EB Nationwide Waste Management Research Ltd Wyvern Environmental Trust Ltd

Review of the Performance of Hydraulically Contained Landfills: Research Sites

Final Report

28 November 2005

Entec UK Limited

Report for

EB Nationwide
Waste Management Research Ltd
Wyvern Environmental Trust Ltd

Main Contributors

Ruth Brown
Ben Fretwell
Anna Hall
Chrissy Rey
Nick Rukin
Phil Scott
Alan Stuart
Neil Wells

Issued by

....
Alan Stuart

Approved by

....
Nick Rukin

Entec UK Limited

Canon Court
Abbey Lawn
Abbey Foregate
Shrewsbury SY2 5DE
England
Tel: +44 (0) 1743 342000
Fax: +44 (0) 1743 342010

h:\projects\hm-250\10744 entrust_brogborough landfill\docs\rr106i1.doc

EB Nationwide Waste Management Research Ltd Wyvern Environmental Trust Ltd

Review of the Performance of Hydraulically Contained Landfills: Research Sites

Final Report

28 November 2005

Entec UK Limited

Document Revisions

No	Details	Date

Contents

1. Introduction

1.1 Background

Operation of sub-water table landfill sites on the basis of hydraulic containment requires the level of leachate within the site to be maintained at a lower level than the surrounding water table or piezometric surface. Under such conditions, there is theoretically little or no risk of groundwater contamination, since any water movement will be into, rather than out of, the landfill site.

Comments made by a senior officer of the Environment Agency in a recent High Court hearing of an application for a judicial review suggested that about 200 non-hazardous waste landfills in England and Wales entail some or all of the sites being below the local water table. Approximately 40-50 of these are understood to operate on the principle of hydraulic containment. In some areas, such as the Midlands and East Anglia, sub-water table landfills may account for as many as two thirds of authorised landfill sites. The hydrogeological conditions in the UK mean that the operation of landfills under the principle of hydraulic containment is not uncommon.

The implementation of European Directives into UK legislation during recent years and associated Environment Agency guidance has in some cases referred to such sites. For example, the Environment Agency's Regulatory Guidance Note 6, which provides their interpretation of the engineering requirements of Annex 1 of the Landfill Directive, identifies requirements to assess the need to prevent groundwater from entering the landfilled waste by risk assessment:

- To ensure that the requirements of the Groundwater Directive are met;
- To ensure there are no unacceptable risks to engineering controls (e.g. the lining and leachate control systems) from groundwater entry;
- To determine the degree of risk on a site-specific basis, considering the geotechnical stability of the lining system, wastes and underlying geological strata, the efficacy of the leachate collection system, the effectiveness of any leachate control systems and the ability to maintain leachate and groundwater management in the long term.

The Environment Agency's position statement on the location of landfills (Regulatory Guidance Note 3) also indicates that they would object to landfilling below the water table in any strata where the groundwater provides an important contribution to river flow or other sensitive surface waters.

In guidance published by the Scottish Environment Protection Agency (SEPA), the requirement for measures to prevent groundwater from entering landfilled waste is interpreted to mean that in most circumstances sub-water table landfills will not be permitted. Whilst the landfill could be designed to prevent groundwater ingress, for example by the construction of drainage blankets, SEPA considers that there are likely to be significant sustainability issues associated with such proposals, which would be subject to careful scrutiny.

The above examples illustrate the concern that has been expressed over the use of hydraulic containment as a means to provide long term environmental protection from the potential impacts from landfill. Existing landfill sites of this type have recently come under close scrutiny since they provide the potential for "direct" discharge of contaminants to groundwater, which is prohibited under the terms of the Groundwater Regulations 1998. The current Agency view is that hydraulic containment is likely to be acceptable only in specific circumstances, particularly if active long-term measures to control leachate and groundwater levels are required. They consider that it is likely to be suitable only in low sensitivity locations.

1.2 Previous Work

In March 2001, Entec UK Ltd released a report on the "Review of the Performance of Hydraulically Contained Landfills", a research project carried out over the period 1999 to 2000 and funded by EB Nationwide, through the landfill tax credit scheme. The study made a significant advance in the understanding of the occurrence of such sites, the legislation controlling them, and the key issues likely to be important in their design and operation. The conclusions of the report were based, however, on limited information and stated that there was a relatively short track record for engineered landfills in the UK on which to base these.

Conclusions from the report included:

- The likelihood that there were approximately 40-50 hydraulically contained landfills in the UK, operated in a range of hydrogeological settings, major, minor and non-aquifers;
- There was evidence of groundwater ingress at several of the sites, dependant on the local differential heads and the nature of the lining, or of the natural strata where non-contained;
- There was no evidence of adverse impacts on groundwater quality, suggesting that the hydraulic containment was being effective in providing groundwater protection;
- In certain circumstances, hydraulic containment may be considered preferable to above water table sites, since risks to groundwater may be less (or zero).

During the study it was identified that there was a clear need for research on example sites where the principals and practicalities of hydraulic containment could be investigated over the medium to long term. The studies needed to have the co-operation and input from both landfill operators and the regulators so that all parties would have confidence in the quality of the research and the results produced. This was to become Phase 2 of the study.

Subsequently, proposals (Entec, June 2001) were submitted to landfill operators that had taken part in the Phase 1 studies to identify sites that could be used in this further research. Three waste management companies offered landfill sites that could be used to support the Phase 2 study. These sites were as follows:

- Brogborough Landfill Site, which is located 3 km to the northeast of the M1 and southwest of Bedford, in Bedfordshire. Funding for the research was secured through the landfill tax credit scheme, with E B Nationwide (Shanks First) acting as the environmental body and Shanks (Waste Services Ltd) providing the third party funding;

- Poole Landfill Site, which is situated approximately 9 km south-west of Taunton in Somerset. Funding was provided through the environmental body Wyvern Environmental Trust Ltd, with third party funding provided by Wyvern Waste Services Ltd;
- Whitehead Landfill Site, which is located approximately 4 km east of Leigh in Greater Manchester. Funding was provided through the environmental body Waste Management Research Ltd, with third party funding provided by Viridor Waste Management.

1.3 Objectives

Initial reports (Entec, 2003a, 2003b and 2004) were prepared for each of the sites which assessed their performance in relation to the fact that they were considered to be hydraulically contained. These reports provided the background to the work and the introduction and basis for subsequent studies and further understanding of each site. They provided details of site development and the environmental setting in order to provide a basis for consideration of issues relevant to hydraulic containment.

Specifically, the initial report for each site addressed the following:

- The site history and how it had developed;
- An understanding of the site hydrogeology and geology;
- The site design and operation, and how this relates to hydraulic containment;
- A conceptual model of the site;
- An assessment of the degree of hydraulic containment (both historical and present);
- A leachate/water balance to investigate the degree of hydraulic containment;
- Review of groundwater quality data to determine any impact of the landfill on local groundwater.

The reports were submitted to the site operators for comment and verification of the factual content. This final report has been prepared with the objective of addressing the other key elements of the project proposals (Entec, 2001). For each of the main subject areas, general information is presented and key issues are identified. Where appropriate, these are assessed and illustrated in relation to available site specific information from the three study sites. Subsequent chapters of this report comprise the following:

- A summary of the environmental setting of each site, based on the initial reports; this chapter is included to present background information for each site sufficient to provide a basis for an understanding of the subject areas covered;

- Consideration of waste stabilisation, taking into account waste types, leachate quality and levels in order to assess the likely period that the site will remain "active" and what factors influence this (Task 5 of the project proposals);
- Review of landfill gas generation and management (Task 6). Gas production is a measure of the degree of waste degradation and therefore is also linked with stabilisation of the waste mass. Data for the study sites, including gas composition and method of extraction and/or utilisation are to be compared with published data from other landfills, including above water table sites;
- Review of engineering and operational issues (Task 7). Landfill engineering and environmental control systems are to be considered in relation to their likely long term effectiveness, and in comparison with typical requirements for above water table sites;
- An assessment of the significance of diffusion (Task 8). Even in hydraulically contained sites, there is the potential for contaminants to migrate from the landfill, driven by the concentration gradient which is likely to exist between high concentrations in leachate and much lower concentrations in groundwater outside the site. Whilst this is considered to be more likely to occur in low permeability environments, the potential for this mechanism at each of the sites being studied is considered;
- Project Overview and Conclusions (Task 9). The overall findings and implications of the studies are assessed and presented. Key issues and results associated with the hydraulic containment principle of operation at the three sites studied are described.

1.4 Acknowledgements

Valuable assistance in the form of data provision, background information on site development, and operational practices, has been provided by the landfill operators of each of the sites studied, Wyvern Waste, Viridor and Shanks (now WRG). Their assistance and contribution is gratefully acknowledged. Funding of the project, in addition to the contributions of the operating companies, has been by the environmental bodies EB Nationwide (Shanks First) (now administered by Grantscape), Wyvern Environmental Trust and Waste Management Research Ltd, and their contributions and patience are also gratefully acknowledged.

2. Summary Site Descriptions

2.1 Introduction

The three landfill sites studied in this project represent a range of site development history, environmental settings and landfill engineering. In broad terms, they can be described as follows:

Brogborough Landfill: Wastes have been deposited into low permeability Oxford Clay strata since 1983. Landfill operations and cell construction standards have evolved since that time, with engineered containment of new areas of the site carried out since 1996.

Poole Landfill: Limited engineered containment was carried out prior to waste disposal at this site, which began during the 1960s. The bulk of waste disposal dates from 1974, when Somerset County Council operated the site. The site was originally developed as a quarry to provide raw materials for an adjacent brickworks, extracting Mercia Mudstone strata.

Whitehead Landfill: This site represents the most modern of the three, with waste disposal beginning in 1998. Wastes are deposited into discrete contained cells comprising a composite liner of engineered clay and a geomembrane artificial sealing layer, together with a leachate drainage layer. Local groundwater in the Triassic sandstone is confined beneath glacial drift deposits which underlie the landfill.

The following description for each site presents further details and represents a summary of information presented in the initial reports (Entec, 2003a, 2003b and 2004), in order to present an introduction for subsequent chapters of this report.

2.2 Brogborough Landfill

2.2.1 Site History

Clay extraction for brick-making was carried out at the site for much of the 20th century and was worked most recently, during the period 1956-1981 by the London Brick Company Ltd.

Excavations for clay went to depths of 25-30 m, through weathered Oxford Clay (known as Callow) into unweathered Oxford Clay (known as Knotts). Some areas were back-filled with spoil, reject bricks and other waste materials.

Landfilling at Brogborough began in January 1983, into an initial void of 23 Mm^3 over an area of 120 ha. Infilling was initially carried out by London Brick Landfill Ltd, with the lease to deposit waste purchased by Shanks and McEwan on 21 April 1986. They have continued to operate the site, recently trading as Shanks Waste Services Ltd, up to their recent acquisition by Waste Recycling Group Ltd (WRG). The site was extended on the northwestern side by an additional ~16 ha in July 2001.

The site has received a range of household, commercial and industrial wastes. The waste management licences, for the main landfill area and the western extension, were modified in 2002, with respect to leachate levels. For the main site area, leachate levels were required to be maintained at least 3 m below the lowest natural ground level adjacent to each filled cell. For the extension area, the levels were to be maintained less than 2 m above the liner, unless a risk assessment agreed with the Environment Agency, could demonstrate that higher levels were acceptable.

As part of ongoing discussions with the Environment Agency concerning leachate level control, the required four year update of the groundwater risk assessment (Entec, April 2002) was prepared for Shanks and submitted to the Environment Agency on 18 April 2002 in accordance with Regulation 15 of the Waste Management Licensing Regulations 1994. The report was based on data available up to January 2002 and superseded previous risk assessments. The Environment Agency commented on this report in a letter dated 22 May 2002, **accepting "the concept of hydraulic containment"** and proposing that leachate levels be maintained at 2 m below piezometric levels in the Kellaways Sand for all of the site.

2.2.2 Site Development

The site has been in operation as a landfill site for approximately 20 years (since January 1983), and as a consequence, landfill operation and cell construction standards have varied considerably. The approximate periods of filling and the designs of different areas of the site are shown in Table 2.1. The layout of the Stages and Cells is shown on Figure 2.1.

Table 2.1 Details of Landfill Operations

Stage	Area (ha)	Design	Dates of Waste Disposal[a] Start	End	Capped
1	14.4	Unlined	Feb 1983	Jun 1987	Yes
2	22.6	Unlined	Jun 1987	May 1990	Yes
3A	7.4	Unlined	Early 1990	Aug 1993	Yes
3B	10.2	Unlined	Early 1990	1994	Yes
3X1	11.1	Basal liner and side seal	Dec 1996	Aug 1999	Aug 2002
3X2a	2.5	Basal liner and side seal	Oct 1997	Aug 1999	Not at present
3X2b		Basal liner and side seal	May 1998	Aug 1999	Not at present
3X3		Basal liner and side seal	Aug 1999	Aug 2001	Aug 2002
4A	15.2	Unlined	1990 (1995)[b]	1992 (1996)[b]	Capped
4B	22.5	Basal liner and side seal	Nov 1995	2001	Capped
Cell 5	8.0	Basal liner and side seal	Sept 2001	March 2003	Temporarily Capped
Cell 6	8.6	Basal liner and side seal	April 2002	March 2005	No

Notes:
a: Dates are approximate.
b: Two separate periods of tipping.

Engineered Barriers

Where constructed, basal liners at Brogborough comprise a minimum thickness of 1 m of engineered Oxford Clay (weathered or unweathered), laid in 0.25 m thicknesses, and compacted to achieve a hydraulic conductivity of $<1 \times 10^{-9}$ m/s. Full side seals have been constructed in recent areas of the site. Previously, side seals were used to line the upper slopes, constructed to extend at least 1 m beneath the interface of the weathered and unweathered clay in order to prevent lateral migration of landfill gas.

Natural Barriers

In addition to areas of the site being engineered to control leachate and landfill gas migration, the site also has natural barriers in place. The landfill lies within a void formed by clay reclamation for brick-making. The geology surrounding the void is comprised of Oxford Clay strata which offer some additional protection due to their relatively low permeability (typically $<1 \times 10^{-9}$ m/s). The minimum thickness of in situ clay beneath the site has been determined to be 2 m, based on borehole and pre-landfill survey information.

Leachate Management

In older areas of the Brogborough site, no leachate drainage facilities were installed in the base of the site and leachate has been abstracted from the waste from vertical 'retrofit' wells. Leachate (~6 m^3/day) from Stage 1 was recirculated into Stage 2 wastes for several years, with no leachate removed off site. More recently, there has been less recirculation, and leachate has been removed from the site (Stages 1 and 2) at a rate of about 4 tanker loads per day (80 m^3/d), five days per week. During the period May 2002-March 2003, approximately 10 000 m^3 leachate were removed.

Wells and fin drains constructed in the area of Cell 3X2b and Cell 3X3 in October 2002 have yielded very low volumes, after initially higher yields, suggesting the leachate is probably localised and perched rather than being connected to a larger more continuous volume of leachate.

Three 50 m deep wells installed in Stage 4a/b typically proved dry wastes towards their base. Whilst these recorded a significant depth of leachate after drilling, they remained dry after pumping, suggesting that leachate may be perched at higher levels in the wastes.

Cells 5 and 6 have been developed with a basal leachate drainage system, which comprises a 300 mm drainage blanket (minimum 40 mm dia stone) on a basal slope of 1 in 50. However, leachate volumes generated in these areas are understood to be small and some absorptive capacity is believed to remain.

2.2.3 Environmental Setting

Geology

The site is situated within Oxford Clay strata, which dip gently between southwest and north-northwest. There are no local mapped outcrops of the underlying Kellaways Sands or older strata. The geological sequence for the area is presented in Table 2.2.

The published geological sheet for the area does not indicate any faulting in the vicinity of the site, and no faults were encountered during operation or engineering of the landfill.

There have been numerous boreholes drilled at the site and in the surrounding area, which can be used to determine the site geology. They also allow definition of the base level of the Oxford Clay, which falls from approximately 35 m AOD on the southern edge of Stage 1 to about 27 m AOD on the northern edge of Stage 2, and to approximately 22 m AOD on the western edge of the site.

Table 2.2 Geological Sequence at Brogborough

Formation/Member	Proven Thickness (m)	Description
Superficial Deposit		
Boulder Clay	Variable 5.0 m maximum	To west and southwest of site.
Oxford Clay		
Callow (Weathered)	~5.0 m	Grey/green/yellow, light brown clay with thin laminations and abundant selenite ($CaSO_4$) crystals.
Knotts (Unweathered)	~35 m	Mid green/grey clay thinly laminated clay with scattered calcareous and pyritised shells and localised shell beds.
Kellaways		
Kellaways Sands	4.0 - 5.0 m	Dense tan/grey/green cemented silty clay to slightly clayey sand.
Kellaways Clay	~0.5 - 1.0 m	Dark grey thinly laminated mudstone with shells and pyrite nodules.
Great Oolite Group		
Cornbrash Limestone	~1.5 m	Dense, hard compact light grey to dark grey micritic limestone with abundant shells and evidence of bioturbation.
Blisworth Clay	3.5 m	Dark mauve/black, thinly laminated to bioturbated, compact mudstone/marl containing scarce shells and scattered pyrite.
Blisworth Limestone	~12.0 - 13.0 m	Dense micritic and shelly limestone with thin bands of silty clay and marl.
Upper Estuarine Series	1.0 - 3.0 m	Greenish grey silty and shelly mudstones.

Comparison of levels of the base of the Oxford Clay with pit base contours prior to waste disposal, allows estimates to be made of the thickness of remaining in situ Oxford Clay beneath the wastes. Most recent areas of the site (western extension area Cells 5 and 6) have been excavated deeper than earlier areas and consequently the residual Oxford Clay thickness is least beneath these areas, approximately 2 m. Clay thicknesses, which include some castback material beneath Stages 1 and 2, have been estimated to be between 12 and 31 m.

Hydrogeology

Groundwater Levels

Monitoring records of groundwater levels held by Shanks (WRG) typically date back to 1993. Piezometric levels measured in the Kellaways Sand in the area show a regional gradient from the southwest to the northeast with a gradient of approximately 0.002 and levels in the order of 50-60 m AOD in the vicinity of the landfill. Piezometric levels in boreholes fall on excavation of an adjacent void and then recover with landfilling; stabilising about ten years after landfilling commenced in that cell.

There are fewer monitoring boreholes for the Blisworth Limestone, but the available data indicate that the hydraulic gradient is similar to that for the Kellaways Sands. Piezometric levels are also similar, but can be greater or less than those of the Kellaways Sands, depending on the location.

Groundwater Flow

Groundwater flow beneath the site occurs slowly within the Kellaways Sand, a cemented, silty clay/clayey sand, and at greater depth within the Blisworth Limestones. Groundwater quality in these strata suggest low flow rates and incomplete flushing of connate waters.

The regional hydraulic gradient suggests groundwater flow towards the northeast of the site, although this gradient has been significantly disrupted by the landfill. There is strong evidence of hydraulic gradients inwards towards the site when it was an open pit, and whilst there has been some recovery with adjustment towards the regional gradient, there is still evidence that the site is locally lowering groundwater levels.

2.2.4 Potential for Hydraulic Containment

Areas Likely to be Hydraulically Contained

Based on monitoring data from deep leachate wells and groundwater boreholes, significant parts of the Brogborough Landfill site are currently hydraulically contained. Leachate levels are typically more than 2 m below predicted piezometric levels in these areas. Given that there has been little leachate extraction to date, the degree of hydraulic containment is not due to site management, although PPC Permit conditions are likely to enforce this degree of future hydraulic containment of the waste and leachate.

Areas Less Likely To Be Hydraulically Contained

In Stages 1 and 2, the base of the landfill is elevated compared to more recently engineered cells. In some areas (on the western edge) the base is above the piezometric level in the Kellaways Sand and so hydraulic containment is not possible. In addition, until recent leachate extraction occurred, leachate levels are likely to have been greater than adjacent piezometric levels and so there would have been no hydraulic containment for a time in the past.

Where the waste is thickest in Area 4B, in the central northern part of the site, leachate levels are higher than the piezometric levels in the Kellaways Sand on the adjacent perimeter of the site. It is possible that piezometric levels beneath these areas are higher than at the perimeter due to loading effects, but without confirmation of this, these parts of the site are potentially not hydraulically contained.

Evidence for Groundwater Ingress

Examination of the rates of leachate level rise in several deep wells has shown that the rate of level rise, taking into account different porosities with depth, appears to be related to the degree of hydraulic containment. Furthermore, predicted inflow rates (up to an equivalent rate of 0.08 m^3/m^2/yr or 80 mm/yr) are consistent with measured hydraulic gradients and likely hydraulic conductivities for the in situ and engineered Oxford Clay.

Summary of Groundwater and Leachate Level Trends

From the examination of leachate and groundwater level data, it has been shown that:

- Groundwater levels are predominantly above the base of the landfill and so the landfill is sub-water table;
- Piezometric levels decrease following excavation of a pre-landfill void and increase on lining and landfilling that void. This means the landfill becomes increasingly sub-water table following landfilling;
- Groundwater levels generally decrease with depth. This means that using the water level in the Kellaways Sand to define the sub-water table extent or the degree of hydraulic containment at the base and edges of the waste is conservative as water levels in the Oxford Clay adjacent to the landfilled void are likely to be higher;
- Leachate levels in deep wells are generally below piezometric level indicating that significant parts of the site are hydraulically contained. The exception appears to be where the pit base is relatively high or waste thicknesses are large;
- Rates of leachate level rise and volume increase, and thus most likely rates of groundwater ingress appear to be linked to the degree of hydraulic containment. The estimated rates of groundwater ingress are consistent with the hydraulic gradient between the waste and underlying Kellaways Sand and the likely hydraulic conductivities of the in-situ and engineered Oxford Clay;
- Average rates of leachate volume increase, based on leachate level rather than leachate level rise data also show good correlation with the degree of hydraulic containment.

2.2.5 Water Balance

A simple water balance was carried out for the Brogborough site in the initial report. Estimates of infiltration during landfilling, (i.e. before the wastes are capped), and through the landfill cap, together with periods of filling, liquid waste inputs, and absorptive properties of the wastes, were used in the calculations.

The results of the water balance indicated that the assessment is dominated by uncertainties, particularly in the absorptive capacity of the waste and effective rainfall input elements. Groundwater ingress at the site cannot be confidently identified, but the results obtained are not inconsistent with the rates of groundwater ingress determined by analysis of the rates of leachate level rise.

2.2.6 Impact on Groundwater Quality

Whilst there is some variation in leachate quality within each stage and between each stage of the landfill at Brogborough, the general composition is consistent with that typically seen at landfill sites receiving domestic and industrial wastes (Environment Agency, 2001). Leachate within the site therefore represents a significant contaminant source, with the potential to impact on groundwater.

Local groundwater quality in the Oxford Clay, Kellaways Sand and underlying Blisworth Limestone is poor, containing high concentrations of a number of major ions. Normal background concentrations of ammoniacal nitrogen, potassium and total organic carbon (TOC), parameters typically found at high concentrations in landfill leachate, are elevated.

Data analysis has shown that only one borehole has shown increasing trends in potassium and TOC concentrations that might be attributable to leachate contamination. However, this borehole has been identified as suspect as a result of leakage of perched groundwater to the Kellaways Sand via the well casing.

There is no clear evidence of contamination of groundwaters at the site, apart from that identified in one of the monitoring boreholes (as discussed above). On this basis, hydraulic containment of the waste and leachate appears to be protecting groundwater in the underlying Kellaways Sand at Brogborough.

2.3 Poole Landfill

2.3.1 Site History

Poole Landfill site was originally Poole Brickworks, run by Steetly Tile and Brick Company, which opened prior to 1880. The Mercia Mudstone Formation, which underlies the site, was quarried for use as brick ‘clay’. The site developed from the north (close to the railway line and the brickworks buildings) southwards. The site layout is shown in Figure 2.2. The clay was excavated to about 30 m below ground level to the southernmost point approximately 10 m north of the Billybrook, a small tributary of Hayward’s Water.

During the 1960s, Wellington Urban District Council started to use the north-west corner of the excavated brickpits for the disposal of municipal waste. Somerset County Council took over landfilling at the site in 1974 and the site has been progressively filled with non-hazardous household, commercial and industrial wastes from north to south following the excavation of the brickpits. The site has never accepted liquid wastes. Wyvern Waste formed in 1992, from the waste section of Somerset County Council, and has continued landfilling at the site until recently, when the site was closed.

2.3.2 Site Development

The development of the landfill can be summarised as follows:

- Initial phase of waste disposal was carried out by Wellington Urban District Council during the 1960s. It is assumed that there was no preparatory engineering carried out. There are no records of the types of waste disposed, volumes, water levels or leachate levels for this period. There are also no records of drainage at the site.

- Somerset County Council took over the site in 1974, and development began in a phased way. The site was divided into 5 phases, with the oldest phase in the north, and most recent phase in the south of the site (Figure 2.2). Phase 2 was constructed on top of the old municipal site used by Wellington Urban District Council.

- Landfilling has recently been completed in Phases 4 and 5. The fifth phase is a small area to the south-west of the site and overlap with waste from Phase 4 onto Phase 3 to provide the correct profile for site final restoration.

Late stage landfilling at the site has been designed to provide appropriate final restoration contours and emplacement of the final clay cap, involving overtipping of waste in the southern portions of Phases 1 to 3 onto the temporary clay cap which has been excavated to a minimum 150 mm thickness.

Natural and Engineered Barriers

The Poole landfill site has some natural and engineered barriers in place. The landfill lies within a void formed by mineral extraction for brick manufacture. The local geology comprises Mercia Mudstone strata which offer some degree of protection to local water resources due to its relatively low permeability. Engineered containment at Poole includes:

- A vertical bentonite slurry trench keyed into the Mercia Mudstone to depths between 1 m and 4 m, which incorporates a central HDPE geomembrane, around the north-western perimeter of the site; this is primarily for landfill gas control;

- Re-compacted clay edge seals around the northeastern and southeastern external perimeters of Phase 4; there was no engineered basal lining, but a drainage blanket was put in place for leachate drainage;

- A welded HDPE membrane covered with 0.75 m clay along the south-west perimeter of Phase 4, to protect an adjacent property from landfill gas migration.

There is no containment for Phases 1-3. According to the Pollution Control Action Plan for the site, capping of the wastes incorporates a jointed LDPE flexible membrane overlying a drainage layer comprising compost and stone drains. Clay is placed on top of the membrane, together with layers of soils to provide a restoration profile suitable for agricultural use.

During the most recent years of operation, waste disposal and restoration capping were carried out over different areas of the site to achieve final contours. Final capping was carried out first on the northern and eastern margins of the site. Waste input continued into the central area of Phase 4 and southern part of Phases 1-3 during 2002 and 2003. In 2004, waste disposal was limited to the western area of Phase 4 and the southwestern area of Phases 1-3. At this time, the eastern half of Phase 4 had been capped and much of Phases 1-3 was in the process of being capped.

Information for the site indicates that the base of the excavations at Poole prior to landfilling may have been irregular. Estimates of the level of the base of the landfill indicate the levels to be in the range 25-35 m AOD, deeper in the central part of Phase 4.

Leachate Drainage, Collection and Treatment

Leachate drainage at Poole landfill is as follows:

- The initial landfilling area (Phase 1) included construction of a basal drainage system together with a pumping station (known as the vertical pump). During the development of Phase 1, several springs were encountered but in general the pit sides were dry. The springs were piped to a sump in the west and a pump was installed.

- During the construction of Phase 2, the existing leachate collection system was extended and pumped from the vertical pump. The basal drainage blanket also had an outfall into Phase 3. A rubble drainage tower was installed in the fill.
- Drainage in Phase 3 was provided by the extension of the existing system in Phase 1. The pit sides were recorded as being generally dry. Vertical drainage towers were also installed in the fill. Groundwater and leachate level monitoring data are available from 1990 onwards. No data are available prior to this time.
- Phase 4 was developed with vertical drainage towers in the fill, with herringbone drains to a basal pumping sump on the eastern side where 2 pipes were constructed up the sides of the pit with each containing a borehole pump (these are known as the inclined pumps).

Leachate Management

Wyvern Waste manage the leachate produced at Poole by pumping it from Phases 1-3 (the vertical pump) and from Phase 4 (the inclined pump) to the on-site treatment plant prior to disposal to sewer. They have provided records of site rainfall, waste inputs and volumes of leachate abstracted and pumped to sewer. During the period April 1996-April 2000, daily volumes of leachate removed from the landfill site were approximately 200 m^3 and 115 m^3 from Phase 4 and Phases 1-3 respectively.

Each of the pumping systems conveys leachate to the site leachate treatment plant, from where the effluent is discharged by gravity to the public sewer to the north of the site. The discharge is carried out under the terms of a Trade Effluent Consent.

2.3.3 Environmental Setting

Geology

The site is underlain by Triassic Mercia Mudstone strata (formerly known as Upper Keuper Marls). These typically comprise silty mudstones with occasional sandstone and gypsum bands. To the north and west of the site, the underlying Otter Sandstone (Sherwood Sandstone) strata outcrop. These dip to the northeast, beneath the site and have been found at depth beneath the Mercia Mudstone. There is also some alluvium present to the north and south of the site, associated with the valleys of the River Tone and the Billybrook respectively. The published geological sheet for the area shows no evidence of faulting at the Poole site.

There have been several phases of site investigation at the site, with numerous boreholes installed, several of which are retained and used for monitoring. A summary of the site geology is given in Table 2.3.

Table 2.3 Poole Landfill - Summary of Site Geology

Name	Presence and Description From On-Site Boreholes
Made Ground	The made ground consists of bricks, ash and sand and clay gravel and is mainly found in the north of the site.
Mercia Mudstone	Red mudstone/siltstone and clays, (termed 'marls') with some thin tea green sandstone bands, The clay, siltstone and mudstone were used for brick making. Thickness was proven to 28 m with a borehole drilled up-dip of the landfill site. It has been calculated that the landfill site at its deepest part has 24 m of Mercia Mudstone beneath the base and therefore the waste lies within the Mercia Mudstone. The transition zone (see below) however may be present at some boreholes due to a suggested fault.
Transition Zone	A transition zone was encountered when a borehole was drilled in 1964 to the west of the site (up dip) and proved 28 m of Mercia Mudstone ('marl and clay'). The base was noted to be predominantly sandy and may mark the transition to the underlying Otter Sandstone. The borehole logs from BH81 and BH1, on the eastern margin of the site (Figure 2.2), indicate a higher proportion of sandstone to the other boreholes and there is a presence of conglomerate in BH81. This suggests that there may be a fault between the predominantly siltstone/marl geology and the sandstone found at a similar elevation.

Hydrogeology

The landfill site has been developed in predominantly low permeability Mercia Mudstone deposits. These contain more permeable sandstone and siltstone horizons. The Environment Agency’s Groundwater Vulnerability map which covers this area, Sheet 42, Somerset Coast (Environment Agency, 1996) designates these strata as Non Aquifers, of negligible permeability. Beneath the Mercia Mudstones lies the Otter Sandstone (Sherwood Sandstone Group), which is classified as a Major Aquifer.

There is evidence for artesian groundwater conditions in the vicinity of the landfill. Boreholes located on the southern and eastern margins of the landfill, and three public supply boreholes within 1.5 km of the site have been identified as artesian. They are likely to intersect high flow zones at depth within the Mercia Mudstone or the underlying transition zone to the Otter Sandstone, confined by the low permeability mudstone strata.

Numerous springs were also identified during development of the site. Two springs were present in the north-east corner of the site prior to the development of the site as a landfill and were used for the water supply to the brickworks. This supply was replaced in 1977 by a borehole in the adjacent field.

In 1977, when the site was a void and thus acted as a sink for groundwater, numerous springs and seepages were noted from the face of the excavation, typically at levels between 36 and 38 m AOD.

Two further springs were documented during excavations in 1976 to the west of Phase 1. The development of the southern part of the landfill during the 1990s gave rise to three documented springs at the south-eastern edge and the southern edge of Phase 4. The springs to the southeast of the phase caused some softening and slumping of the brick pit. This was resolved by placing a thickness of clay over the seepage and the inflow drained to the sump in the phase and pumped out prior to landfilling.

These seepages are likely to coincide with more permeable layers in the strata, the seepage in the southeast of the site was noted to have originated from a water bearing sandstone which was encountered during the construction of Phase 4.

There are three further springs adjacent to Billybrook House to the south west of the site. The first exhibits artesian conditions and flows throughout the year, the second has been sealed, and the third discharges into a settlement pond which has been used in the summer for dust control (Wyvern Waste, 2001).

There is therefore much evidence of groundwater ingress to the site prior to landfilling.

2.3.4 Potential for Hydraulic Containment

Monitoring Infrastructure - Groundwater

There are 62 groundwater monitoring boreholes at the site that have been routinely monitored for water level and quality. Monitoring records held by Wyvern are from 1990 to 2002 and therefore represent a limited time period considering that landfilling commenced in a phased way from 1974. Groundwater level data prior to 1990 are known however from the brickworks abstraction boreholes. Many of the groundwater monitoring wells have an unknown construction, of the 62 monitoring boreholes, logs are available for only 13. The logs show that the boreholes are screened through 'weak marl', 'weathered red brown silty sandstone', 'sandy siltstone' and 'silty clay'. Using these descriptions, it is not possible to distinguish specific horizons within the Mercia Mudstone and therefore no conclusions can be drawn as to the variation in head and properties in these different horizons.

Groundwater Monitoring Data

Groundwater levels at the site range between about 35 m AOD (on occasions in BH59, to the northwest) and 56 m AOD (BH89, to the southwest), with the majority of water levels between 40 m AOD and 50 m AOD. Some boreholes at the site also show artesian conditions (BH18 (near northwestern boundary), 72, 76, 78, 79 82 (all near southern or southeastern boundaries) and 88 (near western boundary)).

A decrease in groundwater levels was observed from 1993 to 1995 in a number of the boreholes around the southern margin of the site, particularly in those marked 100 to 104 (Figure 2.3). The levels recorded in these boreholes are consistent with estimated levels of the base of quarry during landfill development. These water levels have been increasing since 1995, sharply at first and more recently at a slower rate. The timing of this decrease and subsequent increase corresponds with the construction of Phase 4 in 1995 and the subsequent landfilling with waste. BH72, an artesian borehole, seems unaffected by these activities, possibly as a consequence of this borehole being deep (40 m) and obtaining water from a deeper horizon in the Mercia Mudstone strata which is not in hydraulic connection with strata excavated during construction of Phase 4.

More recent data have shown generally consistent water level behaviour in this area of the site, with seasonal fluctuations of approximately 2 m and levels typically in the range 44-46 m AOD. The levels recorded in this area of the site are generally the lowest recorded at Poole. The data therefore suggest a clear link between the observed falls in groundwater levels due to excavation and dewatering, and a recovery in levels associated with landfill development and restricted groundwater ingress. Recent groundwater levels are approximately 14-16 m above the base of the landfill.

Boreholes around the northern margin of the site, adjacent to the earliest landfilled areas, do not show the decline and recovery in levels observed in the southern boreholes. They generally show a limited range of seasonal fluctuations with most showing fluctuations of less than a metre. The main exception in this area of the site is borehole BH59, which has recorded erratic levels, over a range of more than 10 m. Recent groundwater levels are approximately 15-20 m above the base of the landfill in the northern part of the site.

Hydrographs for boreholes along the eastern margin of the site indicate fluctuations in groundwater levels of typically 2-4 m, but no evidence of any long term trends. Boreholes on the western flank of the site show levels in the range 48-56 m AOD. This range is likely to reflect the screening of the boreholes in different horizons within the Mercia Mudstone. Some of the boreholes appear to show an influence from development of Phase 4 of the site during the period 1993-1995. Borehole BH65, which is to the west of Phase 4, recorded a fall in groundwater level of about 6 m, and a subsequent recovery from 1995. The level in this borehole in late 2002 was similar to that measured in early 1991. Infrequent rounds of water level measurement result in the data having peaks rather than smooth seasonal fluctuations.

Leachate Levels

The initial report on Poole reported that leachate levels at the site varied between about 42 m AOD and 59 m AOD over the period of record available, from 1990 to 2002. In general, leachate levels were increasing, although the evidence was inconclusive, due to the limited data and 6-year data gap between 1992 and 1998. Some of the monitoring wells show large fluctuations in leachate levels, GW07 shows a fluctuation of 7.7 m and GW02 of 2.8 m from December 1990 to June 1991. Leachate levels generally decline from central areas of Phases 1 to 3 to the edges of the site. The highest levels measured are in the order of 58 m AOD. Assuming a level for the base of the landfill of about 30 m AOD, this represents a saturated waste thickness of 28 m. The lowest leachate levels have been recorded in wells GW08 and GW09 on the northwestern edge of the landfill. This is near to the location of the pump in this phase and therefore indicates that pumping is reducing leachate levels in this part of the site.

The initial report indicated that there were limited leachate level data for the southern part of the landfill, Phase 4, but that levels of approximately 40-42 m AOD were lower than adjacent groundwater levels. In late 2004, Wyvern Waste installed additional leachate wells (LW21-LW29) across the landfill. These were installed in part to investigate the validity of existing leachate level data and the possibility that the levels measured represented perched leachate. Levels measured in these wells in September 2004 have confirmed the high leachate levels measured previously in the northern area of the site (Phases 1-3). They have also indicated that high levels (>50 m AOD) extend into the southern Phase 4 area, before falling south-eastwards towards the site boundary. The levels measured in the southernmost of the new leachate wells support the previous understanding, that these are below groundwater levels measured in boreholes around the southern margin of the site.

Leachate Quality

Leachate analyses at Poole are principally from the main pumped discharges from each area of the site. They show that the leachate in Phases 1-3 typically contains ammonia concentrations lower than would be expected from an ‘average’ above water table landfill site. Concentrations in more recent areas of the landfill (Phase 4) are however much higher than those in Phases 1-3 and the LandSim v2.5 “most likely” concentration.

Ammonia concentrations measured in the inclined pump (in Phase 4) show considerable variation, which may be linked to seasonality, and range from a minimum of 87 mg/l in late 2000 to 1500 mg/l in September 2001. Concentrations may increase during the summer period and decline during the winter months.

The ammonia concentrations of leachate pumped from the vertical pump (in Phases 1-3) are low, typically in the range 50 mg/l to 90 mg/l, with occasionally elevated concentrations and an isolated maximum of 560 mg/l. The lower leachate ammonia concentrations in this area of the site are possibly due to the age of the waste (waste disposal commenced in 1974) which has allowed the majority of the contaminants to be flushed. A contributory factor for this low concentration may also be dilution by groundwater ingress. The only samples taken of leachate (until recently) are from either the inclined or vertical pumps and there have been no data available for individual leachate wells. There is little data available for chloride with only 2 data points for the vertical pump and four for the inclined pump. The inclined pump (Phase 4) ranges from 4380 mg/l to 8180 mg/l and the vertical pump (Phases 1-3) ranges from 67 mg/l to 2220 mg/l, again reflecting the weaker leachate quality in the older areas of the site.

The leachate wells installed during 2004 have been sampled in order to measure the quality of the leachate within the body of the wastes. Leachate was generally weaker in the wells in the older areas (Phases 1-3) of the site compared with the southern area (Phase 4). Ammonia and chloride concentrations were in the ranges 330-1400 mg/l and 320-3600 mg/l, and 1200-3400 mg/l and 1300-2600 mg/l for Phases 1-3 and Phase 4 respectively. Whilst there is little information with respect to chloride concentrations available for comparison, the ammonia data from the recent investigations suggest that leachate quality within the waste mass is stronger than indicated by the pumped discharges from the site. Consequently, this supports the hypothesis that a large proportion of the leachate removed from the site is diluted by groundwater ingress and that there are likely to be pathways that transport large quantities of groundwater to the discharge points, by-passing the bulk of the waste mass. Further discussion of leachate quality in relation to waste stabilisation at Poole is included in Chapter 3.

Degree of Hydraulic Containment

The measurement of leachate and groundwater levels at Poole has allowed consideration of the degree of hydraulic containment. Leachate levels are controlled to some extent by pumped extraction, but are elevated in the older areas of the landfill. These northern areas of the landfill, Phases 1-3, have recorded leachate levels in excess of 55 m AOD. Whilst these levels decline towards the margin of these areas, they remain above adjacent groundwater levels measured in boreholes around this part of the site, particularly along the northwestern edge. Consequently, there is unlikely to be hydraulic containment in this area of the site.

Groundwater ingress (springs) noted into the void now occupied by Phases 1-3 indicates that on initial landfilling, the wastes would have been hydraulically contained. Leachate and groundwater level monitoring data suggest however that although a significant depth of waste lies below the water table, it has not been hydraulically contained for several years.

There are fewer leachate level measurements for the southern area of the landfill, Phase 4. However, recent measurements in wells LW01 and LW03 and in the recently installed wells indicate levels in excess of 50 m AOD in the northern part of Phase 4, falling to approximately 40-45 m AOD around the edge of the landfill. These levels are lower than groundwater levels measured around the margin of the landfill, particularly along the northeastern and southwestern margins of Phase 4. With leachate levels maintained at about those measured, and adjacent groundwater levels indicating limited seasonal variations, Phase 4 can be considered to be at least in part hydraulically contained.

2.3.5 Water Balance

The initial report presented a water balance for the site. A detailed water balance for the period April 1996-2000 was carried out by Wyvern Waste, and took into account waste and rainfall inputs, estimates of active and restored areas of the site and of the absorptive capacity of the wastes.

The leachate balance showed that the predicted volume of leachate generated for Phases 1-3, which is based on the quantity of infiltration through the cap during the period studied, is much lower than the actual leachate volume extracted. Predicted volumes average approximately 100 m^3/week, with the maximum predicted to be in excess of 1000 m^3/week. This compares with much higher measured volumes of leachate abstracted averaging 115 m^3/**day** (approximately 800 m^3/week). Whilst this difference could be as a result of several factors, the most likely explanation for the difference between predicted volumes of leachate generation and quantities pumped from Phases 1-3 was considered to be that significant groundwater ingress is occurring into these areas of the landfill. On average, the volume of leachate pumped from the landfill exceeds the predicted volume by about 700 m^3 per week, which is approximately seven times the predicted volume of leachate generated by infiltration alone.

The leachate balance for Phase 4 showed predicted leachate generation volumes generally lower than actual volumes pumped except occasionally at times of very high rainfall. Actual volumes pumped and estimated volumes of leachate generated were higher than for Phases 1-3. Larger predicted volumes of leachate generation reflect the higher infiltration estimated over this non-capped area of the site. Weekly predicted leachate generation as a consequence of infiltration was estimated at approximately 600 m^3. Actual volumes pumped from Phase 4 during the period were approximately 80% higher than from Phases 1-3, with about 300 000 m^3 pumped during the 4 year period studied (April 1996-April 2000), equivalent to approximately 1430 m^3/week.

The water balance for Phases 1-3 clearly showed that predicted volumes of leachate generated were significantly below those abstracted, however this was not as clear in Phase 4, where during some periods, predicted volumes of leachate generation were higher than leachate abstraction volumes.

Over the study period, the difference between predicted leachate volumes and pumped leachate volumes is large. There is no evidence for any significant decline in leachate levels in this area of the site, and since the area has been modelled without any low permeability cap, and there are no major inputs to the site from surface waters, it was concluded that the additional volumes pumped are derived from groundwater ingress.

Therefore based on measured volumes of leachate pumped from the landfill and estimates of leachate generation as a result of infiltration, there is evidence of groundwater inflows rather than leachate leakage in both main areas of the site. This is the case both where piezometric levels are in excess of leachate levels, as in Phase 4, and where levels are reversed as in Phase 1-3, where leachate levels are higher than groundwater levels and suggest the potential for leachate leakage out through the base or lateral movement within the waste.

2.3.6 Impact on Groundwater Quality

Overall, whilst the monitoring carried out at Poole has been inconsistent, monitoring data indicate that the landfill is having a minimal impact on local groundwater quality. Elevated concentrations of chloride, a key parameter typically elevated in landfill leachate, have been identified in some of the monitoring boreholes, along the western and northern edges of the site. Concentrations of 30-60 mg/l have been measured, compared with local background values of approximately 20 mg/l. These boreholes are located around the margin of Phases 1-3 of the site. Leachate levels recorded in this area of the site indicate the potential for leachate migration from the site, but leachate pumping and engineered containment appear to be successful in preventing off-site contamination.

There are very limited data for ammonia concentrations in groundwater, despite this being one of the key contaminants in landfill leachates. Slightly elevated concentrations (maximum 1 mg/l) have been identified in a limited number of boreholes only, and do not suggest any significant contamination of groundwater.

In conclusion, there is some evidence of elevated chloride concentrations around Phases 1-3, where there is no hydraulic containment; but there is no evidence of impact on groundwater by chloride around Phase 4, which is at least in part hydraulically contained.

2.4 Whitehead Landfill

2.4.1 Site History

The site was used for tipping of colliery waste between 1912 and 1968, serving a number of deep mines in the local area. During the period 1979-1986, coal was recovered from the spoil by washing, and settlement lagoons were constructed. As part of planning permission for this activity to take place, Wigan Metropolitan Council required the tip to be reformed and restored. This was not completed and much of the spoil area remained unvegetated at that time.

Terry Adams Ltd submitted a planning application in January 1996 for the ‘winning and working’ of the clay at the site and infilling with domestic, commercial and industrial wastes. The plan included reclamation of the old spoil site for amenity, tree planting and agriculture. A number of alterations were made to this original submission, in response to concerns voiced by the Environment Agency, including raising of the base level of the proposed excavation so that the underlying Sherwood Sandstone would not have to be temporarily dewatered. Due to the presence of sands and gravels in the drift deposits, the proposals were revised, with base levels raised further, to at least 3 m above the top of the permeable sands and gravels that had been identified, such that extraction would be limited to the low permeability clay.

Planning permission was granted to Terry Adams Ltd in 1997 and the site passed to Viridor in 2000.

2.4.2 Site Development

Landfilling of waste other than colliery spoil has been undertaken on the site in a phased manner since 1998, with waste deposited within discrete cells. The site layout is shown on Figure 2.4. The design of the cells and the ‘landfill containment system’ is discussed in Wardell Armstrong (1998). The proposed containment at that time was to comprise:

- Underdrainage system (where necessary);
- 1 m thick, engineered clay liner with a maximum permeability of 1 x 10^{-9} m/s;
- 2 mm HDPE geomembrane liner;
- 300 mm leachate drainage blanket, incorporating 180 mm diameter drains;
- Engineered landfill cap (either a synthetic liner or engineered clay layer).

According to Wardell Armstrong, the site was to be divided into 5 containment cells, the base elevations of which lie between 9 and 15 m AOD. The Environmental Monitoring Location Plan (Ref WAS3000/JUN'03) supplied by Viridor indicated that the site was being divided into 6 stages (phases), with landfilling initially progressing from the north of the site along the eastern side. At the end of 2004, Viridor were infilling Cells 4C and 5 of the site, with Cell 6 being constructed. Stages 1-4 had been completed to their final levels, to about 34 m AOD, and were capped or ready to be capped. Approximate stages of development of the site since 1998 are as follows:

- Stage 1 (cells 1A, 1B) - constructed in 1998, waste input began in October 1998 - now restored;
- Stage 2 (cells 2A, 2B, 2C) - constructed in 1999, waste input began towards the end of 1999 - now restored;
- Stage 3 (cells 3A, 3B) - constructed in 2000, waste input began towards the end of 2000 - now restored;
- Stage 4 (cells 4A, 4B) - constructed in 2001, waste input began towards the end of 2001 - now restored;
- Stage 6 (cell 4C) -constructed in 2002/2003, operational at end of 2004;
- Stage 5 (cell 5) - constructed in 2002/2003, operational at end of 2004;
- Stage 5 (cell 6) - under construction at end of 2004.

(Note: Stage and cell designations are inconsistent in comparison with the originally proposed site layout.)

Natural and Engineered Barriers

The site has natural as well as engineered low permeability barriers in place. The landfill lies within a void formed by clay extraction. The geology surrounding the void is comprised of Boulder Clay which offers some degree of protection due to its relatively low permeability. Sand and gravel horizons have been identified within the Boulder Clay. The minimum thickness of in situ clay beneath the site was determined by Wardell Armstrong to be 4.5 m, based on borehole information. In order to take into account concerns of the Environment Agency, the proposed base of the excavation was raised in order to remove the requirement to temporarily dewater the underlying Sherwood Sandstone aquifer and to ensure that the excavations were solely within clay, rather than sand and gravel, deposits.

The wastes are further contained by a composite liner comprising an engineered 1 m thick clay layer (maximum permeability of 1×10^{-9} m/s) and a 2 mm thick high density polyethylene (HDPE) geomembrane placed over the base and side slopes.

Leachate Management

The base of each stage of the Whitehead landfill is covered with a leachate drainage blanket and drain system. The drainage comprises 180 mm diameter drains in a 300 mm thick stone blanket. These drains lead to leachate pumping sidewall risers for abstraction.

Leachate is abstracted and then discharged into a storage lagoon, from where it is piped to a treatment plant. The treated leachate is again stored in a lagoon prior to being discharged under consent to the local foul sewer. Viridor maintain records of weekly rainfall, input volumes of sludges and liquids, and the volumes of leachate extracted and discharged to sewer. During 2002, the volumes discharged averaged approximately 130 m^3/day. Assuming that at this time Stages 1 - 4 were active or had been filled (estimated total area 16 hectares), this volume of leachate equates to an average infiltration over that area of approximately 300 mm/yr.

More recent data provided by Viridor indicate that volumes of leachate pumped to sewer during 2003 were in the order of 100 m^3/day. (This figure is an estimate since the recording flow meter was out of action for part of the year.) Flow records for the period January-September 2004 indicate that the average volume pumped to sewer was approximately 70 m^3/day. This was extracted from cells 2, 3, 4B, 4CN and 5. In addition to the quantities of leachate treated and disposed to sewer, large quantities are recirculated within the wastes. During the above period in 2004, the quantity of leachate recirculated was, on average, approximately 250 m^3/day.

2.4.3 Environmental Setting

Geology

The site is underlain by glacial drift deposits which overlie Triassic Sherwood Sandstone. The sandstone dips to the south and is underlain at depth by the Carboniferous Coal Measures. Due to the dip of the strata, the Coal Measures outcrop some 20 km to the north of the site. There are a number of faults mapped in the area, the most significant being the Bridge Fault, which is recorded to bisect the site.

Wardell Armstrong (1998) summarised the geology, based on site investigation data, as shown in Table 2.4.

Table 2.4 Generalised Geological Sequence at Whitehead (Wardell Armstrong, 1998)

Material	Observed Thickness (m)
Clay	5 - 17
Sand and gravel[1] (layer 1)	0 - 9
Clay	3 - 12
Sand and gravel[1] (layer 2)	0 - 11
Clay	3 - 17
Sand and gravel[1] (layer 3)	0 - 11
Sandstone (rockhead)	

[1] Sand and gravel layers are discontinuous and are not observed in the southern part of the site.

The sands and gravels are variable in nature and range between silty clayey fine sands and silty sandy medium to coarse rounded gravel with occasional cobbles.

Site investigations have recorded up to 12 m of colliery spoil over the western part of the site. Over the rest of the site, the spoil is thinner, with depths ranging between 0 and 2 m. The spoil was disposed of between 1912 and 1976 and has been more recently processed by washing to recover any residual coal. The spoil is described as comprising dark grey clayey siltstone and mudstone fragments.

The drift at the site ranges between 26 and 35 m in thickness and varies from sands and gravels to clay and alluvium. The sand and gravel bands appear to be discontinuous but in places have been logged at up to 12 m thick. The near surface deposits into which the landfill void has been made comprise firm to stiff, silty, brown clay.

Rockhead varies in depth across the site. Near to the southern edge of the landfill it was identified at 24.2 m below ground level (m bgl), equivalent to a level of approximately -4.8 m OD. Further north-east, sandstone was identified at a depth of 17.6 m (approximately 1.4 m AOD), whilst in boreholes along the western fringe of the landfill, the bedrock was at variable levels, generally between -11 m OD and -21 m OD. The borehole data suggest that there may be a valley feature in the bedrock surface trending southwestwards across the central part of the site.

Hydrogeology

The landfill site has been developed in predominantly low permeability drift deposits. These contain more permeable sand and gravel horizons. Beneath the drift deposits is the Sherwood Sandstone, which is classified as a Major Aquifer by the Environment Agency and widely used for water supply.

There are 12 groundwater monitoring boreholes at the site that either have been or are routinely monitored for water level and quality. These boreholes are located around the perimeter of the site as shown on Figure 2.4.

Monitoring records held by Viridor have been provided for the period 1997-2003. Data prior to this, back to 1995, are presented in Wardell Armstrong (1998). Groundwater level data at this time indicated that levels in the upper sand and gravel horizons (layers 1 and 2) were very similar, and marginally lower than those measured in the lowest sand and gravel horizon (layer 3) and the sandstone, which are in hydraulic continuity. The groundwater is confined by the overlying Boulder Clay strata. September 1997 piezometric levels were measured between approximately 18 m and 19 m AOD, with a gentle hydraulic gradient to the south. Wardell Armstrong (1998) considered it almost certain that all of the sand and gravel horizons and the Sherwood Sandstone were in hydraulic continuity in the northern area of the site.

Pumping of minewater from local collieries had ceased by 1970 and it is considered that recovery in groundwater levels in the Coal Measures and the Sherwood Sandstone strata would have been completed several years ago.

2.4.4 Potential for Hydraulic Containment

Groundwater Levels

The majority of the boreholes around the landfill monitor groundwater in the lower sand and gravel horizon or the underlying sandstone, which are considered to be in hydraulic continuity. Other boreholes are completed to monitor groundwater levels in an upper sand and gravel horizon that was identified within the superficial deposits. Borehole data suggest that the occurrence of sand and gravel deposits decreases southwards across the site.

Variations in groundwater indicate that piezometric levels in the sandstone bedrock dominate the local groundwater regime. Levels during the early stages of site development and operation were typically recorded at about 19 m AOD, with levels in the southern area slightly lower than those in the north. Relatively small variations have been observed in most boreholes and recent levels are similar to those recorded initially in 1997 and 1998.

The following observations were made in the initial report on the data that has been collected since 1997:

- A general fall in levels of approximately 2 m was measured in mid-2001. At this time, there had been a major ingress of groundwater during construction works for Stage (Phase) 4b of the landfill, in the southern part of the site;
- Piezometric levels during 2002 were generally between 16 and 18 m AOD, a few metres below natural ground levels;
- Piezometric levels measured during the summer of 2003 showed little variation across the site but were typically in the range 18.5-19.5 m AOD, with levels generally higher in boreholes to the north of the site;
- The lowest piezometric levels (approximately 16 m AOD during 1991 and 1992) are typically measured in borehole AG115, to the east of the landfill; the lower sand and gravel horizon measures slightly higher piezometric levels than the middle sand and gravel horizon;
- Some of the highest piezometric levels and largest fluctuations have been measured in boreholes AG119L and M and AG121L (Figure 2.4), on the western margin of the landfill - levels have fluctuated between approximately 16 and 22 m AOD since

mid 2001. The reasons for these fluctuations may be associated with dewatering in the southern part of the site and/or development of Stage 6 in this area of the landfill.

Piezometric levels are recorded above the landfill base levels in Stages 2, 3 and 4 of the landfill and similar to or slightly below the landfill base level in Stage 1. Therefore Stages 2, 3 and 4 are sub-water table.

Leachate Levels

There are leachate monitoring points within each stage of the landfill, and these facilitate level and quality monitoring. Data from selected leachate wells were used in the initial report to present the variations in levels and the relationship with local groundwater levels.

Levels of the surface of the engineered liner in Stage 1 are typically 16.5-18.5 m AOD. The leachate levels in this area are similar to measured groundwater piezometric levels. In Stage 2, the levels of the surface of the landfill liner are lower, typically between 13.5 and 16.5 m AOD. The data for this area of the site show a greater differential between leachate and groundwater piezometric levels than observed in Stage 1, being in the order of 2-3 m. Development of the landfill southwards, and the associated lowering of the landfill base in this direction to provide appropriate drainage gradients (base levels in Stage 4 for example are typically 11-13 m AOD), has meant that there are larger differentials between managed leachate heads and local groundwater piezometric levels in later, more southerly stages of the landfill.

Leachate levels are below piezometric levels in Stages 2, 3 and 4 and therefore these are hydraulically contained. The degree of hydraulic containment increases southwards across the landfill.

Groundwater Ingress

A feature of the site development at Whitehead was the ingress of groundwater experienced during construction of part of the site.

A significant inflow of water caused problems in the construction and operation of Phase 4 of the landfill. A report by Viridor concluded that this water was sourced primarily from the Triassic Sandstone, which is at a depth of approximately 25m below the site. The construction of Phase 4 began in April 2001 with an initial ingress of groundwater limited to 200 m^3/d. This water was diverted by a small diameter pipe and an additional 3 m of engineered clay fill was placed over the area to restrict the inflow. A second ingress was encountered in August 2001 which led to discussions with the Environment Agency and an investigation into the source of the water. A chamber was installed in this area with pumps used to remove the larger volume of water. At the time of reporting (2001), Viridor indicated that 7000 m^3/d of water was being removed from the south-western corner of Stage 4. Pumping continued throughout 2002 and was terminated in about April 2003. By this time, Stage 4 had been fully engineered and filled with wastes.

Further consideration of this groundwater ingress is included in chapter 5 of this report (Section 5.5.5).

Degree of Hydraulic Containment

Leachate levels at Whitehead are generally controlled by pumped extraction so that heads above the liner are limited and do not exceed licence conditions. Groundwater piezometric levels were recorded typically in the range 16-18 m AOD during 2002, but about 2 m higher than this previously and since. This illustrates that the extent of hydraulic containment can vary over time, particularly where groundwater level variations are large in comparison with leachate level variations.

Early development of Stage 1 in the northern area of the site incorporated the highest basal liner levels. With managed leachate levels similar to local groundwater piezometric levels, this part of the landfill is marginal with respect to hydraulic containment. In the event that leachate levels increased in relation to groundwater levels, there would be no hydraulic containment and the potential for leachate leakage would increase.

Areas farther south, such as Stages 3 and 4, are hydraulically contained. These have been constructed with basal liners at lower elevations than earlier stages. Leachate level control and generally consistent groundwater levels across the site mean that there are increased head differentials between leachate and groundwater than in the earlier stages. Groundwater levels are currently in the order of 5 m above leachate levels in Stage 4.

2.4.5 Water Balance

A water balance was undertaken in the initial report for Stages 1 to 6 (excluding Stage 5). Viridor provided waste input estimates, rainfall data, and details of volume of leachate abstracted. Volumes of sludge and liquid waste disposal recorded for 2002 were also taken into account.

Input data for the water balance included estimates of active and restored areas of the site, approximate waste input rates, estimated infiltration rates for active and clay capped areas, values for the absorptive capacity of the waste and estimates of liquid/sludge inputs.

It was estimated that approximately 36 000 m^3 liquid was introduced into the site in 2002 (infiltration plus liquid inputs), an average of approximately 100 m^3/day. As a consequence of the waste input volumes and using an associated absorptive capacity of 8%, leachate generation during the lifetime of the site and to the end of 2003 was predicted to be minimal. However, when a lower absorptive capacity for the wastes of 5% was adopted, daily leachate volumes of 10-40 m^3 were predicted during 2000-2002. An absorptive capacity of 3% resulted in estimated leachate volumes of approximately 50-65 m^3/day during this period.

These estimates were compared with measured leachate volumes collected and pumped from the site. During 2002, records indicated that an average of 150 m^3/day were extracted during the first quarter of the year, declining to about 65 m^3/day during the summer period July-September. Uncertainties in the input parameters used in the calculations could explain the apparent difference, for example infiltration rates, areas active and restored, liquid inputs and the absorptive capacity of the wastes. During 2002, volumes of leachate discharged to sewer were similar to those reported as extracted, and it is assumed that little or no leachate was recirculated back into the wastes that year.

More recent data for 2003 and 2004 have been provided by Viridor and are summarised together with the earlier data in Table 2.5. Recirculaton of leachate started in mid November 2003 with ~9700 m^3 recirculated by the end of 2003 and a further ~65 420 m^3

recirculated in 2004 up to the end of September. As for the earlier water balance, estimated total water input to the site was much greater than the volume of leachate removed from the site to sewer, indicating that there has been a continuing net gain in the water content within the site. During this period, leachate levels have generally remained within compliance levels, and so this gain has likely been through wetting up the absorptive capacity of the waste.

There is no clear evidence from the annual water balances that significant groundwater ingress is occurring. With a ~4 m head difference between groundwater levels and leachate levels across Stages 2 to 4, groundwater ingress through a 1 m clay liner engineered to 10^{-9} m/s would provide the equivalent of 126 mm/yr infiltration (this assumes that the plastic liner has little effect in the direction of groundwater ingress because it is backed by gravel rather than clay). 126 mm/yr equates to 17.5% of the average total rainfall (717 mm) at the site between 2001 and 2004 or 32% of the average infiltration (398 mm) into the waste over that period estimated by Viridor's annual water balance calculations. With the clay performing at 5×10^{-10} m/s, then the inflows could be ~60 mm equating to 8% of the average annual rainfall or 15% of the average infiltration into the waste. Given the uncertainties in estimating infiltration through uncapped, temporarily capped and permanently capped waste, it is not implausible that some groundwater ingress is entering the site.

Impact on Groundwater Quality

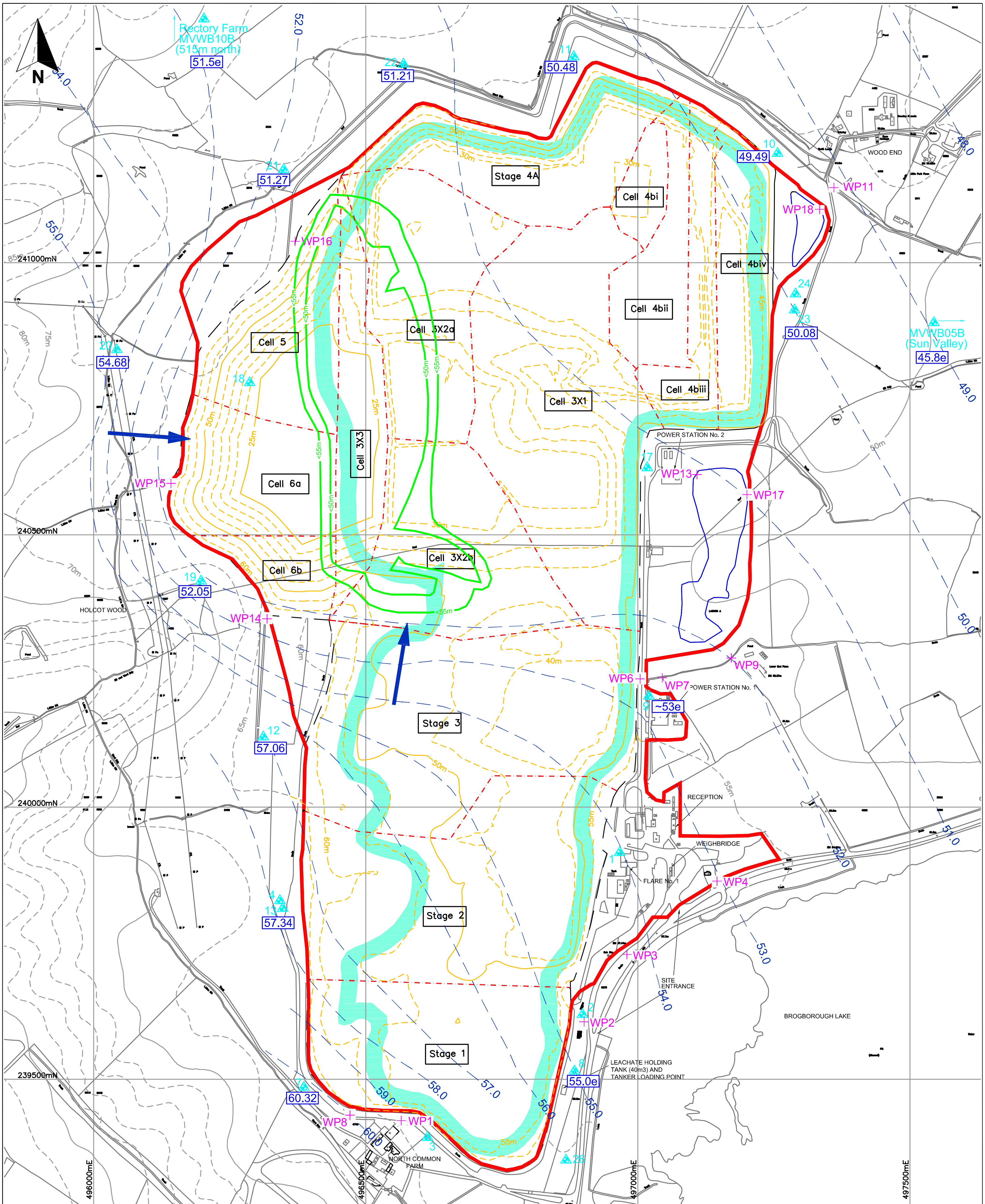
Chloride and ammoniacal nitrogen were selected as appropriate parameters that have been measured at high concentrations in leachate, which would provide evidence of any impacts on groundwater. Monitoring data from boreholes to the north of the site indicated that local background groundwater concentrations for chloride and ammoniacal nitrogen are 30-50 mg/l and approximately 0.1 mg/l respectively. Chloride concentrations above these background levels, in excess of 200 mg/l, were recorded in boreholes AG118L and AG118M, and to a lesser extent in borehole AG126M (up to approximately 100 mg/l). These boreholes are located on the northern boundary of the site, where landfill base and leachate levels are highest compared with local groundwater piezometric levels, and where hydraulic containment is least likely. Whilst the data could be interpreted to indicate some impact from the landfill, early data indicate that chloride concentrations were elevated in 1997, prior to waste disposal. The waste colliery spoil is the most likely source, and elevated sulphate concentrations in these boreholes provide further evidence for this. Ammoniacal nitrogen concentrations did not mirror the observed chloride data, with concentrations typically less than 0.3 mg/l.

Chloride concentrations measured in the southern monitoring boreholes are consistent with background, with no evidence of any significant contamination, although there was a slightly increasing trend in chloride concentrations measured in borehole 119M, on the western margin of the site. Ammoniacal nitrogen concentrations were generally higher than in the northern boreholes, typically in the range 1-3 mg/l, with occasionally higher concentrations measured in boreholes AG122L and AG123L. Again, similar concentrations were measured in 1997, prior to waste disposal taking place.

Overall, the early monitoring data for the site showed no clear evidence that the landfill is impacting on local groundwater quality. Evidence of slightly elevated chloride concentrations in northern boreholes, and of elevated ammoniacal nitrogen concentrations in southern boreholes is not indicative of impact from the landfill since monitoring records suggest that such concentrations were in evidence prior to landfilling taking place.

