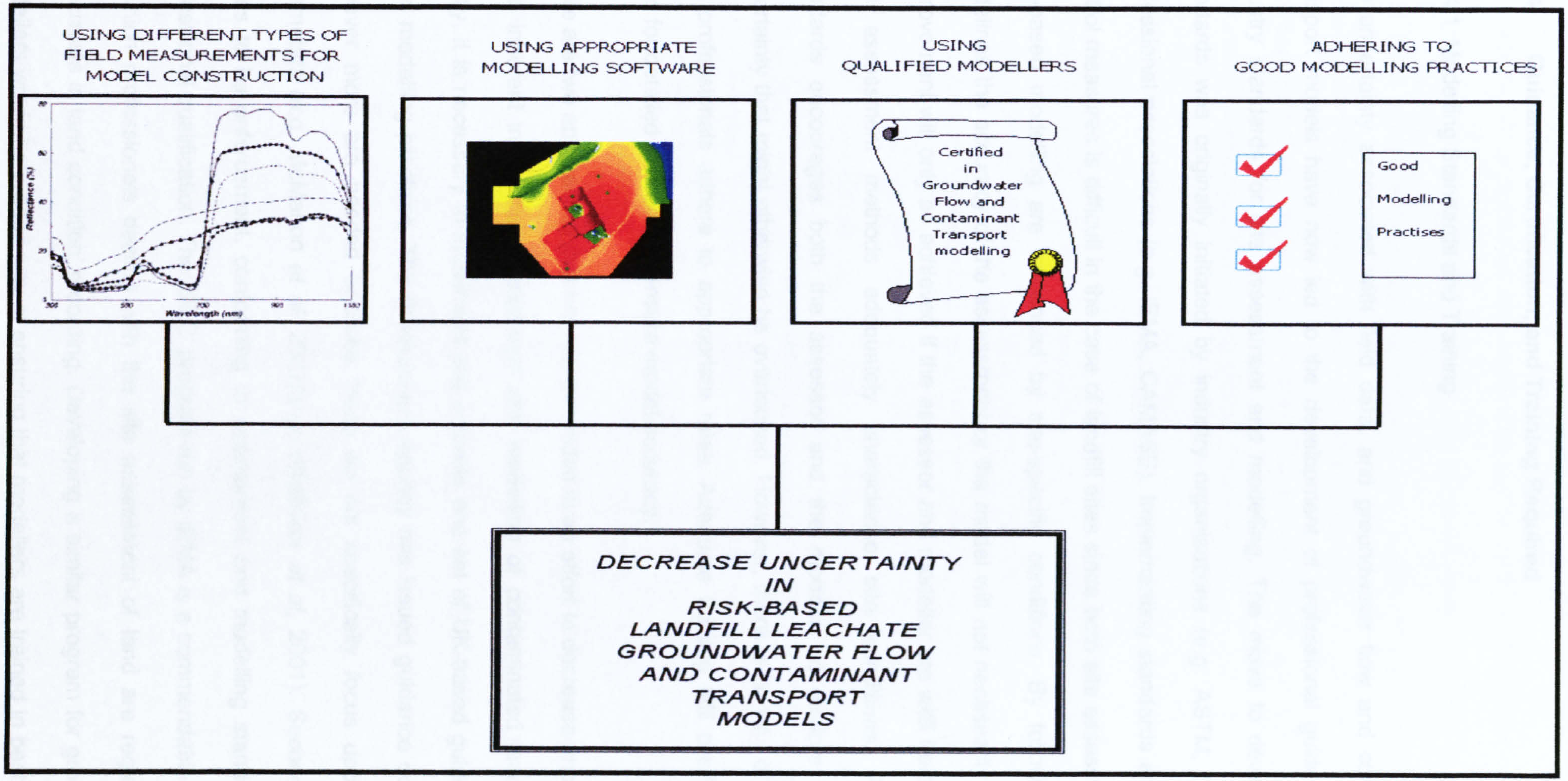
10.3.4 How to Optimise Modelling of Landfill Conditions

To constrain the uncertainty in groundwater flow and contaminant transport models used for leachate risk-estimation, four factors, outlined in Figure 10.2, need to be addressed: (a) availability of detailed and different qualitative and quantitative site-specific data; (b) groundwater flow and contaminant transport models that can simulate site-specific conditions; (c) a modeller that has modelling experience, and is trained in landfill hydrogeology and contaminant transport mechanisms; and (d) adhering to good modelling practises during model construction, noting the model's assumptions and limitations in model reports.

Even if these four factors are taken into account, there is no guarantee that the resulting model will represent site conditions or accurately simulates site-specific landfill risks. Instead, they only provide preventive mechanisms that address inherent uncertainties that are later built into a model. In order to assist modellers in following good modelling practices, future research and development in groundwater flow and contaminant transport modelling should focus upon:

- developing modelling guidelines that assist landfill modellers when selecting parameters to simulate leachate transport mechanisms
- improving the robustness of modelling tools such as automatic calibration models that provide insight into possible parameter ranges
- further research in groundwater and leachate transport under unsaturated soil and landfill conditions and under different hydrogeological and soil-type conditions.