Table 2.5 Summary of Viridor Waste Management's Leachate Balances for Whitehead Landfill Site

hm-250/08975/calcs/WasteStabCalcs/LS-Ratios_nr.xls - Table 2.5 (1 x A4 page)



Key

Licence boundary	Piezometric contour in the Kellaways Sand (mAOD)
Cell boundary	60.32 Piezometric level in the Kellaways Sand (mAOD)
Groundwater borehole	50m Pit base contour (mAOD)
Surface water monitoring point	Direction of groundwater flow in Kellaways Sand
Pit base beneath piezometric level in the Kellaways Sand	50m Level of top of waste (only shown for 50m and 55m contours)

0 m — 250 m

Scale 1:6500 @ A3

Review of the Performance of Hydraulically Contained Landfills

Figure 2.1
Brogborough Site Layout Showing Piezometric Levels in the Kellaways Sand (May/June 2000)

November 2005
10744-S42.dwg wrigs02

Entec

Key

(green)	Phases 1 - 3
(red)	Phase 4
(yellow)	Proposed Phase 5
(hatched)	Location of spring
+	Groundwater pumping station

0 m 150 m

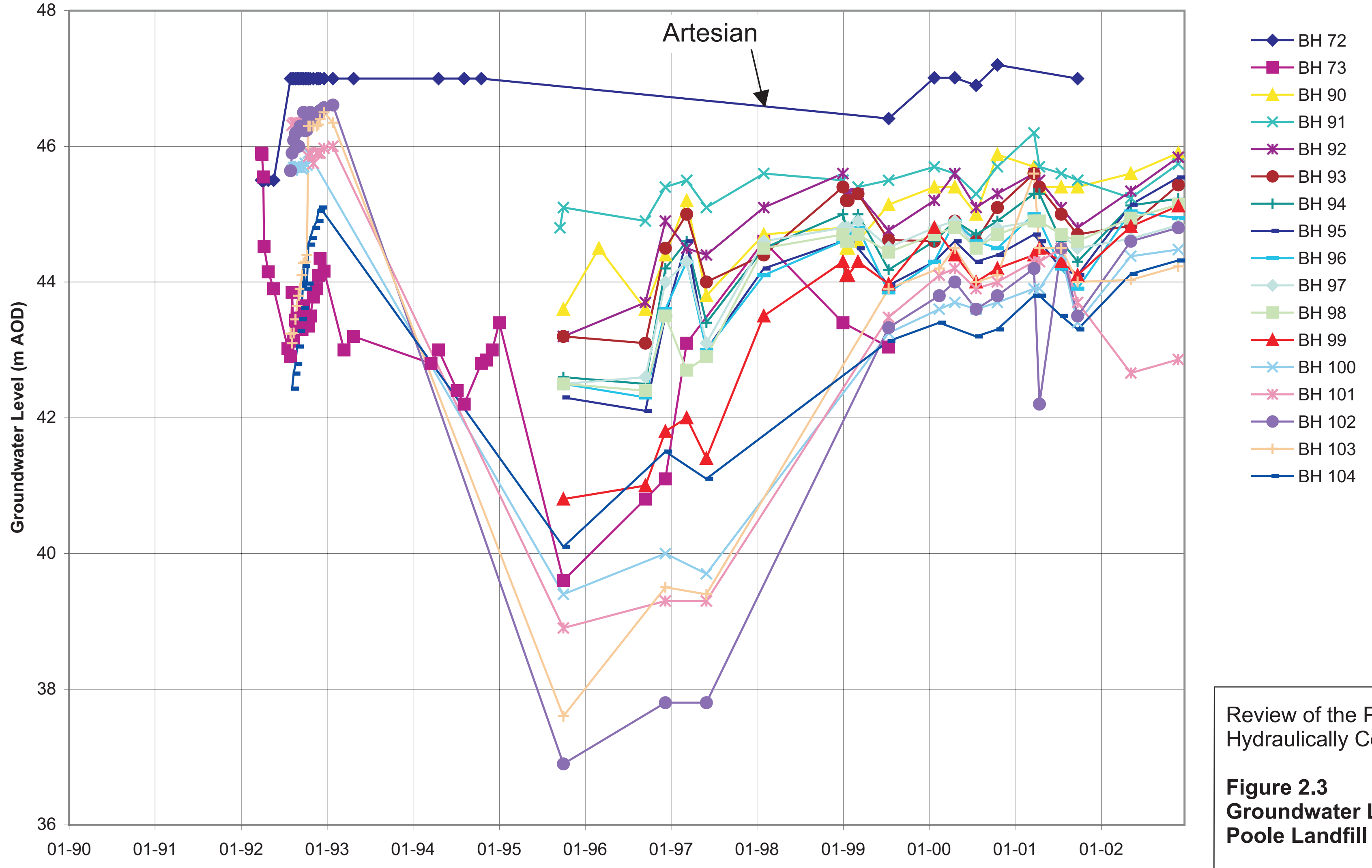
Scale 1:2,500 @ A3

Review of the Performance of Hydraulically Contained Landfills

Figure 2.2
Poole Site Layout

November 2005
10744-S43.dwg wrigs02

Entec



H:\Projects\Hm-250\08465 Wyvern Ph2 HydCon\Data\ASLv2.xls *south*

Review of the Performance of Hydraulically Contained Landfills

Figure 2.3
Groundwater Levels at the South of Poole Landfill Site

November 2005
10744-S44.cdr wrigs02

Entec

Key

- Phases 1 - 3
- Phase 4
- Proposed Phase 5
- Location of spring
- Groundwater pumping station

0 m 150 m

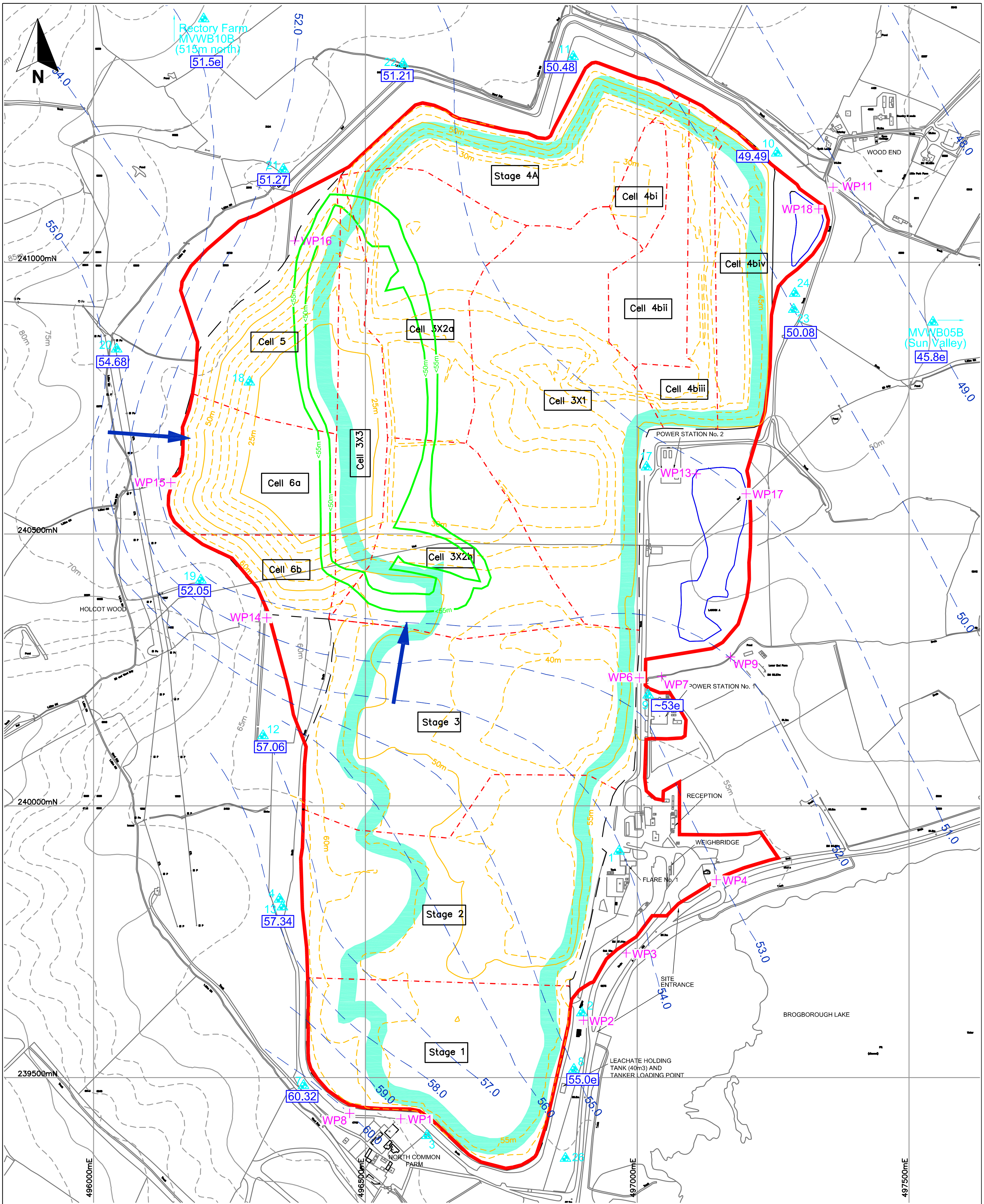
Scale 1:2,500 @ A3

Review of the Performance of Hydraulically Contained Landfills

Figure 2.2
Poole Site Layout

November 2005
10744-S43.dwg wrigs02

Entec



Key

Licence boundary	Piezometric contour in the Kellaways Sand (mAOD)
Cell boundary	60.32 Piezometric level in the Kellaways Sand (mAOD)
Groundwater borehole	50m Pit base contour (mAOD)
Surface water monitoring point	Direction of groundwater flow in Kellaways Sand
Pit base beneath piezometric level in the Kellaways Sand	50m Level of top of waste (only shown for 50m and 55m contours)

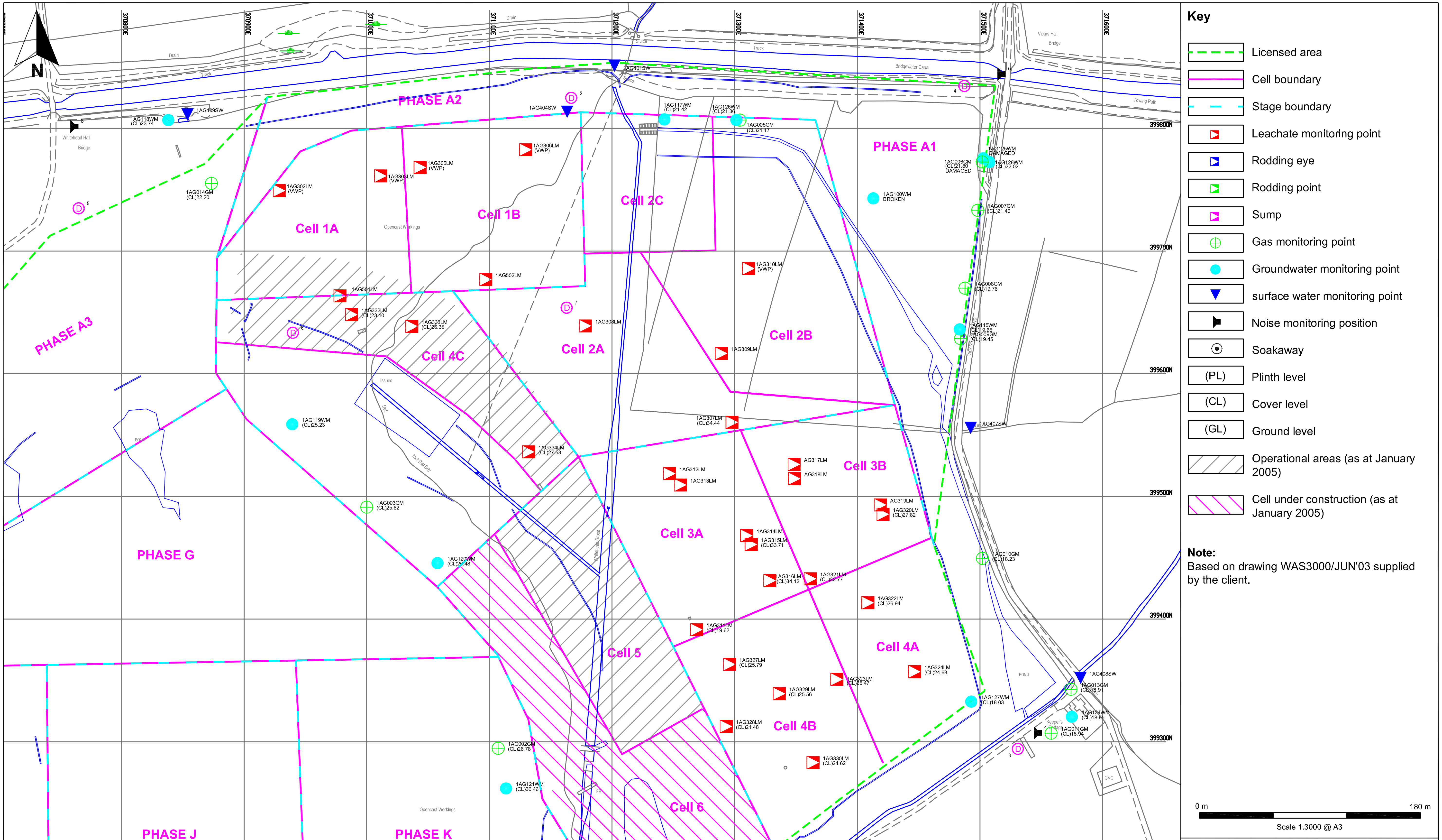
0 m 250 m
Scale 1:6500 @ A3

Review of the Performance of Hydraulically Contained Landfills

Figure 2.1
Brogborough Site Layout Showing Piezometric Levels in the Kellaways Sand (May/June 2000)

November 2005
10744-S42.dwg wrigs02

Entec



Review of the Performance of Hydraulically Contained Landfills

Figure 2.4
Whitehead Landfill Site Plan

November 2005
10744-S45.dwg wrigs02

Entec

3. Waste Stabilisation

3.1 Background

The biodegradation of landfilled waste is facilitated by the presence of water. This understanding has lead to the promotion of the concept of landfills as flushing bioreactors with additional wetting and flushing of the waste with enhanced infiltration (DoE, 1995).

In reviewing groundwater risk assessments for hydraulically contained sites, the Environment Agency has, in the past, expressed concern that the rate of waste degradation and flushing, and thus stabilisation, may be inhibited by high leachate levels in landfills. The rate of stabilisation impacts the length of time landfill sites produce poor quality leachate and therefore potentially increases the length of time it remains a pollution threat.

In "*A review of the composition of leachates from domestic waste in landfill sites*" (Robinson, 1995, p C64), it was noted that bioreactor landfills in the UK have arisen, albeit inadvertently, through the uncontrolled ingress of large quantities of water. Often this has taken the form of groundwater ingress rather than high rates of infiltration and percolation of rainfall. This suggests that groundwater ingress, as expected to occur in hydraulically contained sites, by no means leads to slow rates of waste stabilisation.

Hydraulically contained landfills have the potential for groundwater ingress and thus potential for more rapid wetting and greater flushing of the waste when compared to sites which receive water through cap infiltration only. However, high leachate levels alone only ensure wetting up of the lower waste layers and not flushing of leachate through all of them, unless leachate is extracted from towards the top of the saturated waste.

There are numerous full scale demonstration research projects underway in North America investigating the operation of bioreactor landfills and waste stabilisation, with intensive monitoring and data acquisition (Bioreactor landfills: progress continues, Waste Management, Vol. 24 Iss. 9). 'Managed' bioreactor type landfills are operated and designed so that optimum microbial degradation conditions are maintained through controlled addition and removal of liquid such as leachate, wastewater and storm water, introduction of air, and the design of the liner and cap. These conditions are likely to be different to those in older landfills, which are being operated to minimise leachate levels and thus leachate generation for groundwater quality protection purposes.

Current information on waste stabilisation in hydraulically contained landfills is very limited. In terms of planning for new landfills, and in prediction of the time for waste stabilisation of existing landfills (i.e. the time when the landfill no longer poses a pollution threat), further information on waste stabilisation in hydraulically contained landfills would be beneficial.

3.2 Aim and Methodology

In response to the current uncertainty regarding waste stabilisation in ‘wet’ hydraulically contained landfills described above, and in an attempt to further investigate the efficiency of waste stabilisation, this study aims to see if there is any evidence to suggest whether waste stabilisation, as measured by leachate quality, is any less in hydraulically contained or wet sites than in more typical above-water-table sites. This is undertaken firstly through a review of recent literature on waste stabilisation and literature on recommended approaches for assessment of waste stabilisation, and then through evaluation of available leachate quality and temperature data for the three hydraulically contained landfill research sites used for this study (Brogborough, Poole and Whitehead).

3.3 Current Understanding and Existing Information on Waste Stabilisation

3.3.1 Literature Sources

There is some existing literature on leachate quality and approaches to waste stabilisation evaluation, however Entec is not aware of previous work related solely to waste stabilisation and leachate quality in hydraulically contained landfills.

Keith Knox, an expert in the field of landfill waste stabilisation, was contracted by Entec to provide key papers on waste stabilisation and associated material, and to provide suggestions for an approach to the evaluation of waste stabilisation. The material provided included:

- ‘The Relationship between Leachate and Gas’. Knox (1990);
- ‘Description of a Tracer Test through Waste and Application of a Double Porosity Model’. Beavan, Barker & Hudson (2003);
- ‘Hydraulic Containment of Landfills, Leachate Quality and Waste Stabilisation’. Knox (2004).

In addition to this literature, the following were reviewed for information on leachate quality and waste decomposition processes:

- Waste Management Paper 26B, Landfill Design, Construction and Operational Practice. Department of the Environment. (1995);
- ‘A Review of the Composition of Leachates from Domestic Wastes in Landfills’. Robinson (1995);
- ‘Pollution Inventory Discharges to Sewer of Surface Waters from Landfill Leachates’, Ref: REGCON 70. Prepared for the Environment Agency, by Robinson & Knox (2001);
- ‘Improved Definition of Leachate Source Term from Landfills - Phase 1: Review of Data from European Landfills’, P1-494/SR1. Environment Agency (September 2004).

3.3.2 Content of Existing Literature

Introduction

This section briefly summarises the content of the literature identified above. An overview of processes derived from this literature review is provided in Section 3.4.

Knox (1990)

'*The Relationship Between Leachate and Gas*' (Knox, 1990) assesses whether parameters in leachate, gas and gas condensate can determine the status of decomposition processes, the extent of waste stabilisation and remaining gas potential of the landfill by considering the changes in leachate and gas quality through time. The data assessed are taken from published literature, and supplemented by a sampling programme at eight landfill sites. Brief details on the sampling sites are given and include information on containment type, age of waste, saturated conditions (where known) and sampling details. Leachate quality data are assessed for trends with age of waste, and consistent trends with age appear to exist for COD/chloride, TOC/chloride, ammonia/chloride and acetic acid/total volatile fatty acids (VFA) ratios. Trends are discussed and show a gradual fall in ammonia/chloride, COD/chloride and TOC/chloride ratios and a rise in colour/residual TOC and acetic acid/total VFA ratios with increasing age. It is noted that the relationship between quality parameters and age of waste can be affected by site-specific conditions and that trends of a single parameter on its own can not be relied upon to draw any conclusions about potential connections between the age of waste and parameter concentration. Leachate quality is shown to sometimes correlate well with hydraulic retention time. Gas and condensate quality data are assessed in a similar way. No relationship is found between gas condensate components and age, however a number of gas components show evident trends with age, including aromatics, alkanes, halogenated compounds, alcohols and esters.

Department of the Environment (1995)

Waste Management Paper 26B (WMP26B) (Department of the Environment, 1995) contains a detailed summary of waste decomposition processes, including generalised evolution of gas and leachate composition, and the issues surrounding accelerated waste stabilisation.

Robinson (1995)

'*A Review of the Composition of Leachates from Domestic Wastes in Landfills*' (Robinson, 1995) details leachate quality at a large number of domestic waste landfills, and aims to understand the progression of waste decomposition through leachate composition. The report details time series leachate quality data in an attempt to characterise leachate quality of different categories of landfill. Analytical results of nearly 4000 leachate samples at 72 landfill sites throughout the UK and Ireland are reported, and leachate quality data for eleven categories of landfill based on criteria such as waste input rate, landfill size and shape, degree of water ingress, and waste form (e.g. baled, pulverised) are presented. Trends and detailed analytical results of leachate quality indicators COD, BOD, pH, ammoniacal nitrogen and chloride are presented for each landfill within each category, thus providing an indication of variability in leachate quality and the onset of acetogenic and methanogenic phases of decomposition. Concentrations of other indicators of leachate quality such as heavy metals are also detailed. A summary of the principles of waste decomposition, and the effects of physical and chemical conditions on the biological stabilisation of waste, is included in the report.

Robinson and Knox (2001)

The '*Pollution Inventory Discharges to Sewer or Surface Waters from Landfill Leachates*' (Robinson and Knox, 2001) is a study into the occurrence of substances on the Pollution Inventory list for England and Wales in raw and treated leachates from UK landfills. The results section includes details on findings regarding the correlation of the concentration of selected species in raw leachate with waste input type, moisture regime and leachate biochemical status (acetogenic, transitional, or methanogenic), and also the correlation between leachate species (various metals, alcohols, pesticides, VOCs) and chloride concentration and TOC concentration. It reported no correlation was found between species concentration and waste input type, moisture content or chloride concentration, however zinc and nickel did show a clear correlation with leachate biochemical status, where the metals were higher in concentration in acetogenic leachates (a pH control). This seems to suggest that moisture content was not a significant influence on the concentration of those species assessed.

Beaven, Barker and Hudson (2003)

The '*Description of a Tracer Test through Waste and Application of a Double Porosity Model*' (Beavan, Barker & Hudson 2003) investigates the hydraulic characteristics of waste, through modelling of data collected during flushing of tracer through waste, and through the application of a double porosity model, DP-Pulse. The results of the tracer tests showed evidence for preferential flow routes within the waste, and evidence that the waste was acting as a dual porosity material. These results were successfully modelled using the double porosity model DP-Pulse, and the derived parameters were then used to simulate landfill flushing using DP1D. Changes in the ratio of mobile to immobile porosity of waste was shown to be most important in the ability to flush contaminants from the waste.

Knox (2004)

The information compiled for this project by Knox (2004) reports that leachate quality is not a useful indicator of the extent of waste degradation or stabilisation. Knox suggests that changes in COD and BOD during methanogenesis are mainly attributable to variation in topography, temperature and hydraulic regime of the landfill rather than waste stabilisation. Also, that changes in leachate quality will only occur as a result of flushing, or from a change to anaerobic regime. It is also suggested that in the absence of promotion of aerobic degradation, flushing of the landfill is the only process that will affect leachate quality.

It is reported that general observations show flushing of landfills produces an exponential decline in concentration of conservative species. For example, data for Vestskoven landfill, a hydraulically contained site in Denmark where leachate levels are maintained below groundwater levels, monitored over 30 years, show a decline in chloride concentration with increased flushing (i.e. chloride concentration declined as the site moved from a low to a high liquid:solid (L:S) ratio (Hjelmar and Hansen, 2004). Even with a relatively high leachate production rate corresponding to infiltration of 33% of the precipitation, the monitoring data suggest that final storage quality at the site is hardly achieved after 30 years, although low concentrations have been reached for some leachate components. With adoption of landfill flushing as the process to affect leachate quality, Knox (2004) outlines an approach to the assessment of waste stabilisation based on the liquid:solid ratio (L:S) of a site (detailed and used further in Section 3.8). An additional approach, undertaking a nitrogen balance is also suggested, as flushing of ammoniacal nitrogen is likely to be the controlling parameter for the indication of waste stabilisation at most current UK landfills.

Environment Agency (2004)

A study on the '*Improved Definition of Leachate Source Term from Landfills - Phase 1: Review of Data from European Landfills*' (Environment Agency, 2004) has been undertaken to help understand the future effects on leachate quality and waste stabilisation of UK implementation of the Landfill Directive, in terms of landfill operation. The UK currently has little information derived from UK landfills on leachate quality post-implementation of the Landfill Directive, which requires a phased reduction in the proportion of biodegradable municipal waste going to landfill, pre treatment prior to deposition and the prohibition of certain types of hazardous waste. Methods such as mechanical and biological pre-treatment (MBP) of waste, deposition of ash following incineration, mono-disposal of hazardous waste, and pre-treatment of hazardous waste are employed in the EU, thus this study is a compilation of leachate quality data for landfills using these practices, so to enable application in the UK. Based on the information collected the study also details further research needs.

It is reported there is currently little data available to quantify the effect of pre-treatment on the stabilisation of waste. Data that are available suggest that MBP processes can considerably reduce the organic strength of leachate, avoid the acetogenic phase and produce leachate similar to those derived from municipal solid waste (MSW) landfills in methanogenic phase more rapidly. However, it is reported that landfills containing MBP waste may require an aftercare period similar to that of a conventional methanogenic phase MSW landfill. It is reported that laboratory column tests on mixed and treated organic wastes could be used to predict long-term leachate quality and timescales to achieve waste stabilisation, although further investigations are required.

3.4 Waste Decomposition and Stabilisation

3.4.1 Introduction

Knox (2004) notes that the time taken for waste decomposition and stabilisation is unique to an individual landfill, and controls on waste decomposition and gas production are complex and interlinked. Robinson (1995) highlights moisture content, temperature, waste density, age and composition, waste size, substrate availability, pH, the presence of microbes and nutrient availability as factors in the timescale for waste degradation. One of the most important controls on gas production and therefore a control on the rate of waste degradation rate and hence stabilisation, is moisture content (Robinson 1995) - see also Chapter 4 of this report on landfill gas generation. Robinson (1995) also notes that waste stabilisation in landfills with limited fluxes of water passing through the waste are likely to be decades, or even centuries.

In the following sections, waste decomposition processes are first discussed using information taken from WMP26B (1995). This is then followed by a discussion of waste stabilisation in wet and dry landfills, using information from Robinson (1995), and Knox (2004).

Of the different categories of landfill detailed by Robinson, the two of most interest for this study are 'large landfills with a high waste input rate, deep, wet and "bioreactive"', and 'large landfills with a high waste input rate, (but which are) relatively dry'.

3.4.2 An Overview of Waste Decomposition Processes in Landfills

The following is a brief summary of the five stages of waste decomposition of the organic fraction of waste material within a landfill. As a result of the timing of waste emplacement, and spatial variation in waste characteristics all of these stages may be occurring at different rates at any one time. The process is illustrated in Figure 3.1 and changes in leachate composition associated with each stage are shown in Figure 3.2, both taken from WMP26B, Appendix C (DoE, 1995).

- ***Stage 1 (hydrolysis and aerobic degradation)* -** Initially characterised by aerobic degradation of carbohydrates to simple sugars, carbon dioxide and water. Microbiological activity is intense and can generate temperatures within the landfill of up to 80-90°C. As the oxygen becomes depleted and low oxygen conditions are established, facultative anaerobic micro-organisms dominate the decomposition processes, followed by gradual establishment of obligate anaerobes (methanogens) once all the oxygen has been used up. The duration of this aerobic stage is dependent on the availability of oxygen within the pore spaces of the waste, and therefore can be influenced by landfill practices (i.e. compaction on tipping).

- ***Stage 2 (hydrolysis and fermentation)* -** Characterised by anaerobic hydrolysis of carbohydrates, lipids and proteins to simple sugars followed by fermentation, by bacteria, of soluble intermediates such as volatile acids (i.e. propionic, butyric, lactic and formic acids), acetate, carbon dioxide, hydrogen, sulphate and ammonium. The leachate produced has a high ammoniacal nitrogen content and landfill temperature is typically 30-35°C;

- ***Stage 3 (acetogenesis)* -** Characterised by anaerobic conversion of the soluble acids produced in Stage 2 to acetate, carbon dioxide and hydrogen and conversion by other bacteria of carbohydrate, hydrogen and carbon dioxide to acetic acid. The breakdown of products of fermentation from Stage 2 (butyrate, propionate and ethanol) is only achieved at low hydrogen concentrations. Without this breakdown, there is no food source for methane-generating bacteria, and accumulation of propionic acid results. Low hydrogen concentrations are maintained by sulphate-reducing bacteria (which produce hydrogen sulphide), and further breakdown (oxidation) of Stage 2 products is undertaken by methane-generating bacteria;

- ***Stage 4 (methanogenesis)* -** Characterised by the anaerobic breakdown of acetate and formate, produced prior to this stage, by methane-generating bacteria to form methane and carbon dioxide. Typical pH for this stage is 6.8-7.4, as this is the range that the bacteria are most active;

- ***Stage 5 (oxidation)* -** Characterised by gradual re-establishment of aerobic conditions, with potential for release of metals to leachate.

The decomposition process is aided by the percolation of water through the waste, which removes suspended solids, the soluble constituents of waste and the soluble products of waste degradation. This liquid is termed leachate, and its composition depends on the stage of decomposition.

3.4.3 'Dry' Landfills

'Dry' landfills have limited fluxes of water through them, and therefore limited encouragement of decomposition and hence waste stabilisation through limited contaminant flushing and contact between liquid and waste. Robinson (1995) notes that in the long term, landfills with a high waste input and operated to minimise water ingress will pose a considerable long term problem to regulatory authorities.

The timescale for transition to fully methanogenic leachate quality at the 'dry' landfills detailed in Robinson (1995) varied between 2½ and 10 years, with half the sites reaching methanogenesis within 3 years and almost three quarters after 4 years. One suggestion for the late onset of methanogenesis for the three sites where the transition took longest, was that there were areas of relatively dry, deeper layers of waste beneath perched wetter layers. Also, all three sites are pumped hard for gas, which may draw in air and so prevent or delay the onset of methanogenic conditions.

3.4.4 'Wet' Bioreactive Landfills

Degradation Rates

It is reported by Robinson (1995) that the bioreactive landfills in the UK have arisen inadvertently through ingress of water, either from a rising water table (hydraulically contained sites) or from infiltration of rain through the cap. There is a fine line between water level rise in the landfill, encouraging decomposition and high temperatures, and too much water resulting in the effect of cooling and dilution of the leachate, which will slow down the degradation process. There also appears to be a difference in the decomposition conditions within a landfill between those brought about by an increase in moisture content through groundwater ingress, and those brought about through increased infiltration.

Bioreactivity and high rates of gas generation can be achieved by groundwater ingress. For example, Robinson details three landfills (Aveley, Mountsorrel and Warnham) which are categorised by him as bioreactive, that were subject to groundwater ingress. The three sites experienced rising water levels from the base of the landfill, each with observed rapid temperature rises of >40 C under anaerobic conditions. At Aveley, the temperatures were maintained at this level for up to 13 years. However, there is no evidence put forward for a connection between waste saturation and a rapid establishment of raised temperatures and bioreactive conditions. Note also that one of the characteristics of a bioreactive landfill is a significant depth of unsaturated waste, to help insulate the waste and maintain high temperatures.

The attenuation site at Stangate Landfill shows a relationship between gradual rise in saturated thickness of waste and development of methanogenesis (an indication of the progression of waste decomposition). In contrast to sites with significant groundwater ingress, Robinson reports that those landfills where waste saturation and bioreactive conditions are achieved by high infiltration of rainfall have temperatures only in the range 30-35 C. It is suggested this may be due to an additional increase in acetogenesis as the water percolates down through the waste, and also may be due to cooling effects, though this is thought to be less likely.

Robinson reports that waste saturation from the base may encourage more rapid biodegradation to take place as there is more opportunity for rapid dilution of any added industrial wastes or leachate components (breakdown products) that might cause inhibition of biological processes. Also the leachate does not contain high concentrations of fatty acids, which may arise if a similar amount of water infiltrated from the surface and passed through a layer of waste before reaching the saturated waste layer. In the long term, the increased hydraulic retention time of a saturated landfill may be beneficial for relatively poorly-degradable components to break down, and waste saturation provides much better opportunities for contact between waste and leachate. The water in landfills encourages relatively rapid rates of biodegradation, and physically dissolves and transports waste components out of the landfill for treatment.

Flushing

After much research there is evidence to suggest that the concentration of leachate species in a landfill declines exponentially with flushing due to dual porosity, or multiple porosity characteristics (Knox 2004). The porosity of the part of the landfill which allows significant fluid flow is a small proportion of the total porosity, but water in the non-mobile parts can equilibriate at a fast enough rate to give behaviour similar to that of a mixed reactor. The exponential equation used by the risk assessment tool LandSim 2 (Environment Agency, 2001) models the behaviour of a mixed reactor and therefore is a reasonable simulation of the observed double porosity behaviour (Knox 2004).

Investigations into bioreactor landfills are currently underway in North America, where a number of full-scale projects with extensive data acquisition are in progress (Bioreactor landfills: progress continues, Waste Management, Vol. 24 Iss. 9). These include The New River Regional landfill in Union Count, Florida, Northern Oaks Recycling and Disposal Facility in Harrison, Michigan, Buncombe County Landfill Project in North Carolina, and Maplewood Landfill and King George County Landfills, Virginia.

Two years worth of data are available for the Outer Loop Landfill, Louisville in Kentucky, where a control landfill and two bioreactor landfills are monitored. Findings are reported in 'Landfills as Bioreactors: Research at the Outer Loop Landfill, Louisville, Kentucky - first interim report', EPA/600/R-03/097, September 2003. The interim findings of the preliminary report indicate the aerobic/anaerobic landfill bioreactor (AALB) has temperatures of around 28°C, compared to control cell temperatures of around 16°C. Waste settlement is reported to be greatest in the AALB cells, and thought in part to represent the effects of biological decay. In general the highest rates of change in degradable organics are occurring in the AALB cells, yet there was no downward trend in ammonia concentration in leachate with time.

3.4.5 Summary Characteristics of 'Dry' and 'Wet' Landfills

The characteristics of 'dry' and 'wet' landfills are summarised, in Table 3.1. The information is based on data obtained and presented by Robinson (1995). The summary data for 'wet/bioreactive' sites were obtained directly from the data summary tables, and the data for 'dry' sites were obtained from the summary text on page C57.

Table 3.1 Summary of the Characteristics of 'Dry' and 'Wet' Landfills

Leachate Characteristics	'Dry' Landfill Characteristics		'Wet/Bioreactive' Landfill Characteristics	
	Generally low moisture content, with a waste field capacity <30%		Generally high moisture content, with a waste field capacity of around 30 to 40%	
	acetogenic	methanogenic	acetogenic	methanogenic
pH	5.1 - 7.8	6.8 - 8.2	5.5 - 7.5	7.4 - 8
Temperature	**typically expected to be <30°C**	**typically expected to be <30°C**	**Variable. Rapid initial temperature rise can occur, typically 30-50°C, and be maintained, but is not always the case*. High temperatures appear to be associated with higher rates of gas production.**	
COD	20 000 to 40 000 mg/l	around 2000 mg/l	30 000 to 50 000 mg/l	1 500 to 4 000 mg/l
Chloride	>500 mg/l	>500 mg/l	1 000 to 3 500 mg/l	1 000 to 2 500 mg/l
Ammoniacal nitrogen	mean of <1000 mg/l	around 1000 mg/l	500 to 1 500 mg/l	1 000 to 1 500 mg/l (but as low as 250 to 600 mg/l at Withnell)
Comments	**Onset of methanogenic conditions between 2½ and 10 years after start of landfilling.**		**Variable rate of onset of methanogenesis, 2 to 6 years (where determined) since start of landfilling.**	

Note: *discussed in more detail in Section 3.4.4.

For the majority of characteristics there do not appear to be great differences between those categorised as 'dry' landfills and those as 'wet/bioreactive' landfills. The main differences appear to be temperature within the landfill, the rate of onset of methanogenesis, the rate of gas production and perhaps ammoniacal nitrogen concentrations above 1000 mg/l. Leachate temperature in 'wet' landfills appears to be generally higher than in 'dry' landfills, although it is noted that the higher temperatures are not only/exclusively linked to saturated waste, they are also linked to insulation and thickness of overlying unsaturated waste in some cases. The data are limited for the onset of methanogenesis and may be skewed due to the small data set available, however they seem to suggest that in a 'wet' landfill the onset of methanogenesis is likely to occur sooner than in a 'dry' landfill.

3.5 Approaches to Waste Stabilisation Assessment

3.5.1 Published approaches

This section describes the current modelling approach to waste stabilisation and suggested approaches from literature.

The chemical characteristics of leachate produced in a landfill change with time. The concentration of chemical species in leachate can change due to flushing-out of contaminants as infiltrated water passes through the landfill.

A standard methodology for assessing the risks and uncertainties associated with groundwater contamination from landfill sites situated above the water table, over time, is compiled within probabilistic software packages *LandSim v2* (Environment Agency, 2001) and *LandSim 2.5* (Environment Agency, 2003). Within these packages the potential for waste stabilisation/flushing of contaminants from the waste is calculated using a declining source term equation to model the changing characteristics of the leachate, developed to enable the estimation of leachate concentration at any time. General observations show flushing of landfills produces an exponential decline in concentration of conservative species (Knox 2004). For example, chloride concentration data for (the uncapped) Vesterhoven landfill, Denmark, monitored over 30 years shows a decline in chloride concentration with increased flushing, and is illustrated in Figure 3.3. *LandSim v2* seeks to simulate the declining concentration at any time using the parameters of infiltration, waste thickness, waste field capacity and assumed initial concentration (see Box 3.1).

Box 3.1 LandSim (v2.02, Environment Agency 2001) Declining Source Equation

$$Ct=C_0\exp(-lambda \times t)$$

Where:

Ct = Concentration at time t (e.g. mg/l);

C_0 = Starting Concentration (e.g. mg/l);

lambda = HER/(Wt × Wn);

HER = average infiltration into the waste (m);

Wt = average waste thickness (m);

Wn = water content of the waste under free draining conditions (unitless).

Rearranging this equation:

$$t = \frac{Wt.Wn}{HER}.\ln\left(\frac{C_0}{C_t}\right)$$

An alternative approach to assessment of waste stabilisation is to assess where the landfill lies on an exponential flushing curve by calculation of the liquid:solid ratio (L:S) of a site, where L:S is defined as ‘infiltration to the waste including groundwater ingress, divided by the mass of waste’. In *LandSim 2.5*, species concentration variation with time is linked to the L:S ratio of the landfill and a species and waste-dependent constant Kappa (κ) (see Box 3.2).

Box 3.2 LandSim (v25, Environment Agency, September 2003) Equation

$Ct = C_0 \exp(-\kappa \times LS)$

Where (in addition to the parameters defined in Box 3.1):

κ = Kappa is a species and waste specific parameter (kg/l) derived from leaching tests;

LS is the liquid / solid ratio at time t (l/kg) dependent on the amount of infiltration, waste thickness at time t and density (ρ) so ([HER × area × t]/[Wt × area × ρ]) = (HER × t)/(Wt × ρ).

Rearranging this equation therefore results in:

$$t = \frac{Wt.\rho}{HER.\kappa}.\ln\left(\frac{C_0}{C_t}\right)$$

For both methods, the time to achieve the same degree of stabilisation decreases directly as the average infiltration (including groundwater inputs) to the waste increases.

It is expected that most operational sites will have a low L:S ratio, of ≤0.2, and therefore will be located somewhere near the start of the flushing curve (Knox 2004). It is also suggested that further comparison could be made between the L:S ratios for a landfill and time series data of leachate strength for suggested parameters of ammoniacal nitrogen, COD and TOC.

As an additional approach Knox (2004) suggests undertaking a nitrogen balance, as flushing of ammoniacal nitrogen is likely to be the controlling parameter for the indication of waste stabilisation at most current UK landfills. Comparison could be made of ammoniacal nitrogen inputs (in waste) with outputs (in leachate) to indicate whether a site is moving towards or away from stabilisation. Indication of a site moving away from stabilisation (i.e. waste inputs greater than leachate outputs) would be because of little or no flushing.

3.5.2 Approach for Assessing Waste Stabilisation Adopted for the Study

Following a review of existing literature, and taking into account the type and quality of data available for each of the three sites, it was appropriate to tailor the approach of waste stabilisation assessment to each of the three sites under investigation.

For Brogborough, as leachate quality data extends back to 1990, the assessment allowed a broader empirical review of the data to see what if any evidence there was for waste stabilisation. The approach involved assessment of temporal and spatial trends in leachate quality, and comparison of this quality with waste thickness and with estimated liquid / solid ratios. Despite there being an opinion that leachate quality is not a useful indicator of waste stabilisation it was thought necessary to explore all the options available.

The data sets for Poole and Whitehead research landfill sites are more limited and so the empirical approach was more restricted.

Further details on the approaches used can be found in the relevant data assessment sections which follow.

3.6 Selected Indicators of Waste Stabilisation

The indicators thought to be most useful in the process of looking for evidence of waste stabilisation and increased rates of decomposition are:

- Elevated temperatures in leachate, of more than 35°C, as this is an indication of active decomposition. Elevated temperatures may be associated with a significant saturated thickness of waste and significant thickness of overlying unsaturated waste, and may be spatially variable. Temperatures may also be affected locally by groundwater ingress. Further discussion of temperature variations is also given in Chapter 4, on landfill gas.

- Timing of the onset of methanogenesis, as early onset of methanogenesis could indicate accelerated decomposition processes and hence waste stabilisation, through temporal changes in COD concentration and pH.

- Spatial distribution of leachate quality parameter concentrations, as this may correlate to leachate temperature ‘hotspots’, and thus indicate waste stabilisation ‘hotspots’.

3.7 Site Specific Studies - Brogborough Landfill

3.7.1 Introduction

Investigations undertaken by Entec (2004) show the majority of Brogborough landfill is sub water table and that significant parts of the site are also hydraulically contained, with leachate levels lower than piezometric levels in the underlying Kellaways Sands. Leachate quality data assessed for this study are from individual monitoring wells located throughout the site, and span the period 1990 to 2004. Some of these monitoring points are located above the water table and therefore allow comparison with adjacent sub-water table locations. Leachate temperature data, leachate level, waste thickness and leachate parameter concentrations were assessed.

The approach used to look for evidence of waste stabilisation at Brogborough involved analysis of temporal and spatial trends in the available temperature and quality data, in association with leachate level, waste saturated thickness, total waste thickness and liquid to solid ratios (see Box 3.3).

Box 3.3 Calculation of Liquid/Solid Ratios at Brogborough

For each leachate monitoring well, the liquid input aspect of the liquid/solid ratio has been taken from the water balance inputs (maximum effective rainfall + liquid waste inputs) by July 2003 for each area of Brogborough landfill reported in Table 4.1 of Entec (2004). This approach makes the assumption (supported by discussions with site staff) that waste disposal took place evenly across the whole base of each phase rather than being filled progressively from one side to another. With the latter scenario, water inputs to the last filled waste in that area will be less than that assumed for the whole area. It is also assumed that infiltration through the cap is even, whereas studies at Brogborough (DoE, 1995?) have shown evidence of greater infiltration at the cap edge. Finally, it is noted that any groundwater inputs are not included in this water input estimate.

The solid aspect of the liquid/solid ratio has been estimated from waste thickness (ground level less pit base level) at each well and an assumed density of 1000 kg /m^3. Where the waste is relatively thin (<10 m) the liquid/solid ratio is sensitive to small (±2m) uncertainties in pit base elevation.

3.7.2 Leachate Temperature

From the literature review, leachate temperatures can be highly variable and can reflect the rate of degradation. Temperatures of greater than around 35°C are a reflection of rapid waste degradation, and lower temperatures are a reflection of more 'normal' degradation (Robinson, 1995). Therefore temperature variation across the site (i.e. between new and old landfill areas), with time and with depth, could reflect waste stabilisation. Leachate temperature is also potentially linked to waste saturation, thus comparison of leachate level and waste saturation with temperature was also made. Temperature variation could also reflect degrees of water ingress.

Temporal Variation of Leachate Temperature

Initial analysis of a potential link between leachate temperature and leachate level was made, including how these parameters varied with time and with age of waste. Time series plots of leachate level and temperature were constructed for a number of monitoring locations at each stage of landfilling. Data for monitoring locations within each stage of landfilling were plotted on the same graph, and are shown in Figures 3.4a-3.6b for Stages 1, 2, 3A/3B, 3X1, 4A/B and Cell 3X2 of the landfill respectively. These figures show:

- Some evidence of seasonal variability and some evidence of increased variability where the depth to leachate is lowest nearest the pit edge. Seasonal variability is not clearly related to changes in leachate level (i.e. a recharge affect) and so is more likely due to changes in air temperature above the cap.

- Long terms trends are not clear and where present there is no consistent change with leachate level or within one area of the landfill.

Spatial Variation of Leachate Temperature

To assess whether there was a relationship between leachate temperature and monitoring location within the landfill, temperature data for August 2003 were plotted (Figure 3.7). There were insufficient data for July 2003 - the date used for the leachate level rise analysis and spatial assessment in a previous project report (Entec 2004). Leachate temperature data for earlier years were not plotted as insufficient data were available.

Leachate temperatures for August 2003 range from 17.6°C (monitoring well F8, Stage 2) to 62°C (monitoring well J56, Cell 3X1). Assessment of the spatial distribution of leachate temperature shows the temperature is generally higher in the younger and thicker (northern) parts of the landfill and lower in the older and thinner (southern) parts. The temperature of leachate towards the outer edges of the landfill is generally lower than in central areas, and strong evidence for a thermal gradient within the leachate from the middle to the edge of the waste exists between F 1 and F 4 in the southern part of the landfill and generally in the northern part of the site. Steeper gradients are evident in the north. There are a number of potential explanations for the observed temperature variations across the site, and it is likely that a combination of these factors is at work:

- reflection of variable degradation rates within the landfill;

- variation in the thickness of saturated waste, based on the assumption that liquid in waste can encourage degradation, results in variable degradation rate and hence the observed temperature variation;

- variation in the thickness of overlying unsaturated waste, based on the assumption that unsaturated waste acts as insulation, which results in the observed variable leachate temperature;
- variation in the degree of groundwater ingress throughout the site, results in locally lowering leachate temperature.

Note that a significant temperature difference is evident between C13 and C13B (Stage 1), where temperatures are 18 and 30.5°C respectively, even though the monitoring wells are only about 50 m apart. The lower temperature in C13 will not be a result of groundwater ingress, causing subsequent cooling of leachate, as the pit base at this well is above the piezometric level in the Kellaways Sands measured in August 2003 (i.e. this point is above water table).

As there appears to be some connection between leachate temperature and spatial distribution within the landfill, and to enable further comment to be made on why these patterns in leachate temperature are observed, comparison of leachate temperature with unsaturated and saturated waste thickness and assessment of spatial variation in leachate temperature with waste thickness and saturated waste thickness was undertaken.

Comparison of Leachate Temperature with Waste Thickness

To assess the degree of correlation between leachate temperature and unsaturated, saturated (pit base minus leachate level) and total waste thickness, data were plotted for August 2003 and are shown on Figures 3.8a/b and Figure 3.9. The estimation of saturated waste thickness assumes the waste is fully saturated from pit base to the recorded leachate level. It was necessary to estimate the elevation of the pit base for about three quarters of the monitoring locations from a site contour plan, whilst the remainder were obtained from information previously provided by Shanks. Eleven monitoring locations have negative values for saturated thickness due to the estimated pit base elevation, or stated pit base elevation (at four locations), being above the recorded leachate level. This suggests error in the pit base elevation estimate/measurement.

Comparison of the three plots, suggests that temperature correlates most closely to total waste thickness. For the same total waste thickness, temperatures increase from the older Stages 1 and 2 to the more recently filled areas Stage 3X1 and Cells 3 X2. Above a total waste thickness of about 30 m, there is no further increase in temperature. Temperatures are relatively low at locations J49A, J57, LW02A, LW03A and FL12A - this may in part be due to recirculation of leachate into these areas.

Overall, the stronger correlation of temperature with total waste thickness than saturated waste thickness indicates the degree of saturation of the waste is not the dominant control on temperature in the landfill.

Comparison of Leachate Temperature with Liquid/Solid Ratio

Figure 3.10 compares leachate temperature against the liquid solid ratio (see Box 3.3) for each well. The correlation here is strong and stronger than with waste thickness alone. In particular there is no breakdown in correlation as seen previously between waste thickness and temperature above a temperature of ~30°C. This suggests that the liquid/solid ratio is important and that either maximum temperatures are yet to be achieved in the thicker waste, or whether that without sufficient water the exothermic reactions will take place over a longer period and so perhaps not generate higher temperatures.

The trend shown on Figure 3.10 extrapolates back to temperatures of ~70 to 80°C at very low liquid/solid ratios. From Section 3.4.2 these temperatures are consistent with those generated during Stage 1 (hydrolysis and aerobic degradation) of waste stabilisation.

The liquid/solid ratio of 0.05 equates to an absorptive capacity of 5% for the waste, the point above which free leachate should be produced and so is consistent with free leachate in the wells.

3.7.3 Leachate Quality Variations Across Brogborough

As discussed in Section 3.3.2, leachate quality may not be a good indicator of the degree of waste stabilisation, however for completeness the available data have been assessed.

Spatial Variation in Leachate Quality

Assessment of the spatial variation of leachate quality was undertaken by plotting the minimum, maximum and mean leachate concentrations of BOD, COD, TOC, chloride, ammonia, pH and potassium for 2003 data. There were insufficient data locations to plot leachate concentrations prior to 2003.

In general, concentrations of BOD, COD and TOC are higher in the younger and thicker northern and central areas of the landfill (wells FL12A, J40A, H110) and lower in the older southern areas. Concentrations are also lower in younger areas close to the perimeter of the landfill (wells S1, Q1, H109). The exception is C13, located in the older part of the landfill (Stage 1), where high concentrations of BOD, COD and TOC were recorded in comparison to nearby monitoring locations. This is coincident with a relatively low leachate temperature of 18°C and a mean pH of just less than 7, indicating that degradation at this location is probably slower than other locations in Stage 1. Spatial variation of ammonia, chloride, and potassium is much less marked than that seen for BOD, COD and TOC, although shows a similar pattern.

Temporal Variation in Leachate Quality

Figures 3.11a/b and 3.12a/b show the variation in chloride, ammoniacal nitrogen, COD and BOD concentrations with the age of the waste (time elapsed between start of landfilling in the area of each leachate well and the sample date).

The charts show, with the exception of wells H109-H111, a broad trend of decreasing concentrations with age of waste. There is however significant variability between wells.

Variation in Leachate Quality with Liquid/Solid Ratio

The most recent water quality data for each well, checked for representativeness compared to other data at that well, were selected and a liquid/solid ratio was calculated for the relevant sampling date. This required a correction of the water balance inputs as of July 2003 (see Box 3.3 in Section 3.7.1 including discussion of uncertainties) by adding or subtracting an amount of infiltration to the capped (50 mm/yr) or uncapped (300 mm/yr) waste to take account of sample dates which were earlier or later than July 2003.

Using this approach, concentrations/values of pH, electrical conductivity (EC), chloride, ammoniacal nitrogen, COD, BOD, total organic carbon (TOC) and alkalinity are compared against liquid / solids ratios on Figures 3.13 and 3.14. Each figure has been overlayed by a best estimate location of the different stages of waste stabilisation as shown on Figure 3.2 and described in Section 3.4.2. The qualitative time scale on Figure 3.2 has been converted to a liquid/solid ratio scale on Figures 3.13 and 3.14 by best fitting the COD profile. A perfect match is not expected given the scale is unlikely to be linear.

These figures show:

- The majority of the pH values are above pH 7 and there is a broad trend of decreasing pH with increasing liquid/solid ratio. The low pH of Stages II (hydrolysis and fermentation) and III (acetogenesis) from Figure 3.2 is not apparent in the Brogborough data suggesting the system is well buffered.
- EC rises initially (Stage II to III) and then declines to low values by a liquid solid ratio of ~0.35.
- Chloride concentrations increase initially and then decline, although there is significant scatter in the data. The peak chloride concentration is also at a higher liquid / solid ratio than suggested by the data from trends of Figure 3.2. Chloride concentrations are examined in more detail below.
- Ammoniacal nitrogen concentrations are highest at low liquid/solid ratios equivalent to Stage II and III. From Section 3.4.2, it is Stage II that gives rise to high ammoniacal nitrogen concentrations. The decline in ammoniacal nitrogen concentrations is evident, but there is significant scatter.
- COD increases from Stage II to III and peaks at a liquid/solid ratio of ~0.06 and then shows an exponential decline (± some scatter and anomalies).
- BOD and TOC both show a similar trend to COD, perhaps with a later peak concentration.
- Alkalinity shows a gradual decline from a peak at liquid/solid ratios of ~0.04. There are a few anomalies, in particular wells Q4, Q5 and S1 from Stage 4B.

Overall the changes in leachate quality are broadly consistent with the expected changes as illustrated on Figure 3.2 and discussed in Section 3.4.2.

Evidence for Additional Water Inputs

The variation in chloride concentrations with liquid/solid ratios is examined in more detail in Figure 3.15. Two model predictions of changes in chloride concentration from an assumed peak concentration of 7500 mg/l have been made using the (LandSim v2.5) equation shown in Box 3.2 and kappa values of 7.5 and 18.5*.

Note: *The LandSim v2.5 default kappa value for chloride derived from laboratory column experiments with high flushing rates is close to ~1, so the need to use these higher kappa values to fit the data means either significant water inputs have been neglected in the liquid/solid ratios or that the laboratory experimental results do not translate well to the site scale where flushing rates are much lower.

The different kappa values needed to fit the possible two groupings of data could be related to:

- inaccuracies in pit base data which affect waste thicknesses and thus the solid part of the liquid/solid ratio;
- inaccuracies in the period of landfilling which controls uncapped rainfall inputs;
- higher cap infiltration, recirculated leachate or groundwater ingress in the kappa = 18.5 grouping.

Figure 3.16 marks those wells which have relatively high and low liquid/solid ratios (based on liquid inputs of rainfall infiltration and liquid waste inputs only) for the leachate chloride concentrations measured circa 2003. Recognising that this analysis is stretching the confidence and variability in the data, the following is however noted:

- Those wells with a relatively high liquid / solid ratio versus chloride concentration relationship (on the kappa = 7.5 line on Figure 3.15) are situated on the margins of the site and in a number of cases (C13, C8B, C8 and F16) at locations which are above the 2003 piezometric level in the Kellaways Sand, i.e. are above ‘water table’.
- Those wells with a relatively low liquid / solid ratio versus chloride concentration relationship (around the kappa = 18.5 line on Figure 3.15) are situated in the centre of the site or at the margin, but at locations which have their pit base below the 2003 piezometric level in the Kellaways Sand.

Notable comparisons include C13 / C13B and SE6B/SE7B in Stage 1, F10/F16/H101 in Stage 2/3, but there are also wells on the pit edge for both groups.

Overall the analysis suggests that wells in areas where the waste is sub-water table/hydraulically contained require additional water inputs over and above rainfall/cap infiltration and liquid waste inputs to achieve the same trend in leachate quality (waste stabilisation) with liquid / solid ratios. From the difference in apparent kappa values, it appears that the total water inputs for the sub-water table (presumably hydraulically contained) are (18.5/7.5=) 2-2.5 times the above water table inputs, or that the additional (groundwater?) inputs are about the same as rainfall infiltration and liquid waste inputs. Such inputs are consistent with the estimates of groundwater inputs from leachate level rise analysis and a water balance approach (Entec, March 2004). So although the analysis has a number of uncertainties, the information appears to suggest that **additional groundwater inputs appear to help improve leachate quality/ waste stabilisation**.

3.8 Site Specific Studies - Poole Landfill

3.8.1 Introduction

At Poole Landfill, infiltrated water and water that has entered the landfill through groundwater ingress is pumped out in large (~300 m^3/day) quantities. If this water has flushed through the waste, a high degree of flushing of contaminants is predicted to occur. If however, significant groundwater has entered the basal drainage system and then been pumped out without contact with the waste, limited waste stabilisation would be expected. Instead, significant dilution of the leachate created by cap infiltration interaction with the waste would be expected. This section examines the evidence and makes predictions based on the limited available data.

3.8.2 Data Availability and Approach

Leachate quality data are available for the Poole Landfill, but as a dataset are poor compared to those for Brogborough in terms of number of parameters (chloride has only intermittently been analysed), continuity and availability for the early periods of landfilling. The site operators installed additional monitoring wells in late 2004, and these have allowed collection of improved leachate quality data.

Unlike at Brogborough, there are also no temperature data available for Poole landfill.

Due to the paucity of data, it has not been possible to follow a detailed empirical approach, although available data have been reviewed and interpreted in a similar way to the previous assessment on Brogborough Landfill.

In assessing changes in leachate quality, it is noted that from Table 2.2 of Entec (September 2003), that proportions of household, to commercial & industrial, to demolition and construction varied from 45:20:25 in Phase 1, to 20:35:45 in Phases 2 and 3 (and assumed for Phase 4). More than double the proportion of household waste in Phase 1 may therefore be expected to affect the leachate quality in that area of the site.

3.8.3 Available Leachate Quality Data

Leachate quality data are limited to:

- Pumped samples from the ‘vertical well’ that drains from beneath Phases 1 to 3 and from the ‘inclined well’ that drains from beneath Phase 4. A leachate balance analysis (Entec, December 2003) has shown that the volumes of leachate pumped cannot be provided by rainfall infiltration to the waste alone and it is highly likely that there is a significant groundwater component to the leachate pumped out. This means there is a potential for the leachate pumped from these wells to be diluted. The determinands monitored in these wells include ammoniacal nitrogen, COD, sulphide, with rare measurements of chloride, electrical conductivity and pH. Data are available from 1996-2002.

- Spot measurements from 10 retrofit leachate wells drilled and constructed in June 2004 (Frederick Sherrell Ltd, August 2004). Data were made available for the samples collected in September 2004 and analysed for ammoniacal nitrogen, chloride and $BOD_{ATU.}$

Locations of monitoring points are shown on Figure 3.17 and the data for the spot measurements of leachate are presented in Table 3.2.

3.8.4 Spatial Variations in Leachate Quality

Figure 3.17 shows concentrations of ammoniacal nitrogen, chloride and BOD for the 10 retro-fit leachate wells constructed in June 2004 and sampled in September 2004. This shows:

- ammoniacal nitrogen concentrations are lower in Phases 1 and 2, higher in Phase 3 and highest in Phase 4;

- biochemical oxygen demand (BOD) concentrations are also generally, if not consistently, higher in Phase 1 to 3 than in Phase 4.
- chloride concentrations do not show any obvious spatial trends.

3.8.5 Variations in Leachate Quality with Time

In terms of time series leachate quality data, the longest running data sets are for chemical oxygen demand (COD), ammoniacal nitrogen and sulphide at the vertical well (leachate from Phases 1-3) and at the inclined well (Phase 4) and these are illustrated on Figures 3.18a-c.

Figure 3.18a shows COD concentrations in Phase 1-3 are generally below 100 mg/l O_2. In Phase 4 concentrations fall from 3-4000 mg/l O_2 to values of <500 mg/l O_2 in 1997 and then climbing back to a steadier value of ~1400 mg/l O_2 until the end of the data record in 2002.

Figure 3.18a also shows a dilution factor calculated from the quotient of actual volumes of leachate extracted divided by predicted leachate volumes based on rainfall infiltration alone. Both data sets were discussed and presented in Entec (December, 2003) and are for 13 week (quarterly) rolling averages. This suggests that the initial high values of COD in Phase 4 are likely to be close to the undiluted maxima for the samples, dilution then causing the initial lowering of concentrations and the undiluted COD concentration in 2002 is likely to be approximately (1330 x ~1.5=) 2000 mg/l O_2.

Table 3.2 Leachate Quality and Liquid/Solid Ratios for Retrofit Leachate Wells at Poole Landfill

hm-250\08465\data\quality.xls-Table 3.2 (Page 1 of 2) hm-250\08465\data\quality.xls-Table 3.2 (Page 2 of 2)

Figure 3.18b shows some seasonality in both the Phases 1-3 and Phase 4 leachate ammoniacal nitrogen concentrations. This is likely to be related to dilution. There is no clear evidence of an improvement in leachate quality in Phase 4, but possibly a slight improvement in Phases 1-3. Concentrations in the pumped leachate are significantly lower than those sampled in the retrofit wells (see Section 3.8.4, Figure 3.17) and this further supports the water balance findings that groundwater inputs account for a significant proportion of the pumped leachate volumes and as such provide dilution of the leachate draining from the waste.

Figure 3.18c shows that sulphide concentrations are much higher in the leachate from Phase 4 than Phases 1-3. Neither short term nor long term trends are obvious.

3.8.6 Variations in Leachate Quality with Liquid : Solid Ratio

As for the assessment of Brogborough Landfill, leachate quality has been compared to the liquid to solid ratio for each of the retrofit wells. Box 3.4 describes the approach for calculating liquid to solid ratios for the Poole Landfill and calculated values are presented in Table 3.2.

Box 3.4 Calculation of Liquid/Solid Ratios at Poole Landfill

For each of the retrofit leachate monitoring wells, the liquid input aspect of the liquid/solid ratio was estimated from the duration its area of the landfill was uncapped multiplied by an uncapped infiltration rate plus the time that area of the landfill has been temporarily (unengineered, unrolled) capped multiplied by an assumed cap infiltration. This approach makes the assumption that waste disposal took place evenly across the whole base of each phase rather than being filled progressively from one side to another. Discussions with Wyvern Waste indicate this assumption is valid.

Groundwater inputs have not been included in this water input estimate.

The solid aspect of the liquid/solid ratio has been estimated from waste thickness based on (a) drilled depth (three wells penetrated the pit base and the remainder were reported to be completed in compact waste), and (b) pre-drilling predicted depth of waste (based on ground level less pit base level) at each well. In both cases a waste density of 1000 kg /m^3 has been assumed. In some cases there are significant differences between the predicted waste thickness and drilled depth which may be due to uncertainties in pit base elevation, changes in location or that the drilled wells did not reach pit base.

Liquid solid ratios have not been calculated to compare against the pumped leachate quality of Phases 1-3 and Phase 4 due to the complexity of the infiltration history over a number of different phases and the dilution of the leachate quality by likely groundwater inputs.

Figures 3.19a-c show the change in leachate quality (data only for ammoniacal nitrogen, chloride and BOD) with calculated liquid to solid ratios for the retrofit wells. Each figure also shows the boundaries for different stages of waste stabilisation as deduced for the Brogborough site for comparison. These figures show:

- Ammoniacal nitrogen concentrations generally fall with increasing liquid to solid ratio.
- BOD concentrations fall exponentially in Phase 4 with liquid to solid ratios. Phase 1-3 leachates fit a similar pattern to the Phase 4 leachates except at LW24 which is the area of Phase 1 which has been overtipped. The anomalous leachate quality at LW24 may therefore be related to a mixing of older, higher L:S ratio leachates with younger lower L:S ratio leachates.

- Chloride concentrations possibly fall with increasing liquid to solid ratio in Phase 4, but overall there is poor correlation.
- Where there are changes in leachate quality with increasing liquid to solid ratio, then these changes appear to occur at a higher liquid to solid ratio than at Brogborough (in both cases ignoring groundwater inputs in the liquid aspect of those ratios).

Overall, changes in leachate quality appear to be related to the ratio of infiltration inputs to waste thickness and there is no obvious need to invoke groundwater inputs.

3.8.7 Summary of Leachate Quality / Waste Stabilisation at Poole Landfill

Leachate sampled from retrofit wells completed in the waste has significantly higher concentrations of ammoniacal nitrogen than monitored in the contaminated water pumped from Phases 1-3 (vertical pump) and Phase 4 (inclined pump). This is consistent with the water balance findings (Entec, December 2003) that the water pumped from beneath the site contains a significant groundwater component.

Leachate quality is generally poorer in Phase 4 than in the older phases of landfill at the site and appears to be related to liquid to solid ratios calculated based on rainfall/cap infiltration to the waste rather than including any groundwater inputs. Overall, flushing of contaminants from the waste and waste stabilisation at Poole appears to be related to infiltration inputs and not to groundwater inputs. Groundwater inputs largely appear to dilute the leachate.

3.9 Site Specific Studies - Whitehead Landfill

3.9.1 Background

Data indicate Whitehead Landfill is partially hydraulically contained, with marginal hydraulic containment in the north of the landfill (Stage 1) and hydraulic containment in Stage 2 and southern areas (Stages 3 and 4) (Entec 2003). Data suggest there has been little or no groundwater ingress to the landfill, except between April 2001 and April 2003, during the construction and filling of Stage 4, when large amounts of water believed to be from the underlying sandstone aquifer, were pumped from this area of the site. Apart from rainfall and liquid waste, the only other source of liquid to the landfill has been leachate re-circulation, indicated by site data to have been undertaken in 2003 (around 56 000 m^3) and in 2004 (around 54 000 m^3), where estimated input was greater than discharge to sewer.

3.9.2 Available Data

Leachate temperature data were provided by Viridor Waste Management for a number of leachate wells in 2004. Leachate quality data were also provided for four leachate wells completed in the drainage blankets of four cells and for the combined raw leachate discharge. Locations of wells are shown on Figure 3.20. Data for a wide range of chemical parameters were made available, including the key parameters pH, chloride, ammonia, BOD, COD and TOC. The four leachate wells for which leachate quality data were provided are:

- AG501LM (Stage 1, Cell 1A);

- AG502LM (Stage 1, Cell 1B);
- AG307LM (Stage 2, Cell 2A);
- AG311LM (Stage 3, Cell 3A).

It is understood that these cells were filled separately and are separated from adjacent cells by internal bunds. This, together with there being a leachate drainage blanket means the leachate sampled within each well is likely to be representative of the average quality of leachate within that cell, rather than being the quality of leachate local to a retrofit well completed in waste. Data in 2003 and 2004 are, however, possibly affected by leachate recirculation.

The quality of the combined untreated leachate pumped from the site will predominantly reflect the quality of leachate draining from the earlier cells in the first years and then progressively reflect the quality of the mixture of leachate from older and more recently filled cells and the affect of recirculation.

3.9.3 Leachate Temperature

Figure 3.21 presents the limited leachate temperature data for the Whitehead site and these are summarised in Table 3.3. Locations of monitoring wells with leachate temperatures recorded on 5 October 2004 are shown on Figure 3.20.

Table 3.3 Leachate Temperature at Whitehead Landfill Site

Landfill Area	Leachate Wells	Start of Landfilling	Temperature (^oC) [a]
Stage 1		1998	No Data Available
Stage 2	AG307LM	End of 1999	27.5 – **31.0** – 35.0
Stage 3	AG311LM	End of 2000	23.3-**26.8**-35
Stage 4	AG328LM & AG332LM	End of 2001	23.1 – **27.0** – 31.4

Notes: [a] Data shown as minimum - **mean** - maximum for the period March-November 2004.

With the limited leachate temperature data available, it is difficult to draw any conclusions from the data set out above. Stage 1 is the non-hydraulically contained stage, but no leachate temperature measurements have been made in this area. Where data exist for Stages 2 to 4, all hydraulically contained, but progressively so, the temperatures are comparable, although slightly higher in Stage 2. The waste in AG307LM (Stage 2) is however noted to be ~23 m thick compared to thicknesses in 2004 for the Stage 3 and 4 wells of ~7 to 11 m, so thickness may be the principal cause of any temperature difference. The leachate temperatures for these thicknesses of waste are similar to those found at Brogborough (see Figure 3.9).

3.9.4 Spatial Variations in Leachate Quality

As there are leachate quality data for only four monitoring wells at Whitehead and the sampling dates for these wells do not overlap for more than three wells, detailed assessment of spatial variations in leachate quality has not been undertaken. Recirculation (2003 and 2004) of leachate from one cell into another is also likely to mask leachate quality characteristics particular to one cell for these years.

3.9.5 Variations in Leachate Quality with Time

Figures 3.22a-d and 3.23a-d show the variations in leachate quality for a range of parameters with time. They show:

- Cell 1A and 1B leachate has very similar leachate quality.
- Early leachates from Cells 1A and 1B had low pH values and high concentrations of BOD, COD and TOC characteristic of Stage 2 and 3 (pre-methanogensis) waste stabilisation, as discussed in Section 3.4.2.
- Electrical conductivity, alkalinity and chloride concentrations decrease in Cell 1A suggesting some flushing of the wastes.
- There is a marked deterioration in leachate quality in Stages 2 (AG307LM) and 3 (AG311LM) and in the pumped leachate from the start of 2003. It is likely that the increases in concentrations of chloride, ammoniacal nitrogen, COD, BOD and TOC could be related to recirculation during 2003 and 2004 of untreated leachate. Given the increase in COD, BOD and TOC it also appears likely that a significant volume of leachate being recirculated is from recently filled cells undergoing acetogenic waste degradation. Chloride and ammoniacal nitrogen concentrations are compared to the annual water balance estimates discussed in Section 2.4.5 in Figures 3.24a and b respectively.

Overall the limited time series leachate quality data do not allow identification of any relationships with the degree of hydraulic containment at the Whitehead site.

3.9.6 Variations in Leachate Quality with Liquid to Solid Ratio

It was not possible to gain detailed information on as-built pit base to help define typical waste thicknesses in each area of the Whitehead Landfill. There is also significant uncertainty in the date of filling, restoration and capping and volume of liquids taken. These factors, together with the limited availability of spatial leachate quality data for the site, has meant that it has not been possible to compare leachate quality to the calculated liquid solid ratios as undertaken for the Brogborough and Poole sites.

3.9.7 Summary of Leachate Quality/Waste Stabilisation at Whitehead Landfill

Limited availability of leachate quality and temperature data together with uncertainties related to periods of landfilling and capping at the Whitehead Landfill site mean that it has not been possible to make comparisons between the leachate in the non hydraulically contained Stage 1 and the sub-water table and hydraulically contained Stages 2 to 4.

Significant efforts were made to address these uncertainties and data limitations but without success. Theoretical assessments of the influence of groundwater ingress were also made but without sufficient data and certainty to calibrate these predictions, such assessments do not further the understanding of these sites.

3.10 Summary of Waste Stabilisation

In "*A review of the composition of leachates from domestic waste in landfill sites*" (Robinson, 1995, p C64), it was noted that bioreactor landfills in the UK have arisen, albeit inadvertently, through the uncontrolled ingress of large quantities of water. Often this has taken the form of groundwater ingress rather than high rates of infiltration and percolation of rainfall. This suggests that groundwater ingress, as expected to occur in hydraulically contained sites, by no means leads to slow rates of waste stabilisation.

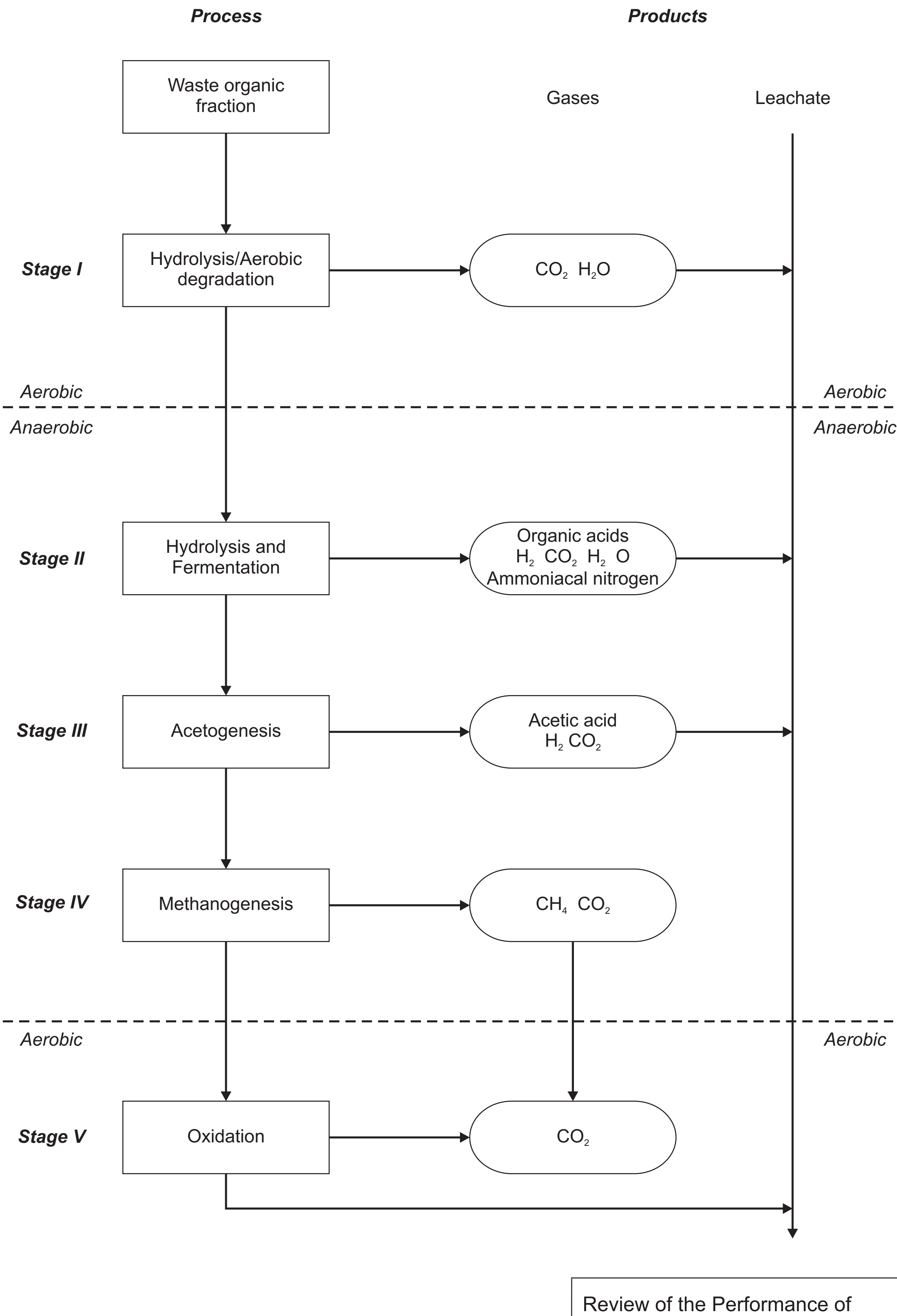
The empirical assessment of the three research sites has shown:

- **At Brogborough -** although there are significant uncertainties in the analysis, it appears that the improvement in leachate quality at wells which are sub-water table/hydraulically contained is quicker than for wells with the same rainfall infiltration + liquid waste inputs to waste thickness (liquid/solid) ratio that are located above the water table. That is, likely groundwater ingress inputs appear to help improve leachate quality in the waste compared to waste that receives only rainfall infiltration.

- **At Poole -** leachate sampled from retrofit wells completed in the waste has significantly higher concentrations of ammoniacal nitrogen than monitored in the contaminated water pumped from Phases 1-3 (vertical pump) and Phase 4 (inclined pump). This is consistent with the water balance findings (Entec, December 2003) that the water pumped from beneath the site contains a significant groundwater component.

  Leachate quality is generally poorer in Phase 4 than in the older phases of landfill at the site and appears to be related to liquid to solid ratios calculated based on rainfall/cap infiltration to the waste rather than including any groundwater inputs. Overall, flushing of contaminants from the waste and waste stabilisation at Poole appear to be related to rainfall infiltration inputs and not to groundwater inputs. Groundwater inputs largely appear to dilute the leachate.

- **At Whitehead -** no conclusions can be drawn due to the limited data set and uncertainties in the details of the water balance and waste thicknesses in different parts of the site.

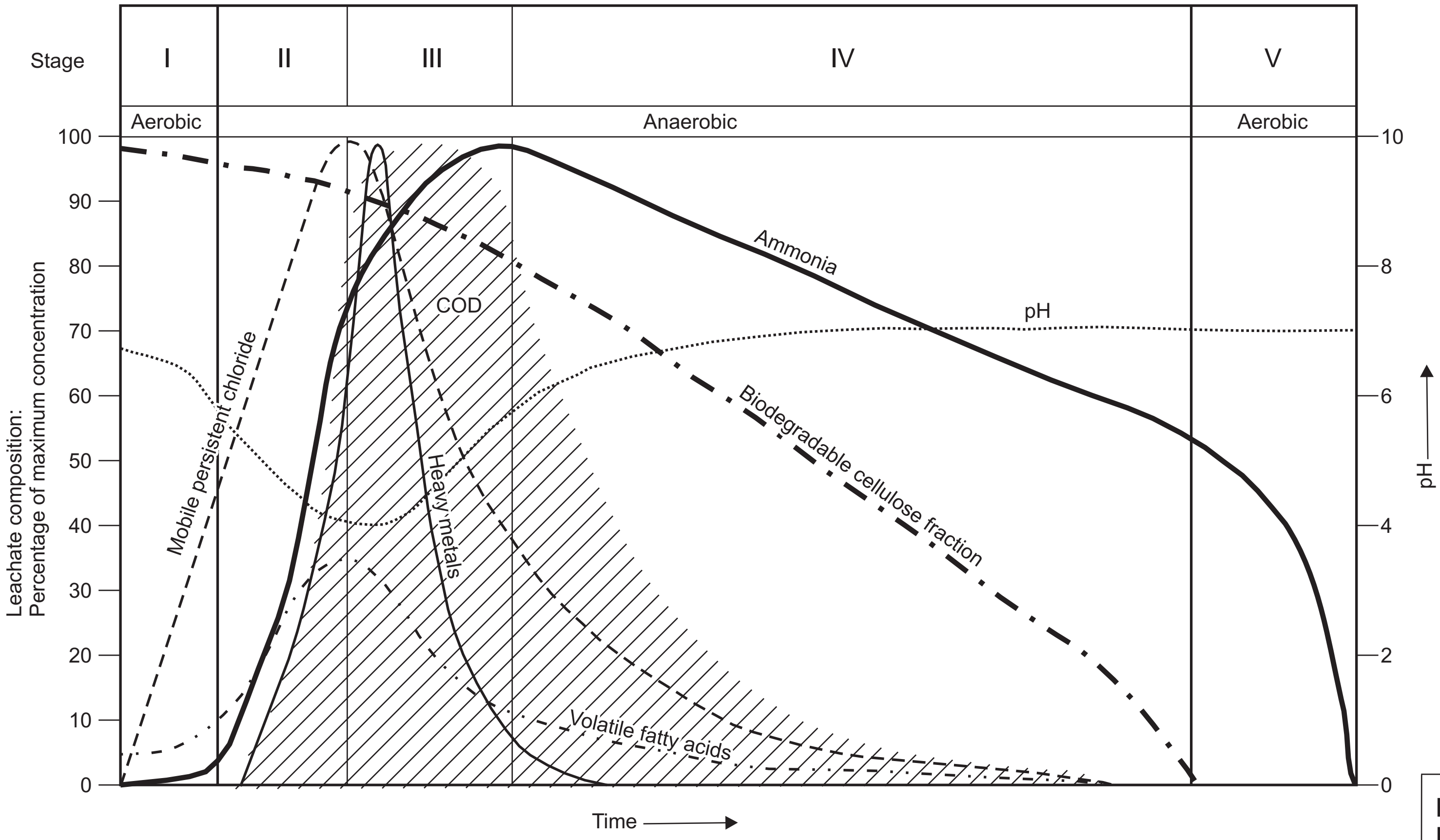


Review of the Performance of Hydraulically Contained Landfills

Figure 3.1
Major Stages of Waste Degradation
(from Waste Management Paper 26B, DOE, 1995)

November 2005
10744-S46.cdr wrigs02

Entec

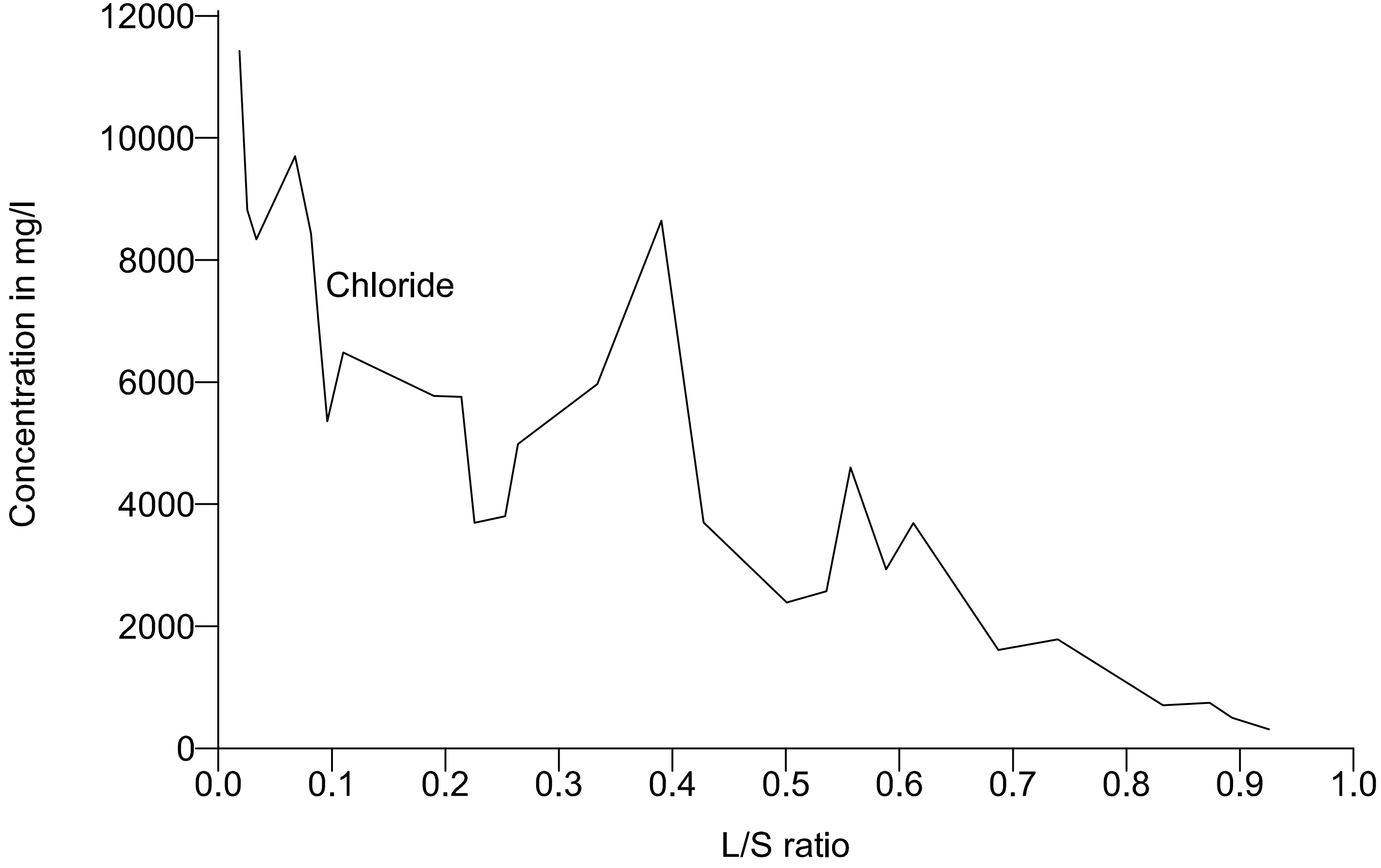


Review of the Performance of Hydraulically Contained Landfills

Figure 3.2
Changes in Composition of Leachate
(from Waste Management Paper 26B, DOE, 1995)

November 2005
10744-S47.cdr wrigs02

Entec



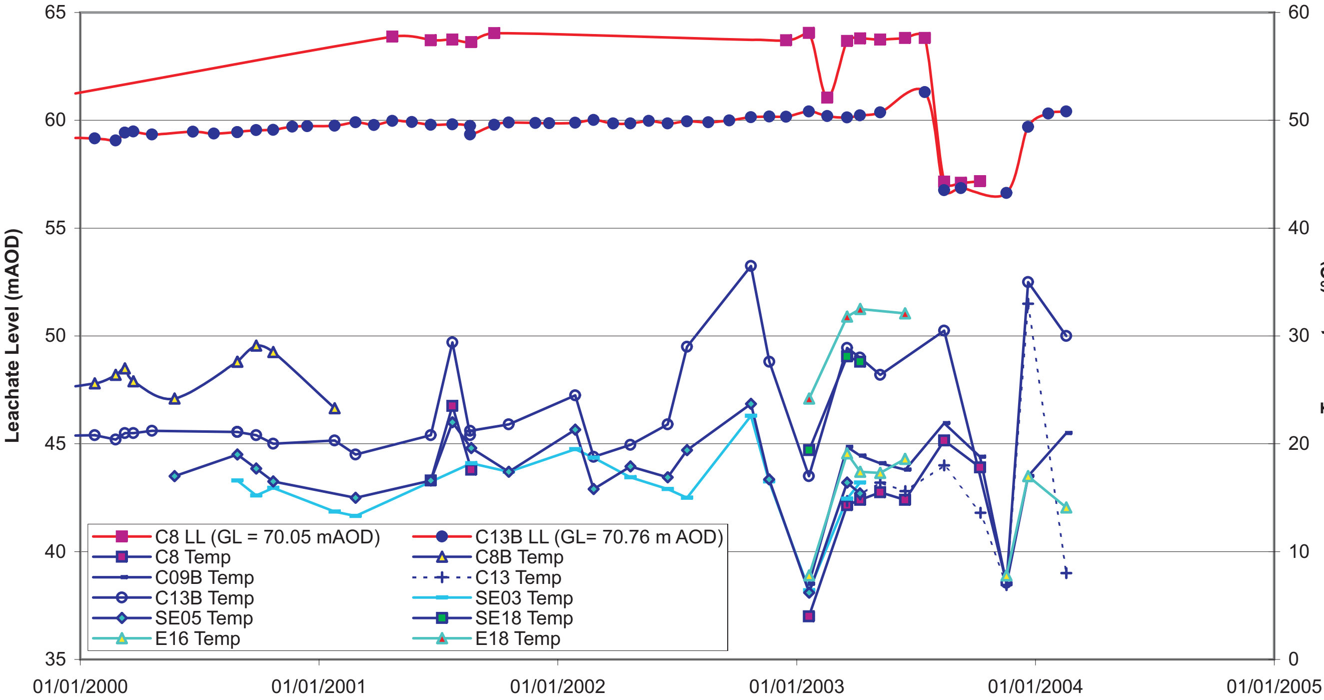
Review of the Performance of Hydraulically Contained Landfills

Figure 3.3
Chloride Concentration at Vestskoven Landfill, Denmark (Hjelmar, 2004)

November 2005
10744-S73.dwg xgeort

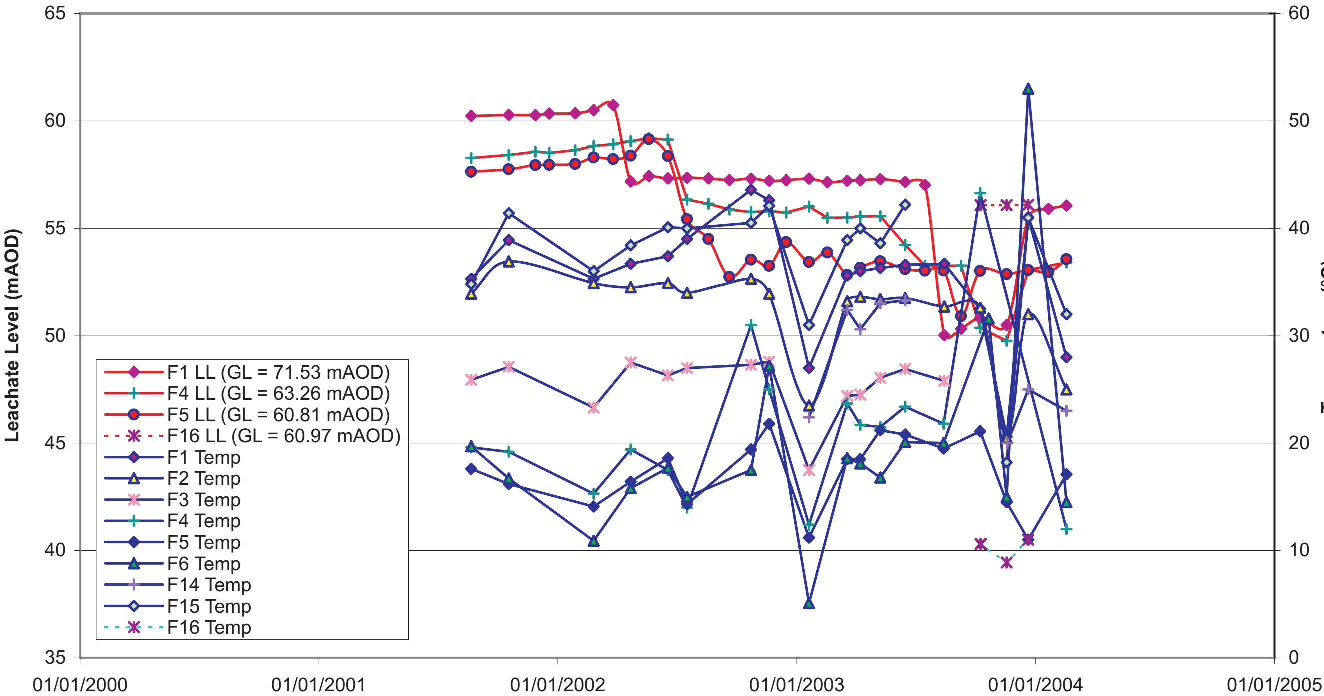
Entec

Figure 3.4a: Stage 1 (Filled Feb 1983 to June 1987)



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Data\Lch Temperature\LeachateLevel V Temp.xls *Stage 1*

Figure 3.4b: F Series - Stage 2 (Filled June 1987 to May 1990)



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Data\Lch Temperature\LeachateLevel V Temp.xls *Stage 2*

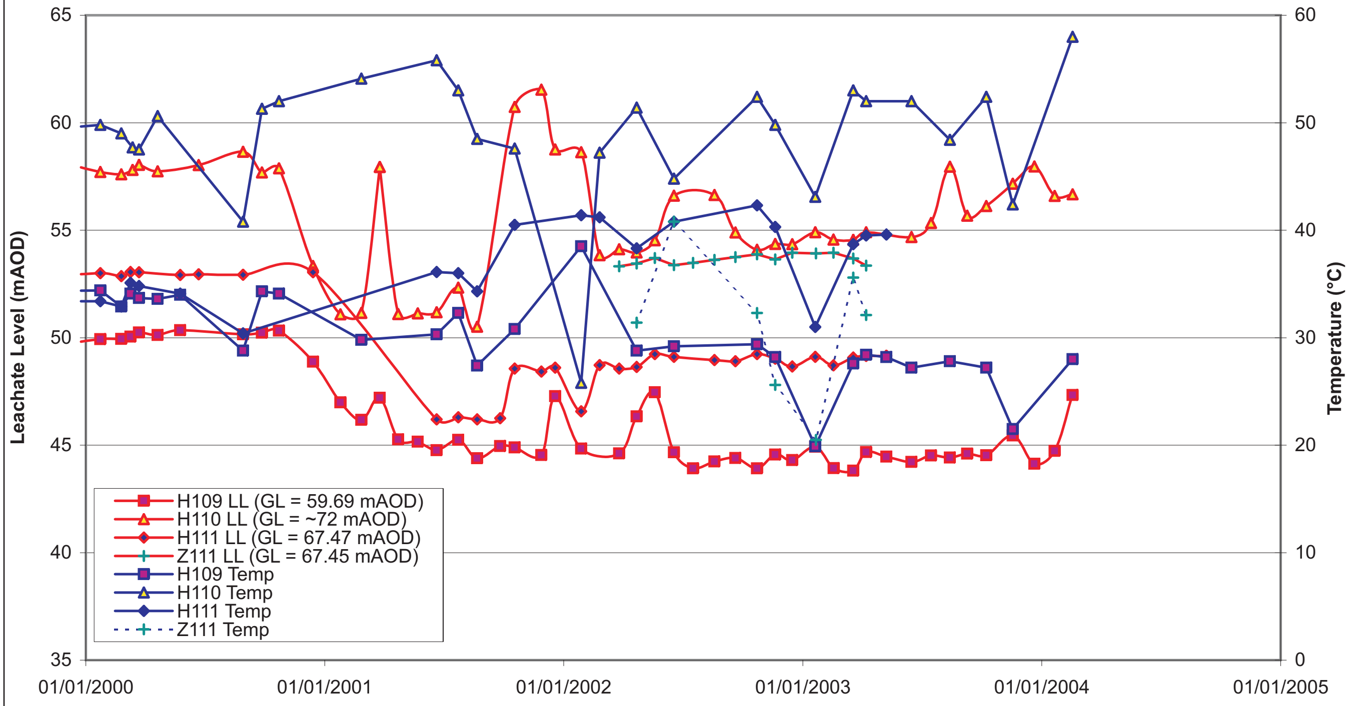
Review of the Performance of Hydraulically Contained Landfills

Figure 3.4
Variation of Leachate Levels and Temperatures in Stages 1 & 2 at Brogborough

November 2005
10744-S48.cdr wrigs02

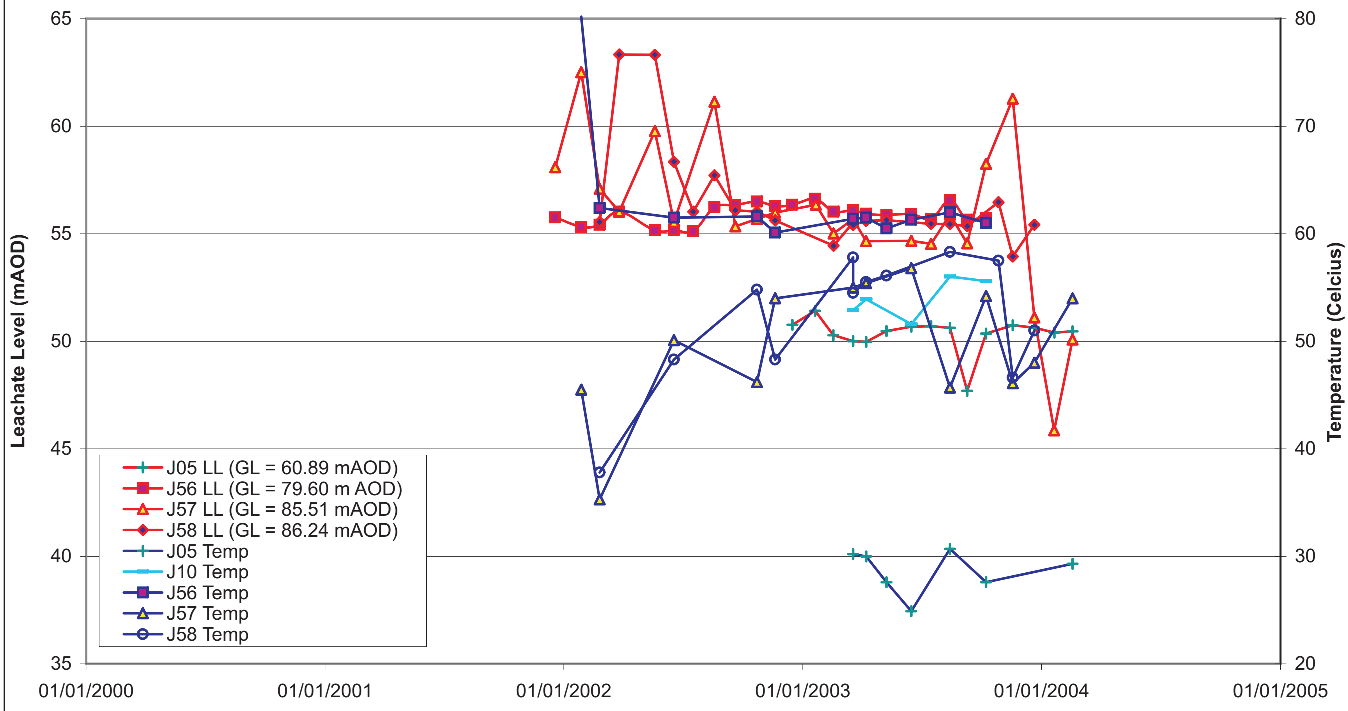
Entec

Figure 3.5a: Stage 3A/B (Filled Early 1990)



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Data\Lch Temperature\LeachateLevel V Temp.xls *Stage 3*

Figure 3.5b: Stage 3X1 (Filled Dec 1990 to Aug 1999)



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Data\Lch Temperature\LeachateLevel V Temp.xls *Stage 3X1*

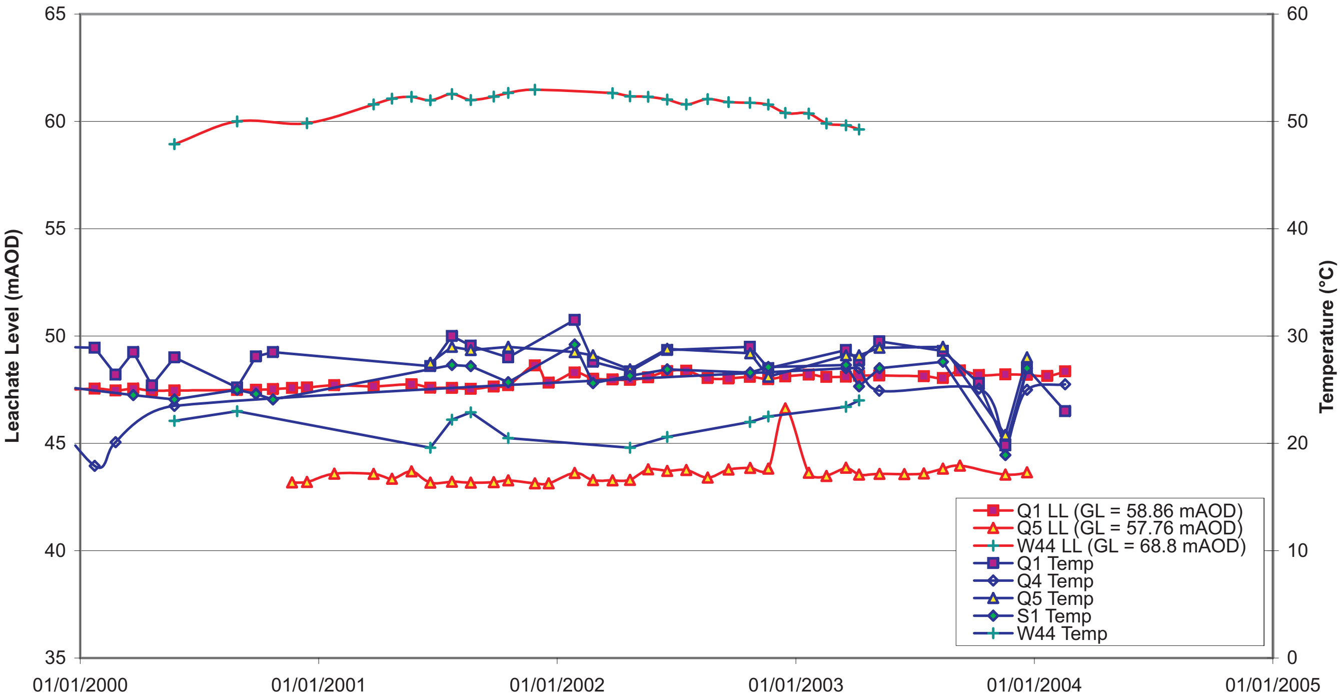
Review of the Performance of Hydraulically Contained Landfills

Figure 3.5
Variation of Leachate Levels and Temperatures in Stages 3A/B & 3X1 at Brogborough

November 2005
10744-S49.cdr wrigs02

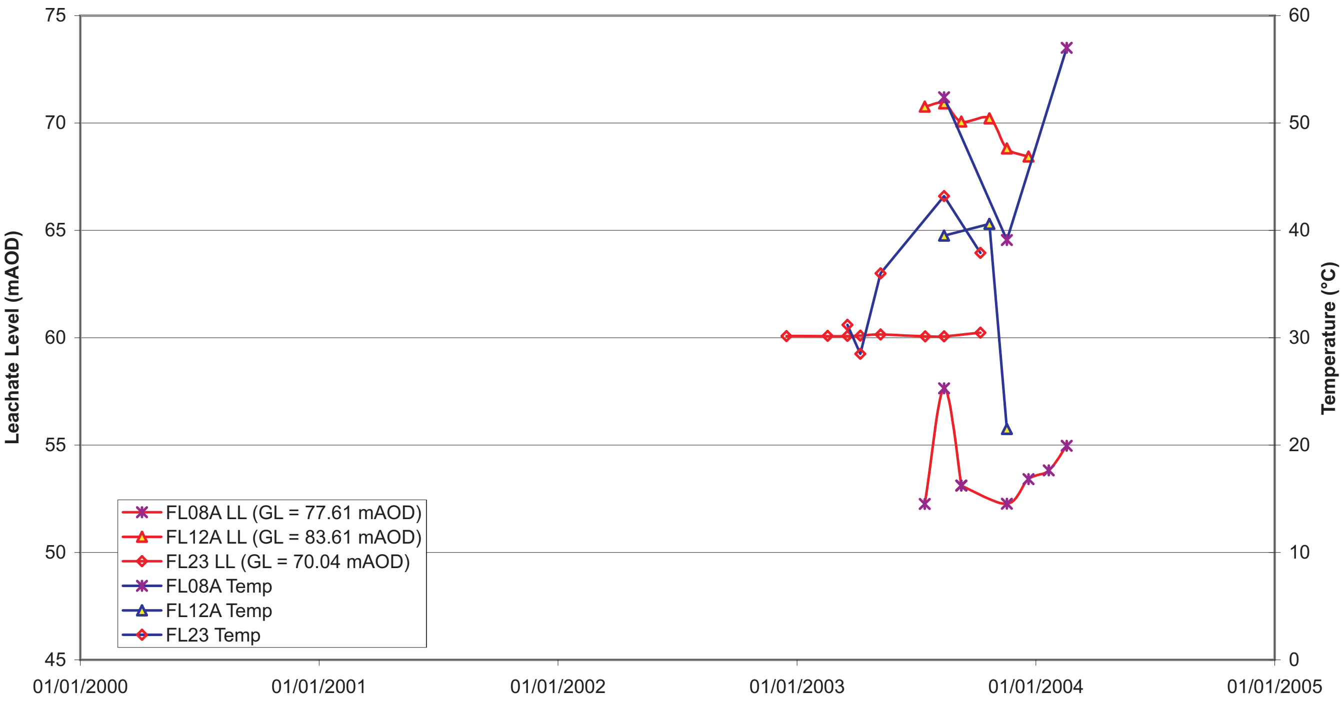
Entec

Figure 3.6a: Stage 4A/B (4A Filled 1990-1992 and 1995-2001, 4B Filled Nov 1995 to 2001)



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Data\Lch Temperature\LeachateLevel V Temp.xls *Stage 4A/B*

Figure 3.6b: Cell 3X2 (Filled Oct 1997 to Aug 1999)



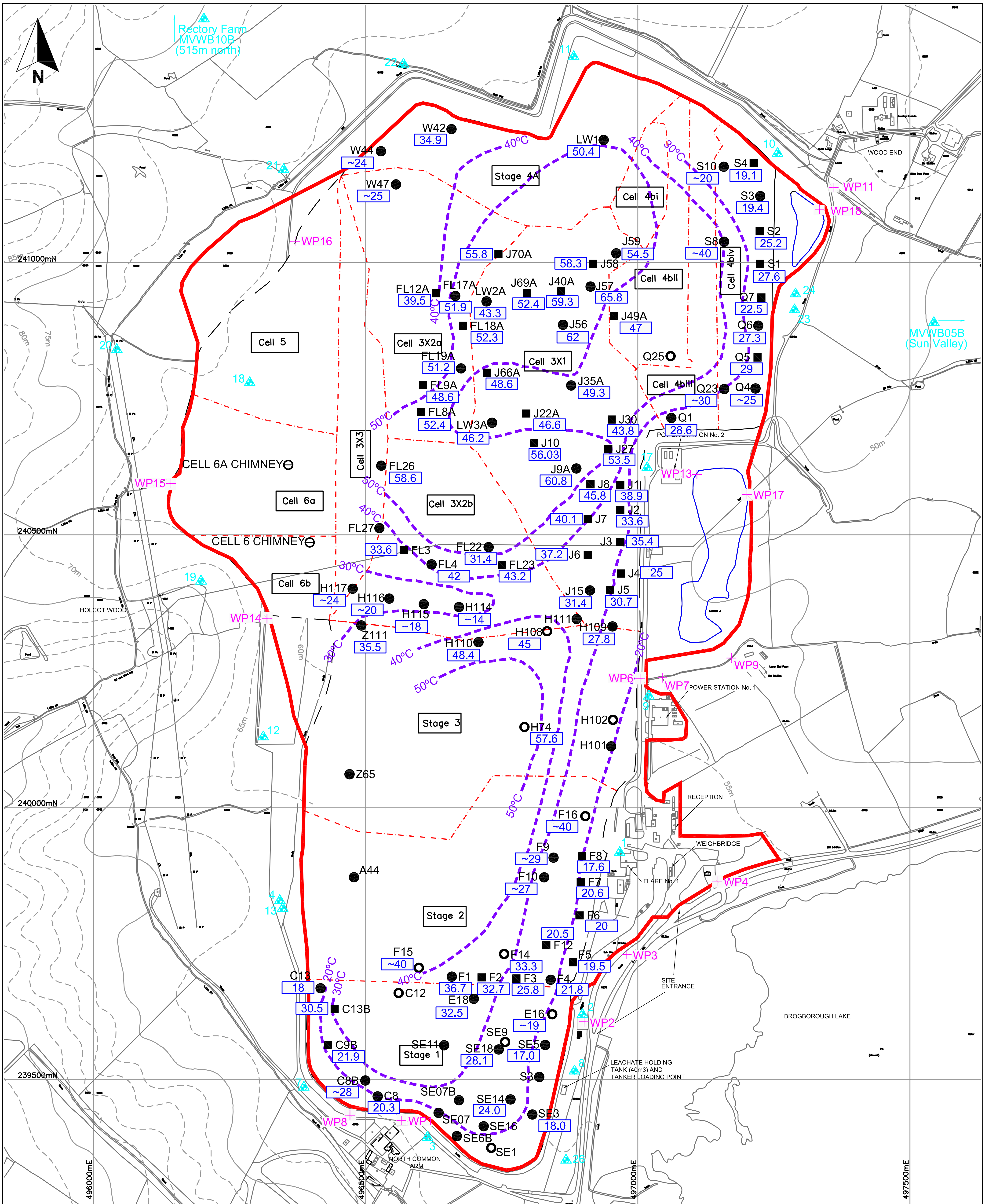
H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Data\Lch Temperature\LeachateLevel V Temp.xls Cell *3X2*

Review of the Performance of Hydraulically Contained Landfills

Figure 3.6
Variation of Leachate Levels and Temperatures in Stages 4A/B and Cell 3X2 at Brogborough

November 2005
10744-S50.cdr wrigs02

Entec



Key

Licence boundary	Leachate sampling well
Cell boundary	Leachate chimney
Groundwater borehole	Well
Surface water monitoring point	47.36 Leachate temperature (°C)
In-waste gas well	Leachate temperature contours (10°C intervals)

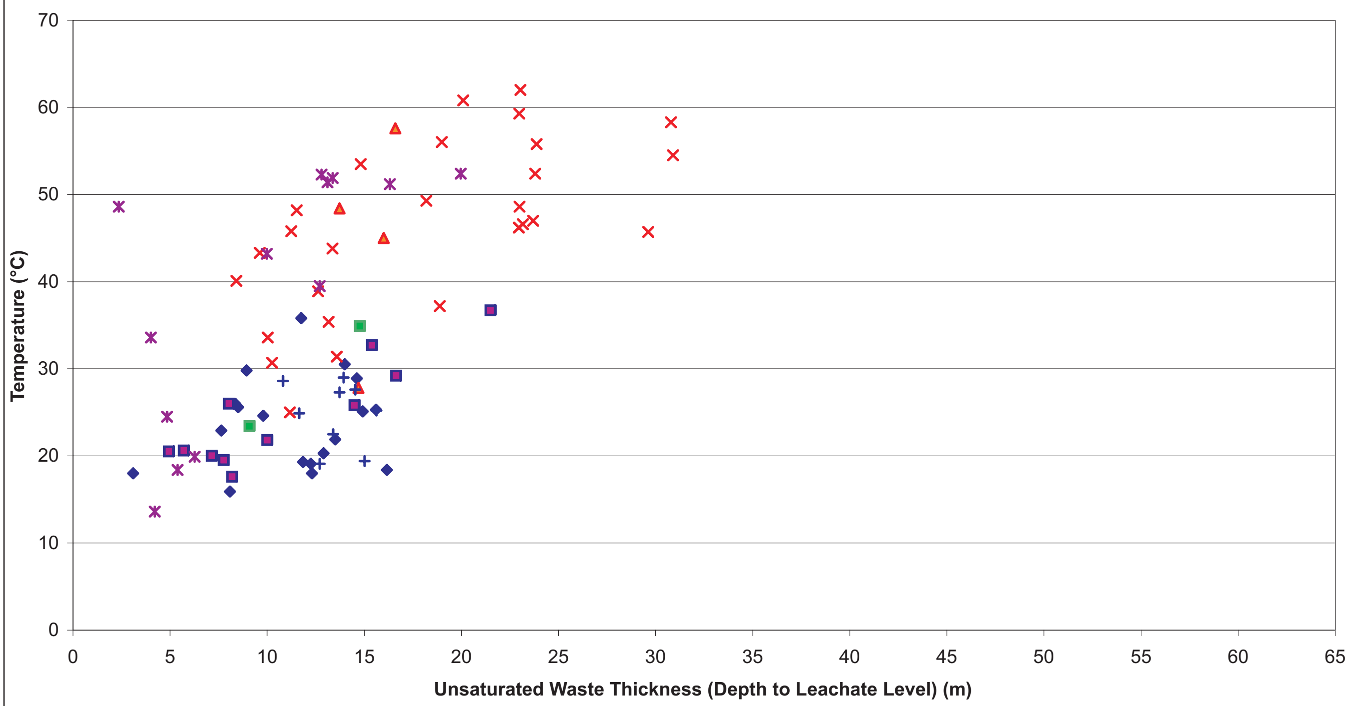
Review of the Performance of Hydraulicaly Contained Landfills

Figure 3.7
Leachate Temperature at Brogborough (August 2003)

November 2005
10744-S51.dwg wrigs02

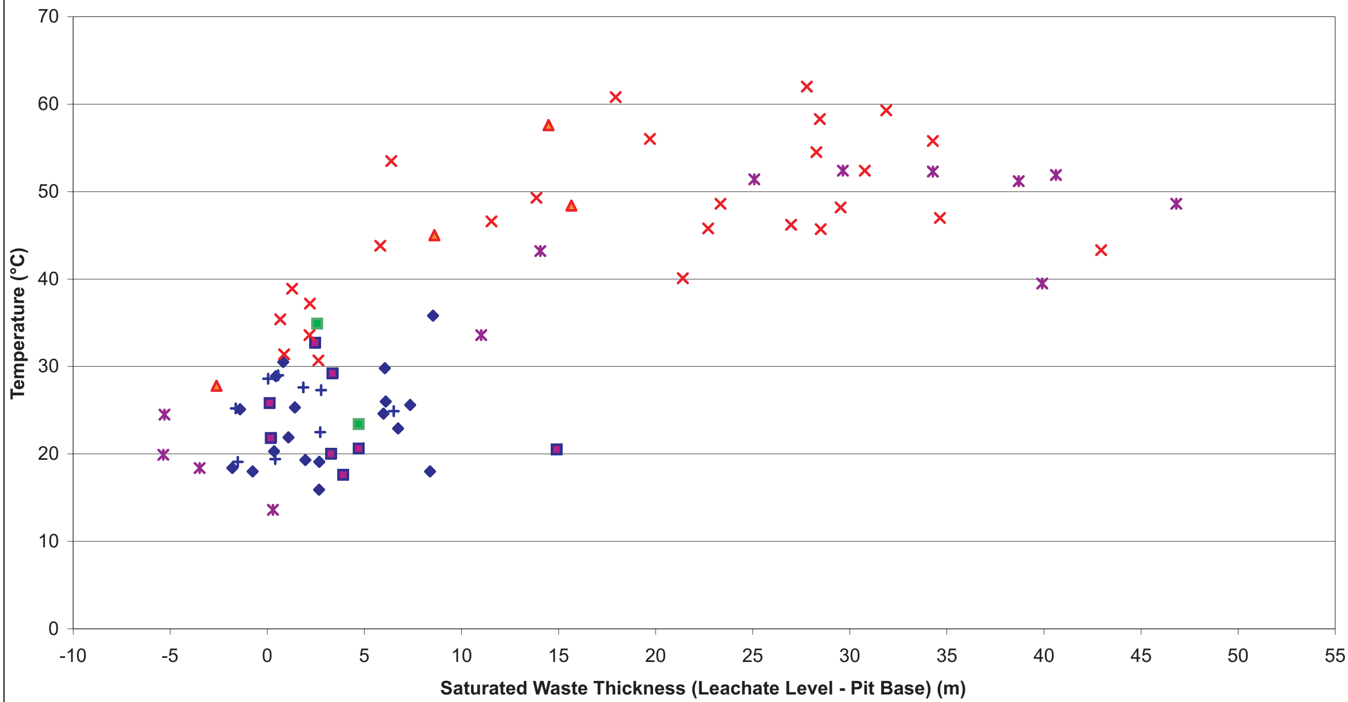
Entec

Figure 3.8a: Unsaturated Waste Thickness Compared with Leachate Temperature at Brogborough, August 2003



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Calcs\temp & quality v thickness of waste.xls *unsatwaste t v leach*

Figure 3.8b: Saturated Waste Thickness Compared with Leachate Temperature at Brogborough, August 2003



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Calcs\temp & quality v thickness of waste.xls *satwaste thickness v leach*

Key

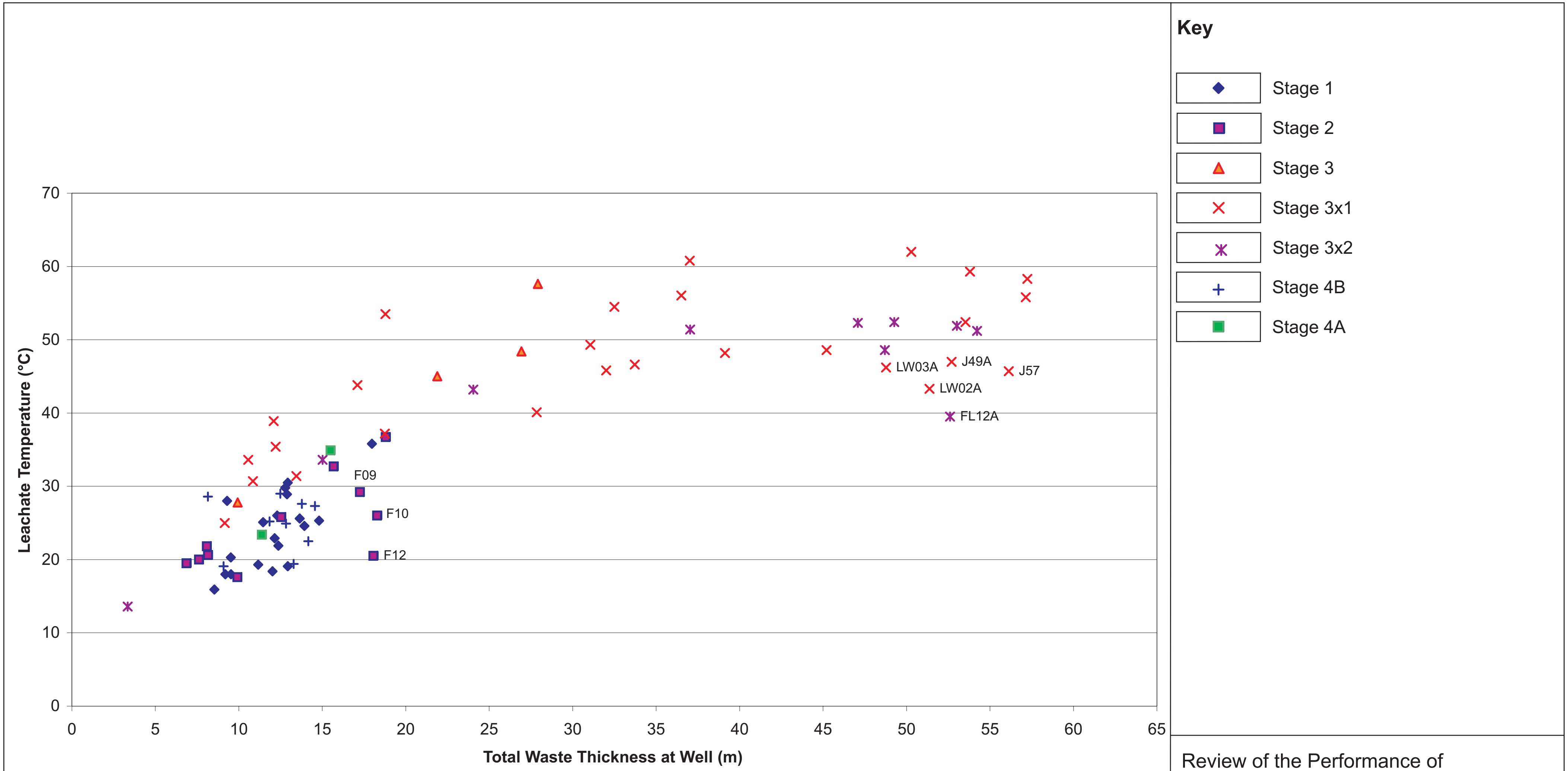
◆	Stage 1
■	Stage 2
▲	Stage 3
×	Stage 3x1
✳	Stage 3x2
+	Stage 4B
■	Stage 4A

Review of the Performance of Hydraulically Contained Landfills

Figure 3.8
Unsaturated and Saturated Waste Thickness Compared with Leachate Temperature, August 2003

November 2005
10744-S52.cdr wrigs02

Entec



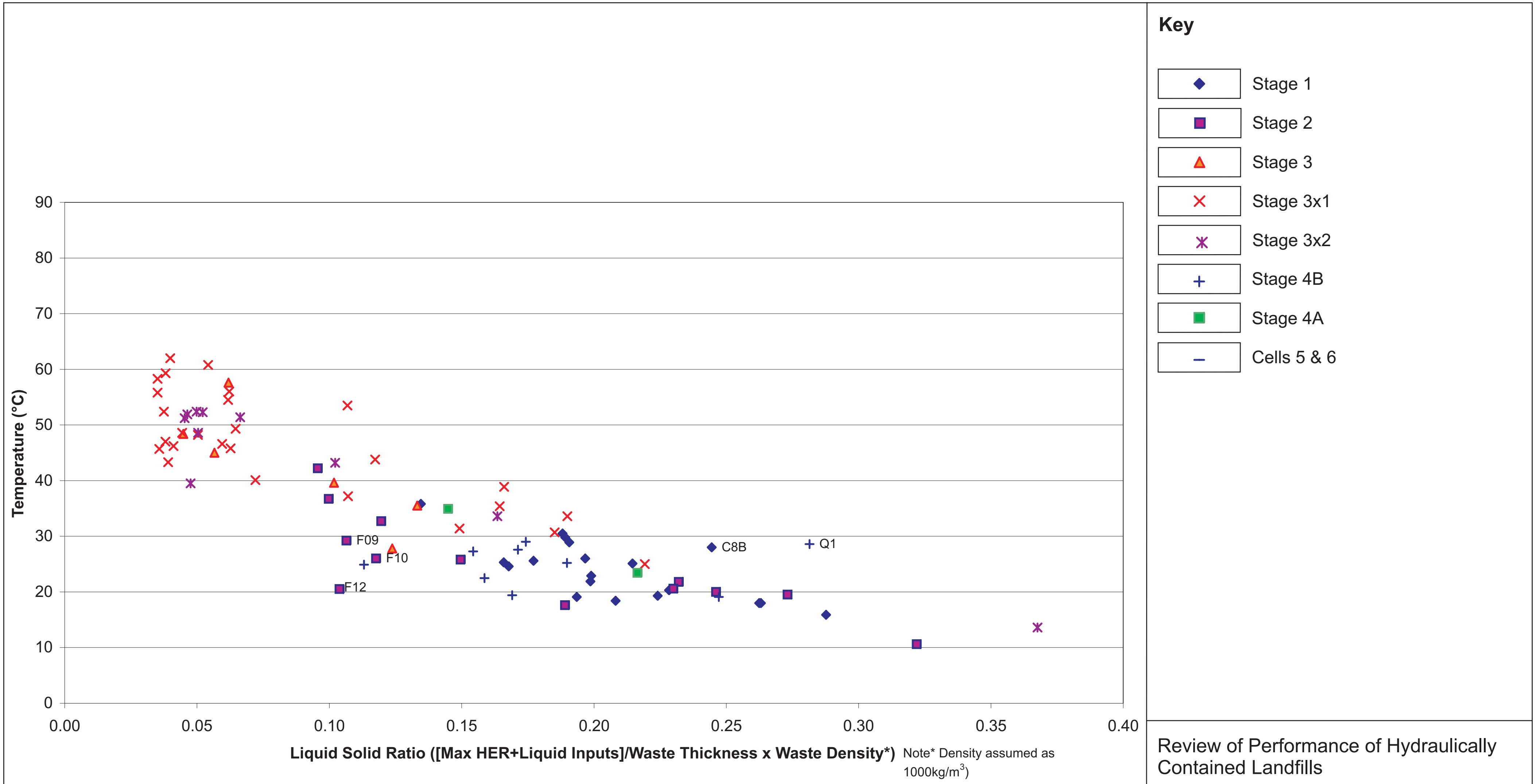
Review of the Performance of Hydraulically Contained Landfills

Figure 3.9
Total Waste Thickness Compared with Leachate Temperature at Brogborough, August 2003

November 2005
10744-S53.cdr xgeort

Entec

H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Calcs\temp & quality v thickness of waste.xlstotwaste thickness v leach t



Review of Performance of Hydraulically Contained Landfills

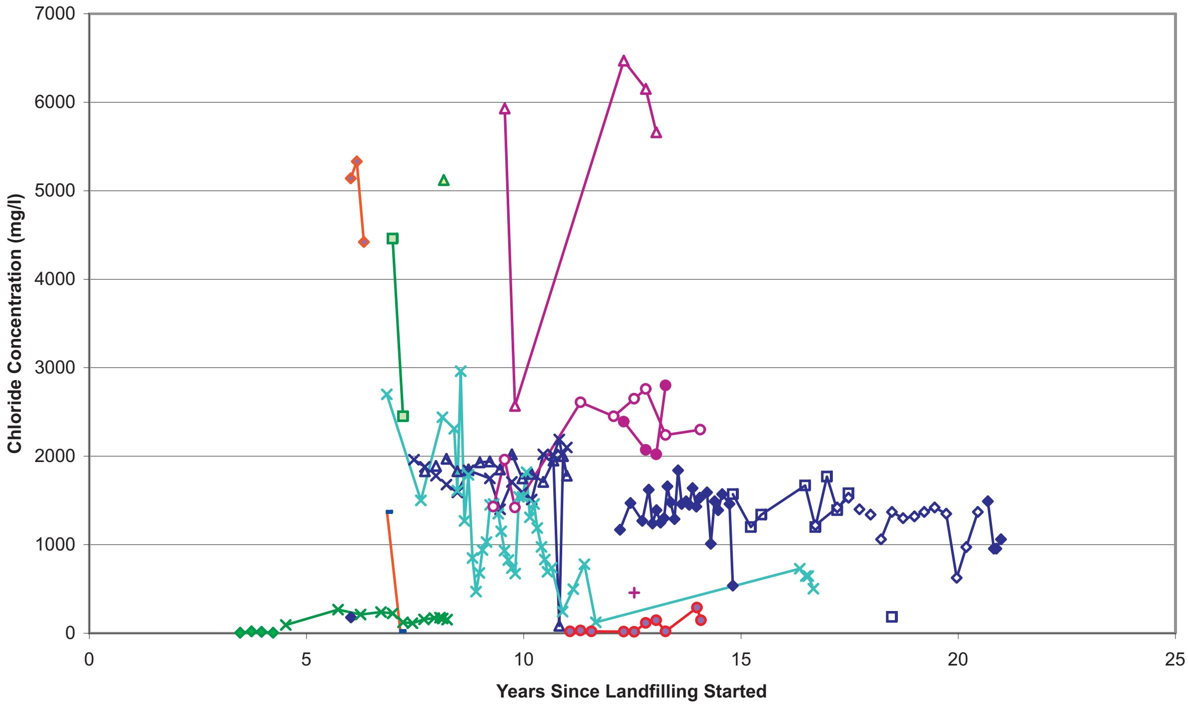
Figure 3.10
Liquid Solid Ratio Compared with Leachate Temperature at Brogborough, August 2003

November 2005
10744-S54.cdr xgeort

Entec

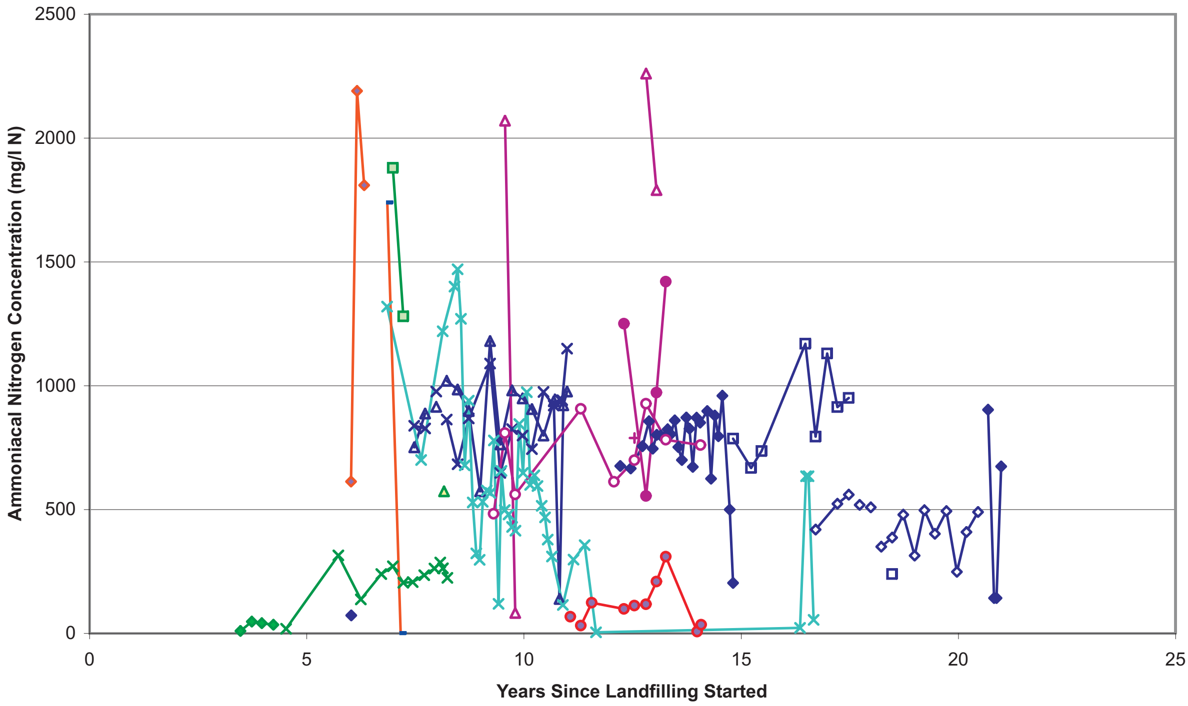
H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Calcs\temp & quality v thickness of waste.xls-Max lambda v T°C

Figure 3.11a: Change in Chloride Concentration with Age of Waste at Well at Brogborough



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Data\Lch Quality\leachate quality summary.xls *Cl v Age*

Figure 3.11b: Change in Ammoniacal Nitrogen Concentration with Age of Waste at Well at Brogborough



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Data\Lch Quality\leachate quality summary.xls *NH3 v Age*

Key

- C013
- C08B
- C013B
- SE03
- SE05
- F016
- Z111
- H109
- H110
- H111
- Q001
- Q004
- Q005
- S001
- W044
- FL012A
- J40A

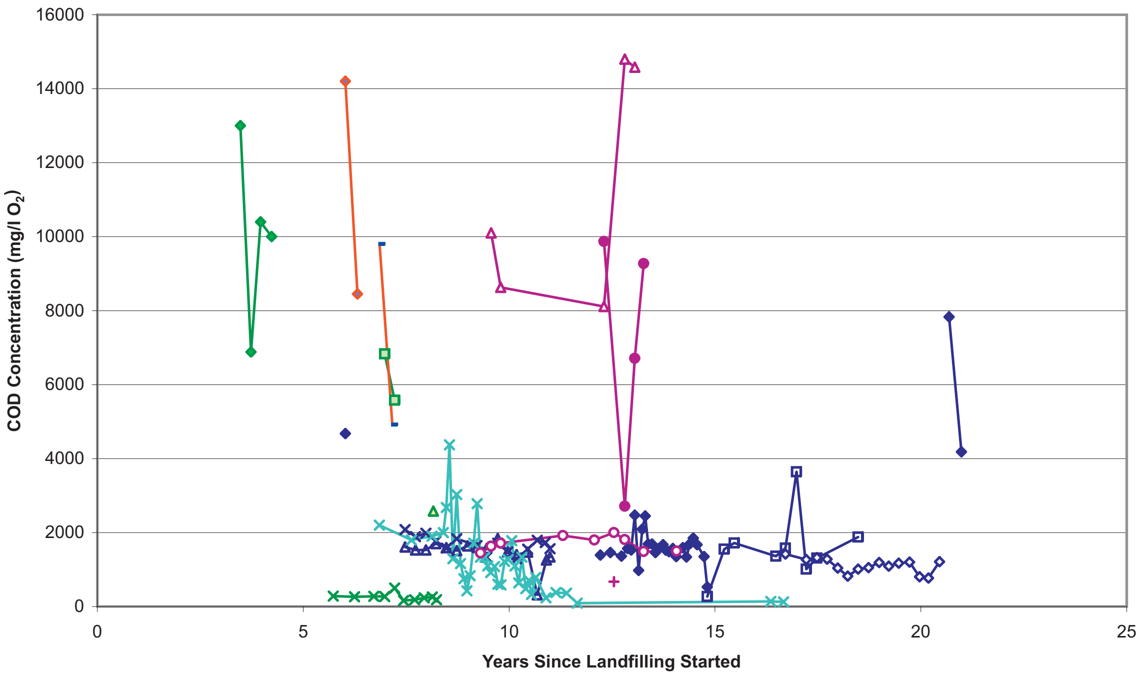
Review of the Performance of Hydraulically Contained Landfills

Figure 3.11
Change in Chloride and Ammoniacal Nitrogen Concentrations with Age of Waste at Brogborough

November 2005
10744-S55.cdr wrigs02

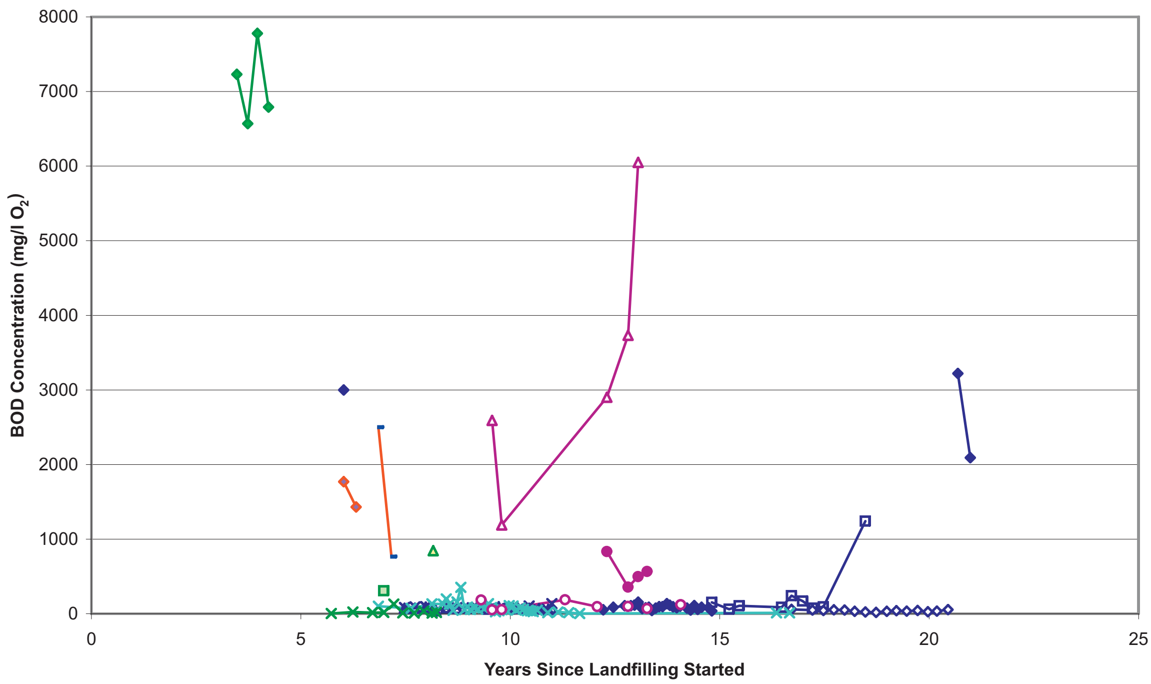
Entec

Figure 3.12a: Change in COD Concentration with Age of Waste at Well at Brogborough



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Data\Lch Quality\leachate quality summary.xls *COD v Age*

Figure 3.12b: Change in BOD Concentration with Age of Waste at Well at Brogborough



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Data\Lch Quality\leachate quality summary.xls *BOD v Age*

Key

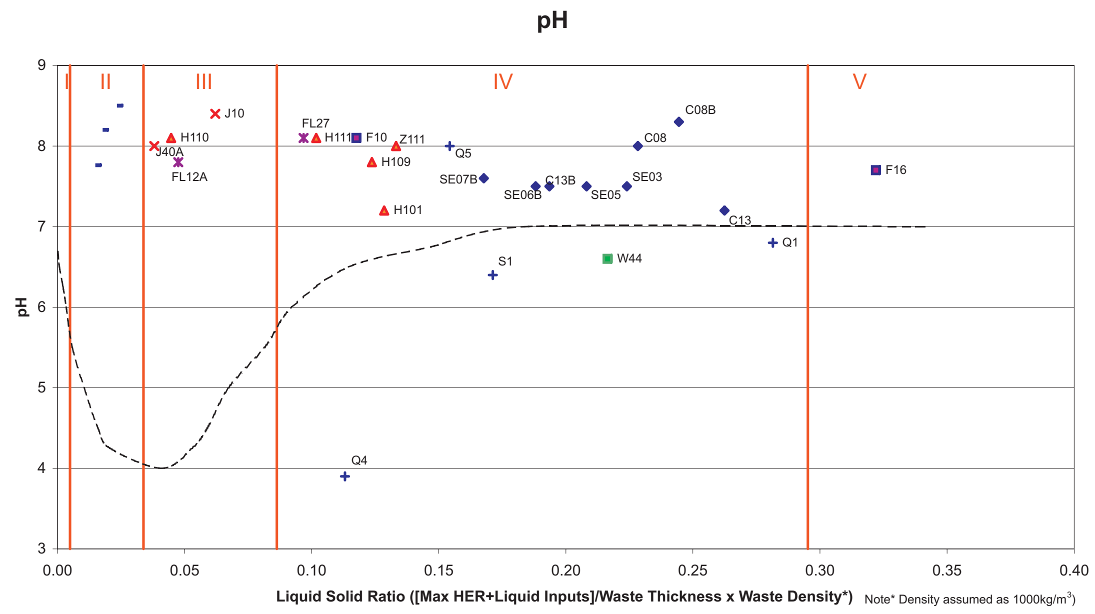
C013	H110
CO8B	H111
C013B	Q001
SE03	Q004
SE05	Q005
F016	S001
Z111	W044
H109	FL012A
	J40A

Review of the Performance of Hydraulically Contained Landfills

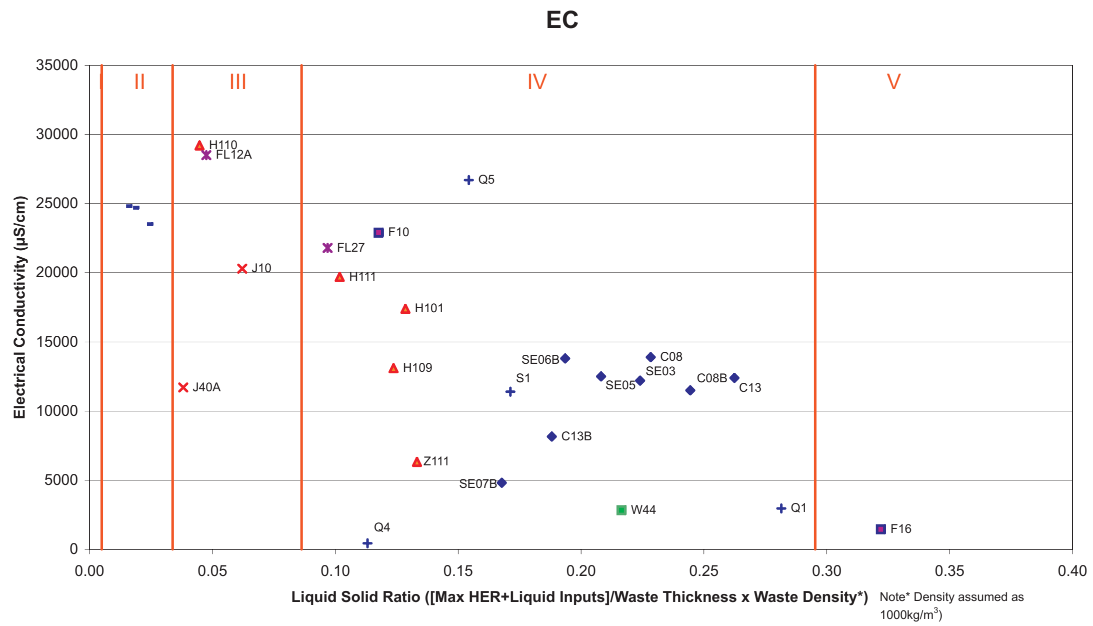
Figure 3.12
Change in COD and BOD Concentrations with Age of Waste at Brogborough

November 2005
10744-S56.cdr wrigs02

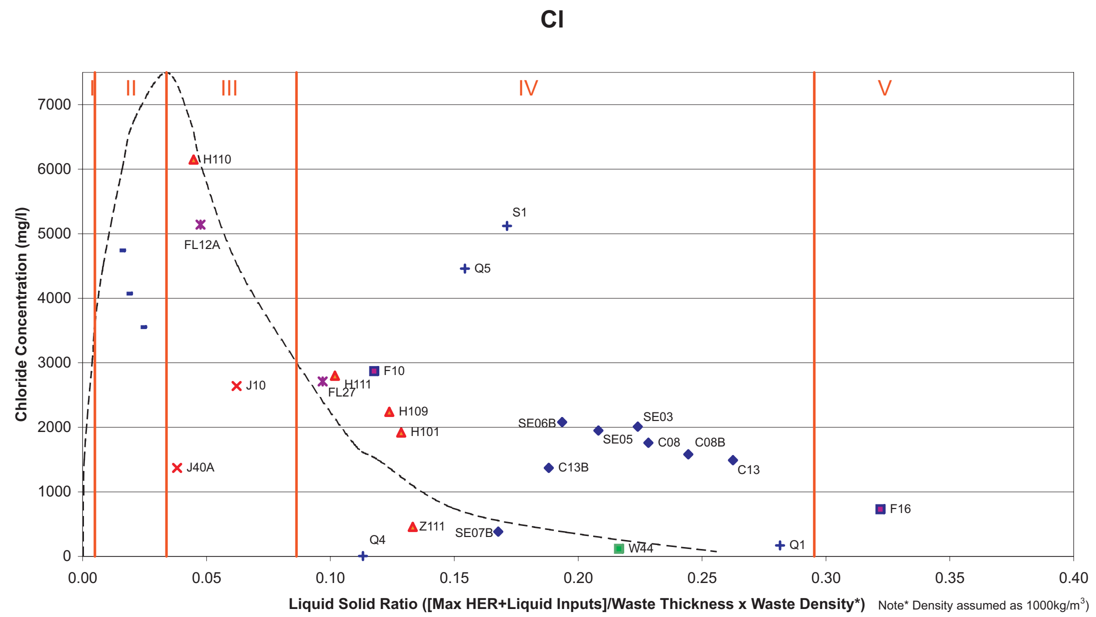
Entec



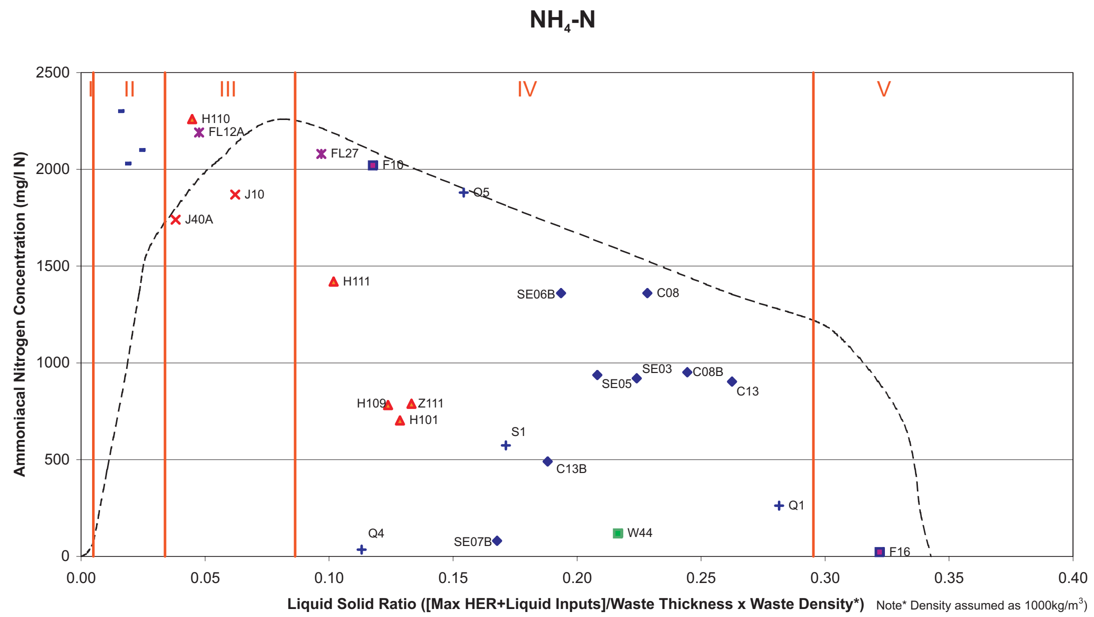
H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Calcs\temp & quality v thickness of waste.xls-lambda v pH



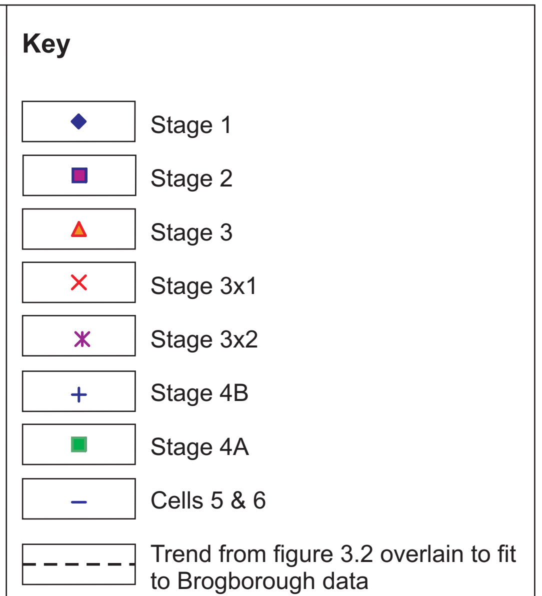
H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Calcs\temp & quality v thickness of waste.xls-lambda v EC



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Calcs\temp & quality v thickness of waste.xls-lambda v Cl



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Calcs\temp & quality v thickness of waste.xls-lambda v NH4

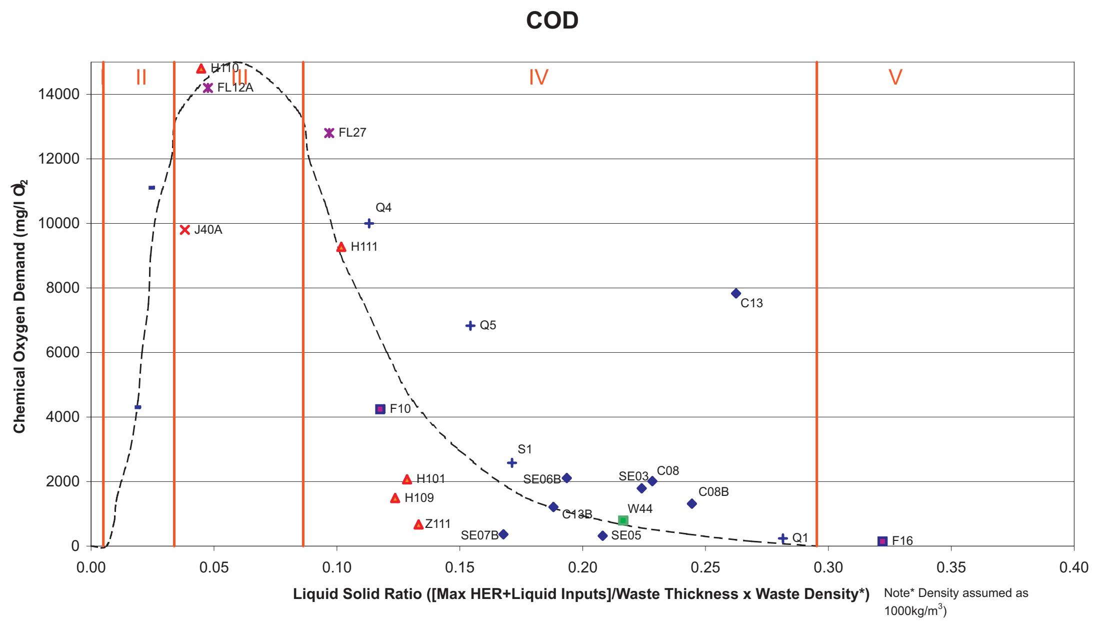


Review of the Performance of Hydraulically Contained Landfills

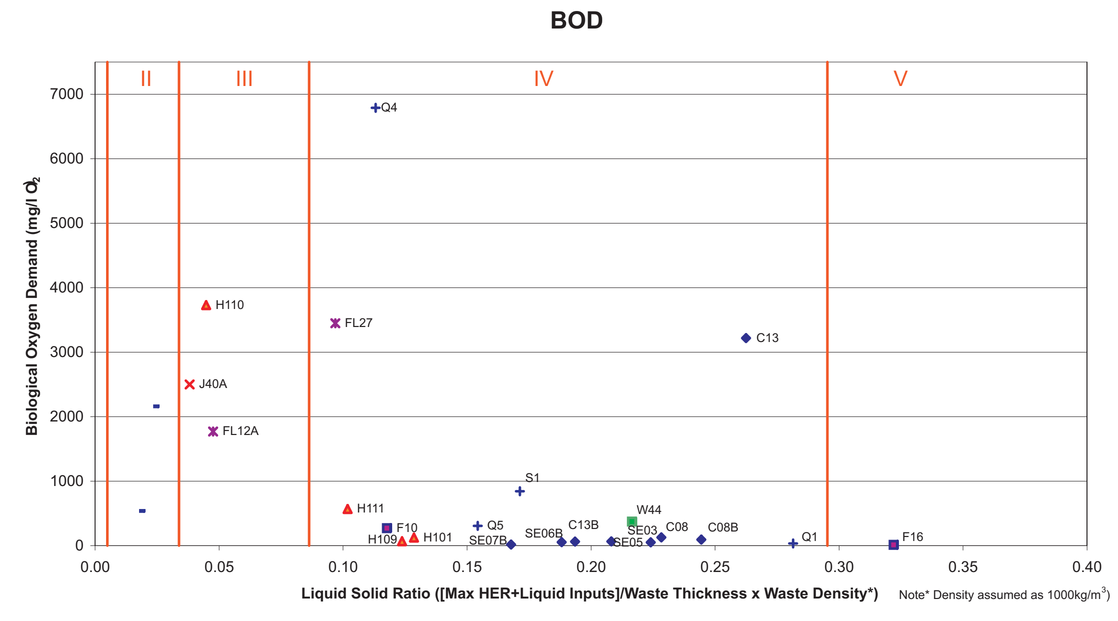
Figure 3.13
Variations in Leachate pH, EC, Cl & NH_4-N Circa 2003 with Estimated Liquid/Solid Ratios at Brogborough