Figure 10.2 How to optimise and reduce uncertainty when modelling landfill conditions



10.4 Guidance, Certification, and Training Required

10.4.1 Modelling Standards and Training

The uncertainty associated with field data, and groundwater flow and contaminant transport models have now led to the development of professional guidelines and industry standards for site assessment and modelling. The move to develop such standards was originally initiated by industry organisations (e.g. ASTM, BSI), and professional associations (e.g. IEMA, CAMASE). Implementing standards and quality control measures is difficult in the case of landfill sites since both site assessment and risk-based modelling are influenced by site-specific conditions. By following such guidelines the accuracy of the assessment or the model will not necessarily improve. Improvement will only be achieved if the assessor and modeller are well trained and if field assessment methods adequately characterise site conditions. Promoting standards encourages both the assessor and the modeller to explore areas of uncertainty that might otherwise be overlooked. However, there is no way of ensuring that professionals adhere to appropriate rules. Adequate training still overlooks the need for detailed field data to ensure model accuracy.

There are two approaches that are recommended in an effort to decrease uncertainties often inherent in the risk assessment and modelling of contaminated landfill sites. Firstly, it is necessary to implement and promote one set of UK-based guidelines for good modelling practices. The Environment Agency has issued guidance documents however more are needed because these do not specifically focus upon landfill parameters (e.g. McMahon *et al,* 2001(a-c); Whittaker *et al,* 2001). Secondly, there needs to be enforcement, conforming to assessment and modelling standards and professional qualification. The 'SiLC' program run by IEMA is a commendable example in which professionals dealing with the site assessment of land are registered as 'specialists in land condition' recording. Developing a similar program for groundwater modellers would be a good way of ensuring that modellers are trained in best practise.

An alternative might be to encourage groundwater modellers to join professional associations that already deal with waste management and contaminated land issues. Organisations such as the IEMA and the Chartered Institution of Waste Management could focus on such modellers as potential new members and consider offering courses that include issues related to contaminant risk modelling. Alternatively, modelling guidelines could be promoted through university and other academic programs in order to reach new generations of modellers.

10.4.2 Certifying and Training Individuals Involved in Landfill Site Assessments

The outcomes of the six investigations carried out in this study suggest that there are at least three ways in which the common errors of field assessment can be avoided. Firstly, professional associations, supported by government agencies and academic institutions, could consider offering industry-focused training courses or certification geared for professionals dealing with site assessment, risks assessment, risk modelling, contaminated land and landfill management. Such courses and certification already exist, however, their scope needs to be expanded. Examples of new topics that should be addressed in such training programs are: assessment methods and tools; good assessment practices; good modelling practices; GIS as effective tools applicable to the risk assessment process; landfill and contaminant hydrogeology and risk communication. If such courses and certification were promoted or supported by government agencies (e.g. Environment Agency, DEFRA etc.), then their legitimacy and importance would become more widely accepted.

Since there is a wide spectrum of government agencies and industry-based groups dealing with contaminated land and landfill sites, a second approach to consider is to promote existing guidance documents dealing with site assessment methods, contaminated land risk assessment and risk-estimation modelling guidelines. An example might be to promote the recommendations of Hooker *et al,* (2000), (sponsored by the British Geological Survey and the Environmental Agency) which outlines and encourages the use of GIS as a tool in managing contaminated land. For publications such as this, it would be useful to organise short training courses geared for local authorities, and waste management professionals. Other examples of useful documents might include the Environment Agency's CLR series (e.g. Table 2.2, DEFRA and EA, 2002(a-d)) and the modelling and risk assessment guidelines (McMahon *et al,* 2001(a-c); Whittaker *et al,* 2001) for contaminated soil and contaminant modelling.

Lastly, communicating risks to stakeholders is an area that has been generally overlooked in the field of contamination and groundwater modelling. Since stakeholders often pay the cost of the assessment and remediation, it is important that they are informed about risks and the decisions that have to be made about remediation. As discussed in Chapter 2, the analytic-deliberative model for risk communication offers positive developments in this area. This is an area that merits further research and additional attention for site assessment.

10.5 Discussion Summary

This chapter has discussed the findings of the research investigations linking them back to the research objectives and then discussing the implications of landfill site assessment methods and landfill modelling when evaluating leachate risks posed by landfill sites. In order to optimise and reduce uncertainty inherent in landfill risk assessments, it is necessary to: (a) use qualified site assessors and modellers, (b) ensure that they adhere to sound assessment and modelling practises, (c) ensure that they select methods and models appropriate for site-specific landfill conditions, (d) ensure that they adhere to professional codes of ethics when communicating the results of their assessments and models and (e) ensure that the use of GPR, Red-Edge remote sensing instruments and kriging are promoted as methods and tools of landfill site assessment.