November 2005
10744-S57.cdr wrigs02

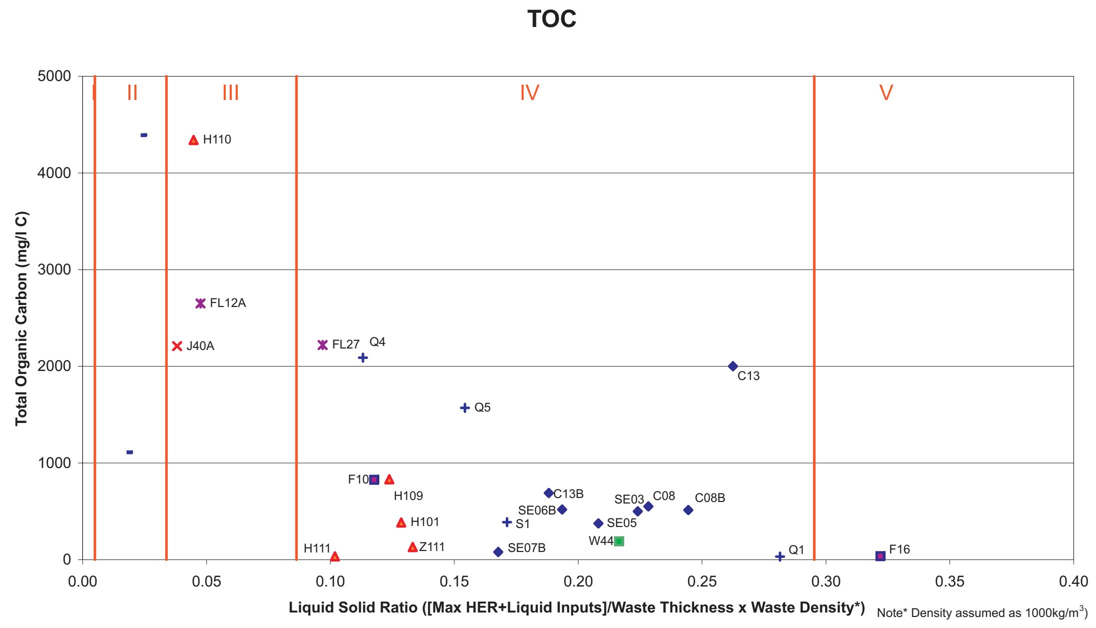
Entec



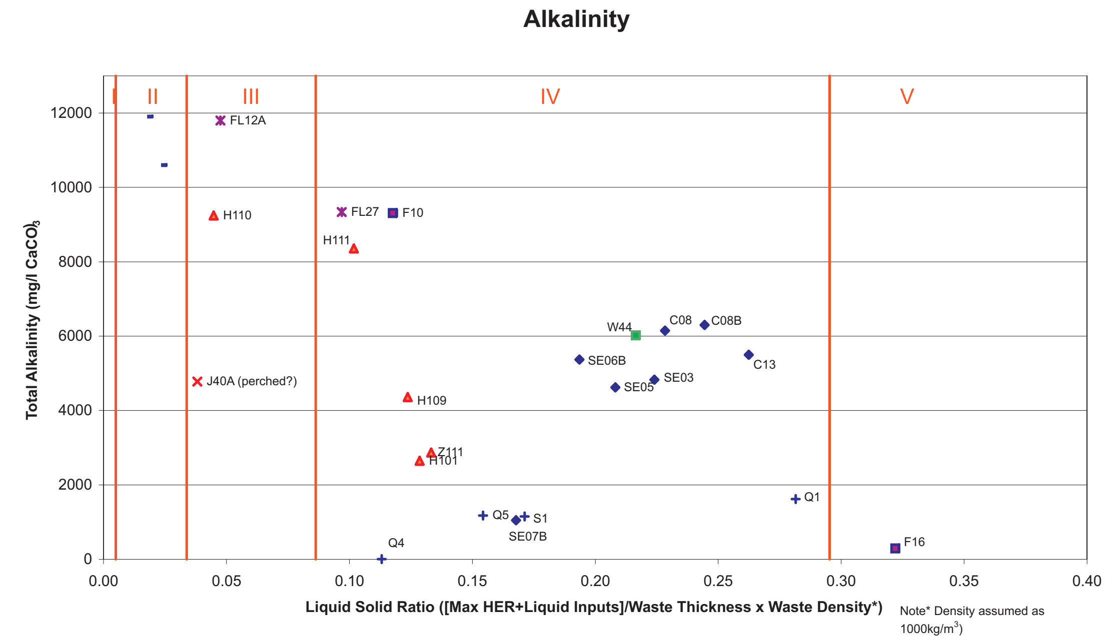
H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Calcs\temp & quality v thickness of waste.xls-lambda v COD



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Calcs\temp & quality v thickness of waste.xls-lambda v BOD



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Calcs\temp & quality v thickness of waste.xls-lambda v TOC



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Calcs\temp & quality v thickness of waste.xls-lambda v Alk

Key

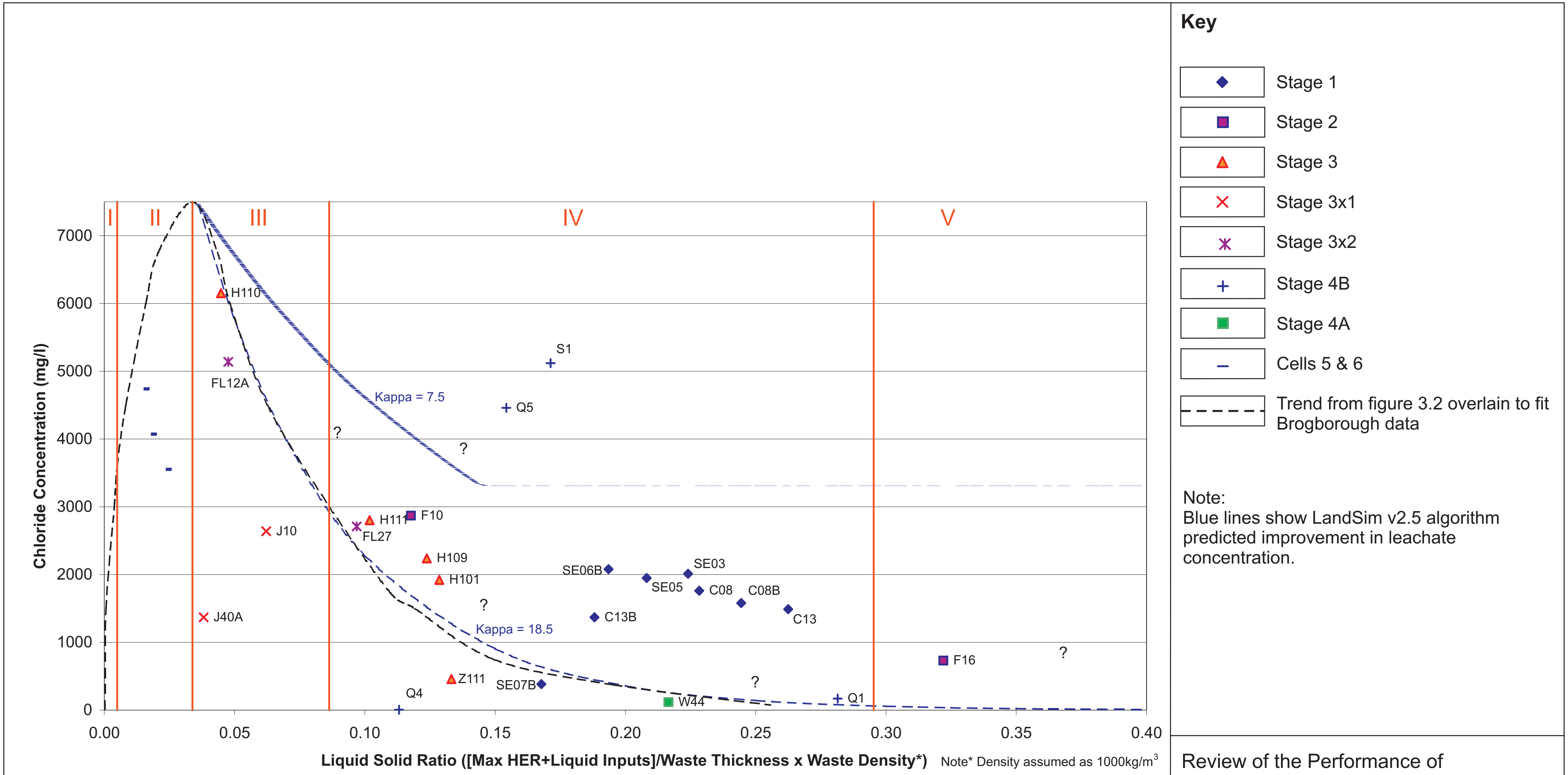
- Stage 1
- Stage 2
- Stage 3
- Stage 3x1
- Stage 3x2
- Stage 4B
- Stage 4A
- Cells 5 & 6
- Trend from figure 3.2 overlain to fit to Brogborough data

Review of the Performance of Hydraulically Contained Landfills

Figure 3.14
Variations in Leachate COD, BOD, TOC & Alkalinity Circa 2003 with Estimated Liquid/Solid Ratios at Brogborough

November 2005
10744-S58.cdr wrigs02

Entec



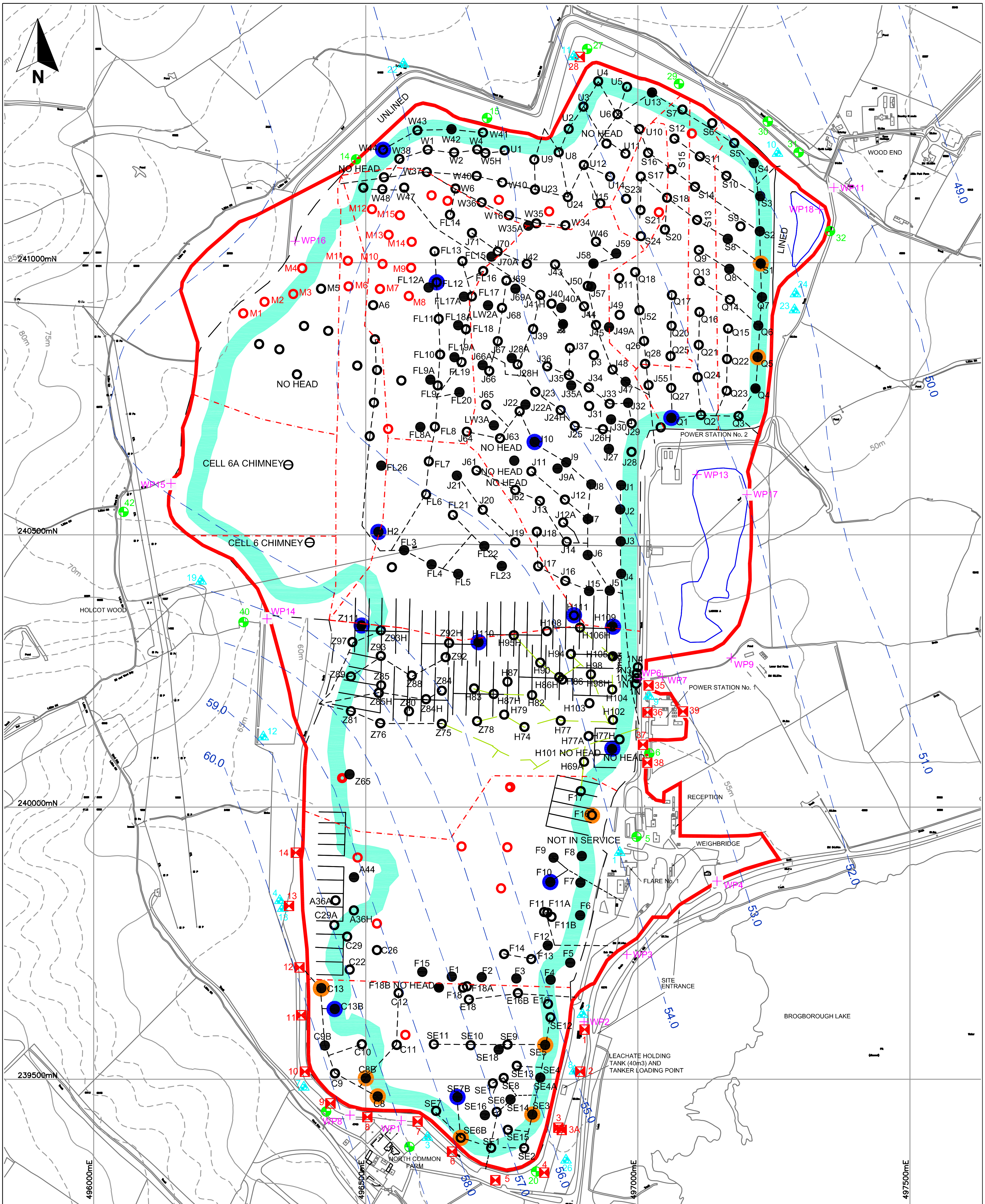
Review of the Performance of Hydraulically Contained Landfills

Figure 3.15
Detailed Analysis of Leachate Chloride Concentrations Circa 2003 with Estimated Liquid/Solid Ratios

November 2005
10744-S59.cdr xgeort

Entec

H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Calcs\temp & quality v thickness of waste.xls-lambda v Cl (2)



Key

Licence boundary	Gas probe	Surface water monitoring point
Cell boundary	In-waste gas well	Indicative gas well
Groundwater borehole	Leachate sampling well	Gas collection line
Pit base beneath piezometric level in the Kellaways Sand	Wells where leachate strength is high compared to liquid/solid ratios	Horizontal gas well
Leachate chimney	Wells where leachate strength is low compared to liquid/solid ratios (ie needs additional water inputs from cap infiltration)	Piezometric levels in Kellaways Sand (July 2003)
Perimeter gas monitoring borehole		

0 m 250 m

Scale 1:6500 @ A3

Review of the Performance of Hydraulically Contained Landfills

Figure 3.16
Leachate Strength Compared with Liquid/Solid Ratios across the Brogborough Landfill Site

November 2005
10744-S60.dwg wrigs02

Entec

Key

- Well constructed in 2004
- Phases 1 - 3
- Phase 4
- Phase 5

Note
Concentration in mg/l

0 m 150 m
Scale 1:2,500 @ A3

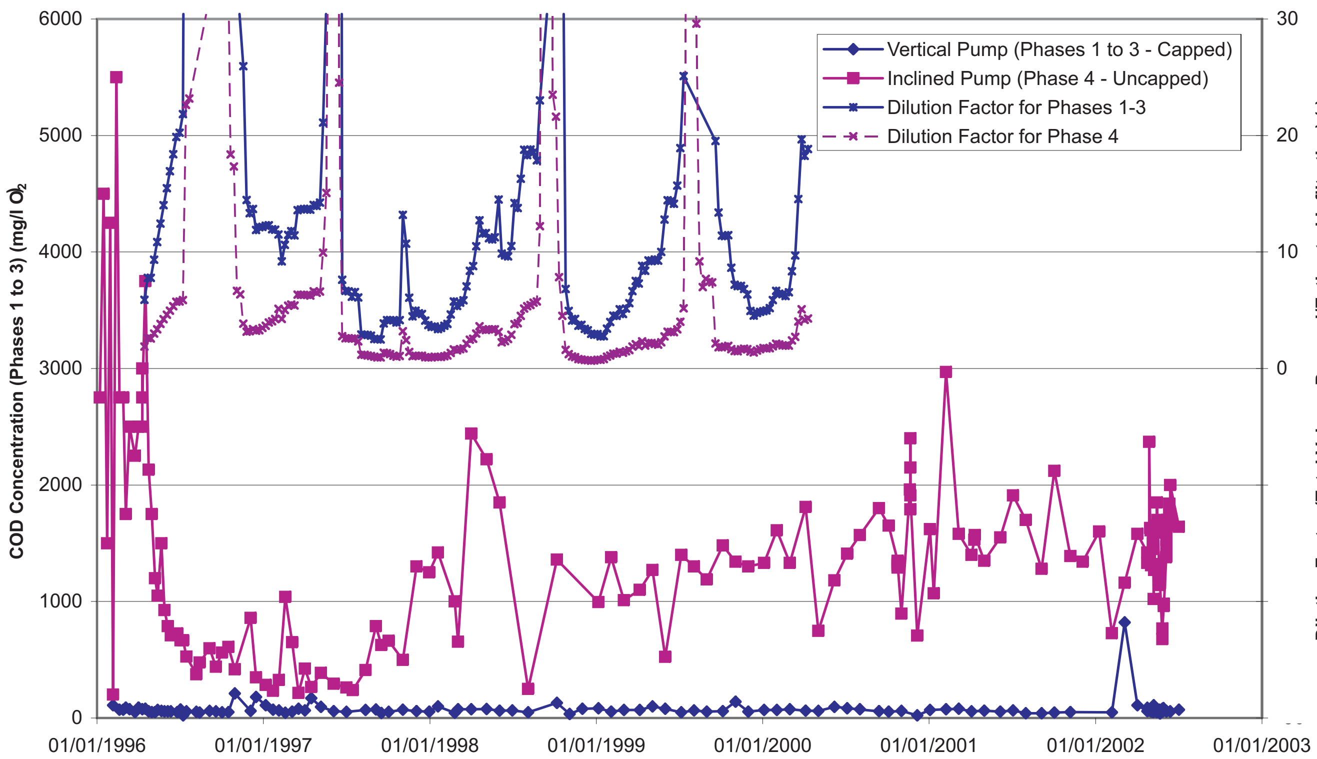
Review of the Performance of Hydraulically Contained Landfills

Figure 3.17
Leachate Quality Across the Poole Landfill Site - September 2004

November 2005
10744-S74.dwg xgeort

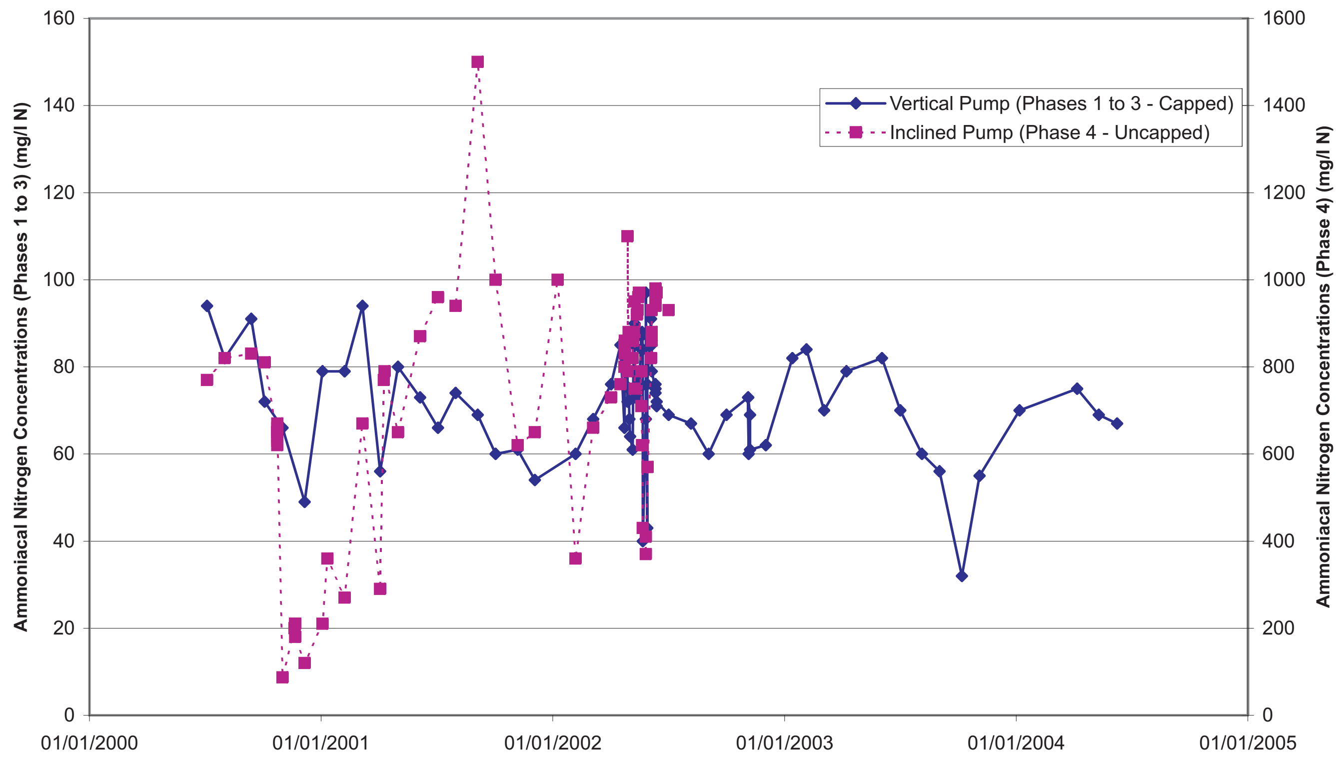
Entec

Figure 3.18a: Variation in Leachate Quality at Poole Landfill - COD



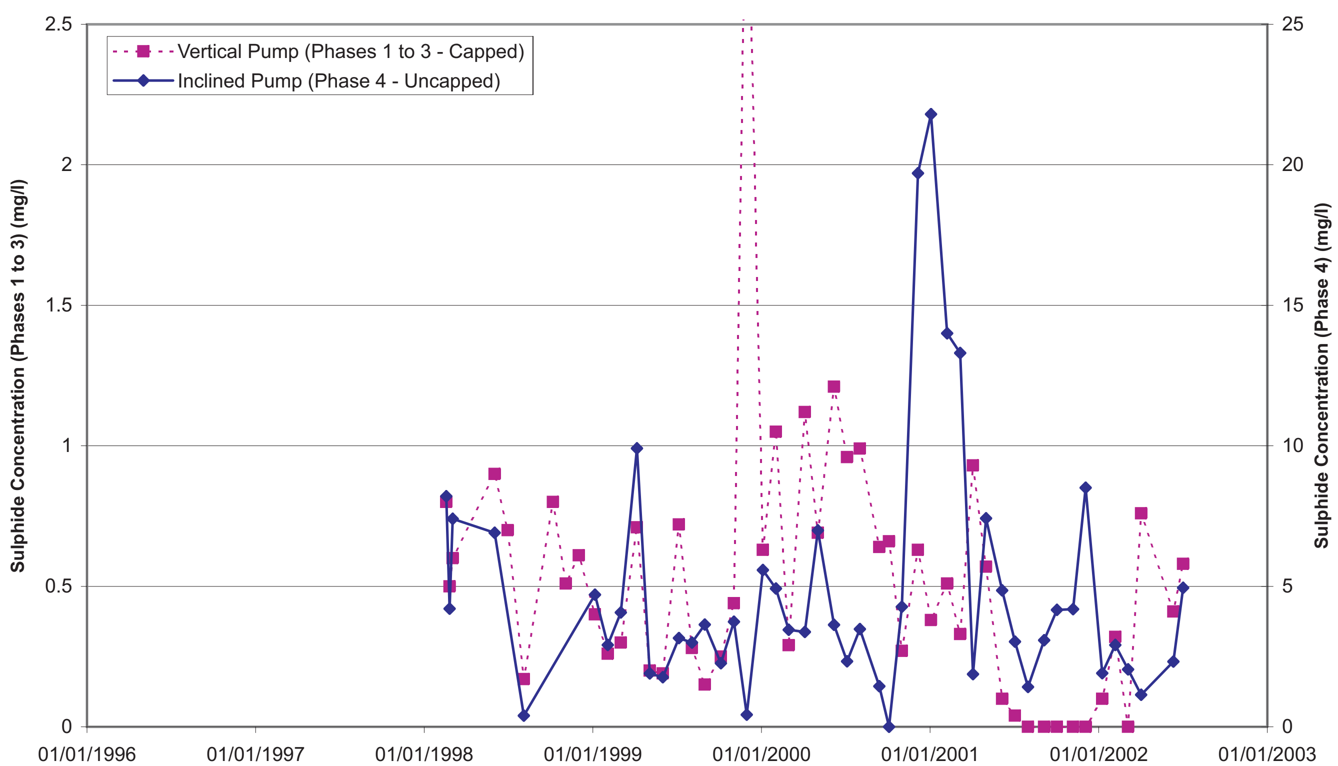
H:\Projects\Hm-250\08465 Wyvern Ph2 HydCon\Data\quality.xls-COD data chart

Figure 3.18b: Variation in Leachate Quality at Poole Landfill - Ammoniacal Nitrogen



H:\Projects\Hm-250\08465 Wyvern Ph2 HydCon\Data\quality.xls-NH4 data chart

Figure 3.18c: Variation in Leachate Quality at Poole Landfill - Sulphide



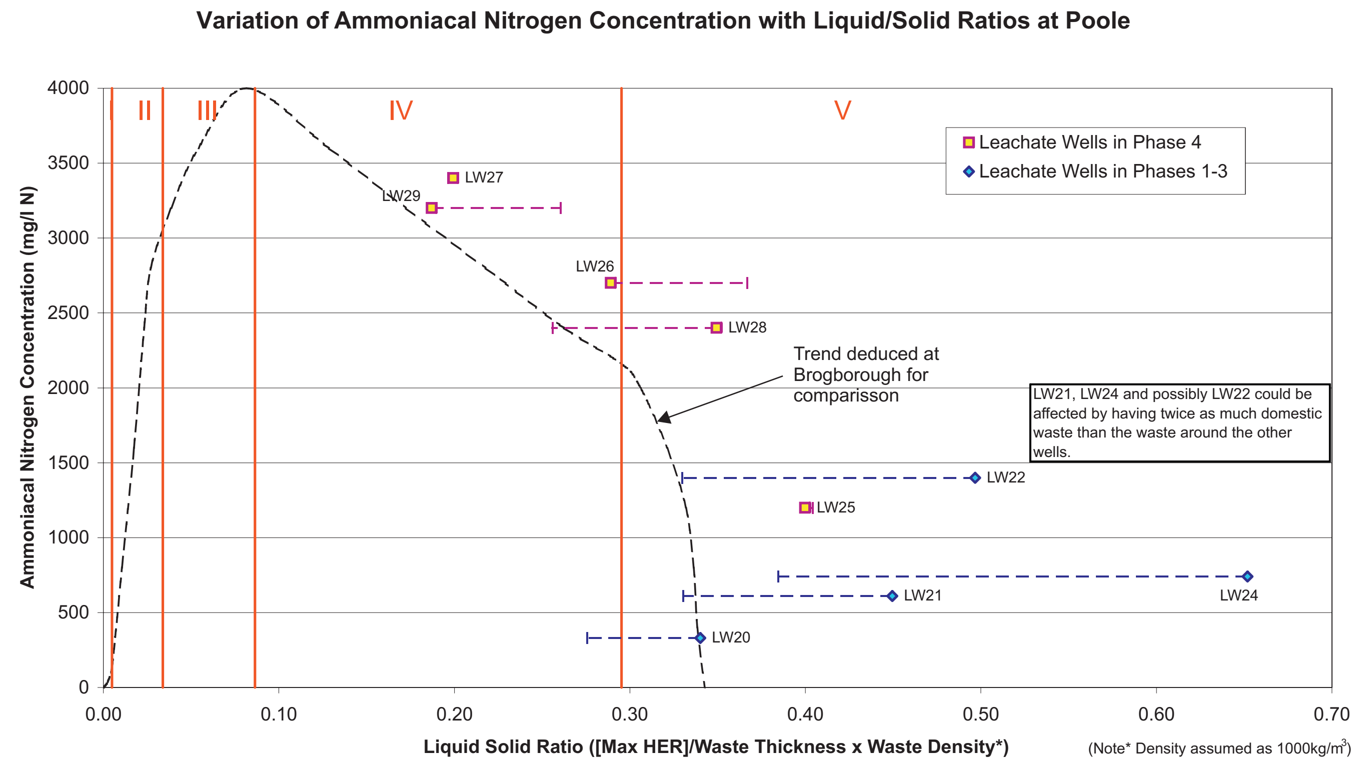
H:\Projects\Hm-250\08465 Wyvern Ph2 HydCon\Data\quality.xls-S2- data chart

Review of the Performance of Hydraulically Contained Landfills

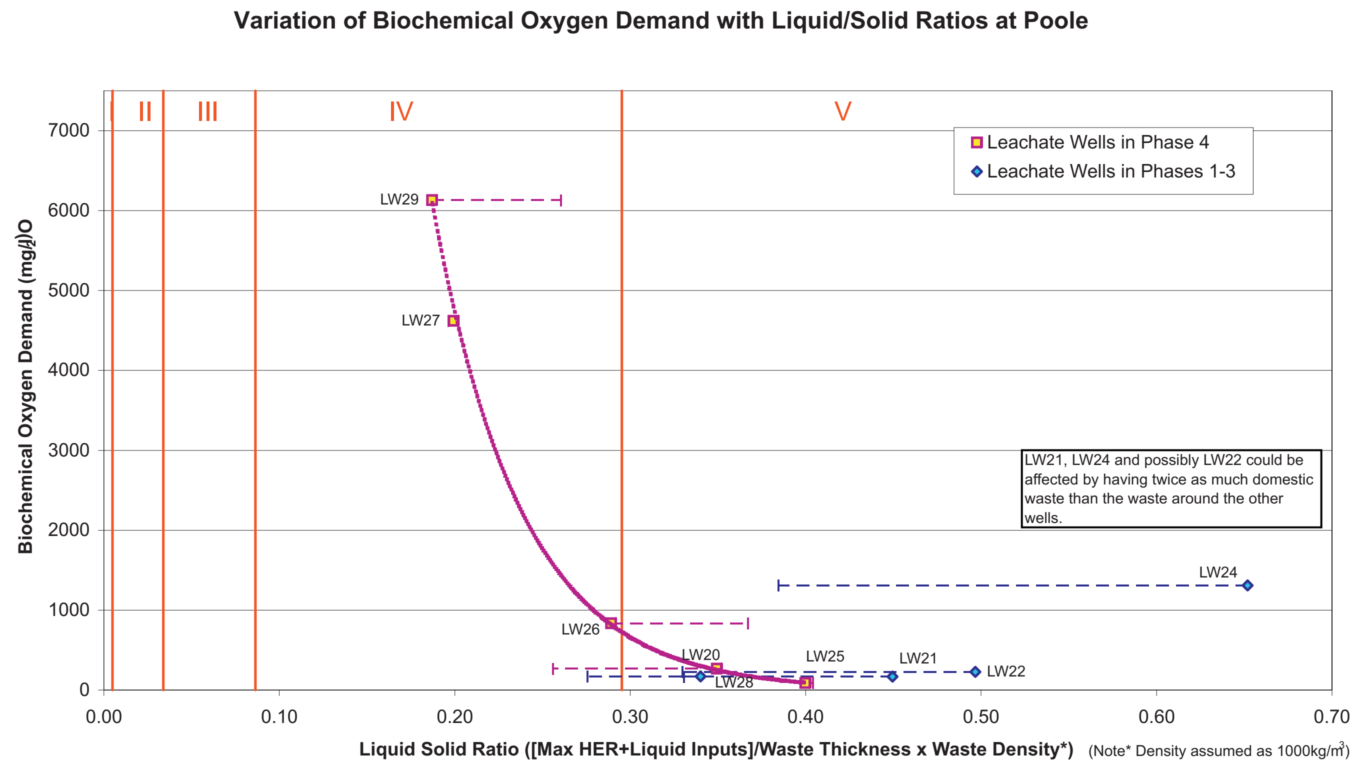
Figure 3.18
Variations in Leachate Quality at Poole

November 2005
10744-S62.cdr wrigs02

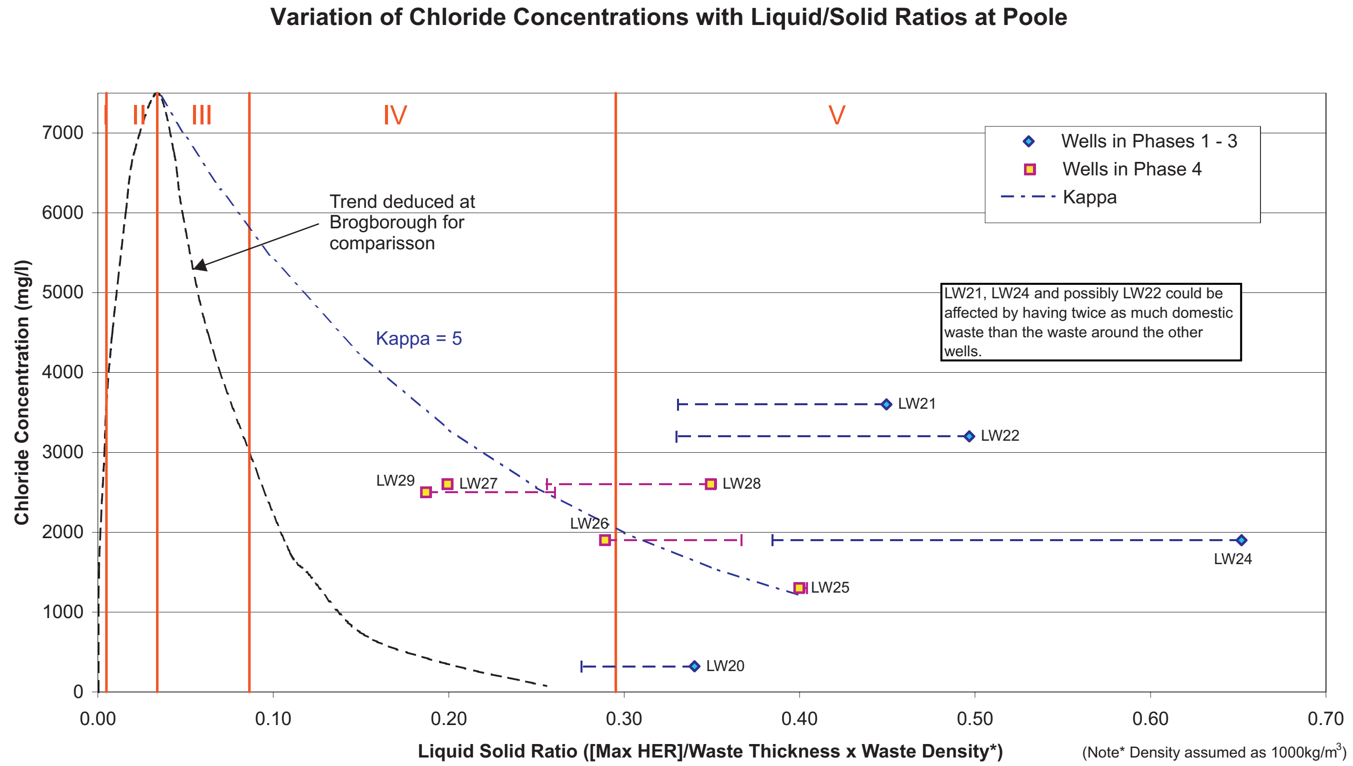
Entec



H:\Projects\Hm-250\08465 Wyvern Ph2 HydCon\Data\quality.xls-lambda v NH4



H:\Projects\Hm-250\08465 Wyvern Ph2 HydCon\Data\quality.xls-lambda v BOD



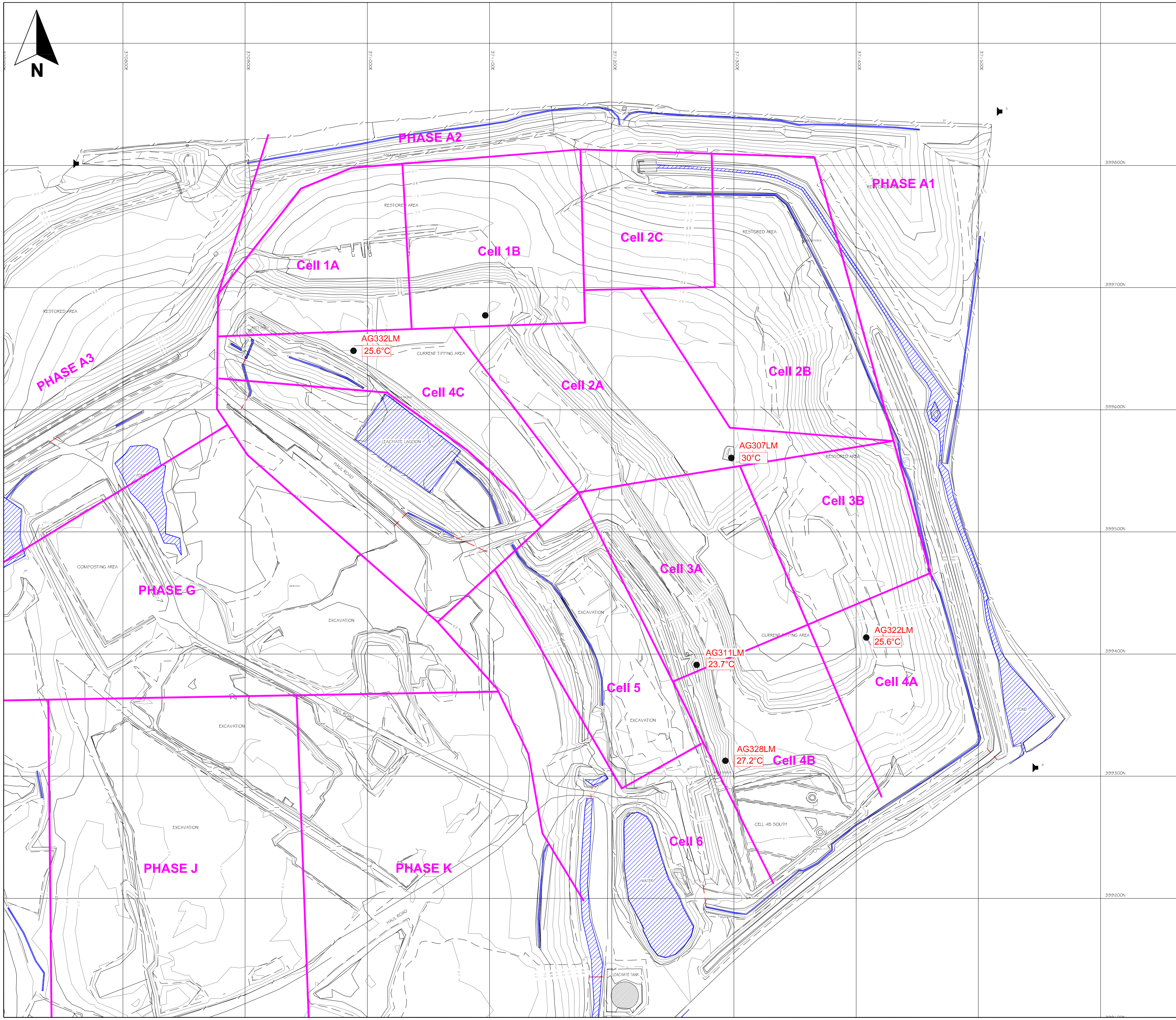
H:\Projects\Hm-250\08465 Wyvern Ph2 HydCon\Data\quality.xls-lambda v Cl

Review of the Performance of Hydraulically Contained Landfills

Figure 3.19
Variation of Ammoniacal Nitrogen Concentration, Chemical Oxygen Demand and Chloride Concentration with Liquid/Solid Ratios at Poole

November 2005
10744-S63.cdr wrigs02

Entec



Key

Cell boundary

Leachate quality monitoring point

23.7°C Leachate temperature 05 October 2004

0 m 180 km

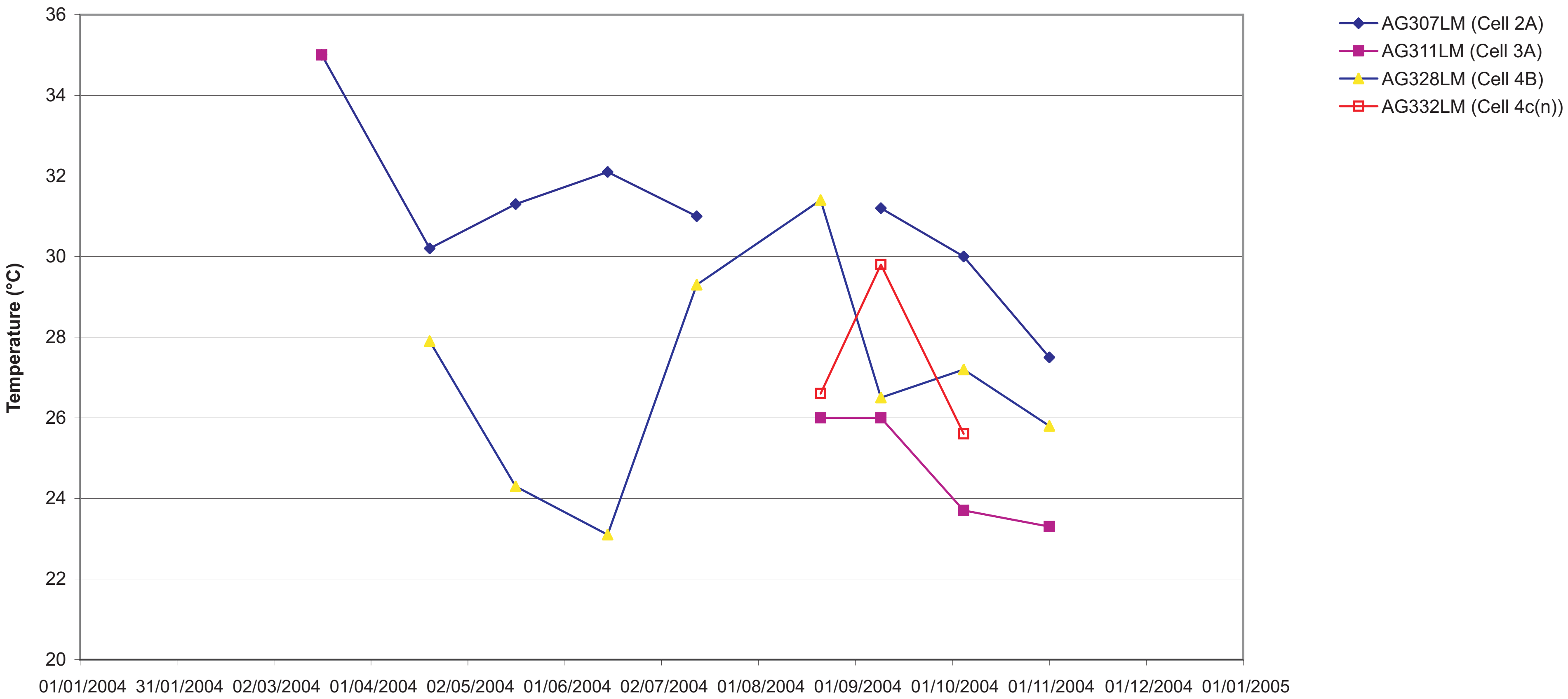
Scale 1:3,000 @ A3

Review of the Performance of Hydraulically Contained Landfills

Figure 3.20
Leachate Quality Monitoring Points and Leachate Temperature at Poole, 05 October 2004

November 2005
10744-S64.dwg wrigs02

Entec



Review of the Performance of Hydraulically Contained Landfills

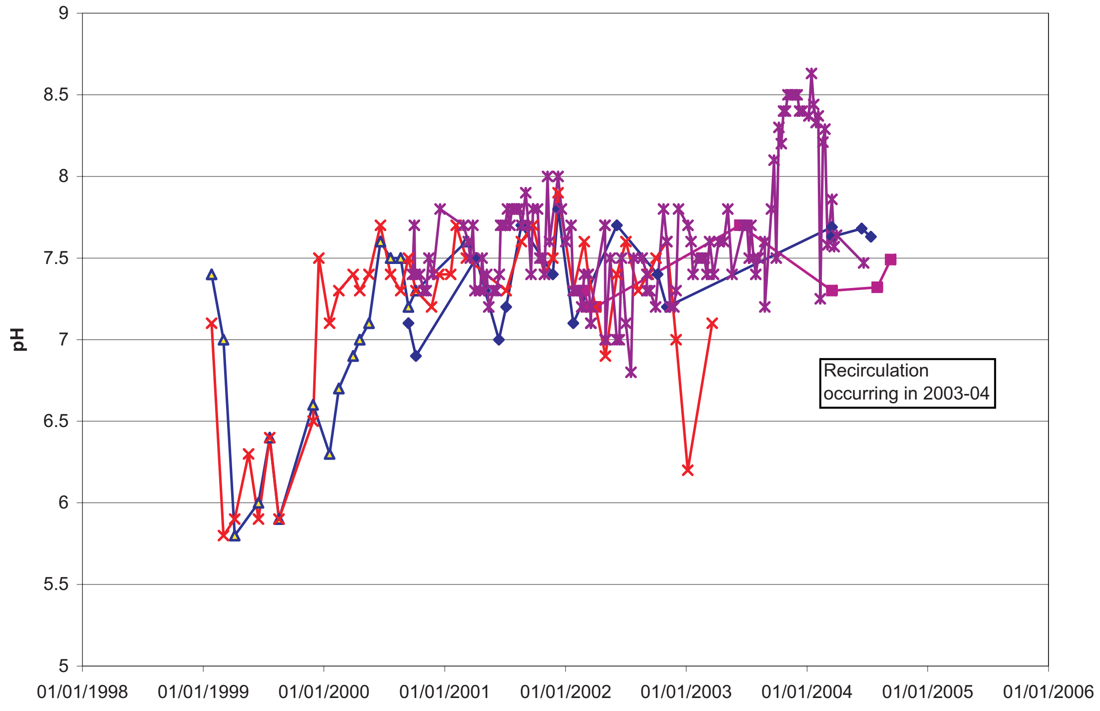
Figure 3.21
Leachate Temperature Data for Whitehead

November 2005
10744-S65.cdr wrigs02

Entec

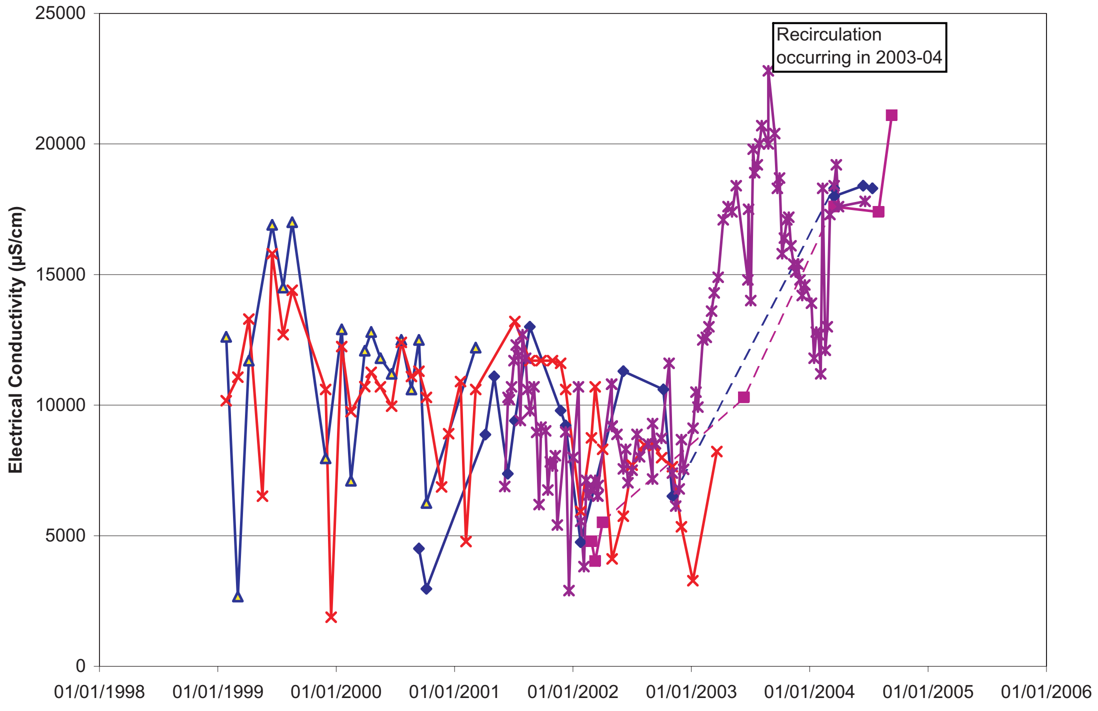
H:\Projects\Hm-250\08975 Viridor ph2 hyd\Data\whitehead temp.xls-Fig 3.21

Figure 3.22a: Leachate pH



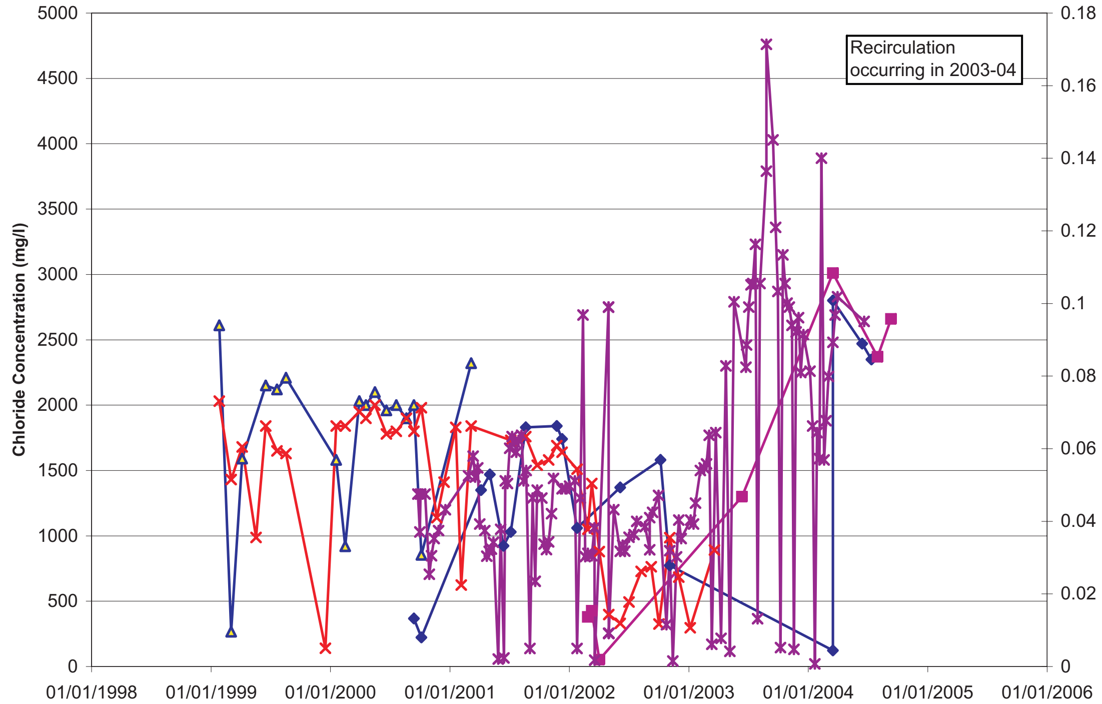
H:\Projects\Hm-250\08975 Viridor ph2 hyd\Data\Leachate Quality.xls-Chart -pH

Figure 3.22b: Leachate EC



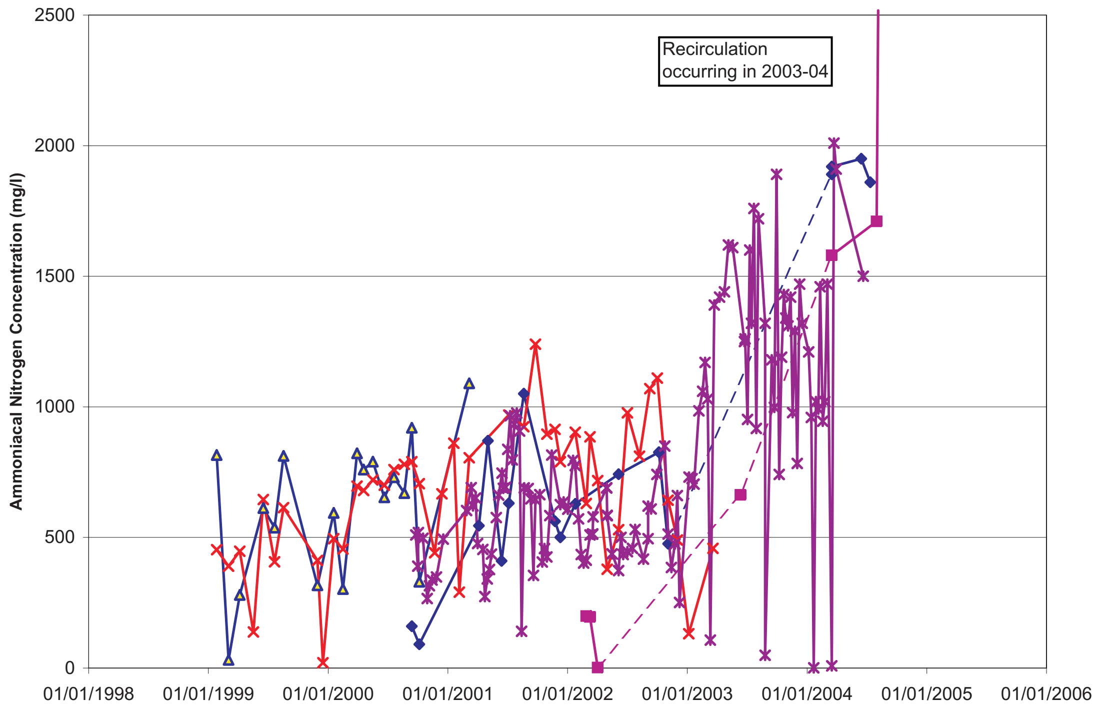
H:\Projects\Hm-250\08975 Viridor ph2 hyd\Data\Leachate Quality.xls-Chart - EC

Figure 3.22c: Leachate Cl



H:\Projects\Hm-250\08975 Viridor ph2 hyd\Data\Leachate Quality.xls-Chart - Cl

Figure 3.22d: Leachate NH_3



H:\Projects\Hm-250\08975 Viridor ph2 hyd\Data\Leachate Quality.xls-Chart - NH3

Key

- AG/307LM
- AG/311LM
- AG/502LM
- AG/501LM
- Pumped Leachate

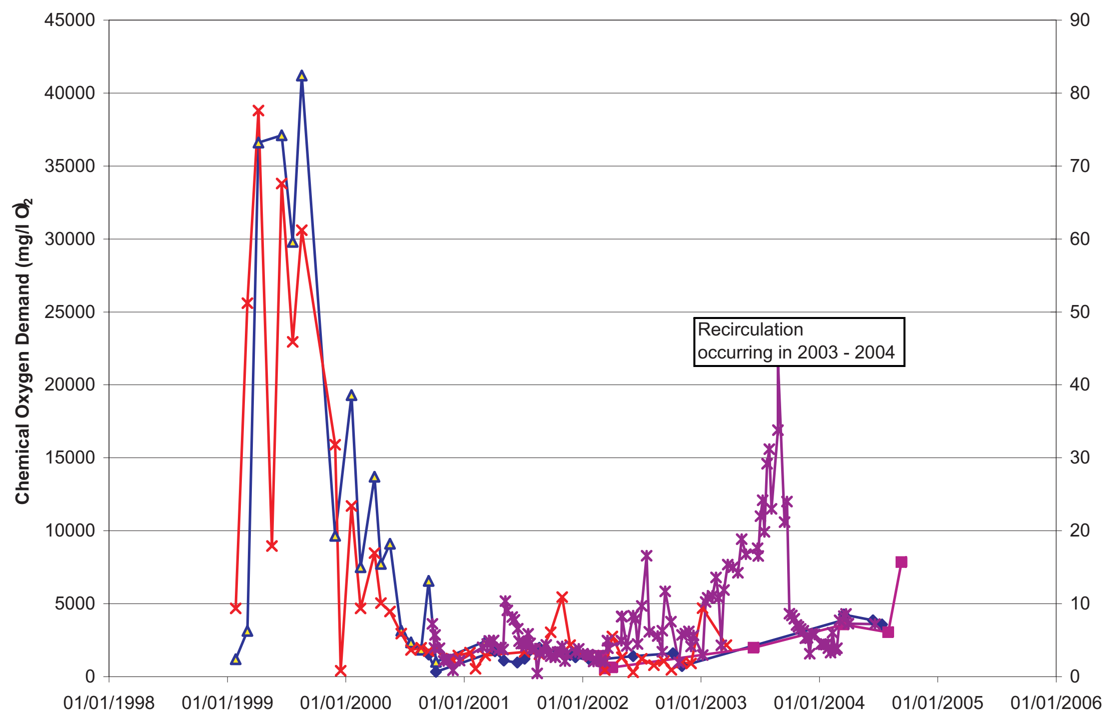
Review of the Performance of Hydraulically Contained Landfills

Figure 3.22
Variations in Leachate pH, EC, Cl & NH_4-N with Time at Whitehead

November 2005
10744-S66.cdr wrigs02

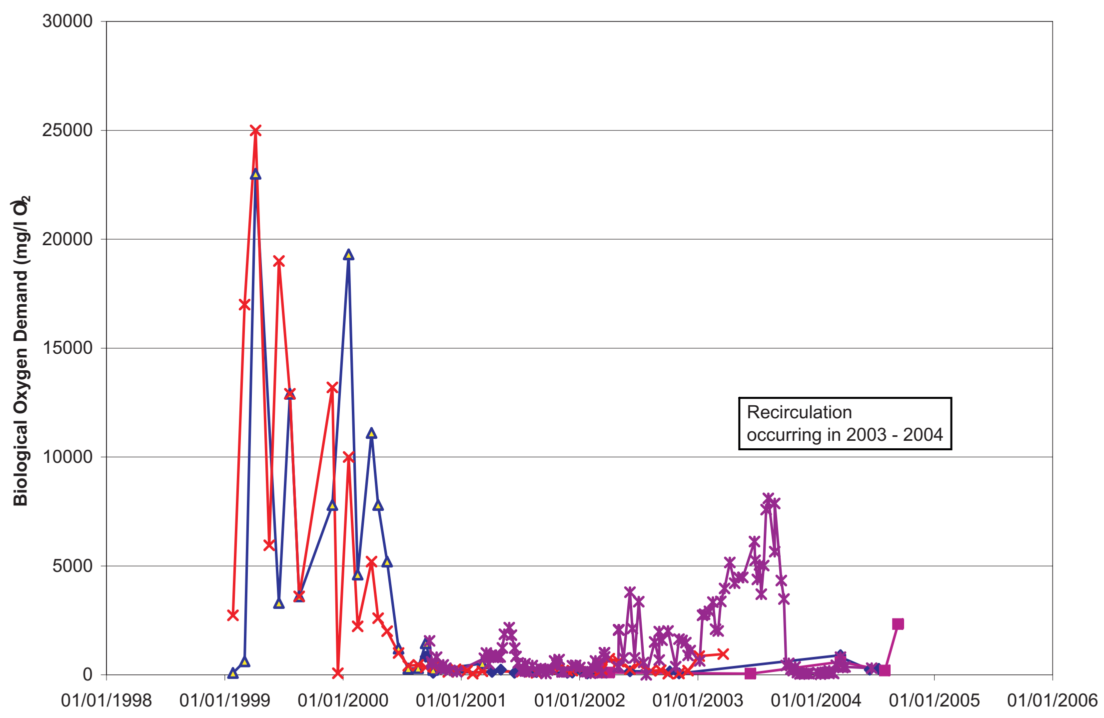
Entec

Figure 3.23a: Leachate COD



H:\Projects\Hm-250\08975 Viridor ph2 hyd\Data\Leachate Quality.xls-Chart - COD

Figure 3.23b: Leachate BOD

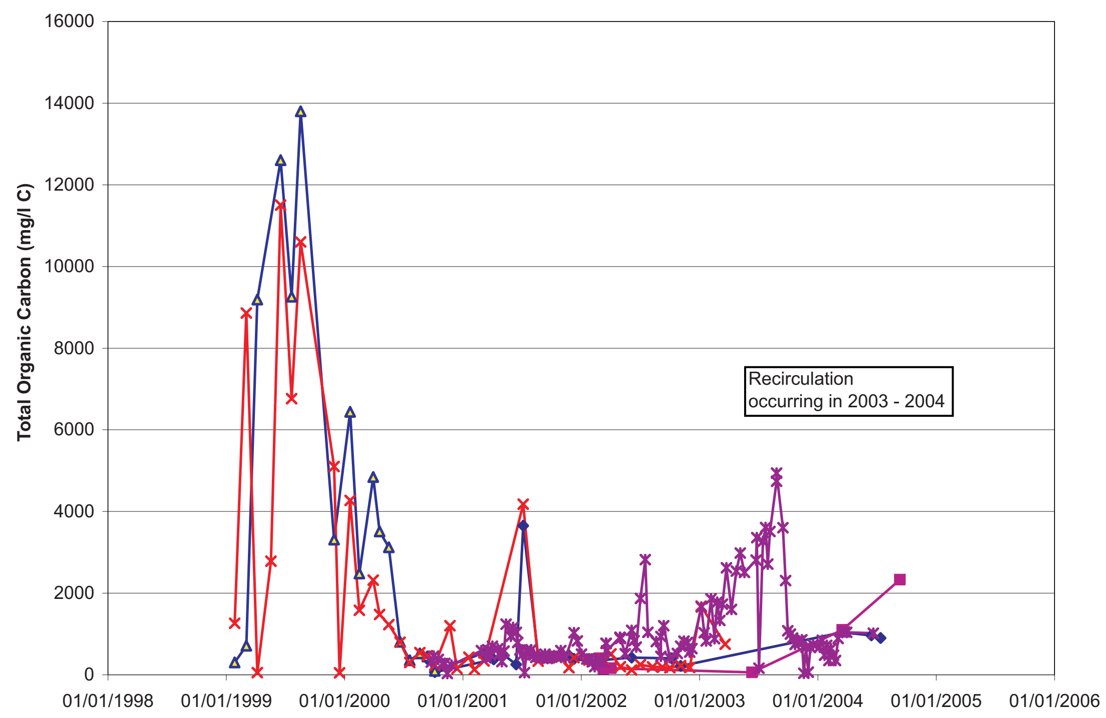


H:\Projects\Hm-250\08975 Viridor ph2 hyd\Data\Leachate Quality.xls-Chart -BOD

Key

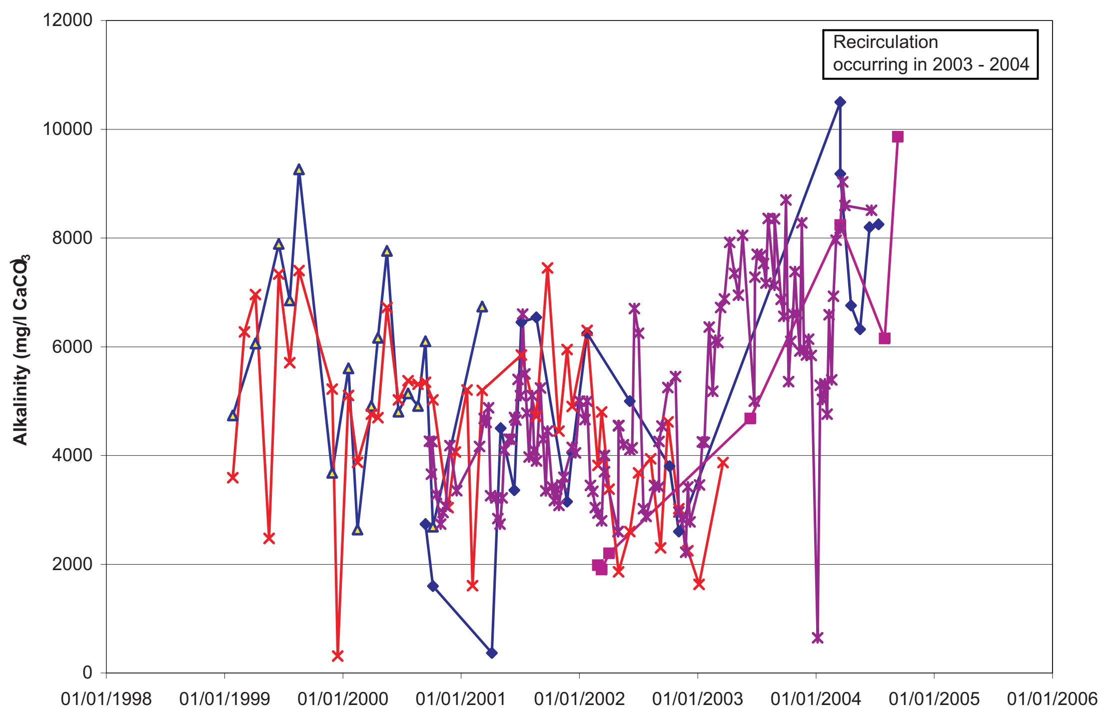
AG/307LM
AG/311LM
AG/502LM
AG/501LM
Pumped Leachate

Figure 3.23c: Leachate TOC



H:\Projects\Hm-250\08975 Viridor ph2 hyd\Data\Leachate Quality.xls-Chart -TOC

Figure 3.32d: Leachate Alkalinity



H:\Projects\Hm-250\08975 Viridor ph2 hyd\Data\Leachate Quality.xls-Chart -Alk

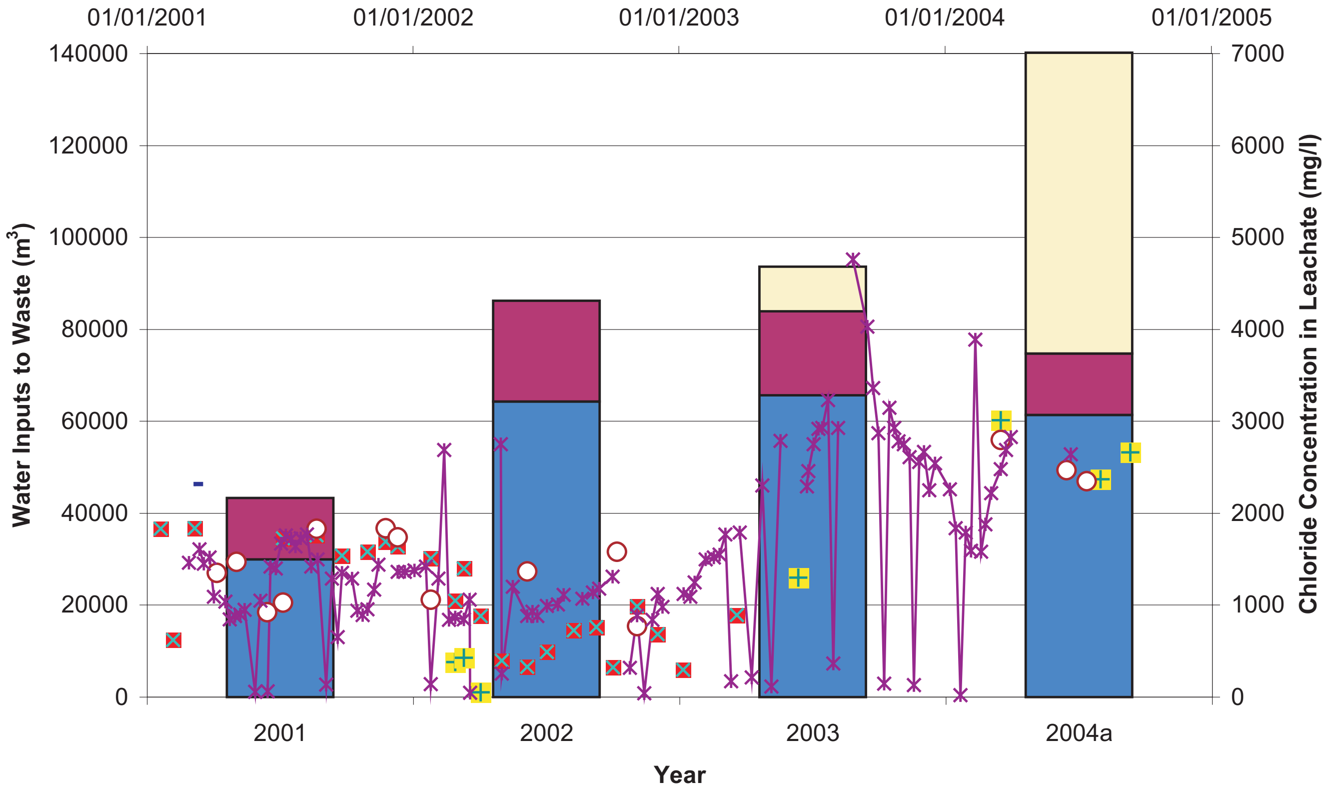
Review of the Performance of Hydraulically Contained Landfills

Figure 3.23
Variations in Leachate COD, BOD, TOC & Alkalinity with Time at Whitehead

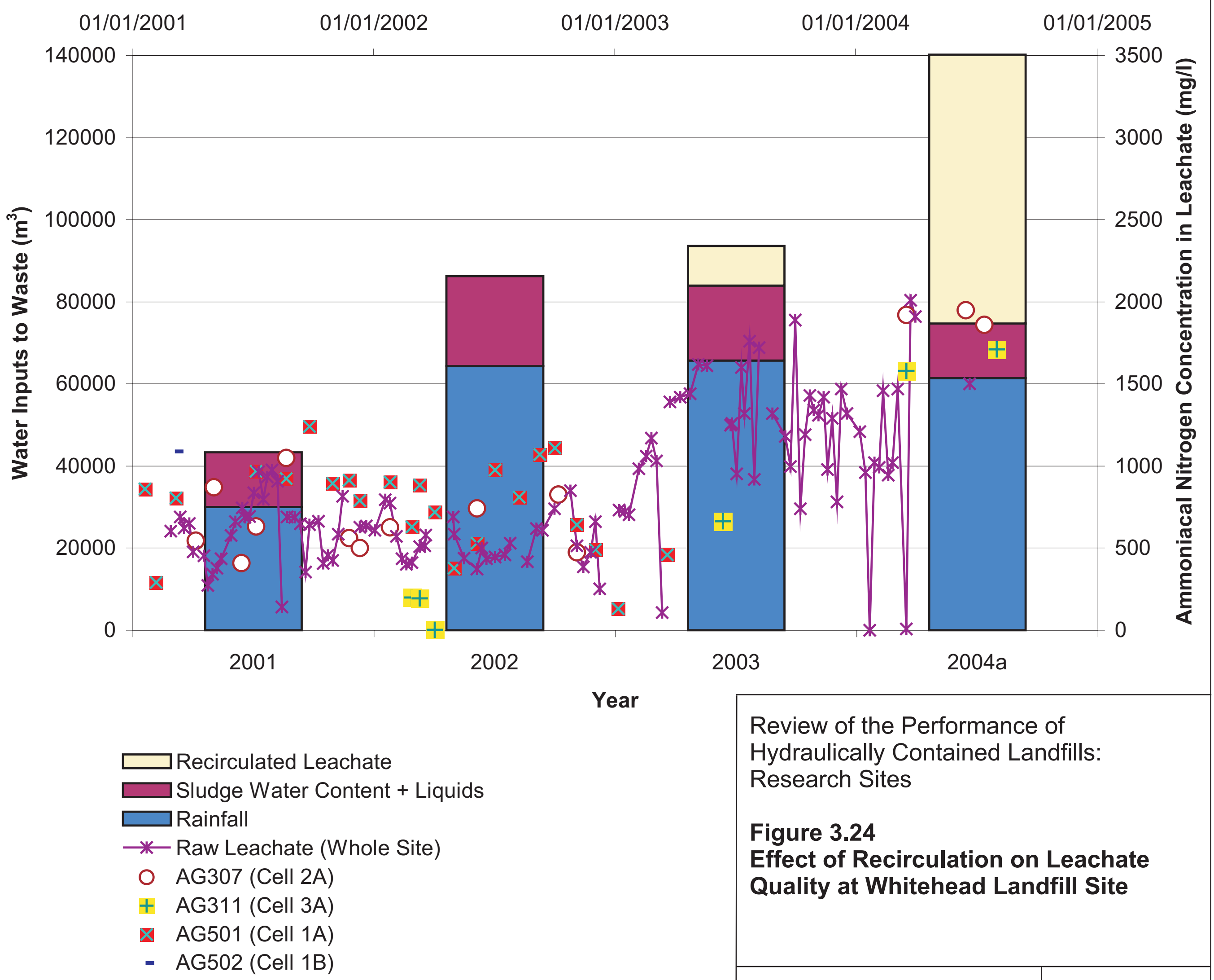
November 2005
10744-S67.cdr wrigs02

Entec

3.24a Effect on Chloride Concentration



3.24b Effect on Ammoniacal Nitrogen Concentration



Review of the Performance of Hydraulically Contained Landfills: Research Sites

Figure 3.24
Effect of Recirculation on Leachate Quality at Whitehead Landfill Site

4. Landfill Gas Generation

4.1 Introduction

4.1.1 Background

This section forms the second part of Task 6 of Phase 2 of the study, as it reviews the effects of hydraulic containment on the production of landfill gas. Part 1 of the landfill gas element of the study was a literature review into such effects and is a stand alone document (Entec UK Ltd, 2005), although a summary of this document is presented here for background reference purposes.

4.1.2 Objectives

This report reviews in-waste and perimeter gas quality monitoring data (gas generation rate data not being available spatially across the individual sites) for each of the sites and determines whether there are any differences between areas that are hydraulically contained and those areas that are not.

4.1.3 Data Collection

Table 4.1 highlights the information Entec had requested from each operator, in order to undertake this study, along with comments as to whether this information could be provided or not.

Table 4.1 Information Requested from Each Operator

Information Required	Comments
Electronic Site Plans	Electronic site plans were requested from each operator, to show the layout of the gas collection system. None of the operators were able to provide Entec with this information, but WRG Ltd and Viridor Waste Management were able to provide a hard copy of the drawings. The site plan does not exist at Poole landfill site, owing to contract negotiations being undertaken between Wyvern Waste Services Ltd and EDL at the time of the study, regarding the provision of landfill gas utilisation at the site.
In Waste Landfill Gas Monitoring	None of the operators were able to provide any landfill gas production data from within the individual stages/phases, as the infrastructure does not exist on the individual wells to afford such monitoring. None of the operators were able to provide Entec with any collection efficiencies within the individual stages/phases. In terms of the landfill gas composition data from within the individual stages/ phases, Most data were provided by WRG for the Brogborough Landfill, limited time series data were provided for the Whitehead site by Viridor together with a broader spatial snap shot dataset. Least data were provided by Wyvern Waste Services Ltd for the Poole Landfill since these data have only recently started to be collected.
Perimeter Landfill Gas Monitoring	Both Wyvern Waste Services Ltd and Viridor Waste Management were able to provide historical perimeter landfill gas monitoring data for Poole landfill site and Whitehead landfill site respectively. WRG Ltd provided composition data for 2004 only, as they had been unable to access previous data on the site which was still held within a former Shanks database.

4.2 Literature Review Summary

4.2.1 Introduction

As part of this study, a literature review (Entec, 2005) was carried out to determine, firstly, the factors which are known to affect the production of landfill gas and, secondly, to determine whether any of these factors will influence gas production from a hydraulically contained landfill site. The pertinent points of this literature review are highlighted in the following sections.

4.2.2 Landfill Gas Production

Much research has been carried out over the years on the mechanisms by which landfill gas is generated. Landfill gas is produced by a series of complex physio-chemical and biological processes within a landfill (Environment Agency, 2002). Landfill gas is generally considered to comprise two distinct fractions: the bulk fraction and the trace fraction.

Bulk Fraction

The composition of the bulk fraction of landfill gas is variable and includes gases of biogenic origin (methane, carbon dioxide and hydrogen), as well as potentially being derived from the corrosion of metals (hydrogen). It also includes the major components of atmospheric air (nitrogen and oxygen), which may be admixed with the other bulk components in varying proportions.

Five stages have been identified in the production of landfill gas (Environment Agency, 2004):

- Phase I Oxygen is consumed and nitrogen is purged from the landfill due to the liberation of other gases (including carbon dioxide);
- Phase II Concentrations of nitrogen reduce due to the continual purging of it from the landfill by hydrogen and carbon dioxide. Alcohols are formed;
- Phase III Acetate is formed;
- Phase IV Methane and carbon dioxide evolve in the ratio of approximately 3:2;
- Phase V A period of endogenous respiration results in the gaseous content of the landfill gradually assuming that of air.

These phases of gas production map directly onto the processes controlling leachate quality discussed in Section 3.4.2 and presented in Figure 3.2, see Figure 4.1.

The period of time it takes for a landfill to progress through all five phases is very variable and is dependant on a wide range of factors, which influence the rate and type of microbial activity within the site. At sites where the rate of microbial activity is high, the five phases may take a few decades.

Landfill gas is produced as a result of the reactions of a number of groups of micro-organisms. These micro-organisms assist in the degradation of the organic fraction within the waste, to form landfill gas. Westlake (1990) states that, "*Bacteria are the most important type of micro-organism involved in these degradative processes and are found throughout landfill.*"

Table 4.2 sets out the typical range of bulk compounds in landfill gas (Environment Agency, 2004).

Table 4.2 Typical Range of Bulk Components in Landfill Gas

Bulk Landfill Gas Components	Typical Value (% v/v)	Observed Maximum (% v/v)
Methane	63.8	88.0
Carbon Dioxide	33.6	89.3
Oxygen [1]	0.16	20.9
Nitrogen [1]	2.4	87.0
Hydrogen	0.05	21.1
Carbon Monoxide	0.001	0.09
Water Vapour	1.8	4.0

Notes: 1 Entirely derived from the atmosphere.

4.2.3 Trace Fraction

In addition to the bulk fraction, landfill gas also comprises a wide variety of trace components. The Environment Agency (2004) have identified over 550 trace compounds within landfill gas, which, in total, make up approximately 1% of the landfill gas by volume. The trace components are formed in one of three ways:

i) By the intermediate biochemical reactions associated with the degradation processes;

ii) By chemical reactions; or

iii) By the degradation or volatilisation of other organic materials deposited in the landfill.

The Environment Agency (2001) has published technical research into the composition and emissions of trace components in landfill gas. In this report, the most cited research (Scott *et al*., 1988) involved testing for trace components of landfill gas at three domestic waste landfills. Sampling of the gas began immediately after the waste was emplaced, and continued for a period of three years. The results of this study showed that a large range of trace components manifested themselves during the different stages of the waste degradation process. The composition of trace landfill gas was subdivided into twelve distinct generic chemical groups, as shown in Table 4.3.

Table 4.3 Generic Groups of Trace Compounds found in Landfill Gas

Generic Group of Trace Compounds	
Hydrogen Sulphide	Esters
Alkanes	Carboxylic Acids
Alkenes	Amines
Cyclic Organic Compounds	Ethers
Halogenated Compounds	Organo-sulphur Compounds
Alcohols	Other Oxygenated Compounds

Source: Environment Agency (2001).

Table 4.4 displays the average concentration of a variety of trace components of landfill gas (Environment Agency, 2004).

Table 4.4 Average Concentration of a Variety of Trace Components of Landfill Gas

Chemical Name	Chemical Group	Median Concentration (μgm^{-3})	Average Concentration (μgm^{-3})
1,1-dichloroethane	Halogenated Organics	13 260	476 223
Chlorobenzene	Halogenated Organics	11 880	246 589
1,1,1-trichloroethane	Halogenated Organics	12 905	189 826
Chlorodifluoromethane	Halogenated Organics	11 570	167 403
Hydrogen sulphide	Organosulphur Compounds	2 833	134 233
Tetrachloroethene	Halogenated Organics	16 640	112 746
Toluene	Aromatic Hydrocarbons	11 995	86 221
Chloroethane	Halogenated Organics	5 190	77 867
n-butane	Aromatic Hydrocarbons	13 623	67 412
Chloroethene	Halogenated Organics	5 600	64 679
Carbon monoxide	Carbon Monoxide	5 822	62 952
Ethylbenzene	Aromatic Hydrocarbons	6 480	37 792
1,2-dichlorotetrafluoroethane	Halogenated Organics	3 200	34 046
Alpha-pinene	Cylco-Alkenes	29 300	33 248
Cis-1,2-dichloroethene	Halogenated Organics	7 700	33 129
Xylene	Aromatic Hydrocarbons	4 700	23 900
Dichlorofluoromethane	Halogenated Organics	3 500	20 131
n-hexane	Alkanes	5 000	19 850
Dichloromethane	Halogenated Organics	1 240	19 054
n-nonane	Alkanes	8 120	19 015
2-butanol	Alcohols	5 400	18 704
1,2-dichloroethane	Halogenated Organics	1 575	16 495
3-methyl-2-butanone	Ketones	1 984	13 614

The differences between the median and average concentrations shown are as a result of extreme values recorded, which influence the average concentrations more than the median concentrations.

4.2.4 Generic Factors Affecting the Production of Landfill Gas

There are many variables affecting the rate of waste decomposition in landfill sites. Micales & Skog (1996) observe that these factors include:

i) Waste management and processing variables - such as the degree of waste compaction and the use of pulverisation;

ii) Waste composition;

iii) Factors influencing bacterial growth - such as moisture, pH, temperature and available nutrients;

iv) Design of the landfill and environmental controls - such as whether there is a gas extraction system;

v) Operation of the landfill - such as cover material and amount of compaction.

The production of landfill gas is influenced by interrelated factors (Pacey & DeGier, 1986). These factors have been widely agreed to include the following parameters as set out in Table 4.5 below and discussed in the following sections.

Table 4.5 Generic Factors Affecting the Production of Landfill Gas

Factors	
Moisture content	Waste type
pH	Density of waste
Waste temperature	Site operational factors
Nutrient availability	

4.2.5 Moisture Content

Moisture content is deemed to be one of the most important factors influencing landfill gas production rates (Environment Agency (2004), Ham (1994), Westlake (1990), Pacey & Dietz (1986), Pacey & DeGier (1986) and Hartz & Ham (1983)). This is of particular relevance to hydraulically contained landfills as the presence of moisture within a landfill site will increase the production of landfill gas, as moisture encourages bacterial growth and will transport nutrients and bacteria throughout the landfill, which in turn increases the rate of biological degradation in the landfill. The Environment Agency (2004) note that an increase in moisture content may promote methanogenesis in a number of ways:

- Dissolution and transport of soluble substrates and nutrients required for methanogenesis;
- Bacterial transport within the waste may be facilitated by water;
- Mixing and buffering within the landfill system will be aided by water;
- Water will dilute the toxic products of acidogenesis, therefore preventing the inhibition of methanogenesis;
- The flow of moisture through a landfill will stimulate microbial activity by providing better contact between insoluble substrates, micro-organisms and the soluble nutrients.

Numerous studies (Ham (1994), Westlake (1990) Pacey & DeGier (1986), Pacey & Dietz (1986) and Hartz & Ham (1983)) have determined the amount of moisture required for optimal methane generation. The consensus from these reports is that maximum methane production occurs when the moisture content (by wet weight) is between 40-50%, although Westlake (1990) recorded peak levels of between 40% and 80%. Table 4.6 highlights the research findings in relation to moisture content.

Table 4.6 Optimal Moisture Content Within Landfill Sites

Authors	Research Findings
Ham, R K (1994)	A moisture content less than approximately 20% on a wet weight basis will greatly inhibit methane generation, ceasing entirely below the 10% moisture level. A low moisture content, or lack of moisture flow, will slow the decomposition process and will favour biological processes, resulting in higher leachate strength and less methane generation. As the moisture content increases to 40-50% on a wet-weight basis, which is at field capacity for most wastes, there will be a steady increase in the rate of methane generation.
Westlake, K (1990)	Moisture levels of between 40% and 80% are required for maximum landfill gas yields. If moisture levels are too high, methane production rates will reduce. Landfill gas production may also decrease, under certain circumstances, if excess moisture is available. The excessive leaching of soluble sugars will lead to the production of large amounts of acidic leachate, resulting in a lower pH, which will inhibit the growth of the methanogenic material and hence the production of landfill gas.
Pacey, J G & DeGier, J P (1986)	Maximum gas production occurs at 40 to 45% (wet weight) moisture content in test landfills.
Pacey, J G & Dietz, A M (1986)	A high refuse moisture content (in the range of 45 to 50% by wet weight) favours maximum methane production.
Hartz, K E & Ham, R K (1983)	Some methane production should occur with moisture levels as low as 10%. Field capacity was found to occur at approximately 40 % moisture content.

The moisture content of the waste can fluctuate, which will affect the production of landfill gas. The factors governing moisture control are given by Pacey & DeGier (1986) as follows:

- Surface-water infiltration (including rainfall);
- Groundwater infiltration;
- Refuse settlement, which will cause an increase in moisture content in the lower portions of the landfill;
- Water released during the decomposition process;
- Liquid additions (such as sludge and process wastes).

How a landfill site is managed will affect the moisture content within a landfill. Landfill sites are typically constructed and filled in a sequential layered pattern, which tends to affect how moisture moves into and through the waste. After periods of prolonged, heavy rainfall, methane production rates often increase. High infiltration rates depended upon whether there is a cap in place on the site and what materials the cap is made from.

Waste compaction will also affect landfill gas production as it increases the density of the landfill contents, therefore decreasing the rate at which moisture can infiltrate the waste.

Moisture content is a factor that can be controlled within a well designed landfill site, with appropriate leachate control systems and management techniques. Most commonly, this is carried out through the recirculation of leachate, as it is generated within the site. The concept of a landfill site with controlled moisture content through leachate recirculation is known as a landfill bioreactor. The aim of a bioreactor landfill is to increase the production rates of landfill gas and to decrease the time required for the stabilisation of the landfill. Studies have shown that the mean rate of methane production nearly doubled when comparing values before and after leachate recirculation.

The results of experiments looking at the effect of moisture content on the rates of production of landfill gas have also concluded that:

- Mature waste produces higher rates of methane production, than newly deposited wastes. This could result from the older waste having a larger, more tolerant methanogenic population. In samples taken from older waste, the maximum methane concentration was recorded when the moisture content was between 40 to 55%, whereas in the samples taken from the newer waste, the maximum concentration was recorded when the moisture content was 75%;
- The work of Knox (1999) and Knox & DeRome (1998) suggests that enhanced leachate levels and moisture content may promote higher CH_4:CO_2 ratios in landfill gas.

It is clear from these results that the on-site management of water can enhance the production of landfill gas. By recirculating leachate, moisture is distributed throughout the waste mass and is “often cited as a system by which waste decomposition can be promoted” (Ham, 1994). The production of landfill gas at many landfill sites may be below the maximum potential, as many sites are operated so as to limit the addition of moisture in an attempt to control leachate generation.

4.2.6 pH

Westlake (1990) describes the impact of pH on the production of landfill gas. He notes that once the methanogenic bacteria are established, the methanogens will remove the acetic acid and hydrogen, formed by the fermentative bacteria. If this process does not occur, volatile fatty acids will be produced, which can accumulate and result in a drop in pH. These acids can then be converted to acetic acid, although if the pH has fallen below the minimum pH required for the growth of methanogens and acetogens, the acid will remain in the landfill. This is referred to as “acid souring”. A low pH may also promote the dissolution of metal ions within the waste mass, which may inhibit methanogenic activity. Therefore, the presence of methanogens within the landfill is essential for the control of pH and to prevent the production of a highly acidic leachate.

Changes in pH will affect the production of landfill gas. Westlake (1990) noted that all micro-organisms are affected by pH and that the methanogenic bacteria within the waste mass will only grow if the pH range is around neutrality (pH of 7.0). The Environment

Agency (2002) commented that the rapid degradation of biodegradable wastes will result in acidic conditions, which may inhibit methane generation.

The effect of pH on refuse methanogenesis was evaluated by Kasali et al (1988), by the addition of a solution of phosphate (0.2 M), buffered to pH levels of between 5.3 and 8.3 to the samples of waste. The authors did not know whether these samples were in an active state of methane production at the time of sampling. The buffer (notably also a source of nutrient) was reported to stimulate acid production, resulting in the accumulation of carboxylic acid, which inhibited methane production. Buffering is particularly important during the early stages of degradation, when excess acids are produced and pH levels can drop quickly.

Research has shown that the optimal pH for the production of landfill gas is between 6.5 and 8.5. A summary of this research is given in Table 4.7 below:

Table 4.7 Optimal pH Within Landfill Sites

Authors	Research Findings
Environment Agency (2002)	Methanogenesis occurs between a pH of 6.5 to 8.5. An optimal pH of 7.4 will occur during Phase 4 of the degradation process (refer to Figure 2.1), when methanogenic bacteria are firmly established.
IWM Landfill Gas Monitoring Working Group for the Institute of Wastes Management (1998)	A pH of between 6.5 to 8.5 is the optimal range for methane production.
Ham, R K (1994)	The optimum pH for methane generation is in the 7 to 8 range.
Pacey, J G & Dietz, A M (1986)	Methanogens require a narrow pH range of 7.0 to 7.2 for optimum methane generation.
Pacey, J G & DeGier, J P (1986)	The optimal pH for methane gas production is near neutral, between 6.8 and 7.2.

4.2.7 Temperature

The temperature of the landfill is another important factor influencing the rate of landfill gas production (Environment Agency (2004), Ham (1994) and Hartz et al (1982)). Yesiller & Hanson (2003) comment that temperature affects the physical, chemical, biological, and mechanical properties and behaviour of wastes and liner materials in landfills. They note that even though temperature has significant effects on various landfill components, there is limited information available on temperatures within wastes, liner systems and the surrounding subsurface.

Research has shown that optimum methane production occurs at certain temperatures and that methane production will cease if the temperatures are too high or too low. The Environment Agency (2002) noted that temperatures as high as 90 ^{o}C could be encountered during Phase 1 of the waste degradation process and that temperatures will reduce as the waste degradation process continues, before stabilising at an optimal 35 to 45 ^{o}C as soon as methanogenesis is well established. Table 4.8 highlights the research findings in relation to temperature and landfill gas production.

Table 4.8 Optimal Temperature Within Landfill Sites

Authors	Research Findings
Environment Agency (2002)	Temperatures will stabilise at an optimum of 35-45°C, once methanogenesis is well established.
Ham, R K (1994)	A temperature less than 20°C will inhibit methane generation. Peak methane generation rate occurs at approximately 40 to 45°C. Temperatures greater than 45°C will produce a marked drop off in methane generation rate.
Hartz, K E, Klink, R E & Ham, R (1982)	The optimum temperature was found to be 41°C, with methane evolution ceasing between 48°C and 55°C.

Changes in atmospheric temperature will have a far greater effect on the rate of gas production in shallow (up to 8 m in depth) landfills, than in deeper sites, as the bacteria are not insulated against changes in temperature. The Environment Agency (2002) observe that landfill gas production will tend to drop when temperatures within the landfill are below 10 to 15°C, and this may result in a seasonal pattern of waste decomposition and landfill gas production at shallow landfills.

Other research has shown:

- Temperatures, in cells where leachate recirculation was occurring, were generally 5° to 10°C higher than found in those cells where it was not practised;
- The rate of temperature increase in cells containing older waste is lower than that recorded in cells containing newer waste;
- A large influx of surface water may cool areas of the landfill site.

4.2.8 Waste Type

The composition of the waste deposited within a landfill will influence both the rate of production and the composition of the landfill gas generated (Environment Agency, 2004). For example, a biodegradable waste landfill will differ from an inert waste landfill site in the composition and quantity of landfill gas it produces.

The implementation of the Landfill Directive (1999/31/EC) will have an impact on the nature and composition of the wastes disposed of within a landfill site. The type of waste deposited in a landfill will impact upon the rate of waste degradation and also the proportion of gaseous compounds in landfill gas mixtures, as certain waste types are more readily degradable than others. Pacey & DeGier (1986) commented that waste containing a high percentage of readily decomposable organic materials (such as food, garden and paper wastes) would result in a high production of methane per unit volume. The more organic waste present within a landfill, the more landfill gas is produced by bacterial decomposition. Ham (1994) concurs that certain waste components degrade more quickly than others. He notes that food waste, for example, degrades very quickly within a landfill, compared to newspapers, which have relatively high lignin contents.

4.2.9 Waste Density

The density of waste is another factor affecting the production of landfill gas (Environment Agency (2004) and Pacey & DeGier (1986)). Waste density is a function of the waste deposited, its particle size and the degree of compaction. In theory, the yield of landfill gas per unit volume of void space increases with waste density. However, waste permeability generally decreases with increased waste densities, thereby inhibiting the free movement of the soluble nutrients required by the bacteria to flourish.

4.2.10 Site Operational Factors

The way in which a site is operated can influence the rate of production of landfill gas. Influencing factors include:

- Site containment;
- Pre-treatment of waste;
- Cellular arrangement for waste disposal within a landfill site;
- Leachate management (recirculation and extraction).

More specifically:

- The use of daily cover may inhibit the movement of leachate through the site, thus affecting moisture contents in deeper waste;
- An engineered cap has been shown to reduce the surface emissions of landfill gas. Johnston et al (2000) discovered that an engineered cap and an effective gas control system can reduce the rate of surface methane emissions by three orders of magnitude, compared with sites without such control measures;
- Once a completed cell has been capped, the amount of gas produced will increase as it is contained within the site (as long as a sufficient liner is in place).

4.2.11 Nutrients

Relating to nutrients, Ham (1994) states that there is little information given to support whether or not nitrogen or phosphorus inhibit or promote the decomposition of waste. Conversely, Pacey & DeGier (1986) note that the carbon-nitrogen ratio is a factor found to affect the production of landfill gas and that the growth medium should exhibit a carbon-to-nitrogen ratio in the order of 16:1 (carbon:nitrogen) to maximise the landfill gas rates.

Filip and Kuster (1979) tested whether certain nutrients (ammonia, glucose, ammonia plus glucose or peptones) could increase the microbial activity within the waste. They concluded that glucose, peptones and ammonia plus glucose stimulated the evolution rate of carbon dioxide. Ammonia by itself had no effect, which led to the conclusion that the availability of carbon, but not nitrogen, limited microbial activity.

Barlaz et al (1989) concluded that ammonia, phosphate and sulphur did not limit the onset of methane production and that the concentrations of these nutrients, present in the accelerated methane production phase, supported a methane production rate of at least 929 litres of methane (at standard temperature and pressure) per kilogram of dry refuse per year. They concluded that more research is needed to determine whether nutrients limit methane production in the final stages of waste decomposition.

Watson-Craik & Sinclair (1995) have studied the effects of added nutrients on methanogenesis, by using six refuse packed lysimeters under varying buffering conditions:- nitrogen (urea), phosphorus, anaerobically digested sludge, and septic tank residue additions. The lysimeters seeded with both nutrients (phosphorus and nitrogen) and a buffer, exhibited methanogenic lag phases 70 days shorter than the control reactors. The continuation of nutrient addition, once methanogenesis had started, did not improve the rate of methane formation when compared with the buffer only controls.

4.3 Factors Affecting the Production of Landfill Gas at Hydraulically Contained Landfill Sites

The factors which are likely to be most important in affecting landfill gas production in hydraulically contained landfill sites, are:

- moisture content;
- pH;
- temperature;
- nutrients; and
- operational factors.

There has been limited research into the production of landfill gas specifically from hydraulically contained landfill sites. Sections 4.2 to 4.6 of the Literature Review (Entec, 2005) discuss specific research relating to the effect the above factors have on the production of landfill gas and relates these findings to hydraulically contained landfill sites. These findings are summarised in the subsections below.

4.3.1 Moisture Content

As discussed in Section 4.2.5, moisture content is considered to be one of the most important factors influencing landfill gas production rates. The presence of moisture within a landfill will promote landfill gas production, as moisture encourages bacterial growth and will transport nutrients and bacteria throughout the landfill.

Within a hydraulically contained landfill site there is a potential for increased moisture content and saturation of the waste as a result of groundwater ingress.

Although the operation of a hydraulically contained landfill site aims to ensure that no leachate can escape from the site, groundwater may enter the wastes from the surrounding strata. In order to maximise the production of landfill gas from a hydraulically contained landfill site, the following conditions are likely to be required:

1. Moisture levels within the landfill site to be maintained at a level between 40% and 55%;

2. Leachate recirculation to be practised to ensure that essential nutrients and bacteria are transported throughout the landfill site. The same logic would seem to apply to encouraging groundwater/leachate movement through the waste by pumping of leachate and allowing some groundwater ingress;

4.3.2 Temperature

The temperature within a landfill site may be affected, if the landfill site is sub-water table, operating on the principle of hydraulic containment. Warmer temperatures increase bacterial activity, resulting in an increase in the rate of landfill gas production, whereas lower temperatures would reduce such activity.

Hydraulically contained landfills may be subject to groundwater ingress. As groundwater temperatures in the UK (typically ~10°C) are lower than that of the landfill leachate, a cooling effect may be produced. This could result in a decrease in landfill gas production. However, this effect is likely to be counteracted, at least in part, by enhanced moisture content and movement of nutrients.

4.3.3 Site Operational Factors

The operational factors described in Section 4.2.10 i.e. use of daily cover, capping of the waste etc above will apply equally to hydraulically contained and non-hydraulically contained landfills. Where there is limited infiltration downwards through low permeability daily covers, groundwater ingress may help to increase the moisture content of the lower waste.

4.3.4 Nutrient Availability

There is a lack of research relating to nutrient availability within landfill sites and its impact on landfill gas production. However, the limited work undertaken suggests that the availability of some nutrients may influence the rate at which methanogenic conditions are established within landfills. At hydraulically contained landfills, groundwater ingress may increase nutrient mobility but if ingress rates are high may also dilute and wash out important nutrients, which could adversely impact landfill gas generation.

4.3.5 pH

It is generally accepted that the optimum pH for methanogenic bacteria is between 6.8 to 7.4 (Zehnder, 1987). It has also been shown that it is necessary to maintain pH values between 6.8 to 7.4, as these conditions are optimal for methanogenic bacteria. The ingress of groundwater with significant alkalinity into hydraulically contained landfills may assist the development of stable pH conditions nearing neutrality, by diluting lower pH landfill leachate. This may favour stable landfill gas production.

4.4 Site Studies - (1) Brogborough

4.4.1 Introduction

Site specific data have been collected from each of the landfills, Brogborough, Poole and Whitehead, and are discussed in the following sections. Background information relating to the environmental setting, site development and hydrogeology for each of the three sites has been presented in Chapter 2, and more detailed information can be obtained by reference to the initial reports (Entec, 2003a, Entec, 2003b, Entec, 2004).

4.4.2 Landfill Gas Collection and Treatment at Brogborough

A site plan detailing the gas installation and monitoring points is shown on Figure 3.16. By 2004, there were 390 wells installed at Brogborough landfill site, which equates to approximately 3.1 gas wells per hectare. The installation pipework is constructed of MDPE (6 bar to 10 bar) of varying sizes. The ring main consists of a 355 mm diameter pipe and the majority of the surface laid collection pipework is either 160 mm or 200 mm in diameter. The wells are spaced at approximately 50 m centres, wherever possible. The well casing is 160 mm for the gas wells and 225 mm for dual purpose gas/leachate wells, to allow for the installation of pumps as necessary.

There are six landfill gas engines, with a combined generating capacity of over 20 MW. In addition, there are four flares installed at the site, with three of the flares having a combined capacity of 7500 m^3/hr. The specifications of the engines and the flares are given in Table 4.9 and Figure 4.2 shows the Jenbacher engine installed at the site.

Table 4.9 Landfill Gas Engines Installed at Brogborough Landfill Site

Manufacturer	Specification
Caterpillar	2 x Caterpillar 1.1 MW engines. Both commissioned in 1999.
MIRRLEES KP7	2 x MIRRLEES 2 MW engines. One commissioned in 1991, the second in 1992 (has since been decommissioned (2003)).
Jenbacher 620	1 x Jenbacher 620 2.4 MW engine (to replace the MIRRLEES KP7 engine which was decommissioned in 2003). Commissioned in 2003.
MIRRLEES KVP16	2 x MIRRLEES KVP16 6.75 MW engines. One commissioned in 1991, the second in 2001.

Note: Information provided by WRG Ltd, 9 November 2004.

The specifications of the flares installed at the site are shown in Table 4.10 and Figure 4.3 shows the three Stirling flares installed at the site.

Table 4.10 Landfill Gas Flares Installed at Brogborough Landfill Site

Manufacturer	Specification
Unknown flare (within Compound 1)	1 flare was installed in approximately 1990. No details exist on this flare.
Stirling (within Compound 2)	3 x Stirling flares, each with a capacity of 2 500 m^3, stack height of 14 m and a burn temperature of between 850°C and 1300°C.

Note: Information provided by WRG Ltd, 9 November 2004.

4.4.3 Leachate Collection and Treatment

In older areas of the Brogborough site, no leachate drainage facilities were installed in the base of the site and leachate has been abstracted from the waste from vertical 'retrofit' wells. Small amounts of leachate (~6 m^3/day) from Stage 1 were recirculated into Stage 2 wastes for several years, with no leachate removed off site. More recently, there has been less recirculation, and leachate has been removed from the site. During the period April 2003-September 2004, approximately 29 302 m^3 leachate was removed from the site for treatment off-site.

Three 50 m deep wells, installed in Stages 4A and 4B, typically showed that the waste was drier towards the base of the Stages. Whilst these recorded a significant depth of leachate after drilling, they remained dry after pumping, suggesting that leachate may be perched at higher levels in the wastes.

4.4.4 In-Waste Monitoring Gas Wells Data Collection

Entec visited Brogborough landfill site on 25 August 2004, in order to attain an improved understanding into how the site is operated. Members of the landfill gas team at WRG provided Entec with all of their available data. It was not possible to obtain all of the data originally requested from the operator, since some is not collected (see Table 4.1).

For example, Entec was unable to obtain any gas production data from the different phases of the landfill. This is because the wells installed in the site have not been fitted with well heads, which enable this parameter to be recorded. The site currently gauge the production of landfill gas in one of two ways; how much gas is used by the engines, or how much gas is being sent to the flares. This does not allow conclusions to be drawn over landfill gas production from within the individual cells, as the gas is being drawn from across all of the gas fields covering the site. In the near future, the site operators are hopeful of gathering more accurate flow data from individual gas lines that lead into condensate knock-out pots distributed around the ring main, using anemometers.

Since landfill gas production data per individual stage of the landfill were unavailable, the following analysis looks for any effects that could be related to the site's sub-water table or hydraulically contained nature on the composition of the landfill gas within the different stages of the site.

4.4.5 Methane Content

Table 4.11 sets out the average methane concentration recorded within each stage and Figure 4.4a compares the average methane concentration per well against the standard deviation for recorded methane concentrations at that well.

Table 4.11 Average Methane Composition from the In-Waste Gas Wells

Landfill Area	Average Methane Concentration (%)
Stage 1	46.9
Stage 2	38.0
Stage 3	51.5
Stage 4A	51.0
Stage 4B	47.7

Figure 4.4a indicates that the average methane concentrations are related to variability in the data and that there appears to be a similar variability in each Stage of the landfill. Variability in methane concentration is significantly related to oxygen concentration (see Figure 4.4b) and so is likely to be related to air entrainment in the well or gas collection system rather more than variability in production of methane. This analysis means that use of methane concentration data, where significant (>2%) oxygen is detected in the same sample, is not appropriate. (Note if the oxygen content is 2% then nitrogen concentrations from air entrainment will be 8%, so this gives a 10% error to the total gas concentration). For this assessment of Brogborough's data, ratios of methane to carbon dioxide have been used to negate the affects of air entrainment or incompletely purged nitrogen from the unsaturated waste.

4.4.6 Controls on Methane to Carbon Dioxide Ratios

Factors to Consider

Section 4.3 has summarised that the key factors controlling landfill gas (methane) production are:

- moisture content;
- pH;
- temperature;
- nutrients; and
- operational factors.

If it is assumed that the waste at Brogborough provides a consistent source of nutrients across the site and that operational factors have remained similar, then methane to carbon dioxide ratios should depend on moisture content, pH and temperature. If any affects of additional water inputs in the hydraulically contained parts of the site are to be revealed, the affect of pH and temperature needs to first be checked.

The following subsections compare methane to carbon dioxide (CH_4/CO_2) (vol/vol) ratios (where O_2 <2%) against pH and temperature of leachate sampled over the period May to December 2003 and the calculated liquid to solid ratio for the waste at each well at this time.

This is the period for detailed evaluation of leachate levels by Entec (February 2004) and for this report's assessment of leachate quality (see Section 3.7).

Relationship between Methane:Carbon Dioxide Ratio and Leachate pH

Figure 4.5a shows CH_4/CO_2 ratios plotted against leachate pH. There are limited leachate quality data when compared to gas composition data to allow a relationship between pH and CH_4/CO_2 to be explored. Although pHs are in the range 6.4 to 8.5, consistent with the range for methane generation discussed in Section 4.2.6, with the exception of two anomalous points, there is no obvious peak at pH 7.4 to 7.5. On this assessment pH does not show an obvious control on CH_4/CO_2 ratios.

Relationship between Methane:Carbon Dioxide Ratio and Leachate Temperature

Figure 4.5b shows CH_4/CO_2 ratios plotted against leachate temperature. There are more data available than for leachate pH, as temperature is recorded on monthly dipping of leachate levels. The plot suggests there is an increase in (CH_4/CO_2) ratio as temperatures fall. This is not consistent with an optimum temperature range of 35-45°C for methane generation and a significant drop off in methane generation below 20°C as discussed in Section 4.2.7. It is consistent with a drop off in methane production above ~50°C. The difference is likely to be due to trying to compare compositional ratios rather than gas generation volumes. It is noted that increasing temperature should decrease the solubility of both CH_4 and CO_2 in water (leachate). For example CO_2 is 1.7 times more soluble at 20°C than at 40°C.

From this assessment, however, it appears that temperature differences could plausibly and typically account for CH_4/CO_2 ratios between 1.1 and 2.3. Anomalously high ratios in a number of Stage 1 samples are noted.

Relationship between Methane:Carbon Dioxide Ratio and Liquid to Solid Ratio

Figure 4.6 shows CH_4/CO_2 ratios plotted against liquid solid ratios. The calculation of leachate solid ratios for leachate temperature and leachate quality data has been discussed in Section 3.7.

In broad terms there appears to be two trends within the plotted data, one with CH_4/CO_2 ratios increasing to ~2 at a liquid solid ratio of ~0.35 and a second with CH_4/CO_2 ratios increasing to >3 at a liquid solid ratio of ~0.25. The latter suggests that a higher CH_4/CO_2 ratio is achieved in certain wells for a given liquid solid ratio or that the liquid solid ratio for these wells does not include all water inputs, e.g. groundwater inputs.

To explore this further, Figure 4.7 also labels the wells for those stages where both trends appear to be operating. This has been used to highlight the locations of wells with relatively high CH_4/CO_2 ratios for a given liquid solid ratio on Figure 4.7. It is noted that these wells are located within the sub-water table part of the Brogborough site whereas the wells with relatively low CH_4/CO_2 ratios for a given liquid solid ratio are located in areas which have not been sub-water table.

This distribution of wells is not inconsistent with a process whereby the wells with high CH_4/CO_2 ratios for a given liquid solid ratio, which are located within the subwater table part of the site, have had additional water inputs from groundwater ingress. Other factors may be involved, but on the basis of this analysis groundwater ingress appears to have lead to higher CH_4/CO_2 ratios than for waste of a similar thickness with only rainfall infiltration inputs.

Summary of Controls on Landfill Gas Composition

Landfill gas composition at Brogborough appears to be affected by operational practice, leading to the entrainment of air, and environmental factors in the waste, in particular pH, temperature and liquid to solid ratio. By using data affected at the most by limited (10%) air entrainment and working with methane to carbon dioxide ratios, it appears that the waste which is in the sub-water table part of the landfill site may be generating gas with a higher ratio of CH_4(%) to CO_2(%) than in the above the water table parts of the site due to additional water inputs from groundwater ingress.

4.4.7 Landfill Gas Monitoring Data - Perimeter Monitoring Boreholes

WRG provided Entec with perimeter gas monitoring data for 2004. They were unable to provide any earlier data due to the change in ownership from Shanks to WRG at the time of this work.

Table 4.12 summarises the landfill gas composition data recorded in the off-site perimeter boreholes and gas probes around Brogborough landfill site. This shows that the only boreholes that have recorded historical elevated concentrations of landfill gas (methane) are BGGB0020 (southeastern edge of Stage 1) and BGGB0014 (northwestern edge of Stage 4A). Neither Stage 1, nor Stage 4A have side wall lining and near both locations there is waste above the water table. There is however, insufficient data to draw a conclusion as to whether subwater table areas have any affect on gas control.

Table 4.12 Landfill Gas Composition Data in the Perimeter Gas Monitoring Boreholes and Probes

Area	Monitoring Points	Comments
Stage 1	Perimeter Gas Borehole BGGB0020 Gas Probes BGGP0001, BGGP0002, BGGP0003, BGGP0004, BGGP0005, BGGP0006 BGGP0007, BGGP0008B, GGP0009, BGGP0010 & BGGP0011	Perimeter Gas Borehole Elevated levels of methane (above 1%) and carbon dioxide (above 1.5%) have been recorded, up to a maximum of 9.1% and 8.7% respectively in September 2004. There is no sidewall lining in Stage 1. Gas Probes No concentrations of methane have been recorded and no elevated concentrations (above 1.5%) of carbon dioxide have been recorded during 2004
Stage 2	Perimeter Gas Borehole BGGB0005 Gas Probes BGGP0012, BGGP0013 & BGGP0014	Perimeter Gas Borehole No concentrations of methane have been recorded in the perimeter gas borehole (GB05), although there has been one incidence of elevated carbon dioxide, when 1.6% was recorded in BGGP0013 in February 2004. There is no sidewall lining in Stage 2. Gas Probes No concentrations of methane have been recorded during 2004, although there has been one incidence of elevated carbon dioxide, when 1.6% was recorded in BGGP0013.
Stage 3	Perimeter Gas Borehole BGGB0006 Gas Probes BGGP0035, BGGP0036, BGGP0037, BGGP0038 & BGGP0039	Perimeter Gas Borehole No concentrations of methane have been recorded in the perimeter gas borehole. Elevated levels of carbon dioxide have been recorded during 2004, with a maximum concentration of 2.1%. There is no sidewall lining in Stage 3. Gas Probes No concentrations of methane or carbon dioxide have been recorded during 2004.

Table 4.12 (continued) Landfill Gas Composition Data in the Perimeter Gas Monitoring Boreholes and Probes

Area	Monitoring Points	Comments
Stage 4A	Perimeter Gas Boreholes BGGB0014, BGGB0015, BGGB0027 & BGGB0029 Gas Probe BGGP0028	Perimeter Gas Borehole No elevated concentrations of methane or carbon dioxide have been recorded in the perimeter gas boreholes, except for a one off of 0.6% of methane in GB14 and ≤1.1% of carbon dioxide in GB29. There is no sidewall lining in Stage 4A. Gas Probes No concentrations of methane or elevated levels of carbon dioxide have been recorded during 2004.
Stage 4B	Perimeter Gas Boreholes BGGB0030, BGGB0031 & BGGB0032 Gas Probes	Perimeter Gas Borehole No concentrations of methane have been recorded in the perimeter gas boreholes. Elevated concentrations of carbon dioxide have been recorded at GB30 (≤2.2%) and GB31 (≤5.0%). Stage 4B has basal and sidewall lining. Gas Probes There are no perimeter gas boreholes surrounding Stage 4B.
Cell 6	Perimeter Gas Boreholes BGGB0040 & BGGB0042 Gas Probes	Perimeter Gas Borehole No concentrations of methane have been recorded in the perimeter gas boreholes. Elevated concentrations of carbon dioxide have been recorded at GB40 (≤1.8%). Cell 6 has basal and sidewall lining. Gas Probes There are no perimeter gas boreholes surrounding Cell 6.

4.5 Site Studies - (2) Poole Landfill Site

4.5.1 Introduction

Information relating to the environmental setting and development of Poole landfill was presented in an earlier report in the study (Entec, 2003a) and summary information has been included earlier, in Chapter 2.

4.5.2 Landfill Gas Collection and Treatment

Details of how the landfill gas management system at Poole has evolved are given below. A site layout plan is included as Figure 2.2.

Phases 1 to 3

Originally gas wells were drilled in Phase 1 of the site, after it had been landfilled and partially capped. These were connected to a landfill gas ground flarestack, rated at 500 m^3/hr, which was located in the compound adjacent to the vertical leachate pump. Subsequently, as Phases 2 and 3 were landfilled, gas wells were constructed and progressively raised to the surface level as part of the tipping operations. These gas wells consisted of stone columns two metres in diameter, with a geogrid used to retain the stone in place. At the centre of the well, a vertical perforated pipe was installed. Once the gas wells had reached their full height and landfilling in that phase had ceased, gas collection pipes were laid over the surface and connected to the existing flarestack. The gas extraction system was left exposed and capable of extension so further waste could be tipped to redress settlement.

Flarestack No. 1 was in operation until 1995, when increased gas generation required the burning capacity to be increased. In consequence, a UKPS1500 elevated landfill gas flarestack, rated at 1500 m^3/hr, with a retention time of 0.3 seconds (minimum) and an optimal burn temperature of 1,100°C, was installed. The flare is currently in operation and incorporates a de-watering tank, to abstract condensate, as well as a flame arrester and slam shut valve. In addition, the flarestack incorporates automatic shut down for high temperature, high oxygen content, low methane content and high condensate level. Since its installation, a telemetry system has been provided which monitors the flarestack and provides alarm calls to the site office and to staff at their homes, in the event of any of the above shut downs or mains power failure. The controls of the flarestack have been further modified with manual overrides, such that it is possible to operate the flarestack as long as mains power is available.

Phase 4

The gas management system for this phase consisted of gas wells, of the same specification as installed in Phases 2 and 3, which were located over, or immediately adjacent to, the basal leachate collection system. In addition to its capability to collect landfill gas, this design allows leachate to drain downwards to the leachate collection system. Temporary connections from the gas wells to the flarestack were made, but these were not satisfactory, due to operational problems with vehicle movements and the difficulty in draining condensate, which caused odour problems. As a consequence, the upward construction of the gas wells ceased in 1998 and they were covered and provided with horizontal gas drains which allowed gas to be collected without the above condensate problems.

In order to overcome the conflict between pipeline routes and vehicle movements, and also to provide temporary additional flaring capacity pending the installation of a power generation plant, a Hoffstetter landfill gas ground flarestack (known as Flarestack No 2) was installed in 1998 in the southern corner of the site, in the vicinity of Billybrook House. This has a rated burning capacity of 250 m^3/hr, but will actually burn in excess of 400 m^3/hr. The optimal burn temperature is rated at 850°C, and it is provided with automatic de-watering of condensate and a flame arrester and slam shut valve. A UV flame sensor is fitted to detect the flame and automatic re-ignition is provided in case of flame failure or mains power failure. The flarestack served the western end of Phase 4 as well as the waste under the metalled road in Phase 5. This flarestack was installed in 1998 and was turned off in October 2003. Subsequently, at the end of summer 2004, further gas wells were drilled in the area of Phase 4 that had earlier been capped. These were connected into Flarestack No 1.

Power Generation

There is currently no power generation at the site, however, during 2000, EDL were contracted to install an electricity generation plant, to utilise the gas produced by the site. During the summer of 2000, the existing gas management system in Phases 1-3 was modified by EDL such that the gas wells on these phases were grouped together so that gas could be collected at one of four well stations. In addition, some of the existing gas pipelines from Phase 4 were connected to the well stations. Each well station contained the control valves and monitoring points to enable the flow of landfill gas to be maintained at its optimum for power generation. Main collector pipes were then laid to convey the gas to the power generator site, with a temporary link to Flarestack No 1. This link is currently still in use, whilst the contractual agreement with EDL is negotiated.

Owing to the ongoing discussions with EDL regarding the collection and utilisation of landfill gas, no on-waste landfill gas monitoring had been undertaken prior to June 2004, as many of the wells have been buried as a result of landfilling operations.

In the early summer of 2001, further gas wells were drilled in Phase 4, with the majority being drilled directly over those gas wells whose construction had been truncated in 1998. This allowed for continuity so that gas could be drawn from the lower levels of this phase and be connected temporarily into Flarestack No 1.

4.5.3 Collection of Monitoring Data

Entec visited Poole landfill site on 29 April 2004, in order to attain an understanding into how the site is operated. Entec was unable to obtain some of the data requested from the operator, as it is not collected, or required to be collected (see Table 4.1). Wyvern Waste were unable to provide any historical on-site gas monitoring data from within the different phases, or any leachate level monitoring data, because the structures were not in place on the site to facilitate the monitoring of leachate and landfill gas. In addition, leachate temperature is not recorded at the site.

This section of the report reviews data available for the site and considers whether the hydraulically contained areas are behaving any differently to the non-hydraulically contained areas. This comparison has been extremely difficult at Poole, and the analysis tenuous, owing to the quality and quantity of the data available.

4.5.4 Landfill Gas Monitoring - On-Site Gas Wells

Landfill gas data from within the individual phases of the landfill site was available only for one set of measurements taken from the retrofit combined leachate and gas wells constructed in June 2004 (Frederick Sherrell Ltd, August 2004). Data collected from the completed wells on 21 June 2004 are presented in Table 3.2 with leachate quality data for those same wells.

Applying the same assessment of gas composition data as undertaken for Brogborough (see Section 4.4.6) has not been wholly possible due to the limited data. The same data selection criteria of <2% oxygen has been used to exclude samples with air entrainment contamination. However there are no leachate pH or temperature data to allow the affects of these controls on methane generation to be evaluated. CH_4 /CO_2 ratios are in the range 1.53±0.04 and there is no obvious relationship with liquid solid ratios (discussed in Section 3.8.6) (see Figure 4.8). In comparison to Brogborough the CH_4/CO_2 ratios are similar to those wells with little groundwater ingress and this is consistent with the conclusions from Section 3.8.6 which indicated that there was little evidence of waste stabilisation from groundwater ingress inputs. Instead, most of the groundwater ingress appears to be collected by the basal drainage system.

4.5.5 Landfill Gas Monitoring - Perimeter Monitoring Boreholes

Wyvern Waste provided perimeter gas monitoring data from 1995 onwards.

Methane concentrations

Tables 4.13 and 4.14 summarise the methane concentrations recorded in monitoring boreholes.

From Tables 4.13 and 4.14 it can be concluded that a similar percentage of boreholes surrounding each phase are recording elevated levels of methane concentrations (56% of boreholes adjacent to Phases 1-3 and 62% of boreholes surrounding Phase 4). However, the data also show that higher concentrations of methane are generally recorded in the boreholes surrounding Phases 1-3 and that these occur more frequently, in comparison with the data recorded in the perimeter boreholes surrounding Phase 4. About 25% of the monitoring boreholes around Phases 1-3 have recorded mean methane concentrations of 30% or more, with no boreholes around Phase 4 recording such concentrations. Phase 4, being more recent, perhaps has somewhat better side seal gas control. This together with the waste being more recent in Phase 4 could explain the lower and less frequently recorded concentrations. Whilst it is also not implausible that high groundwater levels and hydraulic containment for Phase 4 is helping to restrict lateral gas migration, this is not clearly evident given the possible influence of physical containment and age of waste factors. **There is, however, no evidence to indicate that hydraulic containment is leading to poorer gas control compared to where the waste is not hydraulically contained.**

Table 4.13 Landfill Gas Composition Data in Perimeter Gas Monitoring Boreholes - Phases 1-3 (1995-2004)

Perimeter Gas Borehole	Methane Composition[a] (%)	Mean CH_4 / CO_2 Volumetric Ratio	Perimeter Gas Borehole	Methane Composition[a] (%)	Mean CH_4 / CO_2 Volumetric Ratio
BH001	0.0 – **31.3** -82.4	3.39	BH046	0.0 – **11.6** – 70.8	1.75
BH002	0 – **59.3** – 81.5	3.32	BH049	0.0 – **0.01** – 0.7	1.28
BH005	0 – **0.005** – 0.2	0.10	BH050	0.0 – **0.02** – 2.90	0.47
BH006	0 – **0.004** – 0.2	0.16	BH059	0.0 – **46.4** – 66.2	2.08
BH007	0.0 – **0.3** – 65.2	0.97	BH060	0.0 – **0.003** – 0.3	0.14
BH008	0.0 – **0.2** – 45.8	10.51	BH061	0.0 – **0.0** – 0.0	[b]
BH009	0.0 – **2.2** – 46.6	1.09	BH062	0.0 – **0.0** – 0.0	[b]
BH010	0.0 – **0.006** – 0.6	0.50	BH063	0.0 – **0.04** – 8.9	0.13
BH012	0.0 – **0.001** – 0.1	0.11	BH064	0.0 – **0.0** – 0.0	[b]
BH013	0.0 – **0.004** – 0.4	0.11	BH065	0.0 – **8.7** – 53.1	3.98
BH014	0.0 – **0.006** – 0.4	0.33	BH066	0.0 – **0.001** – 0.1	[b]
BH015	0.0 – **0.003** – 0.2	[b]	BH067	0.0 – **0.0** – 0.0	[b]
BH016	0.0 – **0.66** – 2.9	0.45	BH087	0.0 – **6.3** – 51.6	16.40
BH017	0.0 – **73.9** – 94.9	10.22	BH088	0.0 – **30.5** – 70.5	10.10
BH018	0.0 – **0.0** – 0.0	[b]	BH105	0.0 – **52.1** – 75.6	3.15
BH019	0.0 – **9.8** – 62.1	2.36	BH106	0.0 – **57.9** – 71.9	2.52
BH045	0.0 – **11.7** – 80.9	1.93	BH107	0.0 – **46.9** – 80.5	2.54

Note: Data shown as minimum - **mean** - maximum.

Table 4.14 Landfill Gas Composition Data in the Perimeter Gas Monitoring Boreholes - Phase 4 (1995-2004)

Perimeter Gas Borehole	Methane Composition[a] (%)	Mean CH_4 / CO_2 Volumetric Ratio	Perimeter Gas Borehole	Methane Composition[a] (%)	Mean CH_4 / CO_2 Volumetric Ratio
BH047	0.0 – **0.0** -0.0	[b]	BH090	0.0 – **1.0** – 57.7	2.24
BH048	0.0 – **6.1** -80.4	4.81	BH091	0.0 – **1.1** – 67.3	1.35
BH070	0.0 – **0.001** -0.1	0.06	BH092	0.0 – **0.7** – 72.2	1.49
BH071	0.0 – **0.0** -0.0	[b]	BH093	0.0 – **0.7** – 71.1	1.31
BH073	0.0 – **0.2** -6.3	0.65	BH094	0.0 – **0.001** – 0.1	[b]
BH074	0.0 – **0.004** -0.1	1.00	BH095	0.0 – **0.6** – 68.2	1.52
BH074A	0.0 – **0.0** -0.0	[b]	BH096	0.0 – **0.6** – 58.8	1.30
BH075	0.0 – **0.0** -0.0	[b]	BH097	0.0 – **0.5** – 57.4	0.88
BH078	0.0 – **0.0** – 0.0	[b]	BH098	0.0 – **0.6** – 63.3	0.92
BH079	0.0 – **0.0** – 0.0	[b]	BH099	0.0 – **1.1** –63.1	1.43
BH080	0.0 – **0.1** – 12.0	0.43	BH100	0.0 – **0.1** – 15.8	2.19
BH081	0.0 – **0.0** – 0.0	[b]	BH101	0.0 – **2.2** – 66.5	2.54
BH082	0.0 – **0.0** – 0.0	[b]	BH102	0.0 – **2.2** – 64.5	1.78
BH089	0.0 – **18.4** – 80.2	16.63	BH103	0.0 – **3.2** – 52.9	1.62
			BH104	0.0 – **3.5** – 59.9	1.81

Notes:
a) Methane Composition Data shown as minimum - mean – maximum.
b) No concentrations of methane were found within the borehole.

Methane:Carbon Dioxide Ratios

Methane : carbon dioxide ratios recorded within the perimeter gas monitoring boreholes were reviewed, to determine whether there were any differences between the hydraulically and non-hydraulically contained phases. Tables 4.13 and 4.14 display the average methane:carbon dioxide ratios from within the individual perimeter gas monitoring boreholes surrounding Phases 1-3 and Phase 4 respectively.

There is a high degree of variability in methane to carbon dioxide ratios for both Phases 1-3 and Phase 4 and so further interpretation on the significance of these ratios to hydraulic containment has not been undertaken.

4.6 Site Studies - (3) Whitehead Landfill Site

4.6.1 Introduction

Information relating to the environmental setting and development of Whitehead landfill was presented in an earlier report in the study (Entec, 2003b) and summary information has been included in Chapter 2 of this report.

4.6.2 Landfill Gas Collection and Treatment

Gas Collection Infrastructure

Viridor Waste Management provided a copy of the landfill gas plan of the site (see Figure 4.9). The gas wells installed at Whitehead landfill site are constructed of 125 mm diameter slotted pipe, installed within a 300 mm diameter borehole, with non-calcareous stone and a bentonite seal. They are generally 20-30 m deep and sited at 50 m centres. About 6-10 of these wells are connected to a manifold, from which a 180 mm diameter pipe leads into a gas main, which is 315 mm diameter.

There are five knock-out pots spaced around the gas main, which will eventually become the ring main around the site. The knock-out pots contain air-operated positive displacement pumps, which discharge the condensate automatically into leachate well 1.

Power Generation

Table 4.15 sets out the specification of the landfill gas engines installed at Whitehead landfill site.

Table 4.15 Specification of Landfill Gas Engines Installed at Whitehead Landfill Site

Manufacturer	Specification
Caterpillar	2 x Caterpillar 3516 engines. Output - 1136 kW each (Note: the site output limit is 2 MW, as the local network is unable to cope with any more). Throughput - At 100% load and 50 % methane, the throughput of the engines is 600 m^3/hr each. Diameter of exhaust - 250 mm. Stack temperature - 560°C at the engine, not at the silencer.

Table 4.16 sets out the specification of the landfill gas flare installed at Whitehead landfill site.

Table 4.16 Specification of Landfill Gas Flare Installed at Whitehead Landfill Site

Manufacturer	Specification
Hoffstetter	Hoffstetter 500 m^3 flare Installed at Whitehead landfill site in July 2002 Maximum flow rate - 2,000 m^3/hr Minimum flow rate - 400 m^3/hr Stack height - 8.9 m Stack diameter - 2 m Stack temperature - 1,000°C Retention time - 0.3 seconds

4.6.3 Collection of Monitoring Data

Entec visited Whitehead landfill site on 14 October 2004, in order to obtain an understanding into how the site is operated, with particular regard to landfill gas. Some of the data that had been requested for use on the project was not available as it is not collected, nor required to be collected (see Table 4.1). As for the other sites, gas production data from the individual stages of the landfill site were unavailable, because appropriate monitoring facilities are not in place on the site to obtain the data.

Again as for the other sites, as landfill gas production information was not available, Entec attempted to review the effects of hydraulic containment on the composition of the landfill gas within the different stages of Whitehead landfill site.

Gas composition data were made available for in-waste wells AG501 (Cell 1A), AG502 (Cell 1B), AG307 (Cell 2A), and AG311 (Cell 3A) for a period starting as early as February 1999 and ending in August 2002. In addition, gas composition data were made available for the ‘snap-shot’ period August to October 2004 for a much wider range of wells.

4.6.4 Time Series In-Waste Gas Quality Data and Relation to Leachate Quality

Figures 4.10-4.13 show the change in monitored landfill gas component concentrations in the four wells sampled longer term together with leachate quality for comparison. These figures are discussed in the subsections below.

Stage 1 (Above Water Table Landfill)

Figures 4.10 and 4.11 show:

- The presence of methane and carbon dioxide at concentrations >1% from mid 1999 in Stage 1 (above water table with landfill commencing late 1998/early 1999) corresponding to high COD and lower pH values in the associated leachate. Carbon dioxide is generally present in excess of methane and low gas production is suggested by still high levels of oxygen not being purged yet from the waste. These gas and leachate quality data suggest the onset of stage 3 waste stabilisation (acetogenesis), as discussed in Sections 3.4.2 and 4.2.2, within 3-6 months of the start of landfilling.

- The presence of higher methane and carbon dioxide concentrations with methane in excess of carbon dioxide, and oxygen content starting to fall, suggesting displacement of air in the waste through increased gas generation, in Stage 1 from early 2000 in AG501 (Cell 1A) and from mid 2000 in AG502 (Cell 1B). In the leachate, COD and sulphate concentrations are reduced and there is an increase in pH. These gas and leachate quality data suggest the onset of stage 4 waste stabilisation (methanogenesis) following acetogensis (including sulphate reduction) as discussed in Sections 3.4.2 and 4.2.2 within 12 months of the start of landfilling. Methane and carbon dioxide concentrations increase significantly after capping.

Stage 2 (Hydraulically Contained)

At AG307 (Stage 2, Cell 2A), the data are more limited than at AG501 and AG502, but Figure 4.12 shows:

- Similar changes in gas composition can be seen with perhaps acetogenic conditions (carbon dioxide content greater than methane content and not so much gas production) between mid and late 2001, 3-9 months after the start of filling as suggested by the elevation of the well. Methanogenic conditions with more significant gas generation leading to the displacement of oxygen (air) occurs from the start of 2002.

- In comparison, leachate quality does not show the same changes as at AG501 and AG502. For example, there is little reduction in pH during the (gas composition implied) acetogenic stage - COD concentrations are also low (<2000 mg/l) compared to those measured at AG501 and AG502 during the acetogenic stage. The persistence and to a point ongoing rise in sulphate concentrations and a COD which still appears to be rising in mid 2004 are also inconsistent with the gas composition implied move into methanogenic conditions.

Stage 3 (Hydraulically Contained)

Data are most limited for AG311, but Figure 4.13 shows:

- An increase in methane and carbon dioxide and reduction in oxygen concentrations implying (based on data for the other sites) the move towards methanogenesis and the displacement of air from the waste. As for AG307, however, the pH is relatively stable and COD continues to rise after the methane generation starts.

Discussion

In the above water table Stage 1 (wells AG501 and AG502), gas composition and leachate quality datasets both support changes from acetogenic to methanogenic conditions. This is consistent with the mass of waste wetting up sufficiently for waste decomposition to begin and for excess moisture to drain to the base for leachate collection.

In the hydraulically contained Stages 2 and 3 (wells AG307 and AG311 respectively), the gas composition data suggest the onset of methanogenic conditions in the waste, but this is not supported by the leachate quality data, which seems to suggest acetogenic conditions are still progressing. The effect of recirculation on leachate quality in 2003 and 2004 has been discussed in Section 3.9.5, but this does not explain the absence of acetogenic leachate in AG307 (Cell 2A) during 2001. The data for AG311 pre- recirculation are too few to be useful for further analysis.

The 2001 and 2002 data for AG307 show an absence of high COD, BOD and TOC concentrations and low pH values that would be indicative of acetogenic conditions. Options to explain this are a variation in the nature of waste (less putrescibles) or liquid wastes in this area (Cell 2A) or significant dilution by rainfall bypassing the waste or groundwater ingress. Chloride and ammoniacal nitrogen concentrations are however not dissimilar to leachate concentrations in Cells 1A and B where acetogenic conditions occurred. Significant dilution therefore appears unlikely.

4.6.5 In-Waste Gas Composition Variability Across the Site in 2004

As noted in Section 4.6.3, gas composition data were made available for the 'snap-shot' period August to October 2004 for a much wider range of wells.

As for Brogborough (see Figure 4.4a), the average methane concentration in each well appears to be related predominantly to the variability (standard deviation) in the dataset (see Figure 4.14). Methane concentrations in Stage 1 do, however, appear to be lower than those in Stages 2, 3 and 4. Also, as for Brogborough, a significant amount of the variability is related to the oxygen content, i.e. air entrainment in the gas wells or gas extraction system (see Figure 4.15a).

For a given concentration of methane, carbon dioxide concentrations are higher in Stage 1 and perhaps Stage 2C, both above water table when compared to the other hydraulically contained stages (Figure 4.15b). It is not clear why this should be.

4.6.6 Perimeter Gas Monitoring Boreholes

Table 4.17 summarises the landfill gas composition data recorded in the off-site perimeter boreholes around Whitehead landfill site.

Table 4.17 Summary of Landfill Gas Composition Data in the Perimeter Gas Boreholes

Landfill Area	Monitoring Points	Comments
Stage 1	Perimeter Gas Borehole AG014	Elevated levels of methane (above 1%) have only been recorded 4 times since 2000, but at up to 15.9%. Elevated levels of carbon dioxide (above 1.5%) have been recorded on 43 occasions, since 2000. The highest concentrations of methane and carbon dioxide were recorded on 4 October 2004 (16% and 26% respectively).
Stage 2	Perimeter Gas Boreholes AG005, AG006, AG007, AG008 & AG009	Since 1997, no elevated concentrations of methane have been recorded in the perimeter gas boreholes within Stage 2. During the same time period, there have been 16 records of elevated carbon dioxide concentrations, with a maximum of 8% recorded in 1998.
Stage 3	Perimeter Gas Boreholes No off-site boreholes exist around Stage 3	No off-site boreholes exist around Stage 3.
Stage 4	Perimeter Gas Boreholes AG010, AG011, AG012 & AG013	Since 1997, no elevated concentrations of methane have been recorded in the perimeter gas boreholes within Stage 4, although 1% methane was recorded on 22 July 1999 at AG012. During the same time period, there have been 50 records of elevated carbon dioxide concentrations, with a maximum of 3% recorded in March 2004.

The table indicates that there has been only one occasion when elevated (>1%) levels of methane have been recorded in the perimeter gas boreholes surrounding the hydraulically contained stages of the site (Stages 2, 3 and 4), whereas the perimeter gas borehole located near to the non-hydraulically contained Stage 1, has recorded some higher levels of methane on more occasions.

4.7 Summary and Discussion of Results

4.7.1 Gas Generation

Gas generation data were not available for different areas within the three sites and so an analysis of whether hydraulic containment of parts of these sites lead to more or less gas generation has not been possible.

Instead, Table 4.18 compares total gas generation at the different sites, to see if there are obvious differences between Brogborough, for which there appears to be some evidence of groundwater ingress into the waste compared to Poole and Whitehead, where groundwater ingress appears to be collected by the basal drainage blanket and so does is unlikely to affect the waste’s moisture content.

Table 4.18 Comparison of Gas Generation at the Three Studied Sites

	Brogborough[a]	Poole	Whitehead
Start of Landfilling	1983	1974	1998
Filled Area (circa 2004) (ha)	120	~11	~15
Filled Void (Mm^3)	23	~2.2	~2.5
Average Depth of Waste (m)	19.1	20	16.7
Waste Type	Non-hazardous household, commercial and industrial	Non-hazardous household, commercial and industrial	Non-hazardous household, commercial and industrial
Liquids	Low	None	High
Recirculation	Minor	None	High
Estimated Liquid Solid Ratio Range at Monitoring Wells (excluding groundwater inputs)	0.05 – 0.36 (Average of wells in Figure 4.6 = 0.15)	0.28 – 0.65 (from Figure 4.8)	0.18[b]
Power Output (MW)	20	0	2.27
Estimated Gas Flow Rate to Produce Power output (m^3/hr)	10325	0	1173
Flares (m^3/hr)	7500	1500 + ~325	400 - 2000
Total Gas Flow Rate (m^3/hr)	17825	~1825	1573 - 3173
Total Gas Flow Rate per m^3 of waste ($m^3/hr/m^3$)	7.8×10^{-4}	8.3×10^{-4}	6.3×10^{-4} – 12.7×10^{-4}

Notes:

a) Data shown exclude the extension area (Cell 5 and 6) as these areas were not completed for gas collection and utilisation at the time of reporting.

b) Calculated from total (1999-2004) water input (432 701 $m^{3)}$ divided by total waste input (2 449 784 m^3) in Table 2.5 and assuming a waste density of 1 tonne/m^3.

There are a number of uncertainties which affect the estimate of gas generation per m^3 of waste and this makes detailed comparison inappropriate. It would appear however that the three sites have broadly similar gas generation (and collection) rates.

4.7.2 In-Waste Gas Composition

Assessment of gas composition data at the three sites together with estimates of liquid solid ratios and leachate temperature and quality suggest the following:

At Brogborough, it appears that the waste which is in the sub-water table part of the landfill site may be generating gas with a higher ratio of CH_4(%) to CO_2(%) than in the above the water table parts of the site, due to additional water inputs from groundwater ingress. The same relationship is seen at Whitehead landfill with the sub-water table cells having higher methane to carbon dioxide ratios. There is, however, little evidence of this at Poole, although the dataset is very limited.

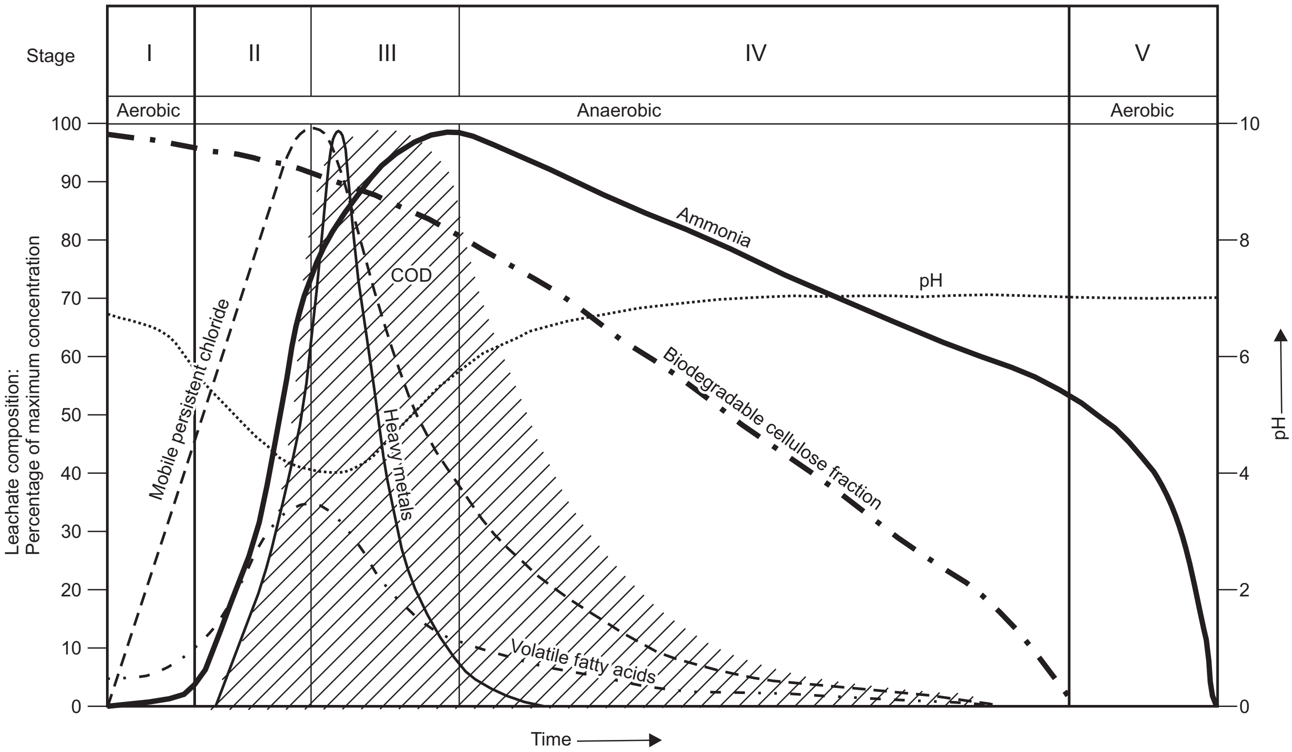
CH_4(%)/CO_2(%) ratios appear to increase with decreasing temperature, and temperatures tend to be higher in the deeper, more insulated and more sub-water table parts of Brogborough. This means that temperature does not explain the higher CH_4(%) / CO_2(%) ratios in the sub-water table parts of Brogborough (and Whitehead). Groundwater inputs with significant alkalinity would increase the solubility of carbon dioxide in the resulting leachate.

So although this study provides only a limited view of the affect of sub-water table landfill on landfill gas composition, it appears that groundwater ingress could lead to higher CH_4(%)/CO_2(%) ratios in the gas produced. Given that there is no clear evidence of gas generation rates being affected, then it is not implausible to conclude that **groundwater ingress into the waste in sub-water table, hydraulically contained landfills can produce similar amounts of gas, but with a higher methane content, than above water table landfills**.

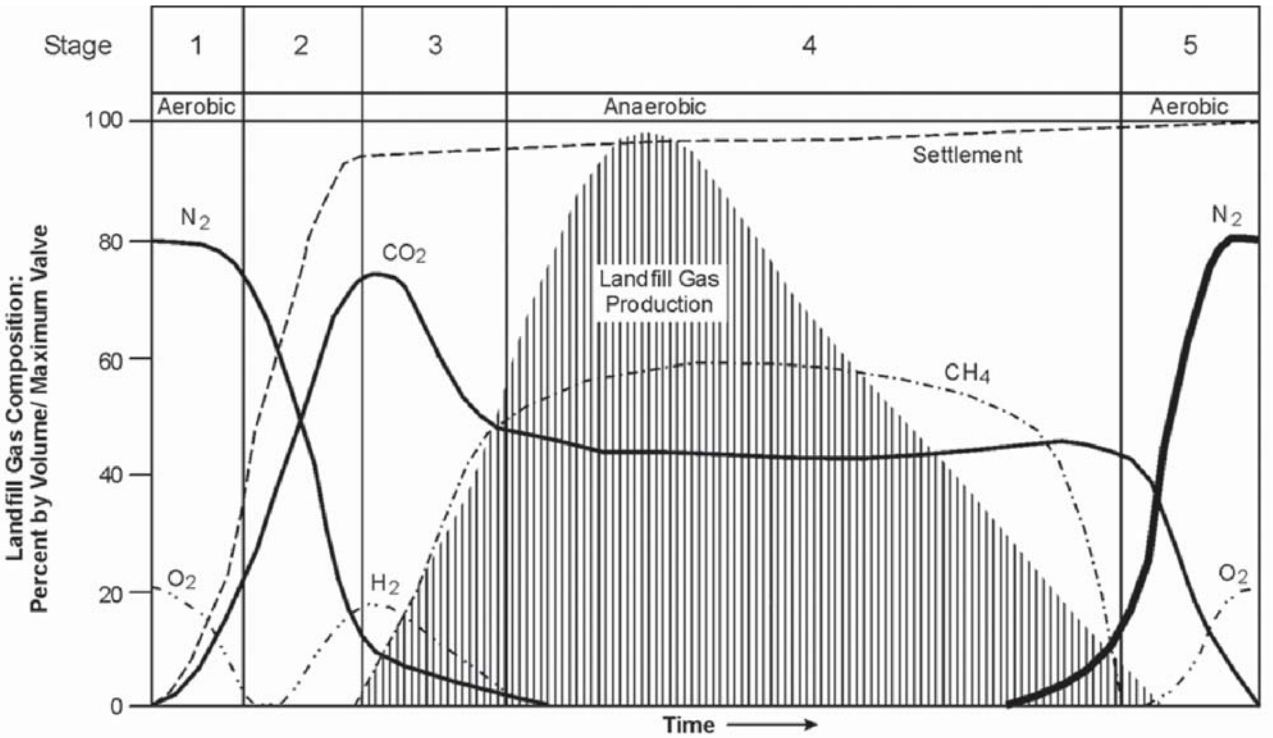
4.7.3 Perimeter Gas Migration

At Whitehead Landfill there is a suggestion within the data that gas migration is greater around the above water table cells than in the sub-water table cells, but this difference could also be related to modification or improvements in side wall engineering and variations in gas generation. At Brogborough and Poole, perimeter migration appears to be related to the extent of side wall engineering rather than being linked to sub-water table conditions. **There is however no evidence to suggest sub-water table conditions are leading to poor gas containment**.