CHAPTER 11: RESEARCH SUMMARY

The overall aim of this research was to demonstrate the effectiveness of combining several novel methods of landfill assessment in order to grasp a better understanding of site-specific risks to soil and groundwater that may be posed by landfill leachate. The field investigations tested three methods that have been used in other environmental and scientific applications but are not regularly applied in landfill management and site assessment projects. The modelling investigations tested the influence of the modeller and data used during model construction.

Among the novel methods, kriging was used to identify new groundwater sampling locations at two landfill sites. The modelling tool was also used to evaluate data-discrepancies in existing landfill sampling grids. The results are a positive contribution to improving landfill site assessments. The models were able to identify the optimal number of additional sampling points required for site-specific conditions and also provided insight into data-uncertainties in existing data sets that could influence both field assumptions and groundwater flow model results. The geostatistical-modelling tool demonstrated reliability and value for both site assessors and modellers when interpreting field data and analysing field data trends that are needed when assessing and modelling groundwater flow and contaminant transport at landfill sites.

Ground Penetrating Radar (GPR) was applied at two landfill sites that faced problems with near-surface leachate migration. The instrument was able to delineate groundwater levels, sand lenses and depths of buried waste at older and unlined landfill cells. It also showed an ability to verify changes in groundwater levels, evaluate the integrity of landfill containment walls and identify paths of leachate migration. In context of the landfill site assessment, it can provide both the assessor and the modeller with detailed descriptions of the subsurface. The combined data is important when estimating and delineating site-specific hydrogeological parameters such as hydraulic conductivity and depth of differing geological strata when modelling leachate flow. GPR data provides an advanced non-invasive landfill management and site assessment tool that can provide detailed subsurface and landfill information rapidly and cost-effectively.

The third novel method utilised airborne and field-based remote sensing instruments that measure spectral reflectance. The integrated approach provides a means to identify vegetation and soil affected by the presence of landfill leachate. The results of this study are of immediate importance to landfill management and site assessment practises for three reasons. Firstly, this approach provides a mechanism for repeat, low-cost monitoring of landfill sites at a multi spatial resolution. Secondly, information about the location of stressed areas in the vicinity of a landfill can be carried out on a regional scale. Thirdly, by analysing soil and vegetation quality, a link can be made between paths of leachate migration and its impacts on the local environment. The remote sensing data sets also provide the site assessor and the modeller with synoptic and planimetric images of paths of near-surface leachate migration. When constructing both conceptual and mathematical models of a landfill site, this information is valuable for understanding the landfill's characteristics and estimating directions of groundwater and leachate flow.

The investigations conducted as part of this thesis also included three modelling investigations that evaluated the impact of: (a) field data used during model construction; (b) modeller assumptions; and (c) modelling practises applied during model construction. The modelling results are noteworthy in terms of reducing the uncertainty associated with landfill risk assessment. They show that by increasing the amount of field data available for model construction, the hydrogeological conditions in each model also change influencing contaminant concentrations and patterns of contaminant migration through time. In total, seventy percent of the models tested produced higher contaminant concentrations when field data used in model construction was increased and when hydrogeological parameter values (linked to modeller assumptions) were raised. These results provide strong evidence of the fact that field data and modeller assumptions can alter model simulations that are used in risk-based landfill site assessments, and change the way risk is perceived.

The results presented in this thesis provide methods and approaches that can improve landfill site assessment and landfill risk estimation modelling. The main conclusions are a reflection of the five research objectives in that field data that is collected during the site assessment and is used to construct risk-based contaminant flow models require close attention during the landfill site assessment, especially when carrying out risk-based analyses. In summary:

- Kriging and innovative technologies such as GPR and remote sensing are capable of providing the large scales of information needed for the landfill site assessment. They provide detailed hydrological and geochemical information about near-surface conditions that cannot be measured using direct methods. They also provide data sets that give a broader understanding of site conditions, thereby ensuring a robust risk assessments of contaminant sources, pathways, and receptors.
- Modelling practices and assumptions formed during model construction significantly influence risk-estimations and require further attention, especially when conducting risk assessments of contaminating landfill sites. The data and assumptions used during model construction inherently influence how site-specific landfill risks will be perceived. This can have cumulative impacts on long-term remedial decisions that are made based on the site assessment and model findings.
- It is essential that landfill site assessors and risk estimation modellers adhere to good assessment practices and modelling practices. There is also a need to further develop and promote professional codes of conduct and perhaps certification of environmental professionals working in fields relating to contaminated landfill assessment, remediation and risk modelling. These professionals need to be trained in order to advance the quality of estimating landfill risks and evaluating remedial options.

By applying several different and innovative methods such as GPR and remote sensing to collect geophysical information about landfill conditions, by adhering to 'best practise' assessment and modelling practices, using a wide-collection of field data for site assessment and risk model construction, a higher level of quality assurance will be achieved in the risk assessment and remediation of contaminating landfill sites. If the recommendations for further research and improvement presented in this thesis are considered, the inherent uncertainties in both the site assessment and in constructing risk models will be reduced. This will enhance the accuracy of the landfill risk assessment and remedial decisions, and improve the outcome for humans, animals and plants in the vicinity of landfill sites, as well as reduce the level of risk of contaminating soils and groundwater supplies.

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