Changes in Composition of Leachate (from Waste Management Paper 26B, DOE, 1995)



Idealised Representation of Landfill Gas Generation (Source: Environment Agency (2004))

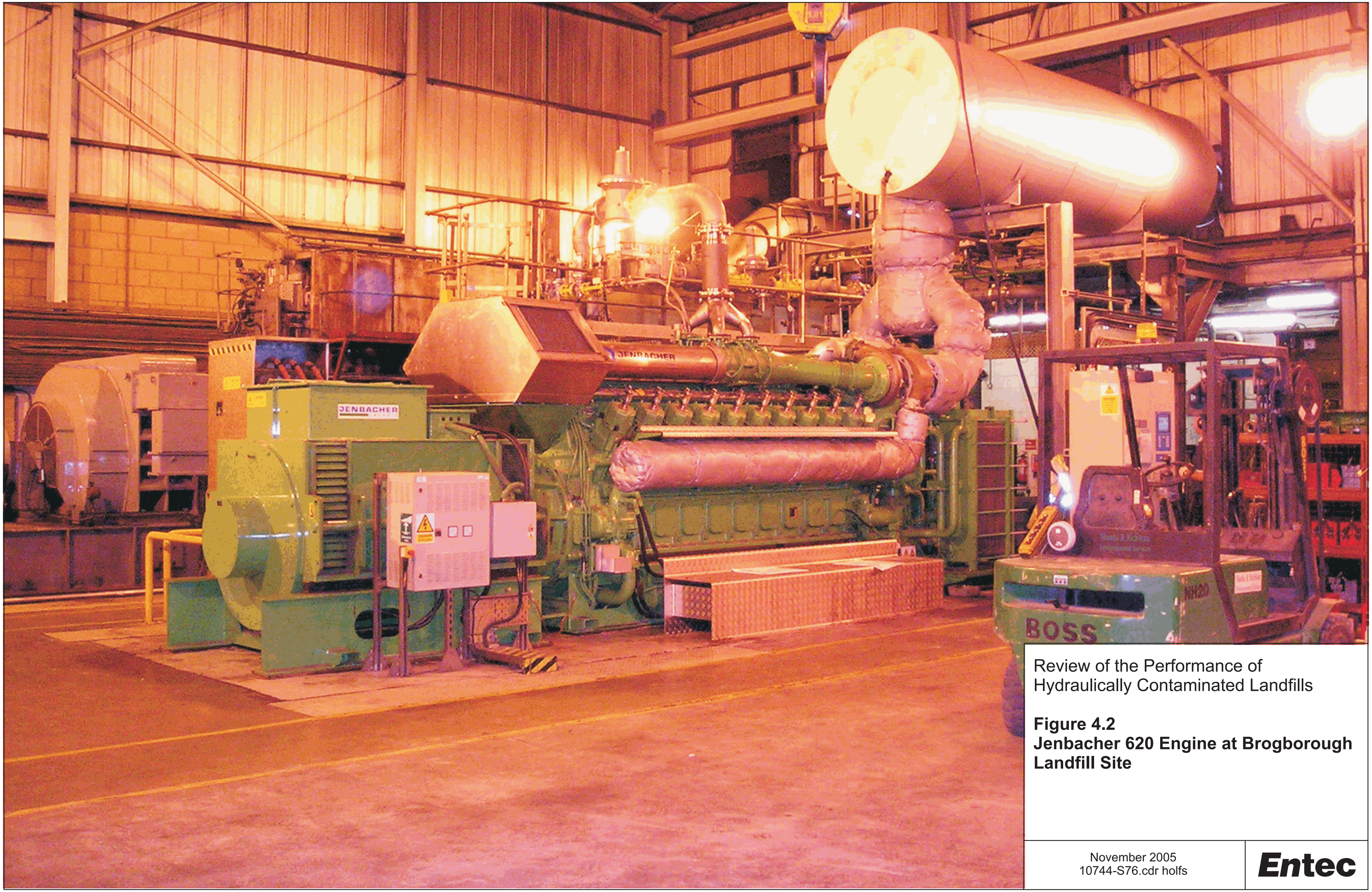


Review of the Performance of Hydraulically Contained Landfills

Figure 4.1
Changes in Composition of Leachate and Idealised Representation of Landfill Gas Generation

November 2005
10744-S68.cdr holfs

Entec

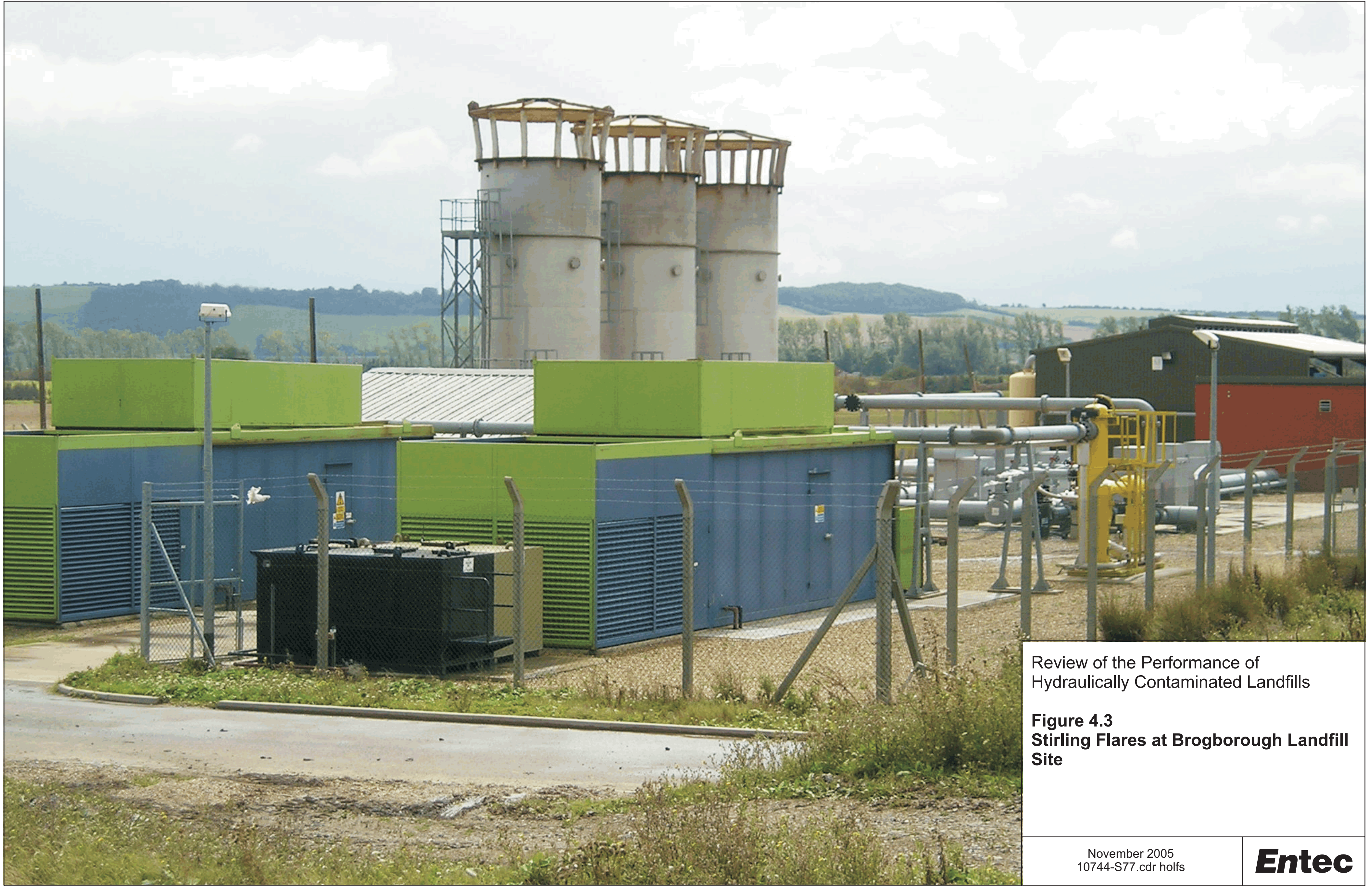


Review of the Performance of
Hydraulically Contaminated Landfills

Figure 4.2
Jenbacher 620 Engine at Brogborough Landfill Site

November 2005
10744-S76.cdr holfs

Entec



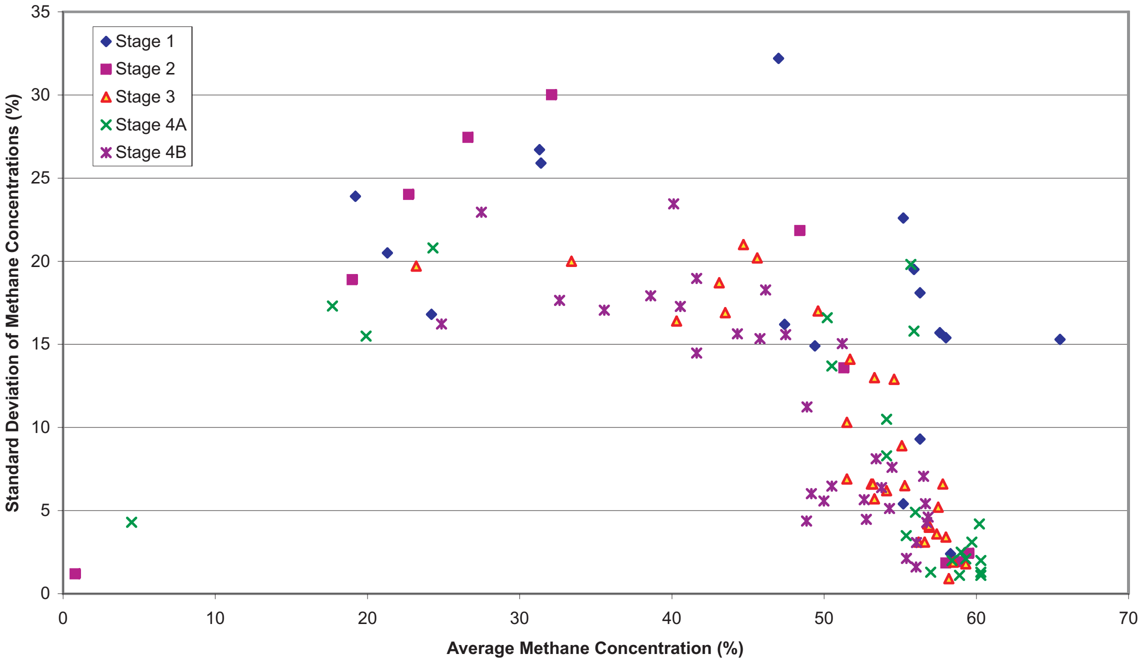
Review of the Performance of Hydraulically Contaminated Landfills

Figure 4.3
Stirling Flares at Brogborough Landfill Site

November 2005
10744-S77.cdr holfs

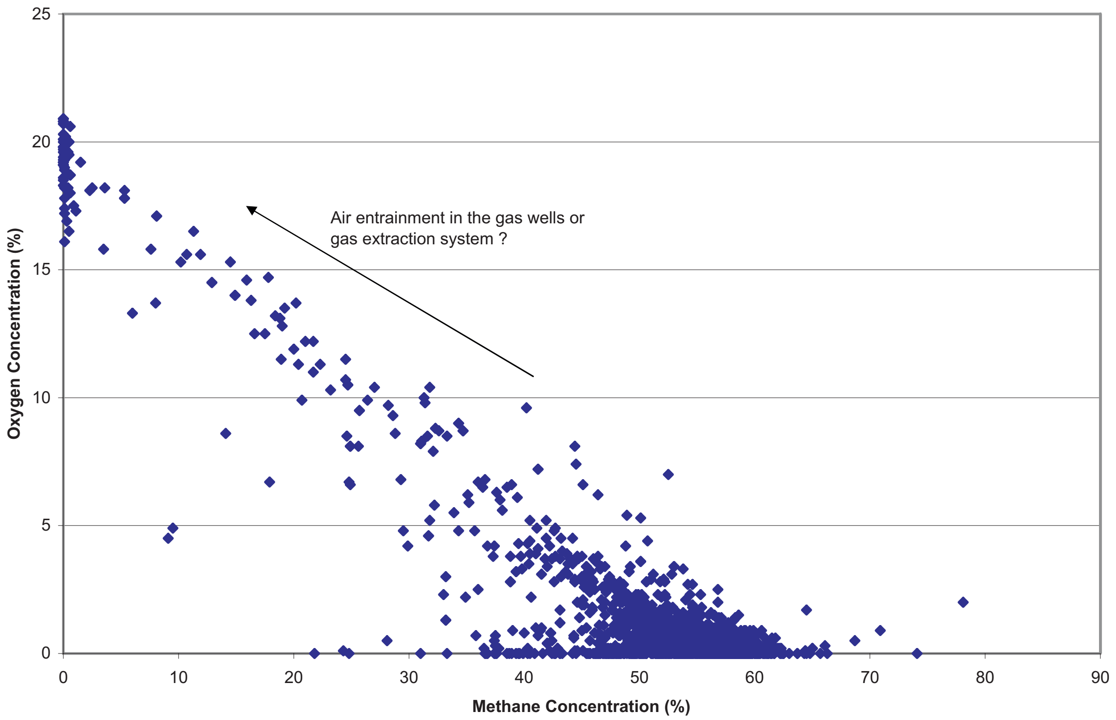
Entec

Figure 4.4a: Methane Concentration Variability



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Data\Gas Boreholes\CH4variability.xls-Chart1

Figure 4.4b: Relationship between Methane and Oxygen Concentrations



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Calcs\
temp & quality v thickness of waste.xls-O2vCH4Fig 4.4b

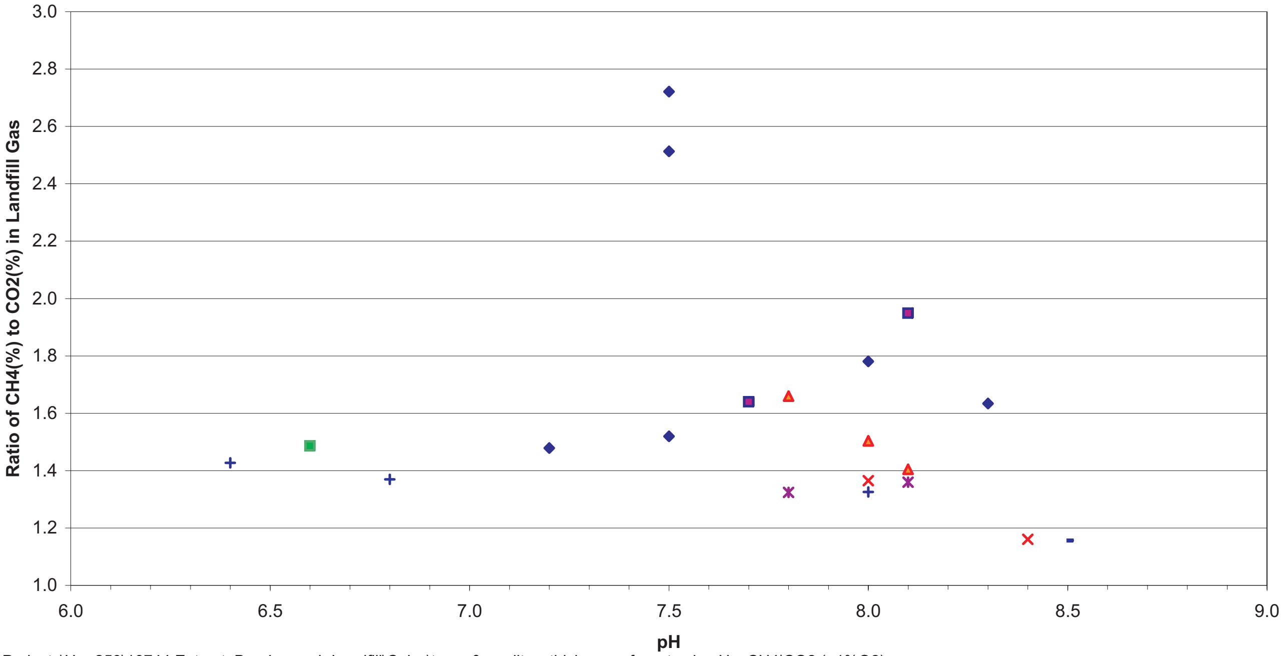
Review of the Performance of Hydraulically Contained Landfills

Figure 4.4
Methane Concentration Variability and Relationship Between Methane and Oxygen Concentrations

November 2005
10744-S69.cdr wrigs02

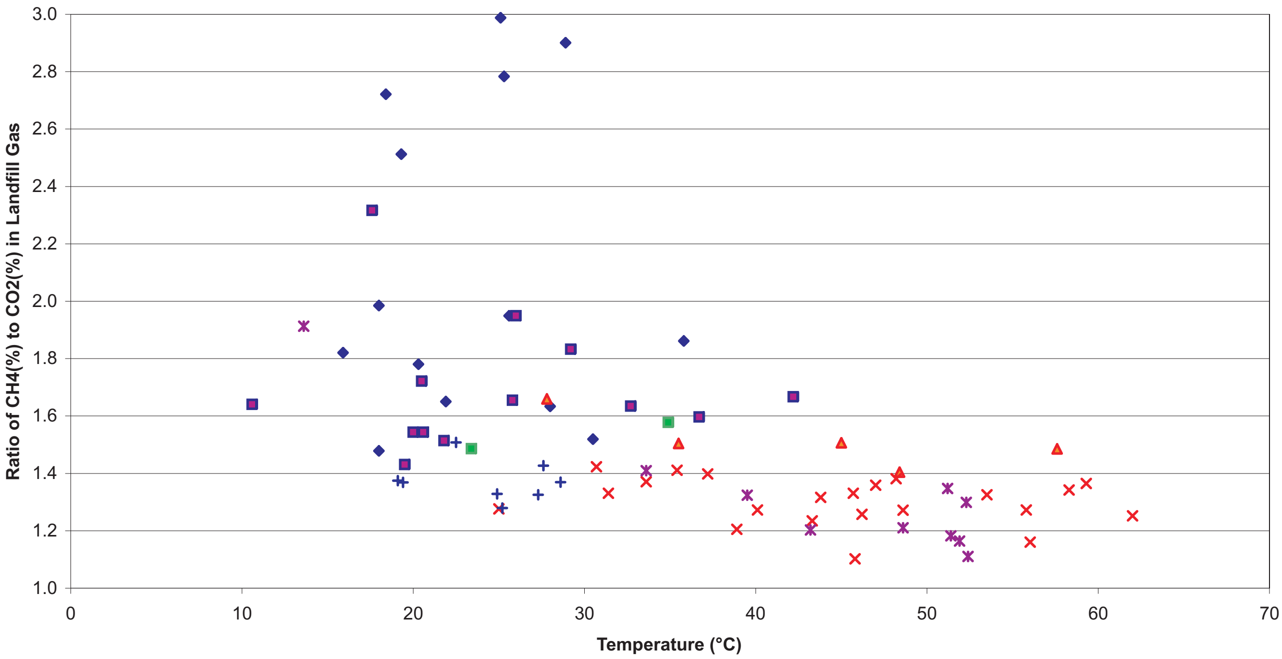
Entec

Figure 4.5a: (CH_4/CO_2) Ratio Against Leachate pH



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Calcs\temp & quality v thickness of waste.xls-pH v CH4|CO2 (<1%O2)

Figure 4.5b: (CH_4/CO_2) Ratio Against Leachate Temperature



H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Calcs\temp & quality v thickness of waste.xls-T°C v CH4|CO2 (<1%O2)

Key

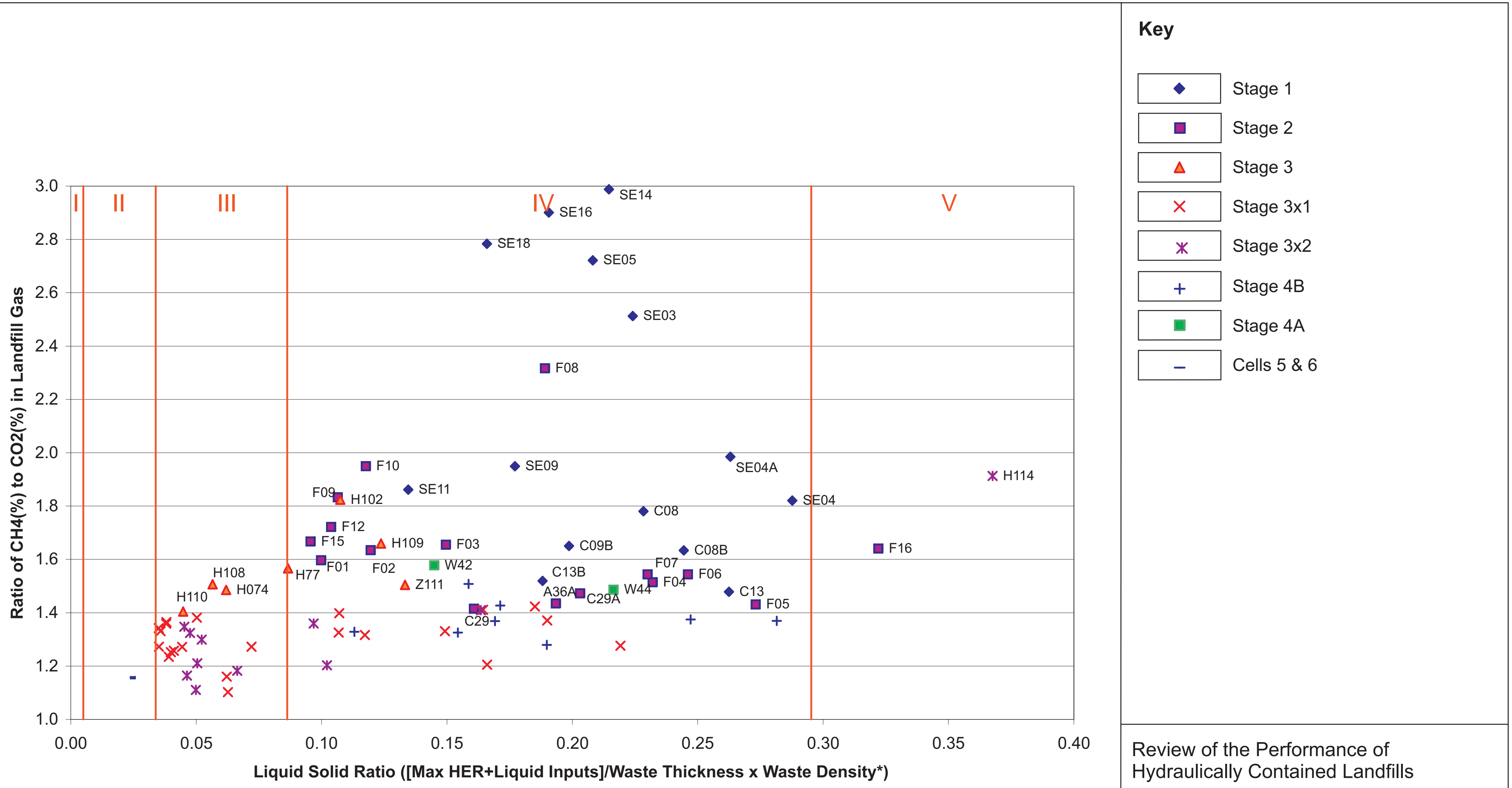
- ◆ Stage 1
- ■ Stage 2
- ▲ Stage 3
- × Stage 3x1
- ✱ Stage 3x2
- + Stage 4B
- ■ Stage 4A
- – Cells 5 & 6

Review of the Performance of Hydraulically Contained Landfills

Figure 4.5
(CH_4/CO_2) Ratios Against Leachate pH and Leachate Temperature at Brogborough Landfill

November 2005
10744-S70.cdr wrigs02

Entec



(Note* Density assumed as 1000kg/m^3)

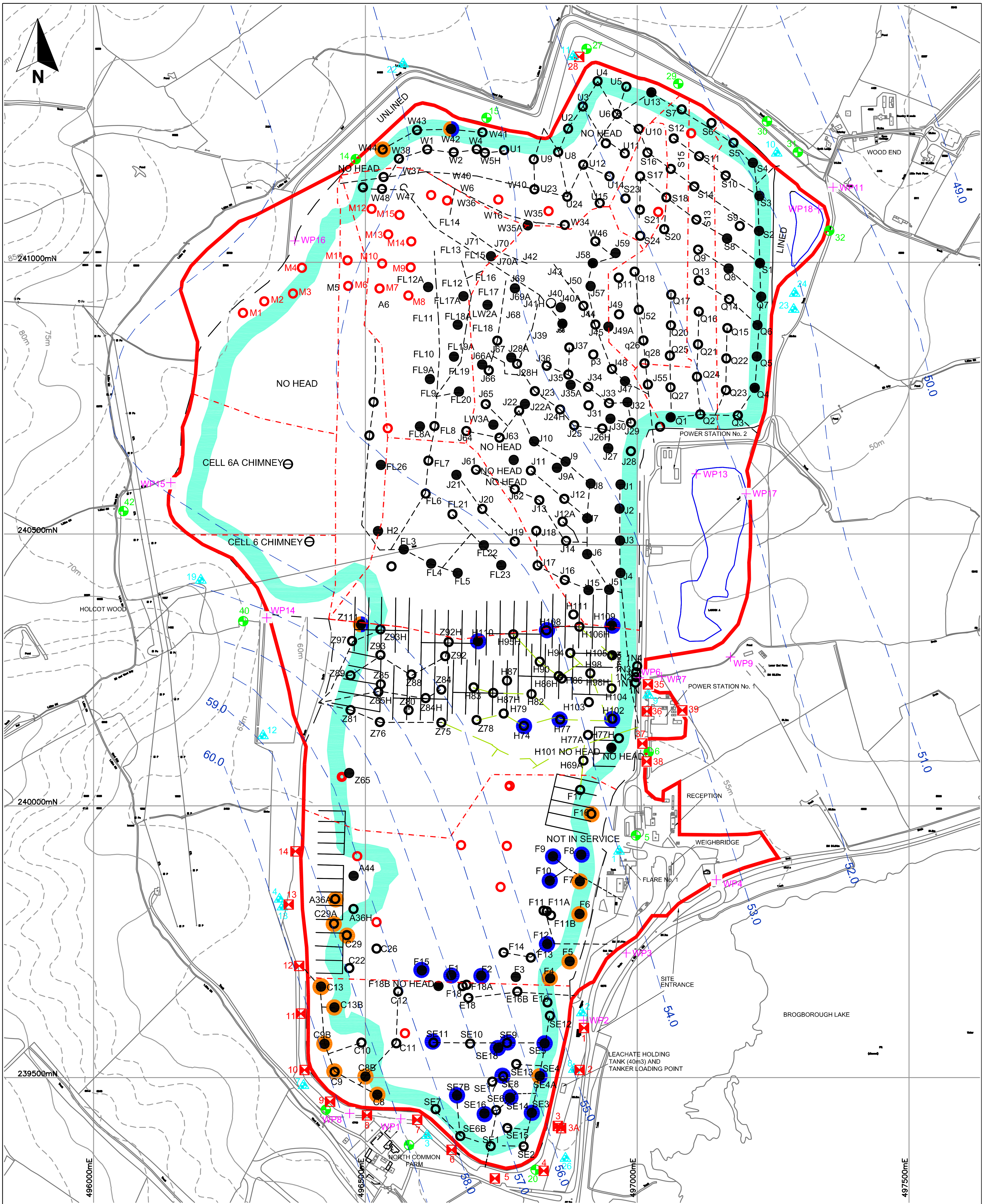
Review of the Performance of Hydraulically Contained Landfills

Figure 4.6
Ratio of CH_4(%) to CO_2(%) in Landfill Gas v Liquid : Solid Ratio

November 2005
10744-S71.cdr xgeort

Entec

H:\Projects\Hm-250\10744 Entrust_Brogborough Landfill\Calcs\temp & quality v thickness of waste.xls-lambda v CH4|CO2 (<1%O2)



Key

Licence boundary	Gas probe
Cell boundary	In-waste gas well
Groundwater borehole	Leachate sampling well
Pit base beneath piezometric level in the Kellaways Sand	Wells where CH_4/CO_2 ratios are low compared to liquid/solid ratios
Leachate chimney	Wells where CH_4/CO_2 ratios are high compared to liquid/solid ratios (ie needs additional water inputs from cap infiltration)
Perimeter gas monitoring borehole	Surface water monitoring point
Indicative gas well	Gas collection line
Horizontal gas well	Piezometric levels in Kellaways Sand (July 2003)

0 m — 250 m

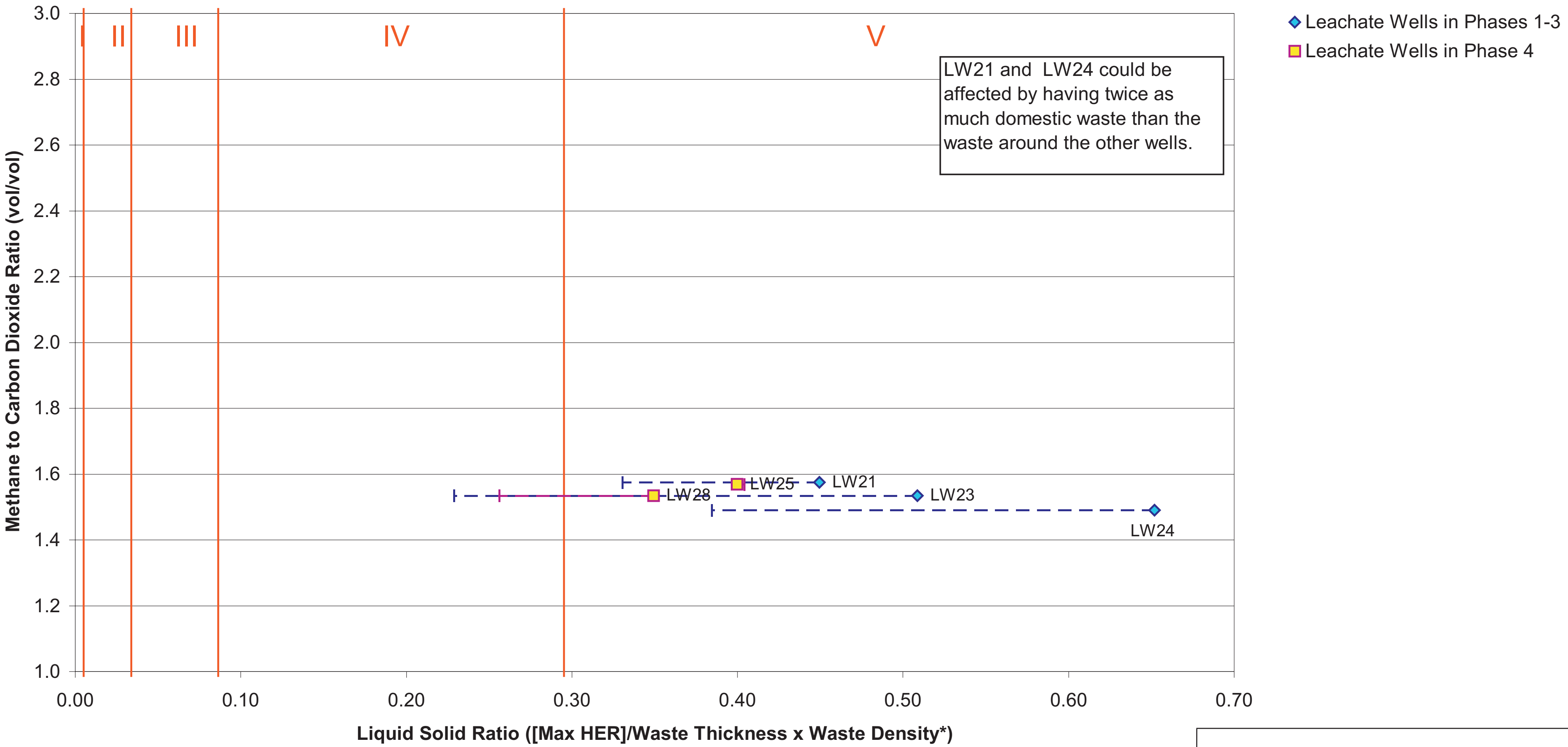
Scale 1:6500 @ A3

Review of the Performance of Hydraulically Contained Landfills

Figure 4.7
Location of Wells with High or Low CH_4/CO_2 Ratios Compared to Liquid:Solid Ratios

November 2005
10744-S72.dwg wrigs02

Entec



(Note* Density assumed as $1000 kg/m^3$)

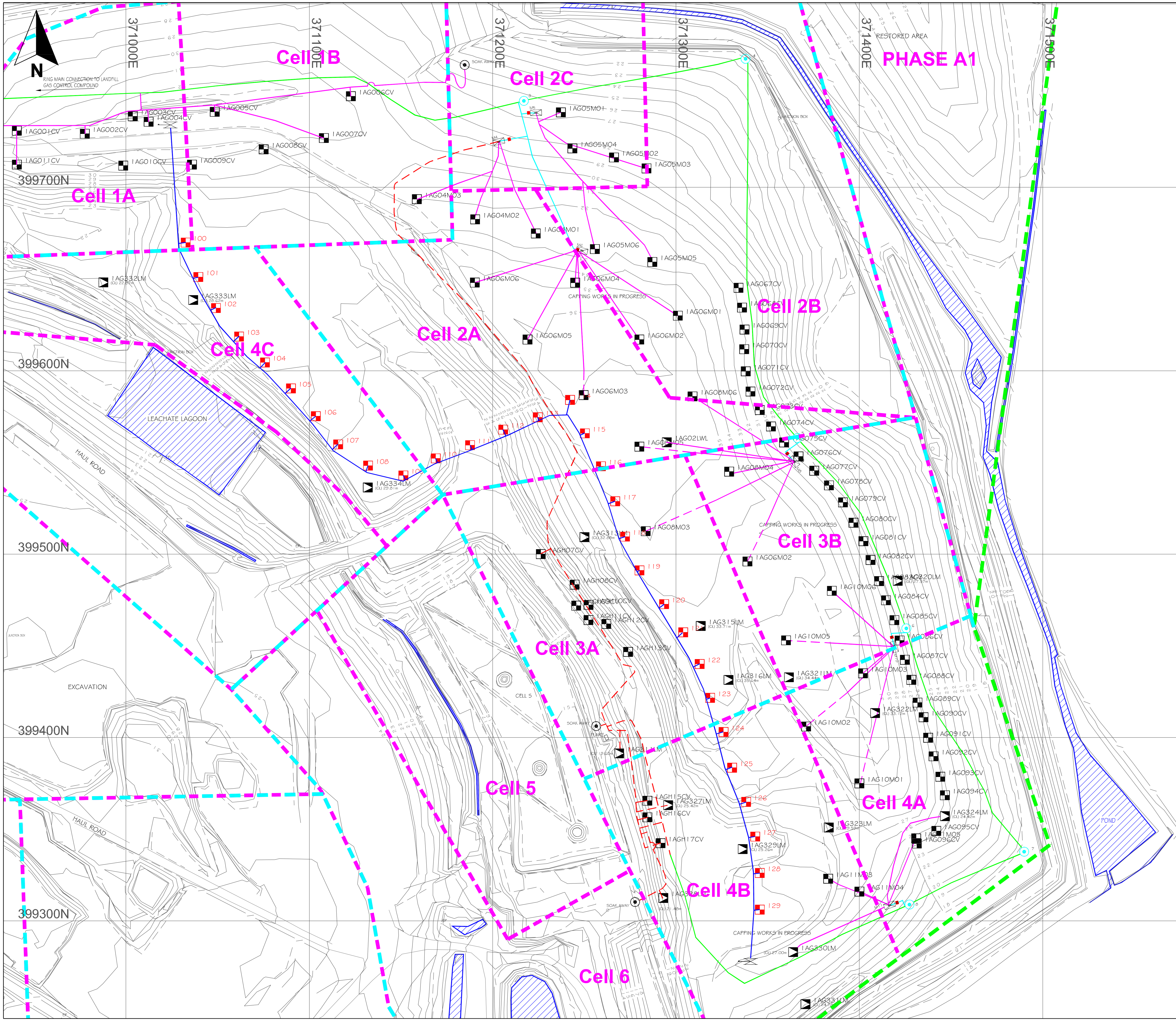
Review of the Performance of Hydraulically Contained Landfills

Figure 4.8
CH4/CO2 Ratios against Liquid Solid Ratio at Poole Landfill

November 2005
10744-S78.cdr holfs

Entec

H:\Projects\Hm-250\08465 Wyvern Ph2 HydCon\Data\quality.xls\lambda v CH4



Key

- Licensed area
- Cell boundary
- Stage boundary
- Gas control well
- Approximate position of new well
- Leachate monitoring
- Manifold
- Soakaway
- Approximate position of new soakaway
- Knockout pot
- Flow monitoring box
- 315mm pipe
- 315mm pipe
- 250mm pipe
- 180mm pipe
- 125mm pipe
- 110mm pipe
- 90mm pipe
- 63mm pipe
- Temporary pipe connection to permanent landfill gas extraction system (size as shown)

Notes:
1. All pipe sizes are O.D.
2. Positions of all pipework and new wells are indicative only.

0 m 100 m

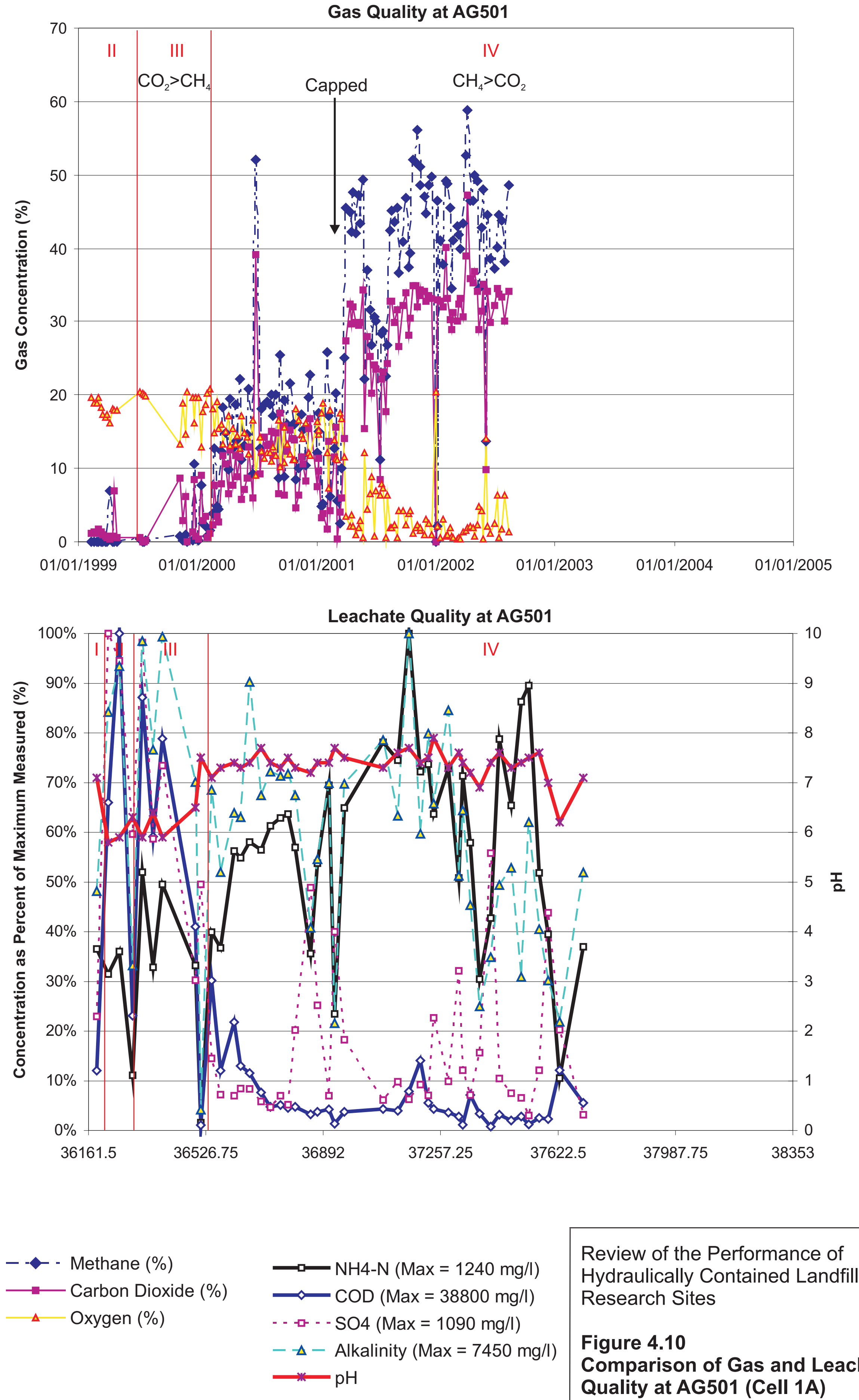
Scale 1:2000 @ A3

Review of the Performance of Hydraulically Contained Landfills

Figure 4.9
Landfill Gas Plan

November 2005
10744-S79.dwg holfs

Entec



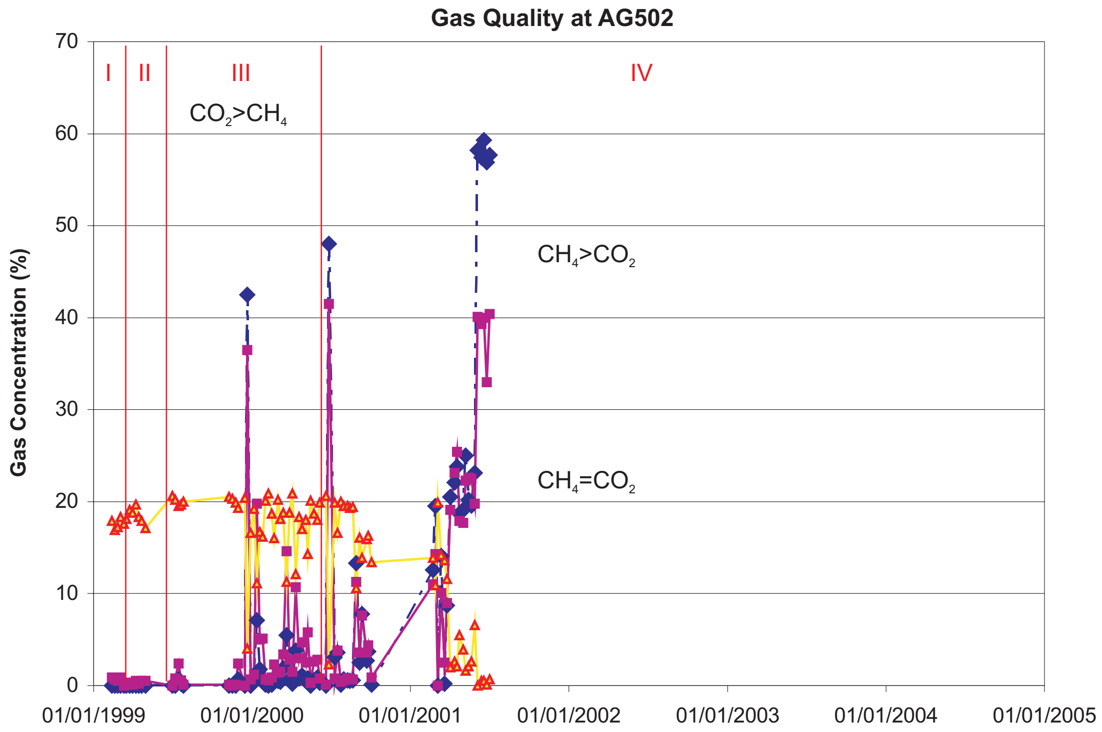
Review of the Performance of Hydraulically Contained Landfills: Research Sites

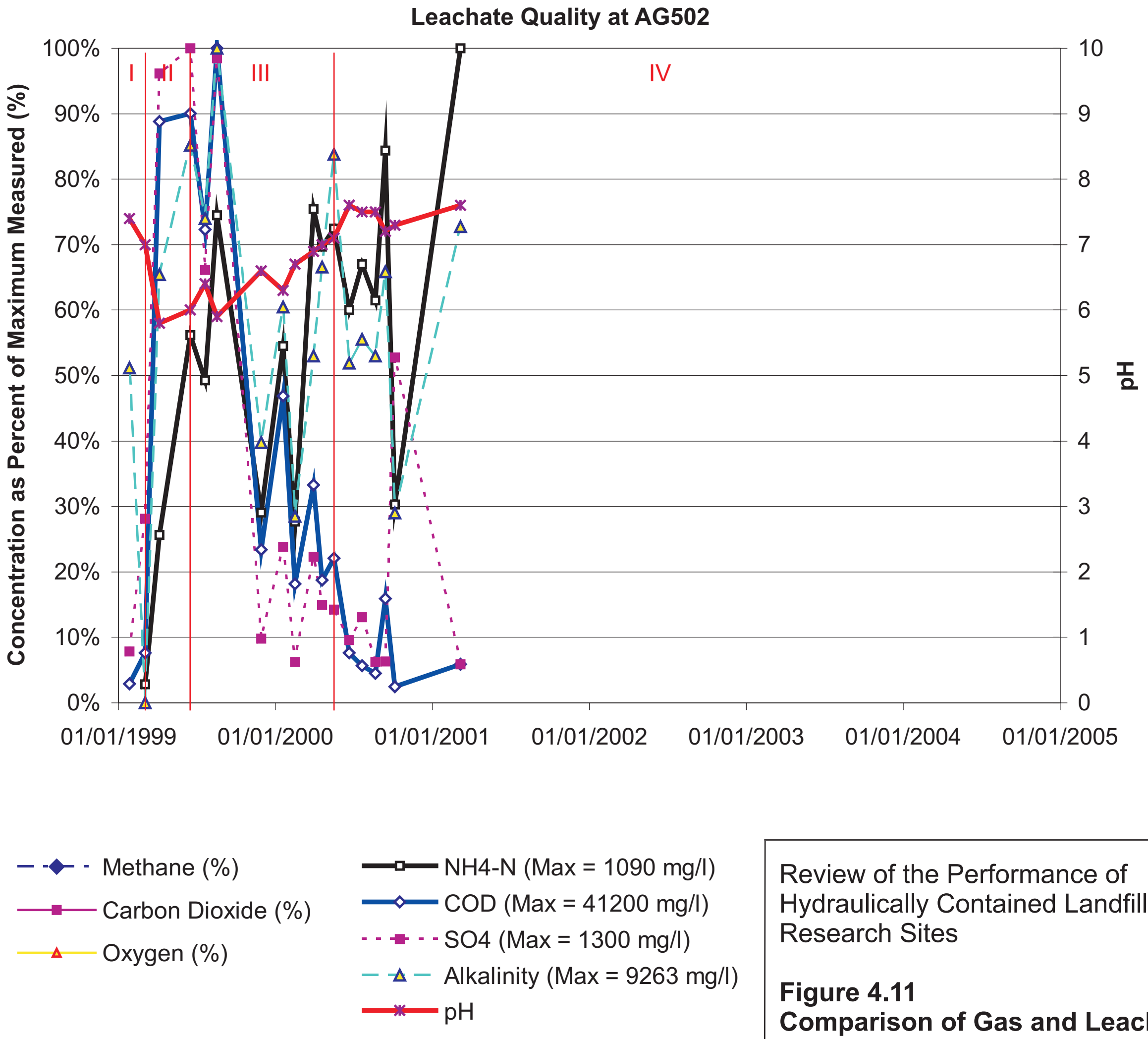
Figure 4.10
Comparison of Gas and Leachate Quality at AG501 (Cell 1A)

November 2005
10744-S80.cdr holfs

Entec

H:\Projects\Hm-250\08975 Viridor ph2 hyd\Data\Gas Quality Data\
Gas Composition Data to 181004.xls\AG501cht & AG501cht_lch





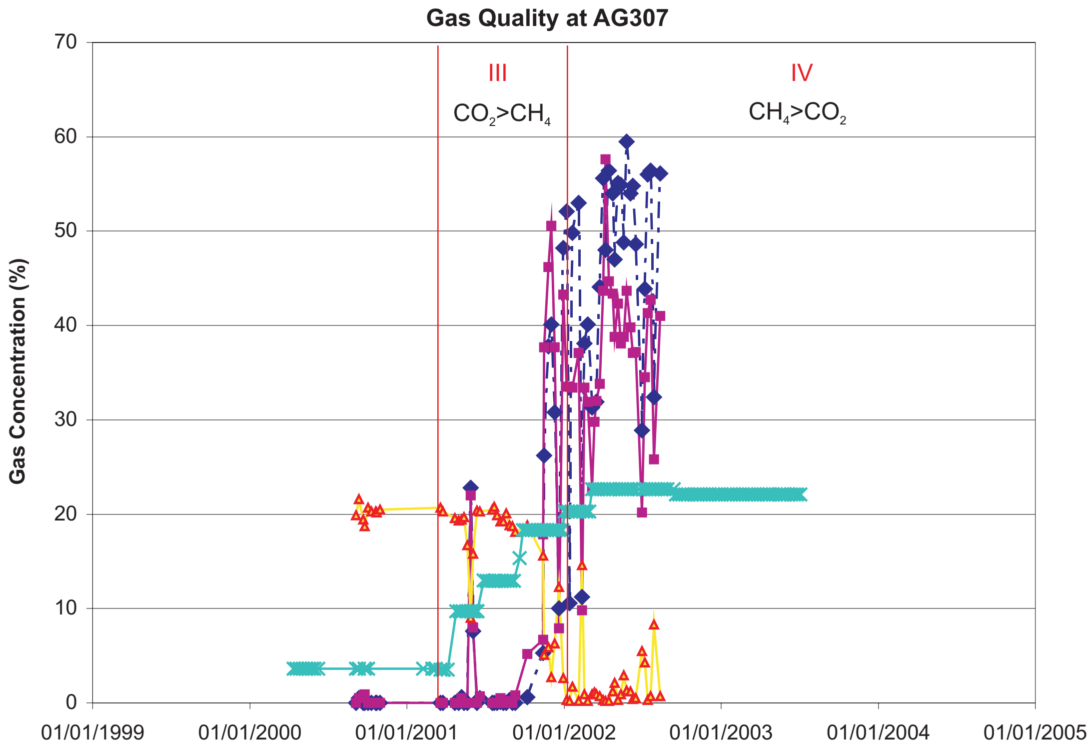
Review of the Performance of Hydraulically Contained Landfills: Research Sites

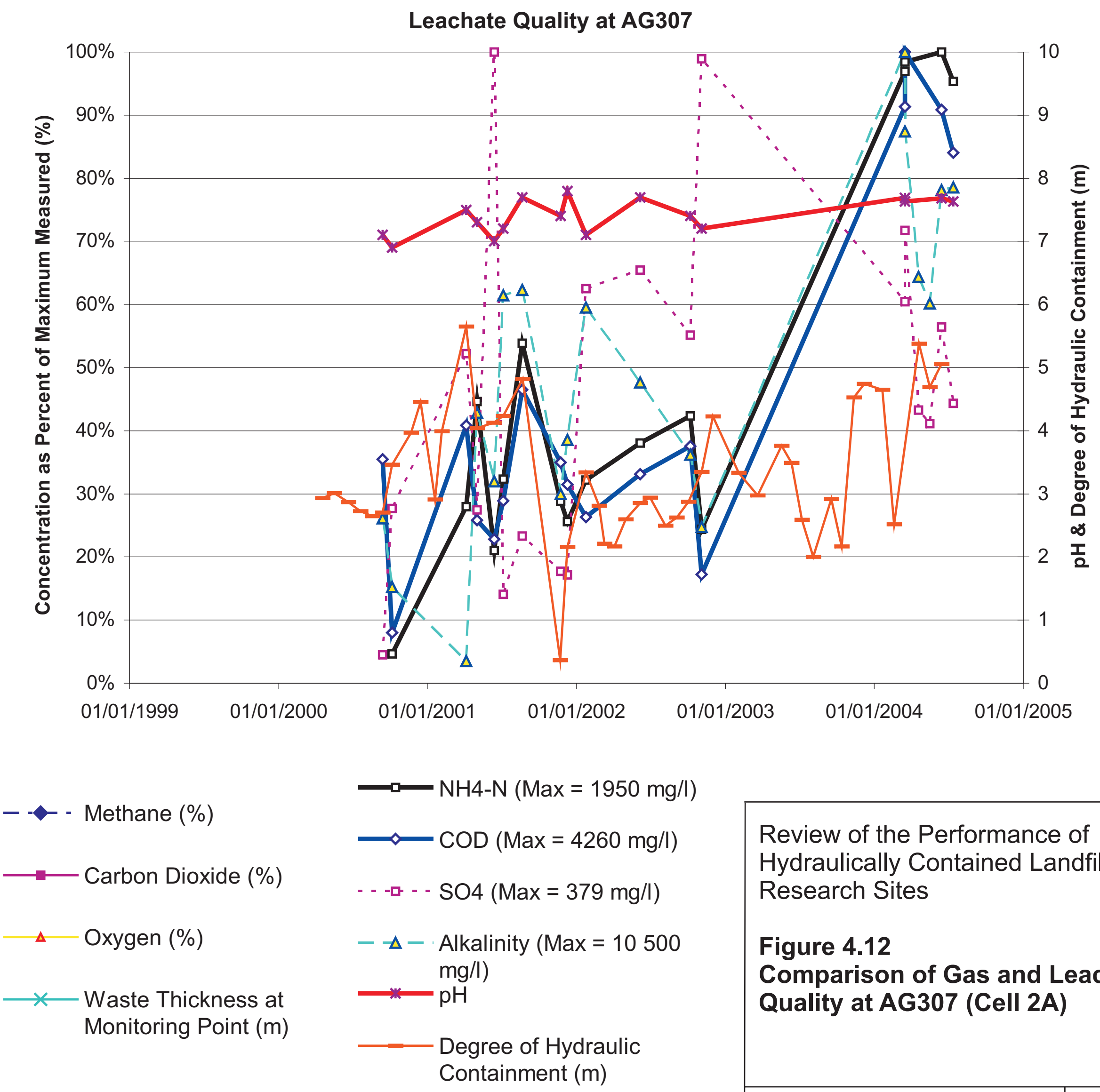
Figure 4.11
Comparison of Gas and Leachate Quality at AG502 (Cell 1B)

November 2005
10744-S81.cdr holfs

Entec

H:\Projects\Hm-250\08975 Viridor ph2 hyd\Data\Gas Quality Data\
Gas Composition Data to 181004.xls\AG502cht & AG502cht_lch





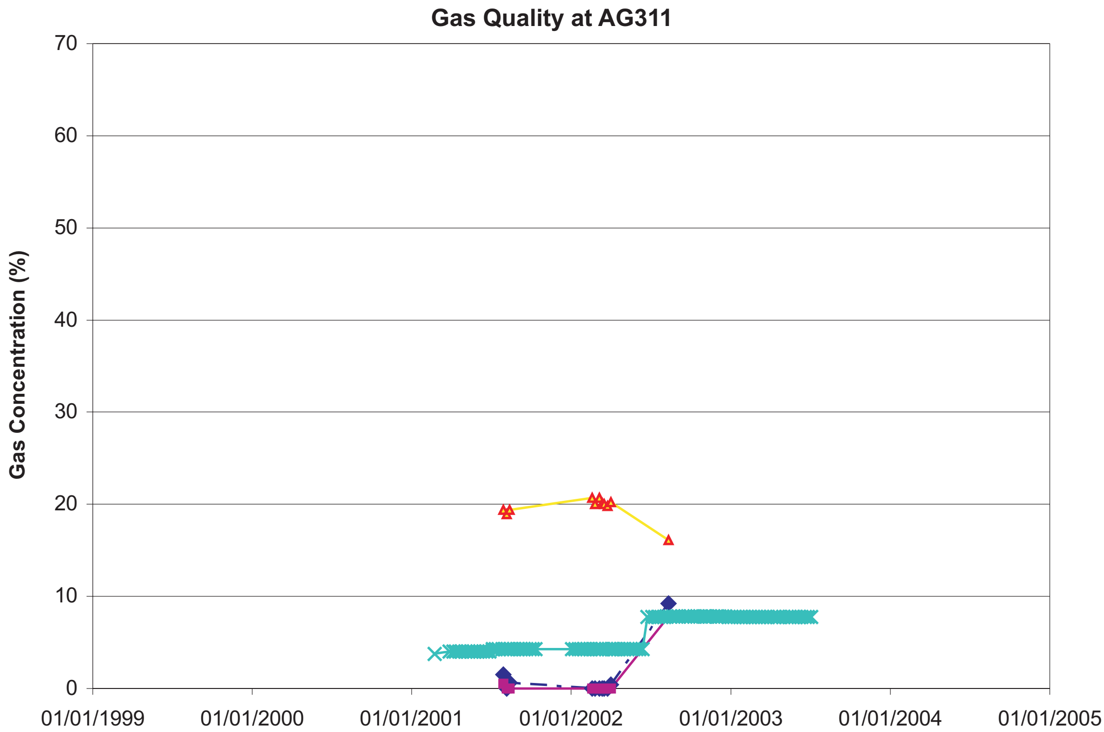
Review of the Performance of Hydraulically Contained Landfills: Research Sites

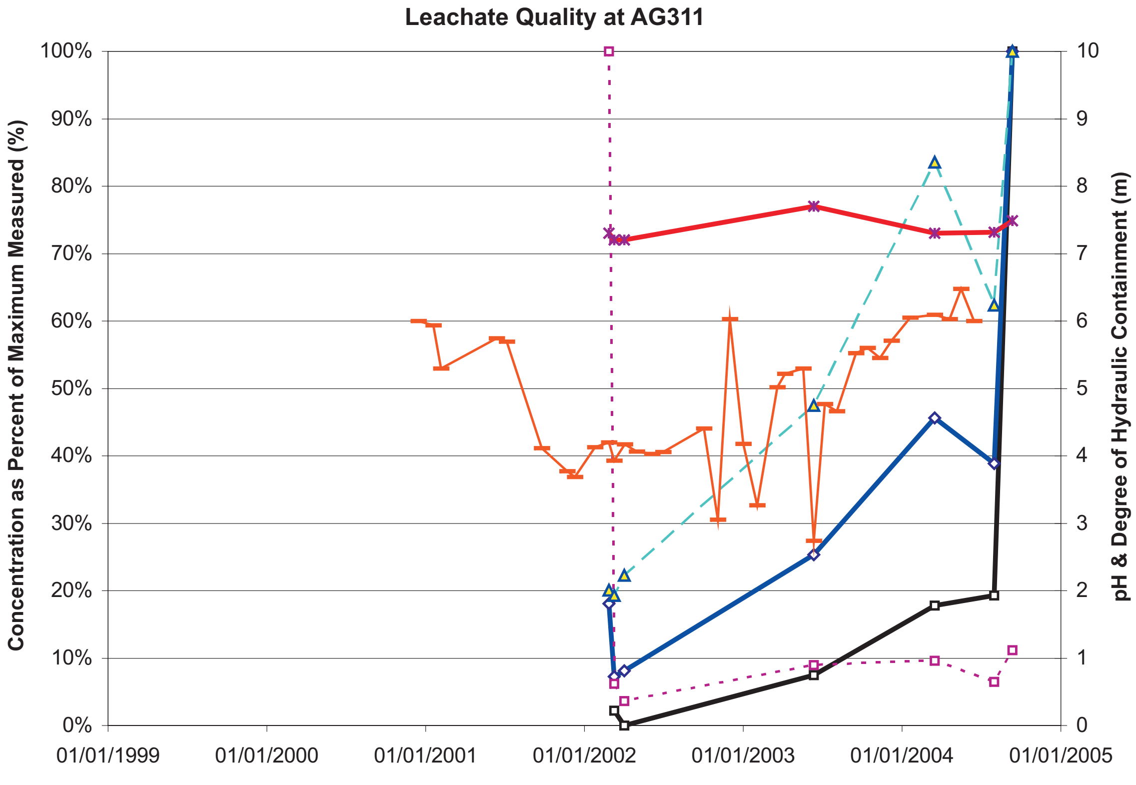
Figure 4.12
Comparison of Gas and Leachate Quality at AG307 (Cell 2A)

November 2005
10744-S82.cdr holfs

Entec

H:\Projects\Hm-250\08975 Viridor ph2 hyd\Data\Gas Quality Data\
Gas Composition Data to 181004.xls\AG307cht & AG307cht_lch





Methane (%)
Carbon Dioxide (%)
Oxygen (%)
Waste Thickness at Monitoring Point (m)

NH4-N (Max = 8880 mg/l)
COD (Max = 7850 mg/l)
SO4 (Max = 2950 mg/l)
Alkalinity (Max = 9860 mg/l)
pH
Degree of Hydraulic Containment (m)

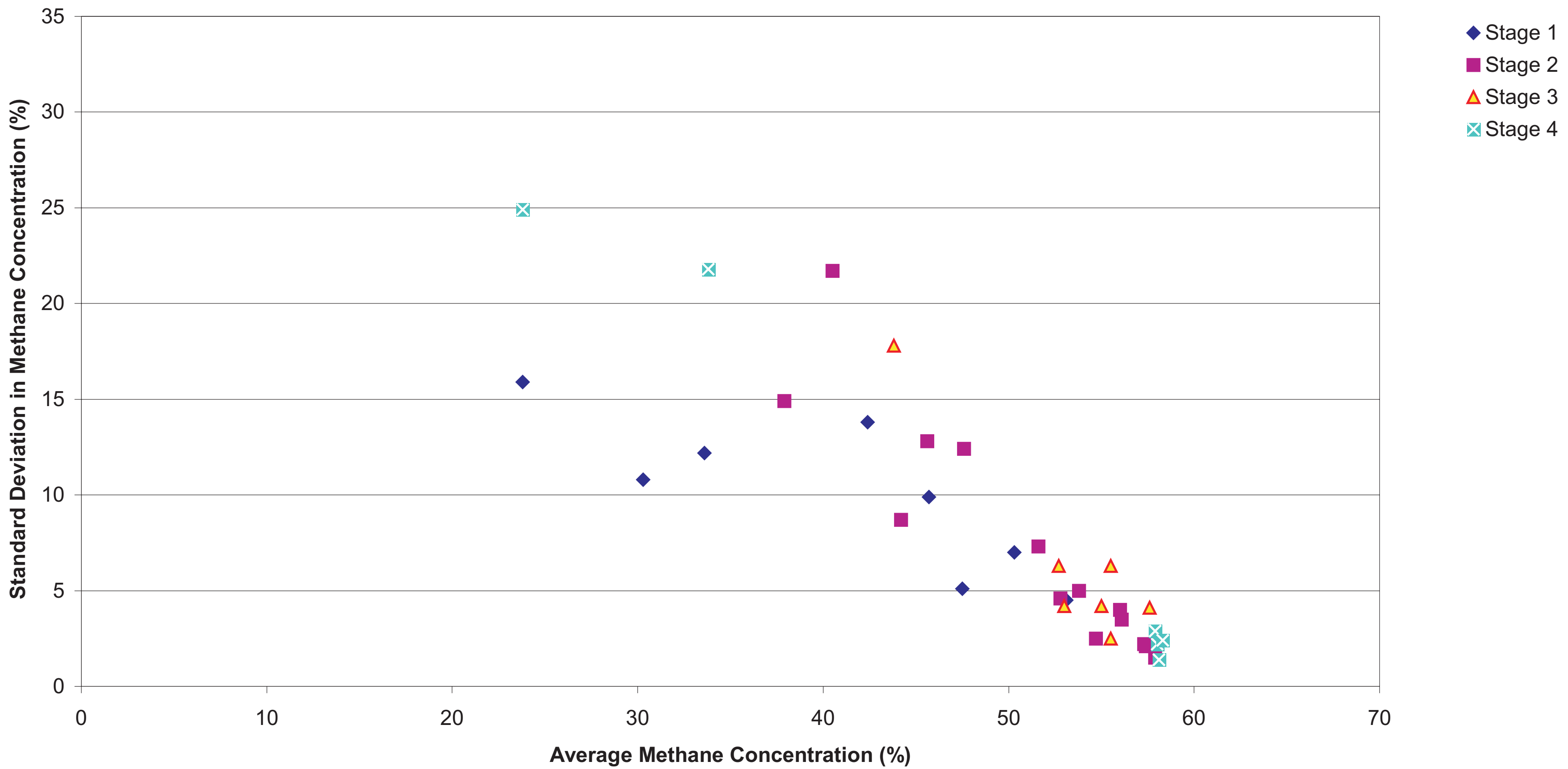
Review of the Performance of Hydraulically Contained Landfills: Research Sites

Figure 4.13
Comparison of Gas and Leachate Quality at AG311 (Cell 3A)

November 2005
10744-S83.cdr holfs

Entec

H:\Projects\Hm-250\08975 Viridor ph2 hyd\Data\Gas Quality Data\
Gas Composition Data to 181004.xls\AG311cht & AG311cht_lch



Review of the Performance of Hydraulically Contained Landfills

Figure 4.14
Variability of Methane in In-Waste Gas Monitoring Wells at Whitehead Landfill Site (August - October 2004)

November 2005
10744-S84.cdr holfs

Entec

H:\Projects\Hm-250\08975 Viridor ph2 hyd\Data\Gas Quality Data\Mean n stddev for onwaste gas.xls\CH4Mean v stddev

Figure 4.15a - Relationship between Methane and Oxygen Concentrations

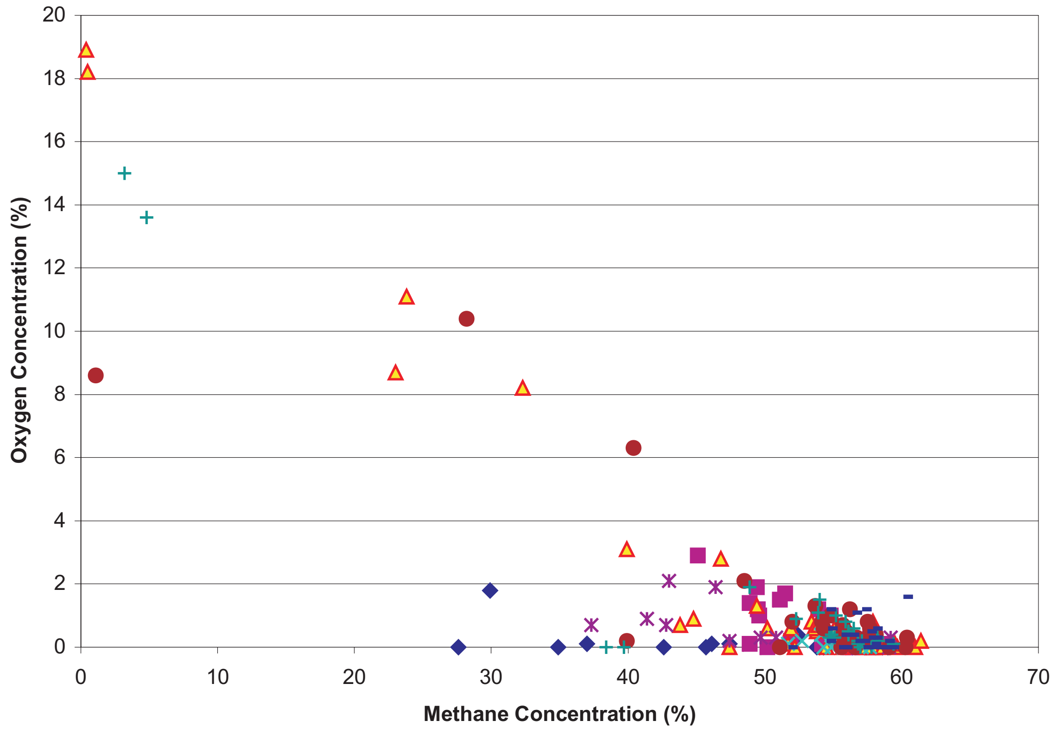
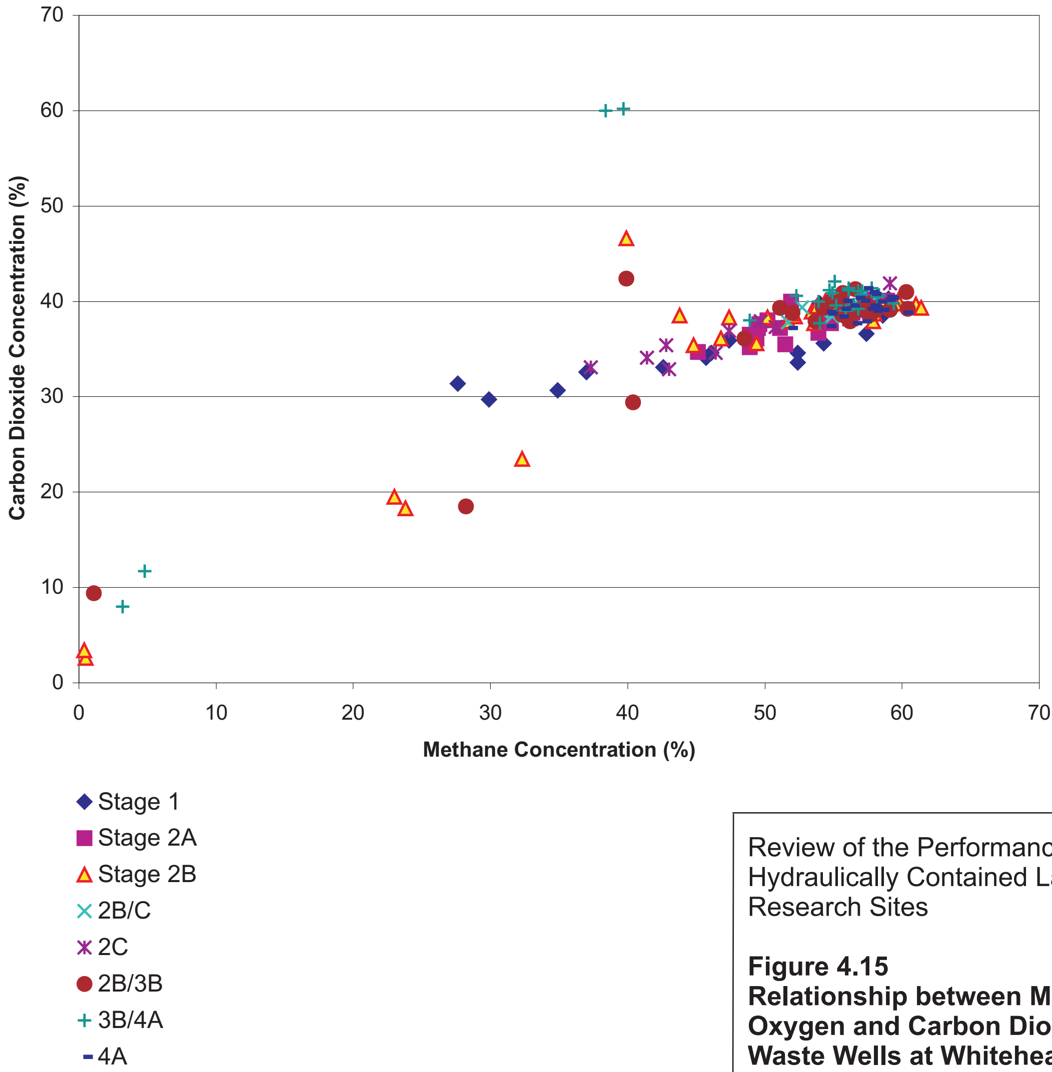


Figure 4.15b Relationship between Methane and Carbon Dioxide Concentrations



Review of the Performance of Hydraulically Contained Landfills: Research Sites

Figure 4.15
Relationship between Methane and Oxygen and Carbon Dioxide in In-Waste Wells at Whitehead Landfill (August - October 2004)

November 2005
10744-S85.cdr holfs

Entec

H:\Projects\Hm-250\08975 Viridor ph2 hyd\Data\Gas Quality Data\In Waste Gas Wells 2004.xls\CH4 v O2 & CH4 v C)2

5. Engineering and Operational Issues

5.1 Introduction

Basic conditions for a landfill site operating on the principle of hydraulic containment require that either:

- the surrounding groundwater level is higher than the leachate level within the landfill; or,
- the surrounding piezometric level is higher than the leachate level within the landfill.

This results in a hydraulic gradient into the landfill body, which in turn prevents leachate migration by advective processes.

5.1.1 Landfill Settings

Landfills developed assuming a principle of hydraulic containment may or may not have significant engineered lining measures. The landfill may be developed in four principal settings:

- entirely in permeable water bearing strata (aquifer), e.g. sand and gravel, sandstone, limestone, etc;
- in permeable water bearing strata, but with the base in low permeability strata, e.g. a clay unit beneath a sand and gravel layer;
- in low permeability strata, above a confined aquifer with significant water pressures, e.g. a sandstone aquifer beneath clay strata, as at Whitehead Landfill;
- in predominantly low permeability strata that contain high permeability layers (e.g. interbedded sandstone and mudstone) above a confined aquifer.

The latter setting may not constitute true hydraulic containment. There is potential for significant groundwater inflow at discrete horizons within the sidewalls of the landfill, or for leachate escape at such horizons should the highly permeable strata not be water bearing.

5.1.2 Groundwater Inflow

Active 'flow' of groundwater into the landfill body will depend on the geological setting, flow rate in an aquifer in direct contact with the landfill and the type and quality of the engineered lining construction. Very low permeability linings will limit the flow velocity into the landfill.

Flow may also be affected by the nature of the leachate collection layer in the landfill. A well-constructed and operated leachate collection system will ensure that inflow is readily dispersed and removed, thereby maintaining a positive head into the landfill. In the absence of a well-developed leachate collection system, the composition of the body of the landfill may influence the dispersion of groundwater inflow. This could lead to perching of leachate that is not adequately drained by the extraction system. In turn, this might cause locally increased leachate levels with respect to surrounding groundwater level.

Depending on the nature of the landfill and its surrounding strata, and the characteristics of the local hydrogeological regime, there is a potential for 'through-flow' of groundwater. Without sufficient low permeability barriers, as currently prescribed in the legislation, leachate could be expelled from a landfill by the passage of groundwater through the site, or could 'leak' from the base of the landfill.

5.1.3 Regulatory Issues

The introduction of the European Union Directive 99/31/EC on the landfill of waste ("the Landfill Directive") and its transposition into UK law through the Landfill (England and Wales) Regulations 2002 (SI 2002/1959) and the Landfill (Scotland) Regulations 2003 (SSI 2003/235) included a requirement for minimum engineering standards for landfills.

Existing and new landfills were classified as inert, non-hazardous or hazardous depending on the nature of the waste to be accepted. The Landfill Regulations also banned the co-disposal of waste of different classes in the same landfill.

For each class of landfill, minimum requirements for lining were specified. For sites other than inert landfills, these included a geological barrier (with a maximum permeability of 1×10^{-9} m/s and minimum thickness of 1 m or equivalent) and an artificial sealing layer. For all sites where leachate may be generated, a leachate drainage layer with a minimum thickness of 0.5 m is also specified.

The Landfill Regulations notes at Schedule 2, paragraph 2(1)(c) that measures are necessary to prevent groundwater from entering into the landfilled waste. However, Paragraph 2(2) indicates that this requirement will be interpreted by the Environment Agency in a risk-based manner.

The Environment Agency has indicated in Landfill Regulatory Guidance Note 6 (Interpretation of the Engineering Requirements of Schedule 2 of the Landfill (England and Wales) Regulations 2002) that in general,

> "*groundwater must be prevented from entering the landfill as far as is necessary to ensure that there is no unacceptable risk to the stability or effectiveness of engineering controls (e.g. the lining and leachate collection system), other environmental protection measures and the environment.*"

The guidance goes on to consider that acceptable risk is to be determined on a site-specific basis. This is to be achieved by showing that, through risk assessments, the requirements of the Groundwater Regulations are satisfied and that the stability of the landfill (lining, waste and surrounding strata) and the efficiency of the leachate extraction system, the groundwater control system and gas extraction system are not compromised.

5.2 Landfilling Engineering and Operations

5.2.1 Void Development

The development of the void as a landfill may require excavation beneath the surrounding groundwater or piezometric level. Alternatively, where development is proposed in an existing void (remaining from mining or quarrying) flow into the void may be evident in the pit walls or base. In either case, groundwater control may be required to ensure the proposed landfill void can be properly developed without resulting in loss of integrity of the lining system.

Softening of Sub-grade

Profiling of slopes within the void, in the presence of groundwater seepage, may result in a softened lining sub-grade (i.e. the surface on which the landfill lining will be developed) where the sub-grade comprises clay rich materials. This has implications for the integrity of the lining developed above. Softened conditions may present difficulty in preparing a compacted clay lining to suitably low permeability.

'Heave'

Where high groundwater pressures exist in confined aquifers beneath low permeability strata, excavation of materials to form the void may eventually reduce the confining load above the aquifer, resulting in basal heave (i.e. the intact strata immediately above the aquifer is disrupted by the upward hydraulic pressure). This may then result in active inward flow of groundwater to the void, making preparation of the sub-grade difficult.

Void development in sub-water table conditions may therefore require dewatering of the surrounding strata to reduce groundwater or piezometric levels if seepage through the void sidewalls or basal heave is to be prevented.

5.2.2 Dewatering Systems

Dewatering of the strata in which a hydraulically contained landfill is to be developed may be required to prevent damage or disruption to the lining system prior to waste tipping, or until sufficient mass has been placed to resist uplift forces.

Dewatering will reduce the piezometric levels around the area to be developed. This in turn may effectively result in a site being developed as an above groundwater landfill during its early life. Consequently, the performance of the lining system must be such that leachate cannot escape under gravity and contaminate the underlying strata and groundwater.

Reduction in piezometric levels may occur naturally as a result of excavation above a relatively low permeability aquifer. The excavation will reduce the effective vertical forces acting on the aquifer, allowing upward flow into the base of the void. If the recharge rate of the aquifer is less than the outflow (by seepage through the base of the void), the surrounding piezometric level will fall. When the aquifer becomes 'confined' once again (following development of a low permeability lining and the addition of waste to the void), the piezometric level will rise once again.

Where artificial dewatering is required, this may include the use of boreholes or pumping sumps created in the excavation. Such dewatering schemes require careful design and planning if they are to be effective in draining strata to allow the lining to be developed for new cells without compromising the degree of hydraulic containment that may already exist in adjacent cells.

5.2.3 Lining Systems

Installation and construction quality assurance of modern landfill lining systems invariably requires dry conditions in the void. As noted above, sub-grade integrity is important in developing the lining system. This may require dewatering of the void prior to development.

High groundwater level/pressures developing behind or beneath the lining system are undesirable without sufficient load within the landfill to prevent disruption of the lining. If these external forces exceed the internal forces exerted by the lining system, waste and *in situ* strata (beneath the base of the landfill), then heave of the base or lining may occur. Such effects will inevitably disrupt the lining, causing loss of integrity and potentially adversely affect stability.

Compacted Clay Lining

Compacted clay lining (CCL) systems require the preparation of a defined thickness of low permeability clay to ‘seal’ the landfill. These usually require the mechanical compaction of clay, placed in discrete lifts, at a pre-determined moisture content to a required density.

Where CCLs are developed on a softened sub-grade, it may not be possible to achieve the required specification, despite increased mechanical effort. The sub-grade may also be compressible, which in turn could cause deflections in the overlying CCL. Deflections and deformations, which develop excessive strains in the CCL, can result in fissures and fractures in the CCL and ultimately loss of integrity. These fissures and fractures could then potentially allow ingress of groundwater or egress of leachate (depending on conditions).

Where CCLs are developed in strata with high groundwater levels, the lining may become fissured or fractured due to high groundwater pressure, again providing potential direct pathways between the surrounding strata and landfill body.

Composite Lining Systems

Composite lined systems (i.e. a CCL and an overlying geosynthetic sealing layer) may become de-laminated as pressure is exerted beneath the geosynthetic lining following disruption to the CCL.

Whilst the geosynthetic lining system itself may remain intact initially, de-lamination provides zones where enhanced leachate migration (or groundwater inflow) could become concentrated in any subsequent defects. De-lamination could produce ‘wrinkles’ in the geosynthetic where stresses will be concentrated subsequently and defects may develop in service.

To prevent excessive water pressures developing, dewatering may therefore be required until sufficient waste has been deposited within the void to ensure forces are balanced.

5.2.4 Leachate Collection Systems

The effective performance of a hydraulically contained landfill typically requires a significant head difference between the surrounding groundwater and leachate levels. This requires an efficient leachate collection and extraction system be provided, whose operation can be maintained until the waste is stabilised and the site no longer poses an unacceptable environmental risk. Without an efficient system of leachate extraction, the potential exists for significant leachate depths to develop locally, or perched leachate to form at high levels in the landfill, reducing the head difference between the groundwater and leachate. Guidance on the design and operation of leachate drainage systems is given in an Environment Agency Technical Report (Environment Agency, 2002a).

Design Issues

The leachate collection system design for a hydraulically contained landfill must be carefully considered if it is to be suitable throughout the complete lifecycle of the landfill.

In common with above water table landfills, the leachate collection system for a hydraulically contained landfill must be able to accommodate the leachate that will be generated due to ingress of precipitation (either falling directly on exposed waste prior to capping, or infiltrating through the landfill cap following closure). However, allowance must also be included for potential groundwater ingress where appropriate.

Without such consideration, the capacity of the leachate collection system might otherwise be insufficient causing increasing levels of leachate in the landfill over the life of the site. Other than including additional leachate extraction wells (by retro-drilling) there will be no opportunity to upgrade the leachate collection system in the landfill following waste deposition.

Construction Issues

Leachate collection systems when installed in modern landfills are normally subject to construction quality assurance (CQA) inspection and approval. As noted above, heave may deform the lining system if external forces are not adequately resisted. Disruption to the lining system may in turn result in disruption of the leachate drainage system (e.g. by deflection of levels and falls to cell bases or pipework, slumping/thinning of leachate drainage stone layers, etc).

Service Life and Performance

Leachate collection systems in the base of landfills are difficult to maintain. Once filling with waste commences the leachate drainage layer and pipework in the base of the landfill cannot be readily accessed. The design of the leachate extraction system must therefore consider degradation of the system over its required service life.

Degradation of the leachate collection system will inevitably reduce its efficiency. The efficiency of the leachate drainage system might be decreased due to:

- ‘clogging’ of separation geotextile, drainage stone voids or pipework by siltation, bio-growth, or precipitation of materials; and
- deformation or collapse of drainage pipework, as a result of weakening by attack from acids, solvents, or oxidising agents.

Consequences of Reduction in Efficiency

Whether reduced efficiency of the leachate collection system arises due to defects occurring during construction (or before tipping) or as a consequence of long-term degradation, the consequences may be the same. Without effective removal of leachate, the head difference between the leachate and surrounding groundwater may be reduced locally, reducing the performance of hydraulic containment.

5.2.5 Landfill Gas Collection and Extraction

Landfill gas generation within the body of the landfill could potentially increase pore pressures within the waste mass, particularly where drainage systems are ineffective. In turn this could increase the outward gas/water pressures acting on the lining system. In turn this might decrease the effective level differences between the groundwater and the leachate.

Recent legislative changes now require that landfill gas must be collected and used, or flared. Consequently effective landfill gas collection and extraction measures are a requirement in all modern landfills. They are normally installed as waste is deposited and operate as early as is practicable following waste deposition.

As with leachate collection systems, gas collection systems can be subject to degradation with time. However, gas collection systems can be maintained from surface (by retro-drilling new wells or refitting collection pipes, etc).

5.2.6 Waste Deposition

Operation of a hydraulically contained landfill should ideally allow for rapid stabilisation of the waste. This will reduce the period during which escape of leachate might present a risk to the environment. This will in turn reduce the dependence on engineered systems that might be subject to degradation and loss of efficiency with time (e.g. the leachate collection system, or the lining integrity).

Municipal waste in plastic bags presents particular problems for waste stabilisation. If the bags remain essentially intact on deposition and subsequent covering, the rate of degradation of the organic component of the waste within may be significantly reduced. Moisture cannot enter the bag to promote degradation.

Intact bags in layers may also provide a barrier to downward percolation of water, forming perched leachate layers in the landfill. If this occurs in the upper layers of the landfill, there is a greater potential for leachate migration through the sidewalls, since the difference in level between the groundwater and leachate will be significantly reduced in such zones. This problem may be compounded if waste is placed in essentially horizontal layers.

Waste placement methods in hydraulically contained landfills should ensure that bags are properly shredded and that the waste is placed in inclined layers (to promote drainage of leachate toward the sideslopes and the leachate drainage layer).

5.2.7 Capping

Capping of the landfill provides a low permeability layer that restricts the amount of water entering the waste mass. It is accepted that all landfill caps leak to some degree (whether they are constructed from compacted clay, geosynthetic layers or both) and this can have significant consequences for large landfills. For a hydraulically contained landfill, to reduce the potential generation of leachate as far as is practicable, the cap should be developed to the best possible standards.

The timing of cap construction and its serviceability long term will affect the volume of water that may pass into the waste mass. Before final capping, a temporary cap may be applied to the landfill to promote run-off of precipitation and reduce the amount of infiltration. Temporary capping may be necessary to allow initial settlement of the waste to proceed. Significant settlement can occur soon after waste deposition is completed in a cell. Settlement effects can damage and disrupt capping layers, causing increased leakage into the underlying waste.

Long term stability may also be an issue. Differential settlement has the potential to damage the cap, and consideration of these effects need to be considered in the design and scheduling of filling operations at the site.

5.2.8 Monitoring Systems

All landfills require effective monitoring to ensure that they are performing properly and are not having an adverse impact on the surrounding environment. Monitoring is required within the landfill body (e.g. leachate level and quality) and surrounding the landfill (e.g. groundwater level and quality, landfill gas migration). Guidance on the monitoring of landfill sites is provided in Environment Agency (2002b).

The nature and extent of the monitoring systems required will reflect the perceived sensitivity of the installation and its surrounding environment. The monitoring systems and schemes should ideally present an 'early warning' system that will allow operational changes to be made to prevent leachate escape. This may include increasing rate of leachate extraction in response to rising leachate level or falling groundwater level.

The monitoring scheme will need to consider the nature of the surrounding strata. Schemes of monitoring groundwater in relatively homogenous strata, where large contaminant plumes might develop in response to leachate escape, could fail in settings where groundwater flow is concentrated in fractures in a rock mass. Consideration of different groundwater regimes surrounding the site would also need to be considered. For example, for a fractured rock consideration of the conditions at depth or some distance from the site would be required and additional measures might be required to monitor different fracture systems that might be intersected by the fractures immediately surrounding the site.

5.3 Site Studies - (1) Brogborough

5.3.1 Introduction

This section considers the landfill construction and operational issues at the site and assesses these factors against general practice for above water table landfill sites. Of particular relevance will be the current requirements and the long-term performance post closure.

The Brogborough Landfill relies on *in situ* low permeability strata to provide containment of the waste. The Oxford Clay, forming the sub-grade to the landfill, has an inherent low permeability. A higher permeability unit beneath the Oxford Clay, the Kellaway Sands, has a piezometric level ~20-30 m above the base of the deeper parts of the landfill. No significant seepage or flow into the excavation is noted.

Landfilling has been taking place at the site since 1983, in prepared cells. Later development has included improved cell preparation and engineering measures, and retrofitting of leachate control systems in the earlier cells.

5.3.2 Development Conditions

Landfill operations commenced at the Brogborough site in 1983 in a significant void developed as a result of clay extraction for brick manufacture. Clay extraction has reportedly taken place during much of the 20th Century, and continued until the early 1980s. Three apparent conditions under which landfilling has taken place, are reported:

- The earliest landfilling operations (1983 to late 1980s) included waste placement directly onto the excavated profile, or backfilled weathered clay arisings rejected from the brick making operations. No preparation of the cells was undertaken.
- Later operations (late 1980s to late 1995) involved the levelling of the base and grading of the side slopes of the void, but no additional engineering was provided prior to waste disposal.
- The recent development of the site (after 1995) has included lining of the base and external sideslopes of the void with engineered clay recovered from the site.

The increasing scope of site preparation and landfill lining probably reflect increasing regulatory control at the site and a general increase in landfill standards with time.

Only the latest cells (those developed after 2001) include any basal leachate collection measures. The other cells have vertical drainage systems installed (mainly as retro-drilled wells).

5.3.3 Waste Management Licence Conditions

Regulatory control, in the form of a Waste Disposal Licence issued in 1992, includes reference to site preparation. This required separation of operational areas by intermediate clay bunds. There was also a condition requiring 'sealing' of the base and sides of the void such that permeability was no greater than 1×10^{-9} m/s. The methods of sealing required agreement with the relevant authority. The Waste Disposal Licence subsequently became a Waste Management Licence following legislative change, and was later modified several times.

The original licence, in addition to requiring 'sealing' of the base and sideslopes of the void, required the installation of a landfill cap (comprising 1 m thickness of compacted clay to a maximum permeability of 1×10^{-9} m/s). There was no specific requirement for leachate abstraction, but a requirement that levels did not rise above a control level 3 m below the lowest surrounding ground level for the relevant cell.

A Waste Management Licence, EA/WML/75021, was issued on 12 July 2001 for the extension area to the site. Relevant conditions affecting design, engineering and operation of the site are included, and make reference to the measures described in the site Working Plan prepared by the operator.

Conditions refer to the standard of containment (including Construction Quality Assurance (CQA)), leachate and landfill gas management systems and capping requirements.

5.3.4 Engineering and Operational Issues

Void Preparation

The landfill has been developed in a void formed by clay extraction to provide a feedstock for brick manufacture. The clay extraction is assumed to have commenced in the early 1900s, and was completed in the 1980s.

There are reportedly artesian conditions in the general vicinity of the site (indicated by piezometric levels ~30 m above the lowest levels of the void). The potential for basal heave (i.e. groundwater pressures beneath the site causing uplift and disturbance in the floor of the excavation) is a function of the groundwater regime in the strata immediately beneath the site, and the thickness of *in situ* material remaining to resist the upward force.

It is noted that there is an apparent reduction in piezometric level in response to excavation in the clay, and that groundwater levels drop significantly in the vicinity of the deeper excavations. This might be a result of basal heave where excavations have removed the 'beam' resisting uplift. For the piezometric levels noted, a beam of ~15 m intact clay is necessary to resist uplift (assuming a density of clay of ~2.0 t/m³, and a Factor of Safety against uplift of 1.0).

In this setting, basal heave is unlikely to be indicated by any dramatic or significant inflow of water. It is considered likely that the reduction in stress level as a result of excavation will result in a gentle "expansion" of the Kellaways Sand as the high pore pressures "push" the overlying clay upward. This expansion of will increase the porosity of the sands, which in turn will lead to a reduction in pore pressure and hence piezometric levels. Given the relatively low permeability of the sands, and therefore a correspondingly low initial porosity, the change in piezometric level may be significant as the porosity increases since relatively little water may be present in the sand.

This type of heave would be very subtle in contrast to the more dramatic failure of a confining beam above an aquifer dominated by fissure flow. If no change in porosity (or fracture volume) is possible as the vertical stress is reduced by excavation, the upward pressure in the aquifer may cause significant failures in the confining layer and produce marked inflows to the base of the void. This mechanism may have resulted in the inflows described at the Whitehead Landfill (included as Study Site 3) and described further in Section 5.5.4.

First Phases (pre 1990)

The conditions of the pit base and sideslopes, and hence the quality of the subgrade, to the site is unknown for the areas subject to the earliest phases of landfilling. It is known that the older parts of the pit were also used for backfilling unsuitable clay rejected from the brick-making process.

Base

For the early stages of landfilling, there has been no preparation of the void prior to waste disposal. Waste has been tipped directly to the excavated void, or over clay waste backfilled to the void during extraction. This material is described as weathered Oxford Clay. Its condition prior to landfilling is unknown. It is assumed to have been relatively loose tipped and probably only lightly compacted by trafficking with plant, or consolidated due to self-weight.

There is a significant thickness of *in situ* Oxford Clay (~10-30 m) between the base of the excavation for the Stage 1 and Stage 2 areas and the underlying Kellaways Sand. The potential for basal heave is considered to be low in this setting.

The base of the void has not been profiled to encourage leachate drainage. Combined with the potential disruption to the exposed clays (through softening or desiccation, slumping of pit walls (see below) and the loose tipping of reject clay, it is unlikely that leachate will drain freely under gravity in the base of the areas used for the early stages of filling.

There is potential for leachate to collect in 'low spots' in the unprofiled base. If there is little inflow to dilute the leachate, concentrations of potential contaminants could rise significantly in such areas, since there will be no flow to mix leachate with other areas.

Sideslopes

The cross sections prepared for the site show an excavation profile with very gentle slopes at the southern margin (~ 1:25 (v:h) in the south-east corner in the Stage 1 area). The slope between the Stage 1 and Stage 2 areas is significantly steeper (~ 1:6 (v:h) measured on the line of section). Side slopes in other areas of the pit are up to ~ 1:3 (v:h).

In northern parts of the site, the steeper pit walls in Stage 4A might be expected to have degraded with time[1]. Although this might affect stability of the pit walls before filling, this has no major implications for quality of the containment in the context of this part of the site.

In the southern areas of the site, softening of the clays, or desiccation and fissuring of the clay due to repeated wetting/drying cycles, may have affected the quality of the sub-grade on the flatter slopes. Again, this has little implication for containment, since these effects would be restricted in vertical depth.

Intermediate Stages (1990-1995)

Base

The base of the landfill has not been systematically profiled during development, but has reportedly been levelled. It is uncertain whether each of the phases that are included in this development stage (i.e. 3A and 3B) were profiled to promote leachate drainage to a particular location.

Inspection of the cross sections prepared for the site suggests that there is ~10 m *in situ* Oxford Clay between the base of the void and the underlying Kellaways Sand. This should provide sufficient mass to mitigate the potential for basal heave.

Sideslopes

The cross sections prepared for the site do not show the excavation profile through the Stage 3 area). However, slopes in the adjacent Cell 6b are ~1:3.5 (v:h). Again, the age of the slopes is unclear and their stability may have been affected by time.

Later Development (post 1995)

Base

The base of the landfill has been prepared in the later stages of development. A regular fall on the base of 1:50 (v:h) is required under the terms of the Waste Management Licence. This has been developed in the later cells prepared in Stage 3, Stage 4B and Cells 5 and 6.

---

[1] Long term stability of the pit slopes will be affected by the drainage of the clays. Clays are typically overconsolidated at depth (due to burial pressure), and during excavation may increase in strength due to negative pore water pressure developing. With time, pore water pressure then increases (due to slow flow in the clay) and eventually reaches equilibrium, or begins to exert a positive pressure with consequent reductions in material strengths. It can take decades for equilibrium conditions to be achieved, but once stable slopes may then become unstable, deteriorate and collapse.

The subgrade to the landfill in these areas has therefore been subject to significant engineering. Soft spots, desiccated areas, fissures etc will have been removed or remediated in preparing the required profile.

It is noted that, for the later stages of landfill development, there was as little as ~2 m of *in situ* clay remaining above the Kellaways Sands. Given the piezometric levels recorded for this unit, it is highly likely that disruption of the basal sub-grade may have resulted from basal heave, unless pore pressures had already reduced in response to other void development in the vicinity.

Sideslopes

Excavation of the clay pit has left sideslopes of ~1:3.5 (v:h) in the areas most recently filled. Again, the age of the slopes is unknown and may be subject to degradation with time. However, the profile identified on the plan of the site appears very regular and it is assumed this development is relatively recent or has been subject to reworking during cell development.

Landfill Lining

The development of the later stages (post 1995) has required that the landfill be lined with an engineered clay barrier, to a minimum of 1.0 m thickness (raised in 250 mm lifts) and with a permeability no greater than 1×10^{-9} m/s. The construction of the lining was also subject to rigorous CQA inspection and approval.

The landfill lining has been applied to the base of the void and to the external sideslopes.

Lining Sub-grade

Although there has been no specific mention of water ponding in the base of the void, and notwithstanding the preparation of the subgrade, there is definite potential that the clay forming the basal subgrade has been disrupted by heave, as noted above. This may have increased the effective permeability of this material, and in particular introduced near vertical drainage pathways (through fissures/fractures developed in the clay).

Lining Quality

The lining prepared for the base and sideslopes of the later stages of development at the site has been subject to a rigorous CQA process. Combined with the low permeability sub-grade (the Oxford Clay) this would be expected to provide a high level of protection to the surrounding ground.

The sub-grade was presumably properly prepared prior to construction of the lining, and its acceptance would have formed part of the CQA programme. Assuming that waste placement followed relatively shortly after construction, the compacted clay lining would not have degraded significantly (by evaporation of pore water near surface). As the groundwater levels rebound, the clay would remain saturated, preventing potential degradation caused by later desiccation (e.g. by reactions with leachate or heating as the waste degrades).

The rebound of piezometric levels may be expected to occur in response to reloading of the Kellaways Sand and a consequent reduction in porosity as water is “squeezed” out as vertical stress increases. An important issue would be the rate of rebound, in response to loading, and the degree of fracturing that might have occurred in the in situ clay above the Kellaways Sand.

If the clay had remained essentially intact, the rebound rate might be seen to be quite rapid, since the water in the Kellaways Sand is expelled under load and contained within the sand unit. Rebound rates might be slower if there is any "leaking" through fissures through the confining beam of clay. Uneven loading of the void might produce enhanced basal heave in unfilled areas.

Water expelled in one area of the void might cause pore pressures to rise locally elsewhere in response to the applied load. The increased uplift forces could cause the landfill lining to become fissured or fractured, thus reducing its integrity and providing direct pathways for leachate egress from the body of the waste should leachate levels exceed groundwater piezometric levels.

Leachate Management

Leachate Collection

It is noted that the base of the landfill has not been profiled many areas of the site and no leachate collection layer is included in the base of the landfill.

A leachate blanket is installed in the base of the later phases and a specification for its thickness and quality is a licence requirement. The base of the landfill is profiled to provide leachate drainage by gravity to a pumping sump.

Vertical leachate drainage measures are included in those areas without a basal drainage layer. These were retrofitted by drilling and are used for leachate abstraction by pumping. Fin drains have also been installed in some areas, to promote abstraction from a wider area of the waste.

Leachate Abstraction

As noted above, in much of the site leachate is removed from the waste by pumping from retrofitted wells. Leachate is abstracted through chimneys prepared in the lined cells, which were raised with the waste as landfilling proceeded.

Leachate Treatment

There has been little recirculation of leachate between cells. Leachate is abstracted from wells in Stages 1 and 2 and pumped to a lagoon prior to disposal from site. Between May 2002 and March 2003, approximately 10 000 m^3 leachate was removed from these areas of the site. Similarly, leachate removal from Stages 3 and 4 has been facilitated by the installation of additional leachate monitoring and abstraction wells. The leachate management strategy for the site is to control leachate levels 2 m below the piezometric level in the underlying Kellaways Sand, with additional leachate abstraction wells installed progressively to allow this to be achieved in areas of the site that are not served by a drainage blanket.

Waste Placement

The method of waste placement is not described in the documents reviewed, but it is assumed to be in relatively thin layers (the early licence refers to a maximum thickness of 2 m). It is not known whether these were placed horizontally or inclined. Daily or intermediate cover is noted, and a minimum thickness of 150 mm is required.

The nature of the waste deposited and the nature of layering is an important issue in the context of leachate generation, and transmission to extraction systems. Horizontal layers may encourage 'perching' of leachate above lower permeability horizons (either waste or cover). It has been noted that, in the vertical drainage wells installed to allow extraction of leachate, there may be evidence of perched leachate (i.e. leachate was initially encountered at high levels in the waste and generated large volumes initially, which have subsequently reduced).

Inclined layers would reduce the potential for perched leachate levels (since leachate would ‘flow’ downslope), but would require adequate drainage at the margins of the void to prevent leachate collecting against the sideslope.

Capping

The waste at the site has been progressively capped with an engineered clay layer following completion of landfilling. The most recent licence specifies a minimum 1 m thick layer, with cover soils. The construction of the cap is subject to a CQA regime similar for the lining system.

The standard of capping would be typical of that for above water table landfills and does not appear to include any additional measures by virtue of the development of the Brogborough site as a hydraulically contained facility. The capping proposals will reduce infiltration and promote run-off from the cap to surface water collection measures.

5.3.5 Comments on Engineering and Operational Issues

The historical development of the Brogborough Landfill has essentially used natural containment of the in situ Oxford Clay. Waste has been generally landfilled in areas of the brick-pit with no sideslope and basal preparation or lining. The later phases have however been engineered with prepared sub-grade and compacted clay linings.

Leachate extraction measures are included across the site, but only the later phases have a leachate drainage layer installed at the base of the landfill.

The issues relevant to the site engineering and operation are considered below.

Site Preparation and Lining

Little or no preparation of the void areas has been undertaken in the earlier stages of the landfill development. This is not of major significance since there is no engineered lining to much of the site.

For the later phases, it is apparent that the void has been developed to a greater depth than the early phases. This has reduced the thickness of *in situ* clay above the Kellaways Sand. As a consequence of groundwater rebound, this may result in basal heave disrupting the strata and possibly the lining installed to the base and sideslopes in these areas.

Leachate Drainage

Leachate is drained from the site via vertical wells retrofitted in the older areas of the waste and from a basal leachate drainage layer in the more recent areas. The leachate extracted is pumped to a holding facility and tankered offsite for disposal.

The limited preparation of the base of the older areas of the landfill may have resulted in areas where leachate will ‘pond’ and may not move toward any abstraction point. Such areas may contain a higher strength leachate than is removed from other wells.

The body of the waste now contains numerous vertical drainage measures (i.e. the retrofit boreholes), which will limit the potential for development of perched leachate occurring.

Capping

The practice of capping soon after final levels are achieved significantly reduces the potential for infiltration of incident rainfall. This minimises the production of leachate through addition of surface water. These measures are typical as those on most landfill sites, irrespective of whether they are developed sub-water table or above groundwater levels.

Comparison With Above Water Table Sites

The development of the Brogborough Landfill illustrates the changing approach to landfill practice with time. The earliest phases show no preparation of the void, with the later phases showing a high level of preparation and engineering. This later development would be typical of the standards expected of above water table sites.

The use of vertical drains to promote leachate extraction from the body of the waste is perhaps slightly greater than would be found on most modern sites. However, older landfilling practice may have required such measures to be retrofitted on modern sites.

The pumped extraction system would appear to be relatively flexible on this site and extraction can be achieved from a number of locations. The disposal offsite by tanker would presumably mean that the holding tanks could be bypassed if required offering flexibility following breakdowns of other parts of the system interruptions of pumping.

5.4 Site Studies - (2) Poole

5.4.1 Introduction

This section considers the landfill construction and operational issues at the Poole Landfill site and assesses these factors against general practice for above water table landfill sites. Of particular relevance will be the current requirements and the long-term performance post closure.

The Poole Landfill might appropriately be considered as an unlined site, in direct continuity with strata which incorporate high permeability zones. Landfilling has been taking place at the site since the 1960s, with little development control in the early years and no significant site preparation/landfill engineering works. Later phases have included some preparation and engineering, including some retrofitting of gas and leachate management measures.

5.4.2 Development Conditions

The site has a long history of landfilling, commencing during the 1960s. Since 1974, the site has been developed in separate phases. The site closed for the receipt of wastes in 2004.

Development of the site has resulted in three apparent conditions for landfilling:

- Landfilling into areas with little or no engineering (typical of areas landfilled prior to 1974);

- Landfilling into areas with no lining, but incorporating simple leachate drainage measures (typical of Phases 1 to 3); and
- Landfilling into partially lined areas, with more extensive leachate drainage systems (Phase 4).

The progression of the development has included apparently increasing engineering standards with Phase 4 including some lining.

Phase 4 was developed from 1995 onwards. This phase might reasonably be considered to reflect recent landfill practice (i.e. after effective implementation of the Waste Management Licensing Regulations). It is assumed that the construction of the side wall lining and a more sophisticated leachate drainage system occurred as a consequence of stricter development and regulatory control than existed for Phases 1 to 3.

Other engineering at the site has included the installation of low permeability geosynthetic barriers at the northern and southern boundaries. These are provided for landfill gas migration control and are apparently installed in the superficial materials above the low permeability Mercia Mudstone strata.

5.4.3 Waste Management Licence Conditions

The current Waste Management Licence, WDL/28/2, was issued on 17 July 2002. Relevant conditions affecting design, engineering and operation of the site are included, and make reference to the measures described in the site Working Plan prepared by the operator.

Conditions refer to the standard of side slope lining, gas extraction, leachate management systems and capping.

The licence requires engineered containment systems for Phase 4. The conditions relevant to the leachate management systems do not apparently describe requirements for basal drainage.

5.4.4 Engineering and Operational Issues

Void Preparation

There has apparently been only limited preparation of the void prior to landfilling.

The landfill has been developed in a void formed by clay extraction to provide a feedstock for the adjacent brickworks. The brickworks opened prior to 1880, and landfilling commenced during the 1960s.

Sideslopes

The cross sections prepared for the site (Entec, 2003a) show an excavation profile with near vertical batters on the northern boundary. The eastern boundary is also assumed to have a steep profile (the pit high wall). The base of the void is apparently formed essentially along a bedding plane, dipping to the northeast. The southern and western slopes appear flatter (the pit low wall). Geological and geotechnical conditions, together with the local hydrogeological conditions, will have dictated the pit geometry at the time of excavation.

The conditions of the pit slopes, and the quality of the subgrade to the site is unknown for the areas subject to the earliest phases of landfilling. In the northern part of the site, the steep pit walls would be expected to have degraded with time. Slumping of the clays would be expected, and collapse of the more competent sandstone/siltstone units would probably have resulted in a flatter profile to the sideslopes. Natural flow paths in the water bearing units may have become disrupted as a consequence of burial beneath collapsed units. This may have resulted in a more 'diffuse' flow into the pit through the collapse material, rather than appearing as springs or linear seepage at the interface between the more permeable layers and impermeable clays.

Long term stability of the pit slopes will be affected by the drainage of the clays. Clays are typically overconsolidated at depth (due to burial pressure), and during excavation may increase in strength due to negative pore water pressure developing[2]. With time, pore water pressure then increases (due to slow flow in the clay) and eventually reaches equilibrium, or begins to exert a positive pressure with consequent reductions in material strengths. It can take decades for equilibrium conditions to be achieved, but once stable may then become unstable, deteriorate and collapse.

The younger and flatter excavated profiles to the south of the void would be expected to be less susceptible to instability, although softening and collapse of the pit walls to the south-east of Phase 4 is recorded. This occurred in the vicinity of springs recorded in the area. It is reported that clay was used to buttress the slope and to restrict inflow in the area.

Base of Void

The base of the landfill has not been systematically profiled during development, and probably has a relatively irregular profile. Assuming development has been essentially along bedding planes, a general fall would be expected toward the north and east of the site.

There are reportedly artesian conditions in the general vicinity of the site. The potential for basal heave (i.e. groundwater pressures beneath the site causing uplift and disturbance in the floor of the excavation) will be a function of the groundwater regime in the strata immediately beneath the site.

Artesian pressures acting on the base of the landfill will act to support the performance of a hydraulically contained landfill, but presumably only if there is sufficient flow upwards toward the base of the landfill. If there is no effective inward flow, leachate could migrate down into higher permeability layers in the floor of the landfill.

Landfill Lining

Phases 1-3 of the site were developed with no lining to either sidewalls or the base of the void.

The landfill area comprising Phase 4 has been developed with an engineered clay lining on the side slopes. No basal lining has been prepared however.

[2] Low permeability clay, by its nature will not drain freely and tends to retain pore water in consequence. As vertical stress is reduced (by excavation), clay particles in the unit will move apart at the free face. The surface tension forces generated by the pore water (held between clay particles) will act to resist this movement since water cannot expand, and the low permeability of the clay prevents flow between pores. Consequently, the trapped water appears to 'suck' particles together, increasing the shear strength of the clay and increasing its apparent stability when exposed in slopes.

The side slope lining comprises compacted clay to a minimum 1 m thickness (measured perpendicular to the slope). The clay lining was raised in effectively horizontal lifts, working from the landfilled waste. The construction was apparently undertaken and monitored in accordance with a defined Construction Quality Assurance plan.

Lining sub-grade

The southern sideslopes of the void forming Phase 4 have been prepared to a profile of ~ 1:3 (v:h) during excavation of the brick-pit. Other slopes (to the east and west) are apparently steeper (maximum ~ 1:1 (v:h)). Some softening and slumping of the walls of the pit has been noted, indicating that groundwater flows into the excavation were occurring at the time of excavation and subsequently. It would appear that no dewatering of the surrounding strata was undertaken during excavation. Any inflow was pumped from sumps developed in the base of the excavation.

Stability of the pit slopes may have been an issue during excavation and subsequently. There is no indication that groundwater drainage measures were installed behind the landfill lining. Given the nature of the surrounding strata (interbedded low permeability Mercia Mudstone and higher permeability sandstone and siltstone horizons), groundwater inflow to the pit would have been concentrated at the higher permeability horizons. This was indicated by the occurrence of the seepage and springs identified in the pit faces.

Lining Quality

The construction of an engineered clay lining directly over springs and seepages will inevitably have resulted in softening of the lining, and potentially a poor interface between the lining and the sub-grade. This may possibly have resulted in an increase in permeability in the lining by piping failures through the clay. This effect may well be exacerbated by the horizontal lifts used in raising the clay lining, since permeability may be locally increased at the interfaces between lifts.

In the context of a hydraulically contained landfill, inflow through the lining would be acceptable and possibly desirable. However, it also suggests that any reverse of flow could potentially concentrate outflow of leachate through the areas of increased permeability in the lining. These areas are potentially associated with the higher permeability units behind the lining and would suggest, therefore, that any escaping leachate would be concentrated in the areas of highest permeability in the surrounding strata.

Leachate Management

Leachate Collection

It is noted that the base of the landfill has not been profiled in any phase to provide a regular fall toward a leachate abstraction point. The profiles included in the cross sections show an apparently uneven base to the landfill.

A leachate blanket is installed in the base of Phases 1 to 3, but its thickness and quality is uncertain. Leachate drainage towers are used to provide the main facilities for leachate extraction from the site. A leachate drainage layer is also included in Phase 4, with the addition of pipes laid in the drainage stone in a ‘herringbone’ arrangement.

It is apparent that dividing bunds have been constructed between some of the phases. These would presumably interrupt the drainage layer in the base of the landfill, although some linkage must exist since Phases 1 to 3 are drained from a single point (see below).

Leachate Abstraction

Leachate is removed from Phases 1-3 from a single collection point on the northern boundary - the ‘Vertical Pump’. Leachate from Phase 4 is removed via upslope risers on the southeastern boundary (the ‘Inclined Pumps’).

The leachate abstraction points are both located at the margins of the landfill. There is no apparent system of leachate extraction from the body of the landfill, other than by gravity drainage through the basal collection layer.

It is uncertain whether the leachate chimneys can also act as additional abstraction points if required. It would appear that there is no ‘back-up’ system for leachate extraction in Phases 1 to 3 if the single Vertical Pump is out of action. Phase 4 has two leachate pumps, but they are co-located in the south-eastern corner of the site. If there is any disruption to the drainage layer in the vicinity of the risers, or damage to the sidewall, there are similarly no apparent back-up arrangements for leachate extraction in Phase 4.

Leachate Treatment

Leachate separately removed from landfill Phases 1-3 and Phase 4 is treated at a single plant. Leachate from the two streams is mixed, treated and discharged via a gravity drain to a public sewer. Very large quantities of leachate are pumped from the site, averaging in excess of 2 000 m^3 per week. Cessation of the pumping, for example as a consequence of pump failure, has implications for the maintenance of hydraulic containment at the site. In the absence of removing such large quantities of leachate from the site, storage capacity within the wastes will be used up more rapidly and leachate levels will increase at a greater rate than at sites where there is less groundwater ingress, with the result that hydraulic containment may be lost.

Waste Placement

Waste placement procedures are described in the site Working Plan. It is assumed that the more recently placed waste was deposited in accordance with the procedures noted. Older waste may not have been placed in a controlled manner.

The procedures note that the landfill is developed in a series of thin layers (300 mm) thick, but it is not clear whether these are placed horizontally or inclined. Daily or intermediate cover is noted, but the thickness and type of material is not described.

The nature of the waste deposited and the nature of layering is an important issue in the context of leachate generation, and transmission to the drainage layers at the base of the void. Horizontal layers may encourage ‘perching’ of leachate above lower permeability horizons (either waste or cover). It has been noted previously that vertical drainage towers have been installed to promote drainage to the base of the void. The construction of the gas wells also suggests that these would act as vertical drains.

Inclined layers would reduce the potential for perched leachate levels (since leachate would ‘flow’ downslope), but would require adequate drainage at the margins of the void to prevent leachate collecting against the sideslope. However, drainage on the sideslope could potentially encourage direct drainage of inflowing groundwater, thereby bypassing the waste mass and being directly pumped from the cell via the basal drainage layer.

Capping

The final capping of the site requires the installation of a geosynthetic capping layer (LDPE membrane) with overlying soil layers. At the end of 2004, various areas of the site were completed and restored to varying degrees, but final capping and restoration of the site is being carried out during 2004 and 2005.

The standard of capping proposed would be typical of that for above water table landfills and does not appear to include any additional measures by virtue of the development of the Poole site as a hydraulically contained facility. The capping proposals will reduce infiltration and promote run-off from the cap to surface water collection facilities.

It is noted that there are no drainage measures proposed within the soil profile. This may cause ponding of infiltrating precipitation if reverse gradients occur at the geosynthetic capping layer. This in turn may lead to locally enhanced infiltration, particularly if it occurs in the vicinity of defects in the cap. Defects may be expected to occur where localised differential settlement beneath the cap is at its greatest (which in turn could potential give rise to reverse gradients).

5.4.5 Comments on Engineering and Operational Issues

The historical development of Poole Landfill has essentially been as a dilute and disperse operation. Waste has generally been landfilled in areas of the brick-pit with no sideslope and basal preparation or lining. The latest Phase has an engineered clay lining to the sideslope.

Each of the Phases reportedly has a leachate drainage layer, linked to a pumped leachate extraction point. Phases 1-3 and Phase 4 are separately drained. Leachate is pumped to the site treatment plant where it is mixed prior to treatment and discharge.

The issues relevant to the site engineering and operation are considered below.

Site Preparation and Lining

Little or no preparation of the void areas has been undertaken. This is not of major significance since there is no engineered lining to much of the site, but potential flowpaths into the site might have been disrupted by slumping and collapse of the pit slopes.

It is uncertain whether such effects will have improved or otherwise the performance of the site as a hydraulically contained landfill. Slumping may have ‘covered’ direct flowpaths into the site, particularly in Phases 1-3. Phase 4, which has been lined with clay on the sideslopes, will not necessarily have direct flowpaths into the site, unless piping failures (due to localised inflow) has compromised the integrity of the clay lining.

Leachate Drainage

Leachate drainage from the site is via single points and may be liable to disruption following mechanical breakdown.

The limited preparation of the base of the landfill may have resulted in areas where leachate will ‘pond’ and may not move toward the abstraction point. Such areas may contain a higher strength leachate than is removed from the wells.

The body of the waste contains numerous vertical drainage measures, to limit the potential for development of perched leachate occurring.

It is unclear how much ‘flushing’ of the waste occurs due to inflow of groundwater. If the flow is impeded by the lower permeability of the waste, a ‘bypassing’ of flow into the drainage layer may occur. The leachate pumped at the extraction point may in consequence be essentially groundwater entering the site. Previous reporting has indicated that the strength of leachate pumped from the site is weak in comparison with that typically produced at landfills receiving a large proportion of household wastes, therefore supporting the hypothesis above. More recent data from leachate wells installed in 2004 have indicated that leachate quality within the waste mass is much stronger, again suggesting that by-passing of the wastes is occurring.

Waste Placement
As noted, it is assumed that waste is placed in essentially horizontal lifts, with intervening cover layers. The installation of vertical drainage measures is assumed to prevent perching of leachate. This method of filling may encourage perching of leachate, but the measures installed should adequately mitigate this.

Capping
The proposed capping (and the intermediate cap already in place) will significantly reduce the potential for infiltration of surface waters. This will minimise the production of leachate through addition of surface water. The measures proposed are essentially those that would be employed on most modern landfill sites.

Comparison with Above Water Table Sites
The Poole Landfill has no particular infrastructure or other features that would not be found on modern landfill sites developed above groundwater levels, although no basal engineering has been provided due to the age of the site.

Leachate abstraction from the site relies on two main facilities, the vertical and inclined pumps. Whilst some vertical drainage towers were incorporated in Phase 4, these are not used for leachate abstraction and reliance is made on the two main pumping locations which are linked to the basal drainage systems. In view of the apparent effectiveness of this arrangement, with large quantities of dilute leachate removed, there has been no requirement to install retrofit wells. This contrasts with the Brogborough site, where large areas of the site are not served by basal drainage facilities. Additional leachate wells installed in late 2004 now provide additional facilities for leachate removal if required in the future.

The pumped extraction system, which is key to maintaining levels, does not appear to be particularly flexible and could be susceptible to breakdown. Given the pumping rate each day, which presumably reflects the inflow rate, there is the potential for rapid rises in leachate level following interruptions of pumping. This may affect the performance of the containment of leachate.

5.5 Site Studies - (3) Whitehead

5.5.1 Introduction

This section considers the landfill construction and operational issues at the site and assesses these factors against general practice for above water table landfill sites. Of particular relevance will be the current requirements and the long-term performance post closure.

Whitehead Landfill only began receiving waste in 1998, and is therefore considered to be an example of modern landfill practice. It includes a composite lining system, comprising a 1 m thick compacted clay layer overlain by a 2 mm high density polyethylene (HDPE) geosynthetic liner. The surrounding strata comprise relatively low permeability Boulder Clay.

5.5.2 Development Conditions

The landfill void has been developed as a result of clay extraction. Previous activities on the site have included tipping of colliery spoil, and later reworking/washing of the spoil for coal recovery.

The landfill is sited above the Sherwood Sandstone aquifer. The void has been formed in the low permeability superficial deposits (Boulder Clay) overlying this unit. At depth in the Boulder Clay, and in proximity to the Sherwood Sandstone, potentially higher permeability units have been identified. These include silt, sand, and sand and gravel deposits interbedded with the Boulder Clay.

The development of the site reflects current landfill practice, requiring a high standard of engineering and containment. All phases developed to date include engineered lining systems on a prepared sub-grade. Leachate drainage and extraction systems are included in all cells.

During excavation and development of an area of the void, groundwater inflows were reported. These required the formation of pumping chambers to remove collected water. Other areas were sealed with low permeability clay to prevent inflow during development.

5.5.3 Waste Management Licence Conditions

Regulatory control, in the form of a Waste Management Licence issued on 9 October 1998 (EAWML 50,009), includes reference to site engineering and operational control.

Conditions are included that require the justification of the design, specifications and methods of construction for each stage of development. Conditions do not specify the minimum standard of containment required, but this must be agreed prior to construction. Similar requirements for the leachate and landfill gas management systems and landfill capping are included. All works are to be undertaken in accordance with a previously approved Construction Quality Assurance (CQA) programme.

There is a requirement in the licence to prevent perching of leachate or disruption of gas collection. This is to be controlled by waste emplacement methods that prevent the formation of low permeability layers in the waste profile. A condition requiring ripping of intermediate cover to increase permeability in advance of further waste emplacement is noted.

5.5.4 Engineering and Operational Issues

Void Preparation

Base

The individual cells within the landfill are formed in a prepared void, with an engineered base to provide drainage to a low point in each cell.

The development of the sub-grade was undertaken entirely within Boulder Clay. This will have avoided any potentially higher permeability horizons in the superficial deposits.

There are reportedly artesian conditions in the Sherwood Sandstone beneath the site (indicated by piezometric levels at ~19 m AOD, relative to levels of ~1 m AOD to -21 m AOD for the sandstone unit). The potential for basal heave (i.e. groundwater pressures beneath the site causing uplift and disturbance in the floor of the excavation) is a function of the groundwater regime in the strata immediately beneath the site, and the thickness of *in situ* material remaining to resist the upward force.

The landfill base levels are significantly above the upper surface of the Sherwood Sandstone in much of the site. Toward the south however, the separation between the base of the landfill and the aquifer is reduced. There may be the potential for basal heave in such areas if there is insufficient mass of *in situ* material beneath the lining to resist the hydrostatic forces causing uplift beneath the site.

Inflow of groundwater was reported in Stage 4 (to the south of the site), which may be evidence of uplift and disturbance (e.g. fracturing and fissuring) of the sub-grade.

Sideslopes

The cross sections prepared for the site show an excavation profile with sideslopes of ~1:3 (v:h). It is assumed that these slopes have been designed based on a slope stability assessment for the site. 1:3 is a typical slope for the installation of composite lining systems in modern landfills. For the slope heights indicated in the sections, and the lining requirements noted, this slope should result in Factors of Safety greater than unity for stability of the lining system under residual conditions (i.e. some time after installation).

In the southern areas of the site, softening of the clays, or desiccation and fissuring of the clay due to repeated wetting/drying cycles if there are fluctuations in groundwater level, may have affected the quality of the sub-grade on slopes. This should not affect the quality of the lining installations if appropriate drainage measures are included behind the engineered clay fill (see below).

Landfill Lining

An engineered clay barrier, to a minimum of 1.0 m thickness and with a permeability no greater than 1×10^{-9} m/s has been installed to the base and sideslopes of the landfill. The construction of the lining was also subject to rigorous CQA inspection and approval. An HDPE geosynthetic liner has then been installed in intimate contact with the clay lining.

Lining Sub-grade

As noted in the Section above, there is the potential that the clay forming the basal or sideslope sub-grade in the southern part of the site has been disrupted by heave due to hydrostatic pressure. This may have increased the effective permeability of this material, and in particular introduced linear drainage pathways (through fissures/fractures developed in the clay). Without appropriate drainage measures between the sub-grade and the engineered clay lining, there is potential for water pressure to build and affect the quality of the engineered clay lining post construction. This could cause a general softening or under extreme conditions result in piping failures through the clay liner.

Lining Quality

The lining prepared for the base and sideslopes of the site has been subject to a rigorous CQA process. Combined with the relatively low permeability sub-grade (the Boulder Clay) this would be expected to provide a high level of protection to the surrounding ground.

The sub-grade was presumably properly prepared prior to construction of the lining, and its acceptance would have formed part of the CQA programme. Assuming that waste placement followed relatively shortly after construction, the compacted clay lining would not have degraded significantly (by evaporation of pore water near surface). The high groundwater levels would presumably ensure that the Boulder Clay remains saturated, preventing potential degradation caused by later desiccation (e.g. by reactions with leachate or heating as the waste degrades).

However, the high groundwater levels (associated with high hydrostatic pressures) have potential consequences for the integrity of the lining. As noted, heave may be a factor in the southern area of the landfill. This could potentially disrupt the engineered clay lining.

The hydrostatic pressures also have the potential to ‘delaminate’ the HDPE lining from the clay lining. If the force acting behind the lining (due to groundwater pressure) exceeds the confining pressure inside the landfill (from the waste mass and drainage layers), the HDPE could be forced apart from the underlying clay lining. This is most likely to occur in the early stages of filling, when there is relatively little waste in the landfill, or on the sideslopes, where the effective vertical stress exerted by the waste is less than that on the base. Any defect then occurring in the lining could allow large volumes of leachate to flow through the lining.

Leachate Management

A continuous leachate drainage blanket is provided in the base of the landfill. This comprises a 300 mm thick granular layer, containing 180 mm diameter collector pipes. The drains lead to inclined risers from which leachate is pumped for disposal.

Leachate is removed from the cells by pumping via upslope risers.

Volumes of leachate produced appear relatively low. Volumes of leachate pumped to sewer during 2003 were in the order of 100 m^3/day. During January-September 2004, the quantity was approximately 70 m^3/day, with large additional volumes recirculated within the wastes.

Waste Placement

The method of waste placement is not described in the documents reviewed, but it is assumed to be in relatively thin layers. It is not known whether these are placed horizontally or inclined. Daily or intermediate cover is noted, and a minimum thickness of 150 mm is required.

The nature of the waste deposited and the nature of layering is an important issue in the context of leachate generation, and transmission to extraction systems. Horizontal layers may encourage ‘perching’ of leachate above lower permeability horizons (either waste or cover). It is noted in the licence that these conditions must be avoided. It is not known whether any of the monitoring points included in the body of the landfill will also act as vertical drainage measures to assist in transmitting leachate to the basal drainage layer.

Inclined layers would reduce the potential for perched leachate levels (since leachate would ‘flow’ downslope), but would require adequate drainage at the margins of the void to prevent leachate collecting against the sideslope. The inclusion of a drainage blanket above the sideslopes will however prevent this occurring.

Capping

Reference is included to a low permeability clay cap in the licence. It is assumed that this has been installed in the completed phases and will be added to those phases under development. The cap is subject to CQA inspection and approval.

The standard of capping would be typical of that for above water table landfills and does not appear to include any additional measures by virtue of the development of the Whitehead site as a hydraulically contained facility. The capping proposals will reduce infiltration and promote run-off from the cap to surface water collection measures.

5.5.5 Comments on Engineering and Operational Issues

The development of the Whitehead landfill reflects modern landfill lining practice. A composite lining system is included and the installation has been subject to CQA approvals.

Leachate collection and extraction measures are included across the site, and drainage is actively promoted from the waste mass.

The issues relevant to the site engineering and operation are considered below.

Site Preparation and Lining

The void has been prepared to accept the lining system. For much of the site there is a significant thickness of Boulder Clay between the base of the landfill and the underlying Sherwood Sandstone. However, in the south of the site significant inflows of groundwater were reported during construction.

This may result have resulted from basal heave disrupting the strata, due to high piezometric levels relative to the base level of the landfill and the reduced thickness of the clay above the aquifer. This high groundwater pressure and inflow may possibly have affected the integrity of the engineered clay lining installed to the base and sideslopes in these areas.

The engineered clay lining has been installed under CQA supervision, as has the overlying geosynthetic lining. The geosynthetic lining is protected by a geotextile and a 300 mm thick leachate drainage layer.

The water pressure developing behind the lining system could require a significant thickness of waste to counter the forces that could otherwise disrupt the lining. This may be especially significant in those areas where there is relatively little separation between the base of the landfill and the underlying aquifer. The HDPE lining will be effectively impermeable to water and excess pressure behind the lining could reduce stability or adversely affect the integrity of the lining system unless countered by waste pressure inside the cell.

Leachate Drainage

Leachate is drained from the site via upslope risers connected to the basal leachate drainage layer. The continuous nature of the drainage layer, with the inclusion of pipes, will allow for effective drainage from all areas of the cells.

Capping

The capping of the site soon after final levels are achieved significantly reduces the potential for infiltration of surface waters. This minimises the production of leachate through addition of surface water. These measures are typical as those on most landfill sites, irrespective of whether they are developed sub-water table or above groundwater levels.

Comparison with Above Water Table Sites

The development of the Whitehead Landfill illustrates modern landfill practice. The lining system is essentially that which would be found in any landfill, irrespective of its location above or sub-water table.

The pumped leachate extraction system is also typical of that included in above water table landfills and should allow effective drainage of the site.

6. Significance of Diffusion

6.1 Introduction

6.1.1 Objectives and Approach

This section considers the role and importance of diffusion to contaminant migration in hydraulically contained landfills. This is done by first briefly describing the diffusion process and then identifying, through the use of simple calculations, the situations in which diffusion is likely to be important. A modelling approach for diffusion developed for the Environment Agency by Buss et al. (2004) to evaluate the likely impact of diffusion from hydraulically contained landfills has been used to assess the three landfills under consideration. The results are used to draw general conclusions regarding the role and importance of diffusion under conditions of hydraulic containment.

6.1.2 Contaminant Transport Processes

Contaminant transport within groundwater takes place by the processes of advection, dispersion and diffusion. These processes can be broadly defined as follows:

- Advection - the movement of solutes within flowing groundwater;
- Diffusion - the process by which solutes move from areas of higher concentration to areas of lower concentration along a concentration gradient.

More extensive descriptions of advection and diffusion, including analytical solutions for particular boundary conditions, can be found in many textbooks (e.g. Freeze and Cherry, 1979) and are therefore not repeated here. A recent review of the role of diffusion in hydraulically contained landfills (Buss et al., 2004) also describes the basic principles of solute transport at length.

It is generally acknowledged that all groundwater moves to some extent and therefore advection occurs in all situations, although groundwater flow rates and hence solute transport rates may be very low. Similarly, diffusion is almost ubiquitous and will occur wherever there are spatial differences in concentration within a saturated porous media, which includes most groundwater situations. Advection and diffusion can occur simultaneously in the same body of groundwater.

Both advection and diffusion should be included in the rigorous treatment of solute transport within groundwater. However, under most aquifer conditions of interest (an aquifer being a reasonably permeable groundwater body), the rate of advective transport is much greater than the rate of diffusive transport, and solute transport by diffusion is generally ignored as a simplifying assumption with little impact on accuracy.

However, diffusion can become an important solute transport mechanism in situations where the rate of advection is low, notably:

- in dual-porosity aquifers, where virtually immobile matrix porewater can act as a store of solutes but does not permit significant advection through the porespace;

- in non aquifer (aquitard) situations, where groundwater flow velocities and hence rates of advection are very low, i.e. contaminant flux may be dominated by diffusion.

The latter situation can apply to flow through engineered landfill liners, which have been constructed to minimise hydraulic conductivity and hence to minimise leachate leakage or groundwater ingress.

In the context of landfills, the solutes of interest are also groundwater contaminants and therefore the remainder of this document refers to contaminants rather than solutes.

6.1.3 Conventional Landfills

In conventional (above-the-water-table) landfills and in sub water table (but not hydraulically contained) landfills, hydraulic and concentration gradients act in the same direction, which is outwards from the landfill. Groundwater (or leachate) flow is therefore from the landfill to the surrounding strata and diffusion and advection act in the same direction and are complementary. Where low permeability engineered liners are present, then rates of flow will be small, although hydraulic gradients may be relatively large, particularly where an unsaturated zone exists beneath the liner, and diffusion may be of importance.

6.1.4 Hydraulically Contained Landfills

Under conditions of hydraulic containment, groundwater heads are greater than leachate heads. As a result, a hydraulic gradient exists from the external environment into the landfill and groundwater flow and advection are into the landfill, although in most modern situations, rates of flow and advection will be limited by the presence of engineered low permeability liners.

Diffusion acts in the direction of the concentration gradient which, for most situations and most substances, will be from higher concentrations within leachate in the landfill to lower concentrations in groundwater. Under these conditions, advection and diffusion are acting in opposing directions and the potential for outward contaminant movement will depend upon which process has the greater influence.

Concentration gradients for substances found at higher concentrations in the external environment and at low concentrations in the landfill, will be in the opposite direction to landfill contaminants. Such substances might include for example dissolved oxygen and nitrate.

Table 6.1 presents a comparison of the features of conventional and hydraulically contained landfills.

Table 6.1 Comparison of Groundwater Flow and Contaminant Transport Processes

Feature	Landfill Type	
	Conventional (above-water-table and sub water table)	**Hydraulically Contained**
Groundwater flow direction	Outwards	Inwards
Hydraulic gradient (head difference)	Can be high, particularly when the water table is below the base of landfill	Generally maintained as low as possible (but may be high in 'sub water table' landfills)
Concentration gradient	Outwards for landfill contaminants	Outwards for landfill contaminants
Advection	Outwards	Inwards
Overall contaminant movement	Outward	Depends upon balance between inward advection and outward diffusion

6.1.5 Diffusion in Hydraulic Containment Landfills

Consideration of diffusion as a contaminant transport mechanism is required as part of a Hydrogeological Risk Assessment (part of any application for a PPC Permit for a landfill) under conditions of hydraulic containment where the outward rate of movement of contaminants by diffusion is greater than the inward rate of movement due to advection. To determine where such a condition exists requires examination of the balance between advection and diffusion.

To assess this, and therefore to evaluate the net movement of contaminants, the following factors require determination:

- *The rate of advective transport*. This will depend upon the rate of groundwater flow, which can be determined from the hydraulic gradient (head difference), the hydraulic conductivity and the effective porosity (to flowing groundwater) of the landfill liner and geological barrier and the presence and design of the liner (composite or simple). In addition, contaminant attenuation properties (retardation and degradation) need to be known. Advective transport rates are also affected by mechanical dispersion, which acts to spread the contaminant longitudinally, laterally and vertically;

- *The rate of diffusive transport*. This will depend upon the concentration gradient between leachate and groundwater (calculated using the concentration difference and distance - dC/dx), the effective diffusion coefficient (see Section 6.2.1), the effective porosity (for diffusion), the attenuation properties of each contaminant and the design of the liner (simple or composite).

Composite liners typically consist of a geomembrane over a low permeability mineral layer (clay or bentonite enhanced sand (BES)). Geomembranes have an extremely low permeability and therefore flow is predominantly likely to be through tears and/or imperfections. Thus flow through the liner can be complex in its behaviour. LandSim (2004) and Buss et al. (2004) discuss the complications arising when considering composite liners in further detail.

For diffusion it will also be necessary to take into account the rate of decline in concentration of the landfill source term. As the source concentration falls, so the concentration gradient to the external environment also declines, and may reverse where peak concentrations occur outside the landfill as a result of diffusion of contaminants in early stages of landfill development. Furthermore, hydraulic gradients may also reverse or reduce - during early stages of landfill development, leachate heads are likely to be kept low, but at later stages may be relaxed (to better balance inflows) and following the cessation of active management, leachate levels may rise above groundwater levels, reversing the flow direction.

It should be noted that in major and minor aquifer situations where no low permeability liner is present, then it is likely that advection will dominate. In non-aquifer locations, such as clay pits, the characteristics of the non-engineered, in situ, low permeability strata (the geological barrier) will determine the potential for, and significance of, diffusion.

6.2 Initial Assessment of Diffusion

6.2.1 General Considerations

An initial assessment of the rate of contaminant movement due to diffusion can be undertaken by considering how the various attenuation processes and variations in free-water diffusion coefficient of different ions and compounds will influence transport rates. Chloride forms the base case, as it is not anticipated to undergo attenuation due to retardation or degradation. It is also a relatively small molecule and has a larger diffusion coefficient than most other ions (Table 6.2). Other contaminants will move at a relatively slower rate than chloride due to the effects of retardation and slower rates of diffusion due to smaller coefficients of diffusion. For contaminants that degrade, then the apparent rate of movement, all other factors being equal, will be slower.

Table 6.2 Diffusion Coefficients for a Range of Contaminants (after Buss et al., 2004)

Substance		Free Water Diffusion Coefficient m^2/s
Chloride ion	Cl^-	2.03×10^{-9}
Sulphate ion	SO_4^{2-}	1.07×10^{-9}
Cadmium	Cd^{2+}	0.594×10^{-9}
Ammonium	NH_4^+	1.96×10^{-9}
Benzene		0.7×10^{-9}
Toluene		$0.413\text{-}0.847 \times 10^{-9}$
Dichloromethane	DCM	0.138×10^{-9}
1,1,1 trichloromethane	TCA	2.37×10^{-9}
Trichloroethene	TCE	$0.439\text{-}0.700 \times 10^{-9}$
Naphthalene		$0.60\text{-}0.69 \times 10^{-9}$
Mecoprop		0.39×10^{-9}

Diffusion within and through porous media is slower than in free water due to the increased tortuosity of the diffusion pathways. For this reason Fick's Law for porous media includes an additional term, ψ, known as the diffusibility (Barker et al., 1995), which accounts for the difference between the diffusion coefficient in water and the effective diffusion coefficient in a porous media.

$$F = -D_0 \psi \frac{dC}{dx} = -D_E \frac{dC}{dx}$$

where

F = mass flux of contaminant per unit area per unit time (kg/s/m^2);

C = contaminant concentration (kg/m^3);

D_0 = diffusion coefficient in water (m^2/s);

x = distance (m);

dC/dx = concentration gradient (unitless).

The effective diffusion coefficient, D_E, in porous media can be defined as:

$$D_E = \Psi D_O$$

where

ψ = the diffusibility = $\delta n_D/\tau^2$; and

τ = the tortuosity (a measure of the flow path followed by groundwater when compared to the straight line distance between two points);

δ = the constrictivity of the pore space;

n_D = the through-diffusion porosity.

From this it is apparent that the effective diffusion coefficient is a function of both the contaminant and the porous medium and it therefore requires measurement for each medium under consideration. However, in practice the constrictivity, tortuosity and through-diffusion porosity are not separately measurable and most treatments, including that of Buss et al. (2004) lump these parameters together in the dimensionless tortuosity term.

For time-dependent diffusion an apparent diffusion coefficient, D_A is required such that:

$$\frac{dC}{dt} = D_A \frac{\partial^2 C}{\partial x^2}$$

This is Fick's Second Law. Barker et al. (1995) relate the various diffusion coefficients through:

$$D_E = \Psi D_0 = \alpha D_A$$

where

α is nR the 'rock capacity factor' or 'fictitious' porosity; and

R is the retardation coefficient, defined as

$$R = 1 + \frac{(\rho_d K_d)}{n}$$

where

K_d = partition coefficient (ml/g);

ρ_d = soil density (kg/m^3); and

n = effective porosity (to diffusion).

Some confusion may arise regarding the term effective porosity and clarification is offered in Table 6.3.

Table 6.3 Definitions of Porosity and Effective Porosity (after Buss et al., 2004)

Type of Porosity	Symbol	Definition	Relative Magnitude
Total	n_{tot}	The total void (air and water filled) space within a porous media.	
Water content	n_{wat}	Porosity measured by water saturation methods. In a clay material, the value measured will depend upon the drying condition employed in the test.	↓
Diffusion	n_{diff}	The porosity through which diffusion occurs.	↓
Advection	n_{adv}	The porosity through which advective flow takes place. This is the effective porosity for water resources and groundwater flow.	↓
Drainable	n_{drain}	Also known as the field capacity or specific yield, it is the porosity from which water can drain freely under gravity.	↓

For clays, where much of the porosity is found in the layers between clay particles, the effective porosity is likely to vary between contaminants, depending upon the charge (neutral, negative, positive) and the size of the molecule. Thus it is a property both of the medium and the solute.

From consideration of the diffusion equations, it is apparent that, in the absence of advection, the rate of diffusion is dependent upon:

- the contaminant properties (free water diffusion coefficient, partition coefficient);
- the porous medium properties (porosity, density, tortuosity, diffusivity and constrictivity);
- the concentration gradient; and
- the distance involved.

In a landfill setting, the sensitivity of diffusion to variations in the above factors varies, and is discussed below and illustrated in Table 6.4 and Table 6.5.

- Free water diffusion coefficients for compounds of interest (e.g. chloride, ammoniacal nitrogen, List I substances) vary across an order of magnitude from around 0.1 to 2.0×10^{-9} m^2/s (see Table 6.2), with higher values associated with monovalent ions (e.g. chloride) and lower values with some organic compounds such as mecoprop. Effective diffusion coefficients will vary according to the porous media (see definition of D_E previously). Reported values of tortuosity/diffusibility (Buss et al., 2004) vary across approximately an order of magnitude, from 2 to 51.

- Partition coefficients show a wide variability, that depends upon the contaminant and the porous media. For example the partition coefficients for chlorinated solvents and DDT are ~20 and >200 000 respectively.

- Porous media properties will vary between sites, but the properties of clay, which make up the bulk of low permeability layers and landfill liners are likely to show only limited variation.

- Concentration gradients will vary between sites depending upon the strength of the source term and the stage of development (landfill leachate generally declines in strength over time once filling is complete). However, diffusion will be of greatest interest (and most rapid) during the early stages of landfill development, when concentration gradients are greatest and diffusion first occurs. Where the source term is weak, even during early stages, then diffusion is unlikely to be a concern. For domestic landfill sites, the scale of variation of the initial source term between sites is likely to be less than a factor of 10 and for the compounds of interest may be smaller. Therefore this is not likely to be a significant factor in distinguishing between the importance of diffusion at a particular site. As the source term declines, so the concentration gradient will also fall.

- The distance over which diffusion occurs is a significant variable. For engineered sites within permeable formations, then this distance is likely to be limited to the thickness of the liner, whereas for sites within thick low permeability strata, the distance to a more permeable formation may be large. The distance will in turn affect the concentration gradient (in the dC/dx term). For a finite source term, the starting concentration will decline leading to a reduction in concentration gradient over time. However, for relatively thin, low permeability layers, the characteristic time for diffusion will be short and therefore the decline in the source term is relatively unimportant.

Composite Lining Systems

Complications can arise at landfills with composite lining systems which incorporate a geomembrane. These can include:

- Advective flow through holes (e.g. cuts, splits, manufacturer defects) in the geomembrane. Landfill risk assessments take into account the fact that even a high-quality geomembrane installation, including construction quality assurance and post-installation geophysical investigation for leaks, will contain a small number of holes;

- Poor contact between the geomembrane and the underlying low permeability layer. Delamination of the geomembrane may occur over parts of a landfill, this is the separation of the geomembrane from the underlying engineered mineral liner. In delaminated areas, the geomembrane may not serve any useful purpose under conditions of hydraulic containment;
- Diffusion through geomembranes. Organic List I landfill contaminants can diffuse through a HDPE geomembrane but inorganic compounds will not. In the absence of holes in the geomembrane, this may represent a significant diffusion pathway.

These complications are discussed in greater detail in Buss et al. (2004).

Table 6.4 Qualitative Assessment of the Importance of Diffusion

Thickness of Low Permeability Layer (mineral liner + geological barrier)	**Inward Groundwater Glow Velocity (V=ki/n)**		
	Low (K<10^{-11} m/s)	**Medium**	**High (K>10^{-9} m/s)**
Thin			
Medium		Diffusion of Decreasing Importance	
Thick			

Degree of Retardation	**Degradation**		
	None	**Some**	**Rapid**
None			
Some		Diffusion of Decreasing Importance	
Significant			

Table 6.5 Qualitative Assessment of Factors Affecting the Rate of Diffusion

Factor	Change	Impact on Rate of Outward Movement of Landfill Contaminants	Explanation
Source concentration C_0	↑	↑	Increased concentration gradient (dC/dx)
Layer Thickness, dx	↑	↓	Decreased concentration gradient
Free water Diffusion coefficient, D_0	↑	↓	Slower rate of molecular movement
Diffusibility / Tortuosity, ψ	↑	↓	Increased diffusion path length
Retardation, R	↑	↓	Increased travel time
Degradation half life, $t_{1/2}$	↑	↑	Decreased attenuation as half-life increases
Hydraulic conductivity of low permeability layer, K	↑	↓	Increased inward advection

6.2.2 Rates of Movement

The rate of movement of a contaminant due to diffusion is typically defined, in one dimension, as the concentration at a location and time and is given by the equation:

$$C_i(x,t) = C_0 erfc\left(\frac{x}{2(D_e t)^{0.5}}\right)$$

where

D_e = the effective diffusion coefficient;

C_i = the concentration at some time, t, and distance, x;

C_0 = the initial concentration (constant);

erfc = the complementary error function.

Retardation is incorporated as follows:

$$C_i(x,t) = C_0 erfc\left(\frac{Rx}{2(D_e tR)^{0.5}}\right)$$

Results of this equation are typically expressed as a fraction of the starting concentration (C_i/C_0), at time t. Figure 6.1 illustrates the characteristic time for diffusion of chloride for various liner thicknesses and Figure 6.2 illustrates the effect of retardation on diffusion using a range of retardation factors and effective porosities. The smaller thicknesses (up to 1 m) are representative of mineral liners in situations where there is no low permeability layer beneath the landfill. The larger thicknesses (>1 m) represent the situation where the landfill is situated within a low permeability geological environment. There is some debate regarding the effective porosity of clay and therefore a range of porosity has been considered in Figure 6.2, up to the likely total porosity of around 40%. Values of Kd have been selected to reflect (a) a poorly retarded compound - Kd of 2 l/kg; (b) a moderately retarded compound - Kd of 10 l/kg; and (c) a more strongly retarded compound - Kd of 100 l/kg. Compounds with higher Kd values are unlikely to be mobile in the low permeability environment of a landfill lining system.

The figures illustrate the importance of both distance and retardation in reducing the impact of diffusion. It is noted that these do not consider inward advection, which will act to slow the rate of diffusion. From these, it is obvious that:

- diffusion will be relatively fast (1-10's of years) for thin liners (or over short distances) and relatively slow (10's to 100's of years) for thicker liners (or large distances);
- retardation is important - strongly retarded contaminants are unlikely to breakthrough at appreciable concentrations.

Figure 6.3 considers the effect of the duration of the contaminant source. In this figure the effect of the source term lasting 50 years at a constant concentration has been evaluated using the superimposition approach (McMahon et al., 2001). It demonstrates that the difference between a constant source and a time-limited source is significant, particularly for the more retarded contaminants.

6.2.3 Time-scales

The time-scale of interest will vary, depending upon the type of landfill. For most domestic, non-hazardous waste landfills, the time-scale of interest will be related to the time for waste to stabilise. Stabilisation occurs as a result of (1) leachate extraction of contaminant mass and (2) degradation of contaminants. Ignoring degradation, waste is generally considered to stabilise at a rate which is a function of the solid:liquid partitioning coefficient, infiltration rate, waste thickness and porosity (Environment Agency, 2001) (Chapter 3 has provided discussion of waste stabilisation).

The definition of the stable condition will vary for List I and List II substances. For List I substance it is likely to require that concentrations in leachate are approaching their minimum reporting values (MRVs). For List II substances, the stable condition concentration will be where there is no further risk of pollution.

Work undertaken by Entec at Brogborough suggests that time-scales to achieve stabilisation with respect to ammonia are in the range 50 to 100 years, but for other sites considerably longer timescales have been calculated. It is noted that ammonia is not generally considered to degrade within the landfill under anaerobic conditions, and therefore stabilisation of degradable organic compounds will be considerably faster. Ammonia concentrations are anticipated to decline as a result of leachate extraction alone.

Where diffusion travel times are of a similar order to the time for stabilisation, then consideration of the rate of decline is required, as this will result in a decrease in the overall concentration gradient between leachate and the external environment. In some cases, the rate of decline may be such that there may be a gradient reversal between contaminants already released and the landfill.

6.2.4 Contaminant Mass Transfer Rates

Diffusion is a relatively slow process where there is any significant thickness between assessment points in terms of the rates of contaminant migration. As a result, it is generally unimportant in terms of mass transfer and is only likely to be of concern in particular situations and for particular (List I) substances (see following Section), whose presence in groundwater is prohibited.

6.2.5 Inward Diffusion

Just as landfill contaminants are diffusing outwards, so substances at higher concentration in the external environment are diffusing and advecting inwards. These substances may include important electron acceptors (e.g. dissolved oxygen, nitrate), which will react with the outward diffusing organic contaminants along a front, thus promoting degradation at the (generally higher) aerobic degradation rate. It may not therefore be correct to assume, as Buss et al. (2004) suggest, that the environment in the liner is anaerobic at all locations. In particular, where the setting consists of a relatively thin liner separating a permeable unit from the waste, then the supply of electron acceptors is likely to be good. However, the rate of inward diffusion will be limited by the concentration gradient, where maximum values are dictated by the external environment, and in the case of dissolved oxygen, by equilibrium considerations.

6.3 Legislation

For the current regulation of landfills, important considerations with respect to contaminant transport are:

- The fate of List I Substances. The Groundwater Regulations 1998 prohibit the discharge of List I substances at the water table. In conditions of hydraulic containment, there is no water table beneath the site and the compliance point is likely to be considered as the base of the liner (see discussion below). The presence of List I substances above the Minimum Reporting Value (MRV) at the compliance point is likely to represent contravention of these Regulations;

- The fate of List II substances. The Groundwater Regulations 1998 require that List II substances (principally ammoniacal nitrogen) do not result in pollution.

Compliance Point

For a hydraulically contained landfill, which is by definition below the water table, consideration of the significance of diffusion is an element of the risk assessment process. A key consideration in assessing the risks of List I substances reaching the 'water table' is determination of the compliance point, as it can no longer be the water table in the strict sense. The selection of the compliance point may have a significant impact on the assessment of a landfill's compliance with the Groundwater Regulations. A number of possible compliance points have been suggested, which are in part dependent upon the hydrogeological setting and in part upon the opinion of regulators. Examples are as follows:

- the base of the artificial sealing layer (where this is a clay or similar);

- the base of the geological barrier (for a composite lining system);

- the top of uppermost aquifer within the vertical sequence beneath the landfill.

This document does not offer a definitive location for the compliance point, but a pragmatic location is suggested as the base of the low permeability element of the system, as this forms an appropriate boundary condition for diffusion. For sites in clay this would be the top of an underlying aquifer, whilst for sites engineered in higher permeability settings it would be the base of the lining system.

6.4 Evaluation of Diffusion

6.4.1 Introduction

To evaluate the effect of diffusion on hydraulically contained landfills, a number of simplifying assumptions are required to form appropriate conceptual models for use with the proposed modelling approach. These assumptions are presented and justified in this Section.

The Buss et al. (2004) approach can model 3 scenarios (shown on Figure 6.4), which are as follows:

- Scenario 1: an unlined landfill excavated within a low permeability unit underlain at depth by an aquifer (e.g. a brick pit or similar);
- Scenario 2: an engineered and lined landfill located “within” an aquifer (e.g. a gravel pit);
- Scenario 3: an engineered and lined landfill excavated through an aquifer and founded in an underlying low permeability unit (e.g. a gravel pit excavated down to London Clay).

The Buss et al. (2004) model uses 2 different lower (or environmental) boundary conditions depending upon the type of contaminant being modelled, as follows:

- For List II substances the concentration at the base of the liner is set as zero and the contaminant flux, rather than the contaminant concentration, is used to calculate the impact on groundwater at the compliance point;
- For List I substances the concentration at infinity is set to zero as this allows the concentration at the base of the liner to be calculated. This approach does however, underestimate the mass flux, which can be calculated by treating the substance as List II.

The approach used here is to classify the 3 study sites under consideration in the above terms.

It is noted that the Buss et al. (2004) model uses a constant source term, which is likely to significantly overestimate the importance of diffusion where the diffusion travel time is long relative to the rate of decline of the source term.

6.4.2 Contaminants to be Modelled

The assessment of diffusion could be undertaken for a large number of contaminants. In practice, by studying a small number of typical landfill contaminants, conclusions can be drawn regarding diffusive behaviour of a large range of compounds. A list of selected contaminants for evaluation is presented in Table 6.6, and justification for their selection given. Their properties are given in Table 6.7. However, the assessment first considers the case for chloride, which as a conservatively behaving substance, is expected to breakthrough wherever diffusion outwards is faster than inward advection. If chloride fails to breakthrough at meaningful concentrations, then the assessment is not carried forward for other contaminants. The principal contaminants of interest are List I substances, for other (List II) substances, the low quantities of mass transferred are likely to undergo dilution and dispersion, which will reduce the impact of diffusion-dominated contaminant transport on groundwater.

Table 6.6 Suggested Representative Contaminants to be Modelled

Contaminant	Justification
Chloride	Present at high concentrations in most leachate but generally low concentrations in groundwater. Behaves conservatively.
Toluene	Frequently found in leachate. A List I substance. Relatively small and poorly retarded molecule.
Mecoprop	Frequently found in leachate. A List I substance. Suspected to not degrade in anaerobic conditions.
Cadmium	List I metal (does not degrade). Retarded.

Properties for the selected contaminants are listed in Table 6.7. It can be seen that the free water diffusion coefficient varies over a range of approximately one order of magnitude. It values also illustrate the higher diffusion coefficient for organic substances through a geomembrane.

Table 6.7 Contaminant Properties

Substance	Diffusion Coefficient[A]		Partition Coefficient		Half life	Decay in Sorbed Phase?
	Free water ($\times10^{-9}$)	Geo-membrane ($\times10^{-12}$)	Clay	Geo-membrane[A]	Clay	
	m^2/s	m^2/s	l/kg	l/kg	days	
Chloride	2.03	0.002-0.03	0	0.00008	0	No
Toluene	0.4-0.8	0.2-0.6	Koc = 131 foc = 1% Kd = 1.31	60-190	365-730	Not known
Cadmium	0.7[B]	Not applicable	120 (Consim at pH 6.8)	Not applicable	Not applicable	Not applicable
Mecoprop	0.39	No information	Koc=20 foc = 1% Kd = 0.2	Not known	Not applicable	Not applicable

[A]Buss et al (2004) and references therein.
[B]Rowe et al. (1997).

6.5 Case Studies

6.5.1 Introduction

The importance of diffusion at the three case study sites is considered. First a brief description of each site is presented. A scoping evaluation is then undertaken, and where appropriate, the diffusion potential is modelled using the approach of Buss et al. (2004). General conclusions are presented at the end of this section. LandSim (Environment Agency, 2001) source term concentrations for the selected contaminants have been used, using 'Most Likely' values, where these are available. For mecoprop and toluene, values have been taken (somewhat arbitrarily) as 0.10 mg/l for illustrative purposes. The source term concentrations are given in Table 6.8.

Table 6.8 Source Term Concentrations

Substance	Concentration (mg/l)
Chloride	2270[A]
Cadmium	0.01[A]
Toluene	0.10
Mecoprop	0.10

[A]Most likely concentration from LandSim.

A number of properties of in situ strata required for the diffusion calculation are not routinely measured and therefore default values have been used. These have been taken from suggested values in Buss et al. (2004) and are reproduced in Table 6.9.

Table 6.9 Default Properties for Use in Diffusion Model

Parameter	Unit	Value	Source
Average pore radius	m	0.00001	Default value
Effective porosity	unit less	0.3	Default value
Tortuosity	unit less	5	Default value
Quality of geomembrane/clay contact		Good contact where present	Assumed
Has part of the geomembrane delaminated?		Yes - where present	No information available
Area of geomembrane delaminated	m^2	500 - where present	

6.5.2 Poole Landfill

Setting

Poole Landfill has been developed within a former brick pit in Mercia Mudstone strata. It is essentially unlined, although some clay side wall liners have been installed. The landfill is only partially hydraulically contained, although there are some uncertainties regarding the site's hydrogeological setting. Large quantities of dilute leachate are pumped from the site, which is considered to be a result of groundwater ingress into the waste.

Site-specific parameters used for Poole Landfill are listed in Table 6.10 and are illustrated on schematic cross-section Figure 6.5.

Table 6.10 Conceptual Model and Landfill Construction: Poole Landfill

Parameter	Unit	Value	Source
Basal width perpendicular to groundwater flow	m	400	See site report
Basal length parallel to groundwater flow	m	400	See site report
Elevation of base of landfill	m AOD	25	See site report
Elevation of top of aquifer	m AOD	-5	See site report
Leachate head inside landfill	m AOD	as groundwater	See site report
Groundwater head outside landfill	m AOD	47	See site report
Thickness of mineral liner	m	Not applicable	No mineral liner
Hydraulic conductivity	m/s	10^{-8} to 10^{-5}	See site report
Dry bulk density	kg/m^3	1.86	Estimate based on porosity
Thickness of geomembrane	m	Not present	

Scoping Calculations

It has been noted previously that for diffusion to be an important mechanism, overall rates of groundwater movement must be slow (whatever direction they are moving in) and a cut-off was suggested at a hydraulic conductivity of 10^{-9} m/s. As a result of the absence of an engineered low permeability liner and the relatively high hydraulic conductivity (10^{-8} to 10^{-5} m/s) of the underlying strata, Poole Landfill does not meet the low-hydraulic conductivity criteria and it can be concluded that diffusion at the site does not result in a net movement of contaminants outwards, as groundwater flow and advection are both inwards at a rate in excess of the outward rate of diffusion. The evaluation is therefore not carried forward to modelling.

6.5.3 Brogborough Landfill

Setting

Brogborough Landfill is a large landfill site that has been developed within a former brick pit in the Oxford Clay. At depth beneath the clay lies the Kellaways Sand, which is classified as a minor aquifer, although it has a low permeability. As a result of this definition, it forms the principal receptor in terms of hydrogeological risk assessments. The site has been operating for many years and incorporates a number of different phases. These phases represent a range of different scenarios, whose diffusion risk could be measured separately. The principal difference between the various hydraulically contained phases, in terms of the diffusion risk, relates to the thickness of in situ clay remaining at the base. To simplify the assessment the end members of this range are considered, as follows:

- Older cells with a shallow base and a considerable remaining Oxford Clay thickness (Stage 3A and 3B) between landfilled waste and the Kellaways Sand;
- Recent deep cells (Cells 5/6) with limited remaining Oxford Clay thickness between waste and the underlying Kellaways Sand.

Other phases can be represented by these end members, or lie between them and are not, therefore explicitly modelled. Table 6.11 gives details of the site parameters required by the model and a schematic section across the site is presented as Figure 6.6.

The two areas have been modelled under existing groundwater and leachate heads and also under a condition of hydraulic containment in which leachate heads are 2 m below groundwater heads.

Model Results

Using the values of hydraulic conductivity given in Table 6.11, which are based on measured values, the model indicates that diffusion does not result in significant movement of contaminants from the landfill to the base of the geological barrier under existing conditions. Under conditions of ideal hydraulic containment (leachate heads maintained at 2 m below piezometric levels), additional contaminant mass leaves when compared to the current situation. It should be noted that, as stated earlier, where no significant impact from chloride occurs, then other contaminants were not modelled because chloride is present at high concentrations and is more mobile than other contaminants. It should also be noted that chloride concentrations in the Kellaways Sand beneath the site are elevated and therefore the concentration gradient will be significantly less than that used in the model, which assumes limited concentrations in the external environment.

For Cells 5 and 6, the calculations predict large inflows, however, these cells are reported to be dry. A calculation using a lower value of hydraulic conductivity ($<10^{-11}$ m/s) has therefore been undertaken to better match the observed low inflows.

The results of the modelling for Cells 5 and 6 (Table 6.12) indicate that the importance of diffusion is largely dependent upon the value of hydraulic conductivity used, with diffusion only becoming significant when hydraulic conductivity values are very low.

The condition of hydraulic containment aims to maximise leachate head (to 2 m below groundwater level) and so to minimise inflows. The results show that inflows under conditions of hydraulic containment are predicted to be much smaller than under existing conditions.

Cells 3A and 3B do not indicate significant breakthrough of chloride even under conditions of hydraulic containment and therefore have not been considered for other contaminants. The assessment suggests that the thicker clay beneath these cells provides sufficient protection to the underlying aquifer.

Table 6.11 Conceptual Model and Landfill Construction: Brogborough Landfill

Parameter	Unit	Value		Source
Basal width perpendicular to groundwater flow	m	1970 (total site) 620 Stage 3A&3B 715 Cells 5 & 6		Site plan
Basal length parallel to groundwater flow	m	1118 (total site) 250 Stage 3A&3B 250 Cells 5 & 6		Site plan
Elevation of base of landfill	m AOD	Stage 3A&3B Cells 5 & 6	40 - 47 26	Site report. Table 3.2 10744rr015i1
Elevation of base of aquifer	m AOD	13-32		Site report. Figure 2.3-2.5 10744rr015i1
Leachate head inside landfill	m AOD	Stage 3A&3B Cells 5 & 6	45-55 <28	Site report. Figure 3.16 10744rr015i1
Groundwater head outside landfill	m AOD	Stage 3A&3B Cells 5 & 6	53-59 54	Site report. Figure 3.6 10744rr015i1
Thickness of mineral liner (reworked Oxford Clay)	m	Stage 3A&3B Cells 5 & 6	no liner 1 m eng'd clay	Site report.
Thickness of geological barrier (Oxford Clay)	m	Stage 3A&3B Cells 5 & 6	5 2	Site report.
Hydraulic conductivity	m/s	In situ Oxford Clay 6×10^{-9} Engineered Oxford Clay 2×10^{-10} Kellaways Sand 5.6×10^{-7}		Site report. 10744rr015i1
Horizontal hydraulic gradient		0.002		Site report.
Dry bulk density	kg/m^3	1700		Estimate based on total porosity estimate of 35%
Thickness of geomembrane	m	Not present		

Table 6.12 Brogborough Results

Area	Scenario	Flow into Landfill m^3/a	Starting Conc'n (mg/l)	Time (years) to:		Peak		Comments
				Break-through	Peak	Conc'n (mg/l) at Top of Aquifer	Mass (kg) Leaving Landfill per Annum	
Chloride								
Stage 3A&B	Existing	559	2270	630	7000	2.9	0.8	Limited mass entering aquifer
	H-C		2270	900	>10000	74	11.7	Limited mass entering aquifer
Cells 5&6[A]	Existing	34000		4	13	<0.00001	<0.01	Limited mass entering aquifer
	H-C		2270	15	363	80	94	Significant mass entering aquifer
Cells 5&6[B]	Existing	1700	2270	20	750	185	171	Significant mass entering aquifer
	H-C	112	2270	20	>10000	1900	706	Significant mass entering aquifer
Cadmium								
Cells 5&6[A]	Existing		0.01	3000	8000	<0.00001	<0.01	
	H-CA		0.01	4000	4000	<0.00001	<0.01	No significant breakthrough
Cells 5&6[B]	Existing		0.01	8000	>10000	<0.00001	<0.01	
	H-C		0.01	5000	>10000	<0.00001	<0.01	
Mecoprop								
Cells 5&6[A]	Existing		0.1	10	27	<0.00001	<0.01	
	H-C		0.1	100	500	<0.00001	<0.01	No significant breakthrough
Cells 5&6[B]	Existing		0.1	100	750	<0.00001	<0.01	
	H-C		0.1	250	>10000	0.04	<0.01	Breakthrough
Toluene								
Cells 5&6[A]	Existing		0.1	40	100	<0.00001	<0.01	
	H-C		0.1	100	300	<0.00001	<0.01	No significant breakthrough
Cells 5&6[B]	Existing		0.1	100	300	<0.00001	<0.01	
	H-C		0.1	100	300	<0.00001	<0.01	

Notes:
H-C= hydraulically contained and assumes a 2 m head difference between groundwater and leachate (assumes groundwater remains at present levels).
[A]hydraulic conductivity of 10^{-10} m/s.
[B]hydraulic conductivity of 10^{-11} m/s.

6.5.4 Whitehead Landfill

Summary Description

Whitehead landfill is a former area of low lying ground into which colliery spoil has been tipped and which has subsequently been developed for landfilling. The geology consists of superficial deposits (boulder clay, sands and sands and gravels) over the Sherwood Sandstone aquifer. It has been developed as an engineered, contained landfill in a series of cells (phases), with a composite liner consisting of 1 m of engineered clay, overlain by a 2 mm geomembrane. The engineered liner has been founded on boulder clay strata, several metres above the confined sandstone aquifer. A schematic section across the site is included as Figure 6.7. Site-specific parameters for Whitehead Landfill are listed in Table 6.13. Groundwater flow direction in the Sherwood Sandstone has not been determined for this assessment.

Table 6.13 Conceptual Model and Landfill Construction: Whitehead Landfill

Parameter	Unit	Value	Source
Basal width perpendicular to groundwater flow	m	400	Site plan
Basal length parallel to groundwater flow	m	600	Site plan
Elevation of base of landfill	m AOD	10	Initial report
Elevation of base of aquifer	m AOD	-5 to -20	boulder clay minimum 4.5 m
Leachate head inside landfill	m AOD	13-20	1 m leachate head on base
Groundwater head outside landfill	m AOD	15-20	approximate range 16-20
Thickness of mineral liner	m	1.0	Initial report
Hydraulic conductivity of clay liner	m/s	10^{-9} to 10^{-10}	Maximum value required by the specification to likely hydraulic conductivity
Dry bulk density	kg/m^3	1750	Estimate
Thickness of geomembrane	m	0.002	m

The presence of a geomembrane at Whitehead Landfill requires consideration of flow and diffusion through it. Table 6.7 presents geomembrane properties used in the assessment. These have been taken from Buss et al. (2004) without modification, because no site-specific information is available.

Initial Assessment

None of the three Buss et al. (2004) scenarios represents the Whitehead landfill scenario exactly. The most appropriate conceptual model is Scenario 1 ‘clay over aquifer’ but this does not include the option to model the role of the geomembrane. Ignoring the geomembrane, the situation with respect to diffusion at Whitehead is essentially similar to Brogborough (also Scenario 1). The relatively more permeable engineered liner at Whitehead means that diffusion is likely to be relatively less important than at Brogborough as advection will be greater. For this reason, modelling of the situation at Whitehead is considered only briefly, for the case of chloride.

Results

The results of the assessment are shown in Table 6.14 and indicate that a clay liner with a hydraulic conductivity of 10^{-9} m/s (ignoring the role of the geomembrane) does not show any appreciable vulnerability of groundwater to diffusion, but decreasing the hydraulic conductivity to 10^{-10} m/s results in a substantial increase in the mass transfer of chloride. However, this increase is insufficient to result in substantial movement of the List I substances considered for Brogborough.

Table 6.14 Whitehead Landfill Results (Chloride)

Area	Scenario	Flow into Landfill m^3/a	Starting Conc'n (mg/l)	Time (years) to		Peak		Comment
				Break-through	Peak	Conc'n (mg/l) at Top of Aquifer	Mass (kg) Leaving Landfill per Annum	
Whole site	K=10^{-9} m/s	6300	2270	363	1300	<0.00001	<0.01	No significant mass entering aquifer
	K=10^{-10} m/s	630	2270	1000	>10000	87	7	Moderate mass entering aquifer

6.6 Conclusions

A scoping investigation indicates that diffusion is only likely to be of importance for hydraulically contained landfill sites where certain conditions are met, namely:

- a very low permeability engineered barrier;
- a relatively thin engineered or geological barrier;
- a limited hydraulic gradient between leachate and groundwater.

Examination of the study sites has revealed that:

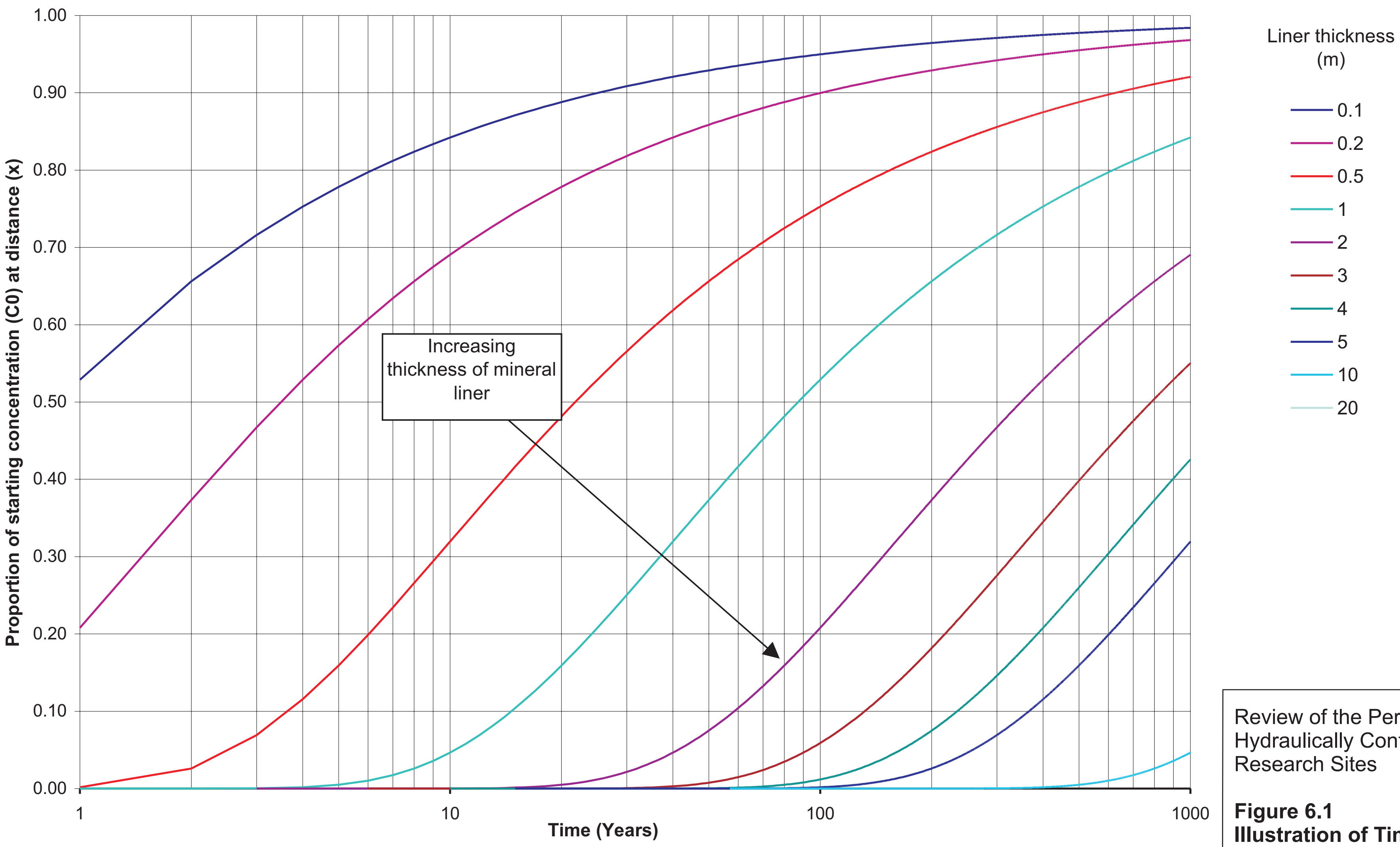
- inwards advection dominates over diffusion at Poole landfill;
- diffusion may be a consideration for the deeper/newer parts of Brogborough landfill, where the thickness of in situ and engineered clay is limited, but is unlikely to be a concern for the older parts, where there is a considerable thickness of in situ and 'cast back' clay between the landfill and the underlying Kellaways Sand;
- the thickness of clay beneath Whitehead is sufficient to prevent significant diffusion at the specified maximum hydraulic conductivity. .

General conclusions regarding the role of diffusion in hydraulically contained landfills can also be drawn, as follows:

- Where scoping calculations identify that diffusion is a potential problem, then leachate management strategies can be adopted that can limit the impact by encouraging advection, i.e. by keeping leachate levels low compared to external piezometric levels in areas of the site where leachate concentrations are high;
- care is required in selecting parameters, particularly hydraulic conductivity. Typically in risk assessments it is assumed that higher values are more pessimistic, but for diffusion, lower values present a higher risk.

It is noted that there appears to be little or no field data on diffusion from landfills to support these conclusions, or those of Buss et al. (2004).

The Buss et al. (2004) approach is likely to be overly pessimistic because it does not consider the effect of a declining source term. It is also limited to the three scenarios provided, although each can be used to consider individual components. Of the three sites examined here, one (Whitehead) presented a situation (geomembrane over clay over aquifer) that is not directly covered by the model. This restriction can potentially limit the usefulness of the model application to real sites.



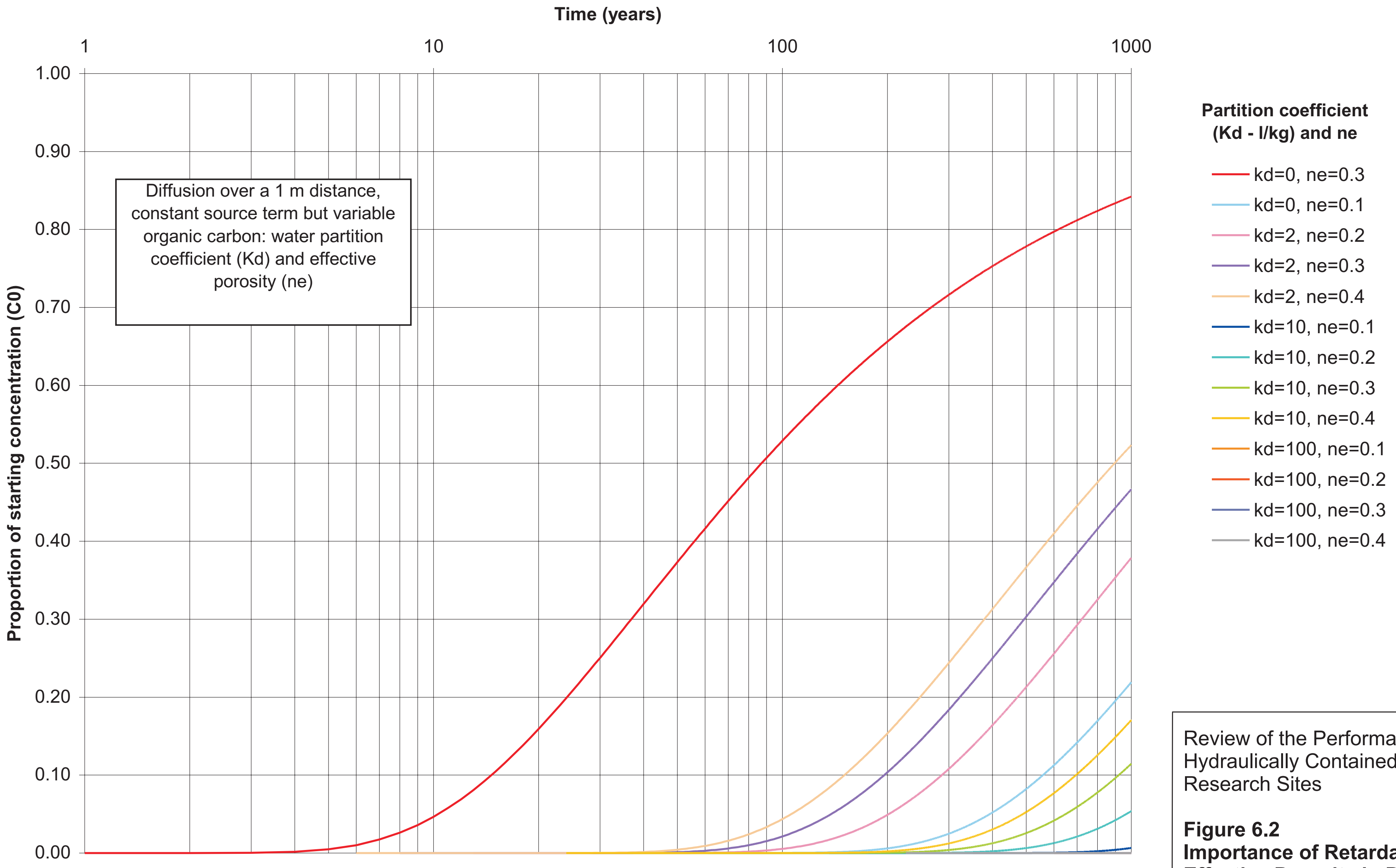
Review of the Performance of Hydraulically Contained Landfills: Research Sites

Figure 6.1
Illustration of Time to Diffuse Across Liners of Different Thicknesses

Hm-250/8465/calcs/diffusion_calcs.xls unretardedvsdistance

November 2005
10744-S35.cdr parkj

Entec



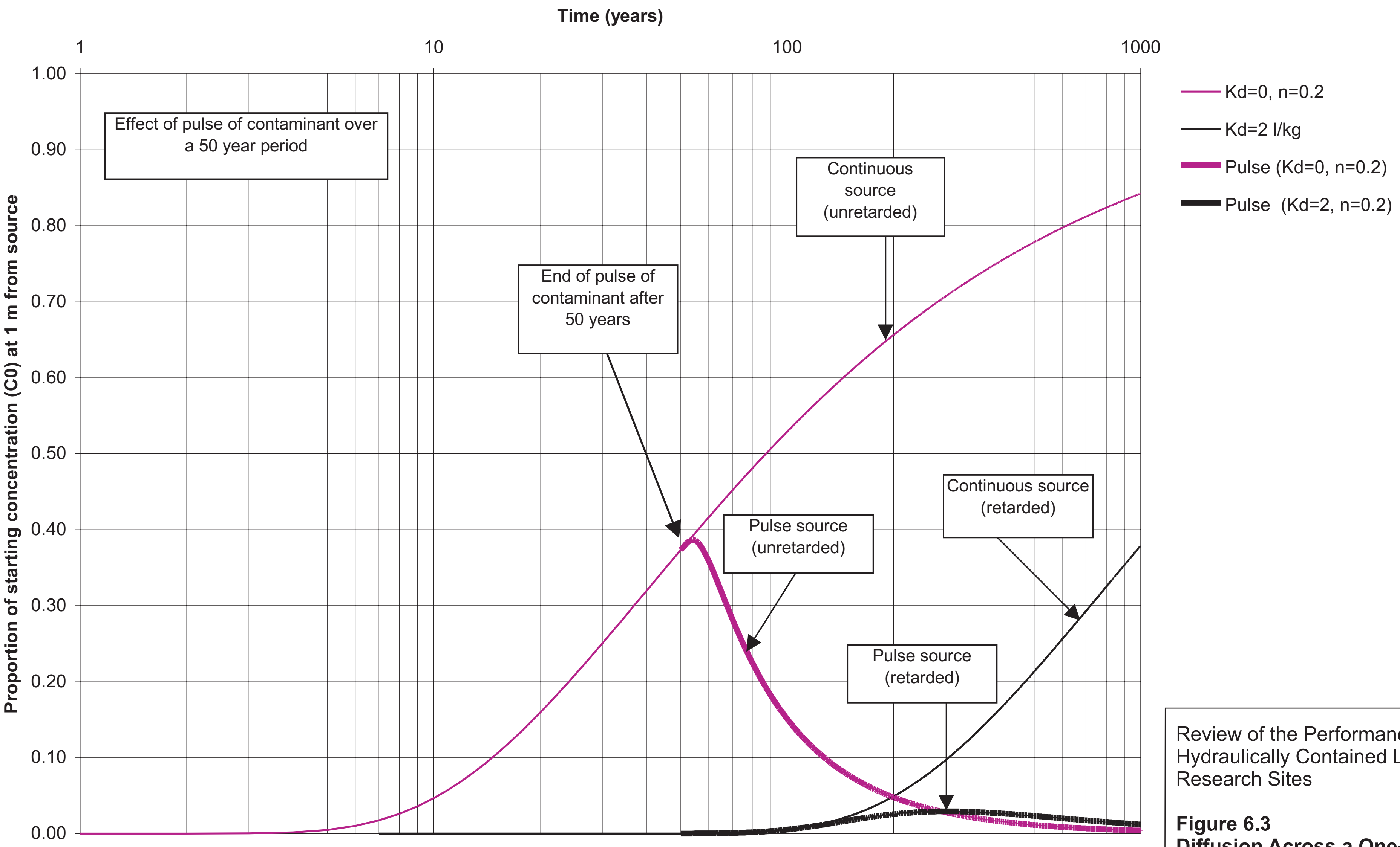
Review of the Performance of Hydraulically Contained Landfills: Research Sites

Figure 6.2
Importance of Retardation and Effective Porosity in Diffusion Calculations

November 2005
10744-S36.cdr parkj

Entec

Hm-250/8465/calcs/diffusion_calcs.xls uvariable kd+ne



Review of the Performance of Hydraulically Contained Landfills: Research Sites

Figure 6.3
Diffusion Across a One-metre Liner for a 50-Year Pulse

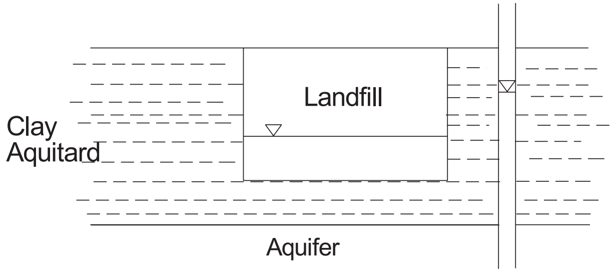
November 2005
10744-S37.cdr parkj

Entec

Hm-250/8465/calcs/diffusion_calcs.xls superimp-1

SELECT LANDFILL CONSTRUCTION SCENARIO

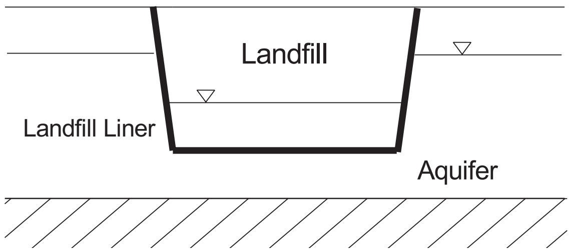
Scenario 1



The landfill is constructed in a clay pit, underlain by a confined aquifer. Water and contaminant fluxes occur across the bottom of the landfill only.

Select Scenario 1

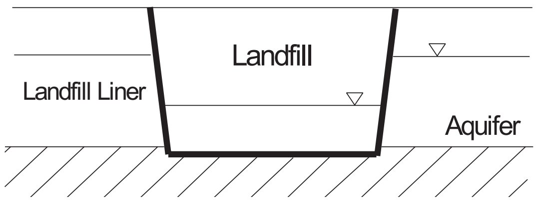
Scenario 2



The landfill is lined and located in a permeable formation a finite distance above an impermeable layer. The water and contaminant fluxes can occur through the base and sides of the landfill.

Select Scenario 2

Scenario 3

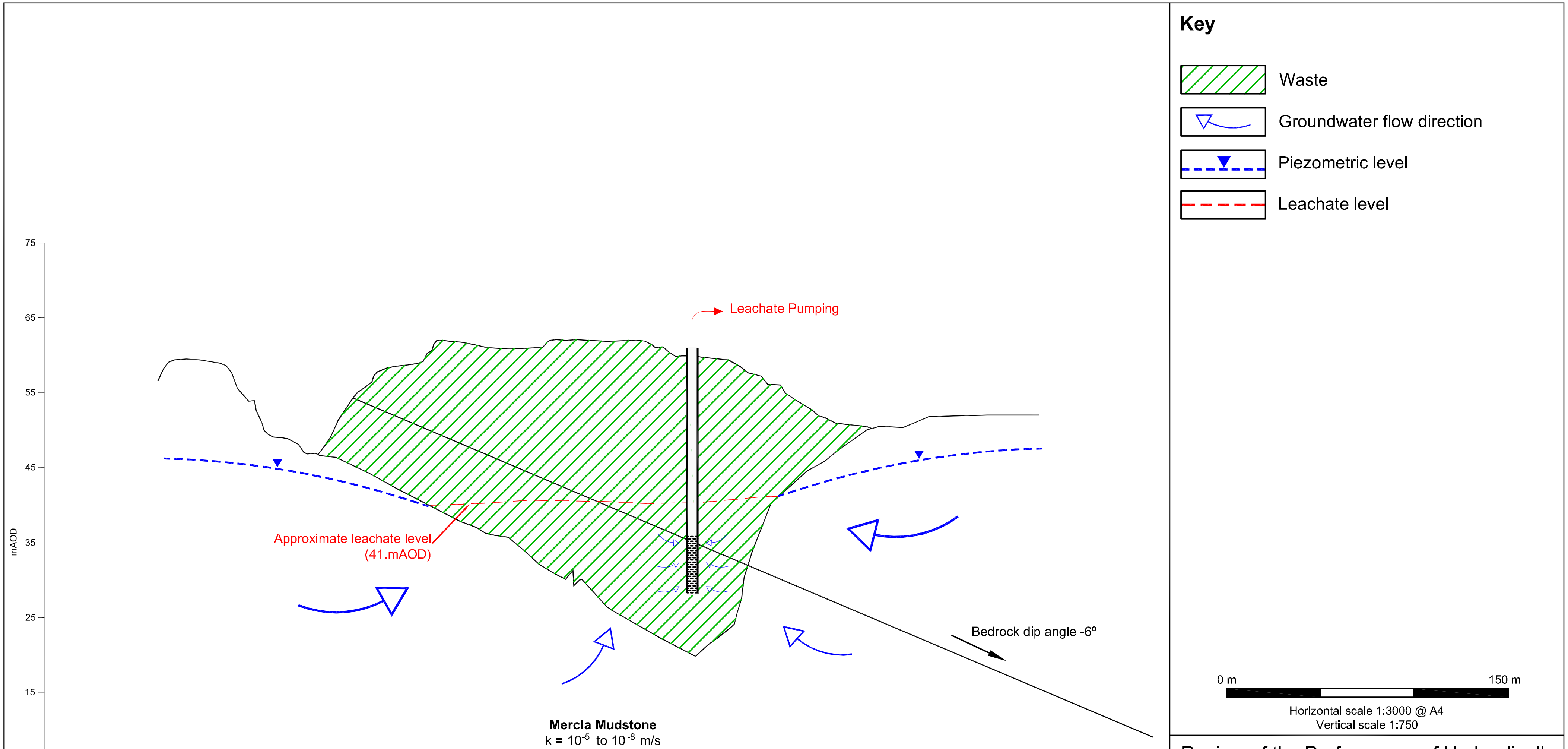


The landfill is lined and located in a permeable formation a finite distance below an impermeable layer. The water and contaminant fluxes can occur through the sides of the landfill only.

Select Scenario 3

Review of the Performance of Hydraulically Contained Landfills: Research Sites

Figure 6.4
Hydraulic Containment Scenarios (from Buss et al, 2004)

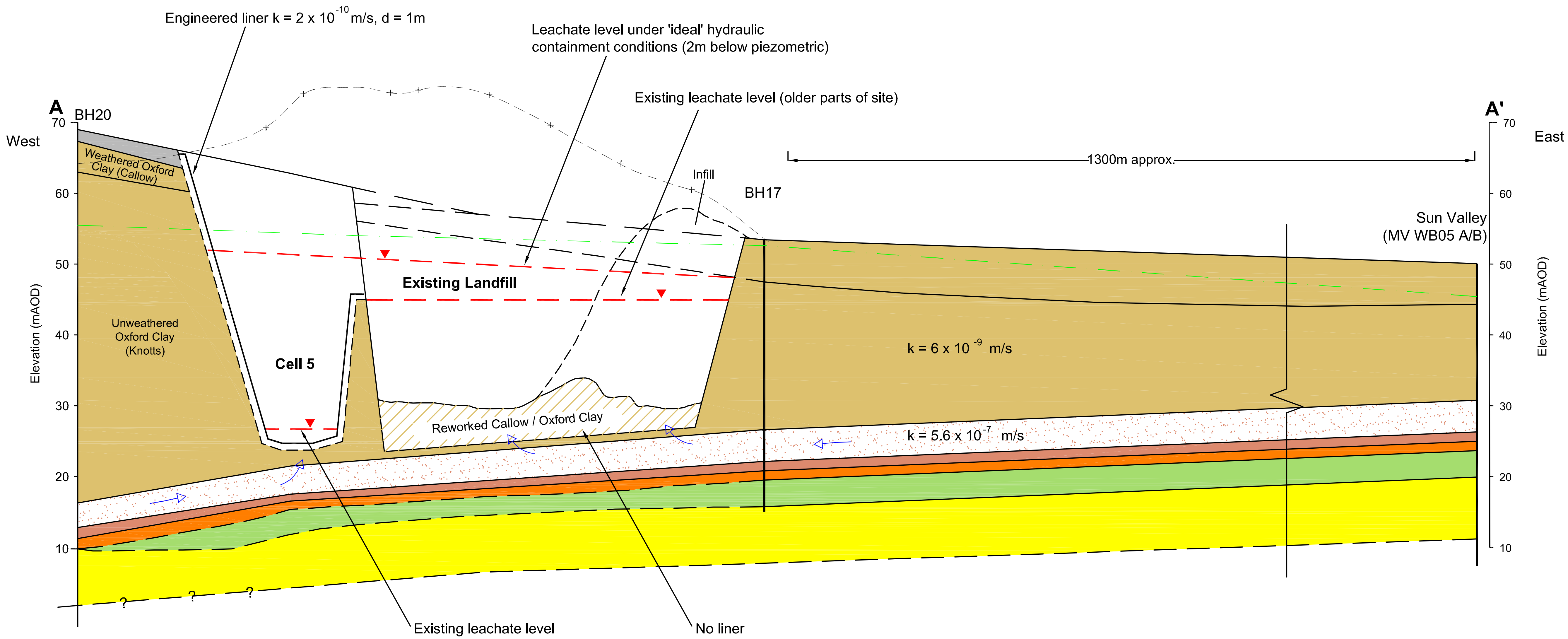


Review of the Performance of Hydraulically Contained Landfills: Research Sites

Figure 6.5
Schematic Cross Section through Poole Landfill

November 2005
10744-S39.dwg xgeort

Entec



Key

- Proposed post settlement contours
- Piezometric level in Kellaways Sands (July 2003)
- Made ground/ superficial
- Oxford clay reworked* ('insitu' geological barrier)
- Oxford clay formation* ('insitu' geological barrier)
- Kellaways sand (potential receptor)
- Kellaways clay*
- Cornbrash formation
- Blisworth clay formation*
- Blisworth limestone formation (potential receptor)
- Groundwater flow direction
- Leachate level

Note:
* Strata with significant attenuation capacity.

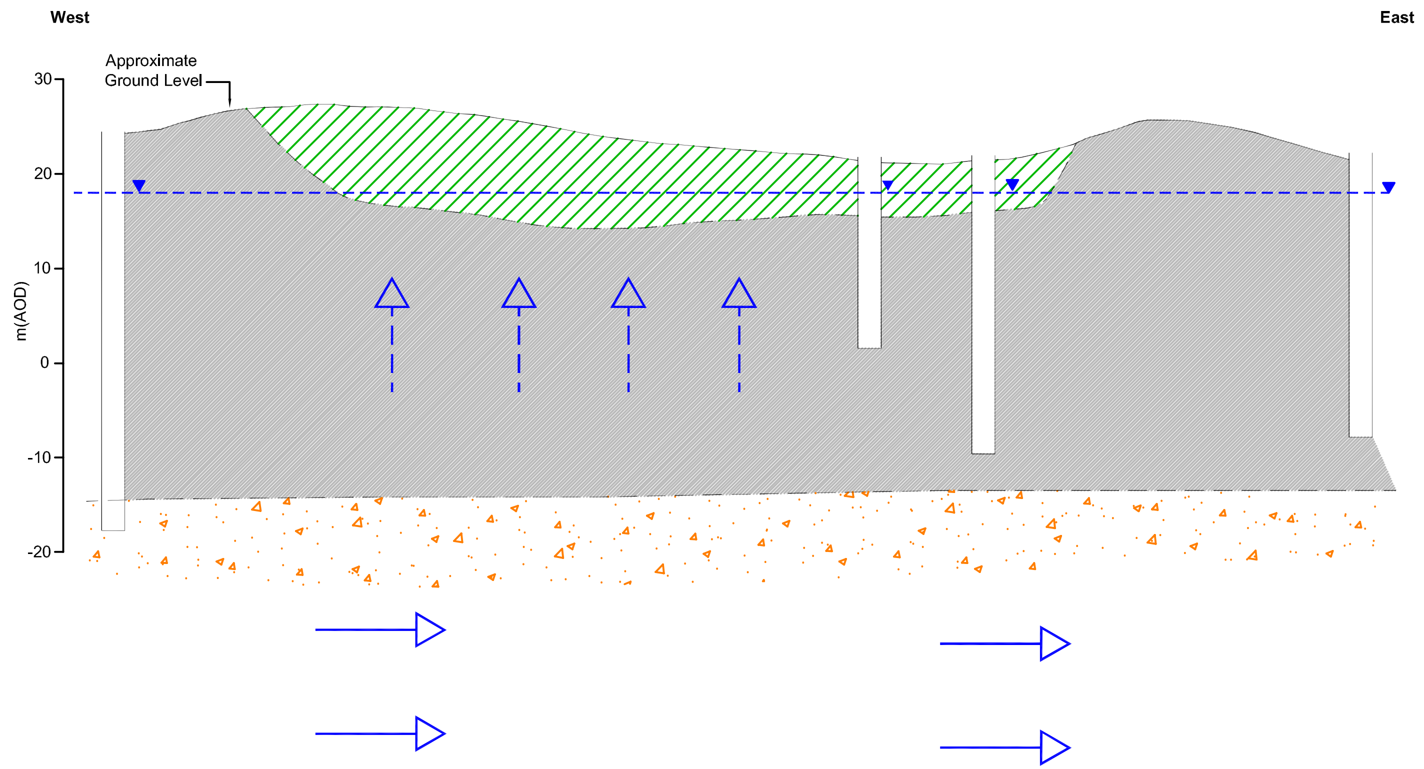
0 m — 250 m
Horizontal Scale 1:6500 @ A3
Vertical Scale as shown

Review of the Performance of Hydraulically Contained Landfills: Research Sites

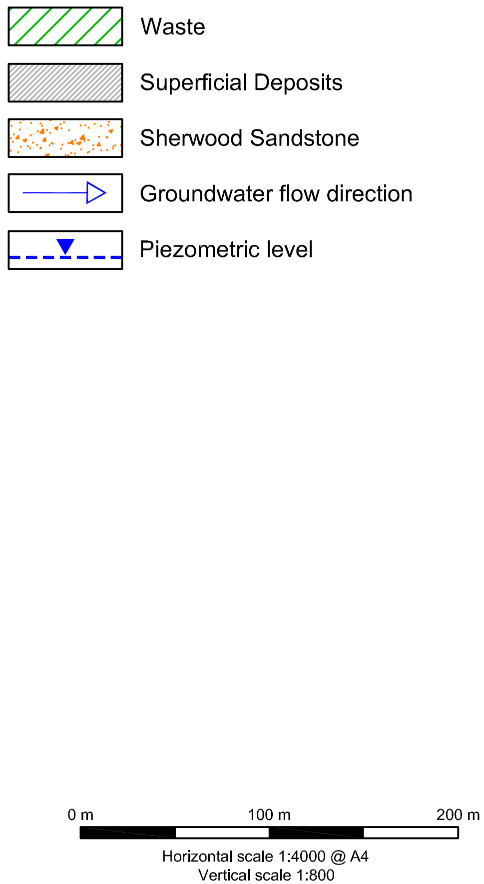
Figure 6.6
Schematic Cross Section through Brogborough Landfill Site

November 2005
10744-S40.dwg parkj

Entec



Key



Review of the Performance of Hydraulically Contained Landfills: Research Sites

Figure 6.7
Schematic Cross Section through Whitehead Landfill Site

November 2005
10744-S41.dwg xgeort

Entec

7. Overview and Conclusions

7.1 Overview

This project has reviewed many of the key issues associated with the development and operation of hydraulically contained landfill sites and where possible has used the three study sites offered by operators to illustrate these issues.

Hydraulically contained, or sub-water table landfills, have been identified more as "special cases" in the UK in recent years. The implementation of European Directives into UK legislation and associated Environment Agency guidance in some cases refer to such sites. For example, the Environment Agency's position statement on the location of landfills indicates that they would object to landfilling below the water table in any strata where the groundwater provides an important contribution to river flow or other sensitive surface waters. While there may be certain circumstances when a precautionary approach to the development of such sites is appropriate, it is our view that in many ways, such sites should be treated no differently to other "above water table" landfills. Support for such an approach is provided in other Environment Agency guidance, which, in relation to the potential for entry of groundwater into a landfill, advocates determination of the degree of risk on a site-specific basis, considering:

- the geotechnical stability of the lining system, wastes and underlying geological strata;
- the efficacy of the leachate collection system;
- the effectiveness of any leachate control systems; and
- the ability to maintain leachate and groundwater management in the long term.

Each existing or proposed landfill site should be assessed on its own merits, or lack of them, once regulatory constraints have been addressed.

The study sites used in this project cover a range of hydrogeological environments and site histories, and have provided illustration of evolving engineered landfill development in the UK. Key characteristics for the sites, which show hydraulic containment in different forms, are as follows:

- Poole Landfill: Waste disposal began in the 1960s, with the bulk of waste disposal dating from 1974. No basal containment engineering was carried out, but leachate drainage facilities were incorporated. The landfill was permitted on the basis of its location within predominantly low permeability strata (Mercia Mudstone), although there are high permeability horizons within these. Hydraulic containment is provided by the pumping of large volumes of leachate from the site to maintain leachate levels below groundwater levels in the surrounding mudstone.
- Brogborough Landfill: Waste disposal began in 1983, with wastes initially deposited into worked out clay pits with no preparatory containment engineering or leachate drainage facilities. The low permeability Oxford Clay provides natural

containment. Landfill operations and cell construction standards have evolved over time, and more recent areas have incorporated engineered clay basal and sidewall lining and leachate drainage blankets. Hydraulic containment is to be achieved by the maintenance of leachate levels 2 m below the piezometric level in the underlying Kellaways Sand aquifer.

- Whitehead Landfill: The most recent of the three sites, with waste disposal beginning in 1998. The landfill has been developed in low permeability drift strata which overlie and confine groundwater in Triassic sandstones. Modern landfill engineering measures have been incorporated, with discrete cells lined with a composite liner of engineered clay and geomembrane overlain by a leachate drainage blanket. Leachate drainage and removal and the maintenance of limited leachate heads beneath piezometric levels in the sandstone, mean that most of the site is hydraulically contained.

No new landfill development similar to that at Poole would be permitted at the current time without much more extensive containment engineering, in compliance with the requirements of the Landfill (England and Wales) Regulations 2002. However, despite the absence of a comprehensive, low permeability liner, monitoring data have shown that there is little or no evidence of groundwater contamination in the vicinity of the site. This situation has been and will continue to be achieved by the pumping and discharge of very large volumes of dilute leachate from the site. The large quantities involved result from significant quantities of groundwater entering the landfill, in the absence of low permeability lining, which add to the volumes of leachate that are generated by rainwater infiltration. Thus whilst there are no significant adverse impacts on local water resources, long term leachate management of this type, with associated operational costs and secondary impacts on the receptor of the pumped effluent, is not desirable.

Information from the Poole site has also emphasised the importance of accurate and appropriate monitoring data collection at landfill sites. Leachate quality is likely to be a key factor in the process of licence or permit surrender and representative data are required for the assessment of the status of a site. Until additional leachate monitoring wells were installed at Poole in late 2004, it was possible to infer from the data for the quality of leachate discharged from the site that the wastes were in an advanced state of stabilisation. Leachate samples from the new wells indicated that the quality of the pumped leachate was misleading, affected by large volumes of groundwater ingress which diluted contaminant concentrations. Analyses of samples from the new wells indicated that leachate quality within the wastes was much stronger and more representative of that produced from above water table sites. Flushing of the wastes by large volumes of water, as could be postulated by the leachate volumes and quality measured, was not occurring. This means the site could face pumping of large volumes of dilute leachate for as long as it would take an above water table site of the same waste thickness to stabilise.

Brogborough Landfill represents a stark contrast to the Poole site. Developed in naturally low permeability strata, and underlain by groundwater of limited resource value, this and several other landfills in this area of the country can be considered to be sited in suitable, low risk locations for landfill. Volumes and rates of groundwater flow are low, and whilst there is leachate extraction from the site, quantities are proportionately much less than at Poole. Because of the hydrogeological conditions, the development of the landfill, incorporating liners and drainage blankets, is not made any more difficult by the fact that the base of the site is below local piezometric levels. For the same reasons, the quantities of groundwater ingress are small (no greater than cap infiltration) and manageable in conjunction with the leachate produced from rainfall infiltration to the wastes. Consequently the volumes of water available for flushing of the wastes to encourage stabilisation, or to promote increased landfill gas production, are limited.

It is unlikely that stability was a major issue considered at the planning stages for the Poole and Brogborough sites. As a result of occurrences at other sites, where disruption of liners has allowed groundwater ingress and caused leachate management problems, greater attention is now given to the potential for instability at sub-water table sites. A stability risk assessment is required as part of the PPC Permit submission for all landfill sites. At Whitehead landfill, a Boulder Clay thickness of approximately 20 m separates the landfill from the underlying confined sandstone aquifer. Despite this, large volumes of groundwater entered the southern part of the site during development. Works were carried out to address the problem and the landfill was constructed as planned, however, this occurrence illustrates the potential difficulties of such sites, which at Whitehead could have had more serious consequences if it had happened during the operational phase of the site. In extreme circumstances, groundwater upwelling subsequent to liner construction and waste disposal could result in rupture of the liner, groundwater ingress to the leachate drainage blanket, and additional and more onerous long term leachate management requirements.

7.2 Conclusions

Conclusions from previous reports (Entec, 2003a, 2003b and 2004) on the three sites have been summarised in Section 2 of this report and are not repeated again here. In terms of the work reported in this document, the key findings from the studies are discussed in the individual chapters of the report, but the conclusions drawn are summarised in the following sections.

7.2.1 Waste Stabilisation

Indicators that have been used in determining the evidence for waste stabilisation and increased rates of decomposition are:

- Elevated leachate temperatures - temperatures above 35ºC indicate active decomposition of the waste and may be associated with a large saturated thickness of waste and a large thickness of overlying unsaturated waste. Temperatures can be affected locally by groundwater ingress;
- Timing of the onset of methanogenesis, measured through COD and pH determinations, since this may indicate accelerated decomposition and hence waste stabilisation;
- Distribution of leachate quality parameter concentrations, which may correlate with elevated temperatures and thus indicate areas of enhanced waste stabilisation.

The above were considered at each of the three study sites, although the quantity of available data was variable. For each of the three research sites, the assessment of the data has shown:

Brogborough - whilst there are uncertainties in the analysis, there is evidence that improvement in leachate quality in wells which are sub-water table/hydraulically contained is quicker than in wells with the same rainfall infiltration and liquid:solid ratio that monitor leachate in above the water table parts of the site. Hence likely groundwater ingress appears to improve the leachate quality (increase the liquid:solid ratio) compared to that which receives only rainfall infiltration.

Poole - leachate samples from retrofit wells into the waste have indicated much higher concentrations of ammoniacal nitrogen (and other contaminants) compared with those measured in leachate pumped from the different phases of the site. This is consistent with the findings of a water balance for the site, which indicated that the pumped discharges contain a significant component of groundwater. Leachate quality is generally poorer in Phase 4 than in the older phases of the site and appears to be related to the liquid:solid ratios calculated using rainfall/cap infiltration to the waste rather than incorporating groundwater inputs. Flushing of contaminants from the wastes and waste stabilisation appear to be related to rainfall infiltration and not to the much larger volumes of groundwater ingress, which dilute the leachate.

Whitehead - there are limited data available for this most recent of the three sites and no conclusions can be drawn. However, in view of the site setting and construction, it is anticipated that the site is likely to perform in a similar way to an above water table site.

7.2.2 Landfill Gas

A large amount of information has been published which discusses the mechanisms by which landfill gas is generated. Generic factors affecting the production of landfill gas have been identified and discussed in the report, as a basis for considering the significance of hydraulic containment on these processes.

The factors which are likely to be of most significance in affecting landfill gas production in hydraulically contained landfill sites have been identified as:

- Moisture content;
- pH;
- temperature;
- nutrients; and
- operational factors.

No data quantifying the generation of landfill gas in different areas of the three research sites were available, hence a broad assessment and comparison between each of the sites was carried out. This was done on the basis that there was evidence for groundwater ingress to a large proportion of the wastes at Brogborough compared with Poole and Whitehead, where basal leachate drainage and collection means that a smaller proportion of the waste is affected.

Using available information relating to power outputs from gas generated at each of the sites, the number of flares and their approximate capacities, and estimated waste volumes, the total gas flow from each site was estimated. Whilst there are a number of uncertainties in the estimates, the information suggests that the three sites have similar gas generation and collection rates per unit waste volume, and hence there is little evidence for enhancement of, or reduction in gas production as a result of higher moisture content through groundwater ingress.

Additional data analysis has been carried out, using gas composition, liquid:solid ratios, leachate temperature and leachate quality data. The data suggest that gas with a higher ratio of CH_4 to CO_2 is being generated by wastes in the sub-water table area of the Brogborough site compared with above water table parts of the site. This relationship is also seen at Whitehead, but there is inadequate data for Poole and less distinction between sub-water table and above water table areas of this site for this statement to be made.

Comparison has also been made between CH_4 : CO_2 ratios and temperatures within the wastes. The ratios appear to increase with decreasing temperatures at Brogborough, with the highest temperatures measured in the deeper parts of the site (i.e. those parts furthest sub-water table). Hence temperature does not appear to be the key factor in producing higher CH_4 : CO_2 ratios in the sub-water table areas of the Brogborough or Whitehead sites.

Whilst the study provides only a limited view of the affect of sub-water table conditions on landfill gas concentrations, to some extent as a result of the limited data available, the studies suggest that groundwater ingress could lead to higher CH_4 : CO_2 ratios in the landfill gas produced. In view of the similar gas generation rates estimated for the three study sites, it is possible to conclude that groundwater ingress into wastes in sub-water table sites can produce similar amounts of gas, but with a higher methane content than gas derived from typically drier, above water table sites.

7.2.3 Engineering

Requirements for landfill engineering are now significantly more prescribed than in the past, principally to meet the requirements of the Landfill (England and Wales) Regulations 2002 and the Landfill (Scotland) Regulations 2003. A geological barrier, artificial sealing layer and a leachate drainage layer are required at non-hazardous and hazardous landfill sites, unless some of these requirements can be reduced on the basis of risk assessment. These "modern" requirements are reflected in the containment engineering and leachate drainage measures used at Whitehead landfill, but less so at Poole and Brogborough.

Conclusions drawn in relation to engineering for each of the study sites are as follows.

Brogborough - the development of this site illustrates the changing approach to landfill practice with time. The site's historical development has used the natural containment provided by the Oxford Clay, with no sideslope or basal preparation or lining. Later phases have been engineered with prepared sub-grade and compacted clay linings, no different to what would be required for an above water table landfill. Similarly, only later phases of the site have a leachate drainage layer installed at the base.

The later phases have been developed to greater depths, with the result that the thickness of the underlying in situ clay is reduced and the landfill base is at greater depth beneath the local piezometric level in the Kellaways Sand. This means that as a consequence of groundwater rebound, there is an increased risk of basal heave disrupting the strata and possibly the lining installed to the base and sideslopes in these areas.

Retrofit wells are used for leachate abstraction over large areas of the site, whilst more recent areas are equipped with a drainage blanket. Again, these measures are no different to those that are typically required at above water table sites.

Poole - Limited containment engineering was carried out at the site prior to landfilling. This is consistent with the age of the site, more than 30 years old, and reflects the view of the time that the natural strata (Mercia Mudstone) provided protection to local water resources as a result of its predominantly low permeability. However, the site illustrates the potential problems associated with such strata, which are interspersed with zones of higher permeability.

With limited engineering to prevent it, large quantities of groundwater enter the landfill. The studies have shown that the leachate drainage facilities are effective in transmitting these volumes to the abstraction wells and that most of the groundwater ingress by-passes the waste mass. Had engineering requirements been different at the time the site was developed (for example, as required under current regulations), detailed assessment of the stability of the basal and sidewall liners would have been required taking into account the pressures exerted by groundwater in the surrounding strata. Where compacted clay linings are used in strata with high groundwater levels, the lining may become fractured or fissured due to high groundwater pressure, potentially providing pathways into the landfill body. The same conditions can result in delamination of a composite liner (typically compacted clay and an overlying geosynthetic sealing layer) following disruption to the clay component. Were the brickworks at Poole to be developed today, dewatering may also be required to prevent excessive water pressures developing, at least until lining was completed and sufficient waste has been deposited to ensure that forces are balanced.

Whitehead - this site began receiving wastes in 1998 and provides an example of modern landfill practice. It includes what can be considered a standard composite lining system, used widely for above water table and sub-water table landfills, comprising a 1 m thick compacted clay layer overlain by a 2 mm high density polyethylene (HDPE) geosynthetic layer. Additional natural containment is provided by low permeability superficial deposits (Boulder Clay). Leachate drainage and extraction systems are included in all cells, again typical of above water table and sub-water table sites.

In the south of the site, significant inflows of groundwater occurred during early stages of construction. This may have been caused by basal heave disrupting the strata, due to high piezometric levels (associated with the underlying sandstone aquifer) and a reduced thickness of clay above the aquifer. Remedial works were carried out to prevent this ingress prior to construction of the landfill liner in this area. This has provided further illustration of the potential risks associated with external pressures acting on engineered liners. Had the groundwater ingress not been identified and addressed at an early stage, there was the possibility of later disruption of the liner and the consequential impact on leachate management as a result of enhanced water entry to the site.

7.2.4 Diffusion

Under most aquifer conditions, the rate of advective transport is much greater than the rate of diffusive transport, and for a long time, solute transport by diffusion has generally been ignored as a simplifying assumption with little impact on the accuracy of risk and impact assessments. The Environment Agency has published guidance and a methodology for evaluation of the likely impact from diffusion on hydraulically contained landfills and this has been used to assess the three research sites used in this project.

Diffusion can become an important solute transport mechanism in situations where the rate of advection is low. This can be the case with flow through engineered low permeability landfill liners, which are designed to minimise leachate leakage or groundwater ingress, as required by current Regulations.

Under conditions of hydraulic containment, a hydraulic gradient exists from the external environment into the landfill, and consequently groundwater flow and advection are into the landfill. Diffusion acts in the direction of the concentration gradient, that is, usually from the higher concentrations within leachate in the landfill, to lower concentrations in groundwater.

Hence in these circumstances, advection and diffusion act in opposing directions, and the potential for an impact “outside” the landfill will depend on the magnitude of each of the processes and which of them is dominant

The site specific assessments of the research sites indicated that:

- Inwards advection dominates diffusion at Poole;
- Diffusion may be a consideration for the deeper, more recent parts of Brogborough landfill site, where the thickness of the underlying in situ and engineered clay is limited, but only when assuming a 2 m degree of hydraulic containment and very low (10^{-11} m/s) hydraulic conductivities. Diffusion is unlikely to be important under current conditions in these areas. Diffusion is unlikely to be a concern for the older parts of the site, which are underlain by a considerable thickness of clay between the wastes and the underlying aquifer;
- The thickness of the clay strata beneath the Whitehead site is sufficient to prevent significant diffusion at the specified maximum hydraulic conductivity.

Diffusion is only likely to be of importance for hydraulically contained sites where:

- The engineered barrier is of very low permeability;
- The engineered or geological barrier is relatively thin; and
- There is a limited hydraulic gradient between leachate and groundwater.

In nearly all circumstances, a low hydraulic conductivity engineered barrier is considered desirable (and is required in current Regulations for hazardous and non-hazardous landfills), since it will generally reduce the risk of groundwater contamination. In the assessment of risk from diffusion, lower values of hydraulic conductivity represent a higher risk.

8. References

1. Barker, J A, Kinniburgh, D G & MacDonald, D M J (1995) *Groundwater Modelling and Modelling methodology: A review*. R & D report 295/20/A, National Rivers Authority, Bristol.

2. Barlaz, M A, Ham, R K & Schaefer, D M (1989), *Inhibition of Methane Formation From Municipal Refuse in Laboratory Scale Lysimeters*, Applied Biochemistry and Biotechnology, 20 p. 181.

3. Beavan, R P, Barker, & Hudson, (2003), *Description of a Tracer Test Through Waste and Application of a Double Porosity Model.* Proceedings Ninth International Waste Management and Landfill Symposium, Sardinia, 2003.

4. British Geological Survey (1989). *Hydrochemistry of Porewaters from Jurassic Oxford Clay, Kellaways Beds, Upper Estuarine and Upper Lias Formations of the Elstow Site, Bedfordshire.* B G S Report WE/89/28, Ross et al.

5. Buss, S R, Herbert, A W, Green, K M & Atkinson, C (2004). *Contaminant Fluxes from Hydraulic Containment Landfills: a Review.* Environment Agency Science Report No. SC0310/SR.

6. Department of the Environment (1995?). *Hydrological Study of a Landfill Cap at Brogborough*. DoE contract reference No PECD 7/10/252. Report prepared by P B Leeds-Harrison, S F Tyrrel & K S Harrison of Silsoe College, Cranfield University and M J Lowe of Shanks & McEwan (now WRG). Report undated, but has a reference for 1995 as in press.

7. Department of the Environment (1995). *Waste Management Paper 26B, Landfill Design, Construction and Operational Practice.*

8. Entec UK Ltd (2001), *Review of the Performance of Hydraulically Contained Landfills: Proposal for Phase 2 - Research Sites* Report Ref 06161\docs\n001.

9. Entec UK Ltd (2002), *Brogborough Landfill Site: Groundwater Assessment*. Entec Report No 07673RR004i2, April 2002.

10. Entec UK Ltd (2003), *Brogborough landfill Site: Conceptual Model and Hydrogeological Risk Assessment (part of PPC Permit Application).* Report Ref 07673RR004i6, September 2003.

11. Entec UK Ltd (2004), *Review of the Performance of Hydraulically Contained Landfill Sites,: Geology, Hydrogeology and Level of Hydraulic Containment at Brogborough Landfill Site*. Report Ref 10744/RR015i1.

12. Entec UK Ltd (2003a), *Review of the Performance of Hydraulically Contained Landfill Sites,: Geology, Hydrogeology and Level of Hydraulic Containment at Poole Landfill Site*. Report Ref 08465RR036i1.

13. Entec UK Ltd (2003b), *Review of the Performance of Hydraulically Contained Landfill Sites,: Geology, Hydrogeology and Level of Hydraulic Containment at Whitehead Landfill Site*. Report Ref 08975RR054i1.

14. Entec UK Ltd (2005), *A Review of the Factors Affecting the Production and Composition of Landfill Gas at Hydraulically Contained Landfill Sites*.

15. Environment Agency (2001), *LandSim Manual Release 2.0.* Prepared for the Environment Agency by Golder Associates.

16. Environment Agency (2001), *Pollution Inventory Discharges to Sewer or Surface Waters from Landfill Leachates.* Report of contract: REGCON 70, May 2001.

17. Environment Agency (2001), *Investigation of the Composition, Emissions and Effects of Trace Components in Landfill Gas*, R & D Publication P1-348, pp. 3-4.

18. Environment Agency (2002a), *Landfill Engineering: Leachate Drainage, Collection and Extraction Services*. R & D Technical Report P1-397/TR ISBN 1 857059 921, September 2002.

19. Environment Agency (2002b), *Guidance on the Monitoring of Landfill Leachate, Groundwater and Surface Water.* Environment Agency, Bristol.

20. Environment Agency (2003), *The Development of LandSim 2.5.* National Groundwater and Contaminated Land Report GW/03/09, Environment Agency, Solihull.

21. Environment Agency (2004), *Guidance on the Management of Landfill Gas*, Environment Agency LFTGN 03, September 2004.

22. Environment Agency (2004), *Attenuation of Mecoprop in the Subsurface*. Environment Agency Science Group Report NC/03/12, March 2004.

23. Environment Agency (2004), *Improved Definition of Leachate Source Term from Landfills. Phase 1: Review of Data from European Landfills.* Science Report P1-494/SR1.

24. Filip, A. and Kuster, E. (1979), *Microbial Activity and the Turnover of Organic Matter in a Municipal Refuse Disposed of in a Landfill*, European Journal of Applied Microbiology and Biotechnology, 7, p. 371.

25. Freeze, R A & Cherry, J A (1979), *Groundwater.* Prentice Hall.

26. Halliday, D M, Joseph, J B & Mather, J D (1997), *Engineered Landfill Containment - Lessons from the Oxford Clay.* In: Sardinia International Symposium - Vol 3, Lining, Drainage and Stability in Landfills, pp 141 - 150.

27. Ham, R K (1994), *Decomposition and Leachate and Landfill Gas Generation*, IN: SWANAS Annual International Solid Waste Exposition, 32, pp. 95-104.

28. Hartz, K E, Klink, R E & Ham, R (1982) *Temperature Effects: Methane Generation from Landfill Samples*, IN: Journal of the Environmental Division, ASCE, 1982, pp.629-638.

29. Hartz, K E & Ham, R K (1983), *Moisture Level and Movement Effects on Methane Production Rates in Landfill Samples*, Waste Management and Research, 1, pp. 139-145.

30. Hjelmar, O & Hansen, J B (2004), *Towards Final Storage Quality in Landfilling: An Example,* Proceedings of Waste 2004, Stratford-upon-Avon, UK, September 2004.

31. Institute of Hydrology (1988), *Hydrological Data UK. Hydrometric Register and Statistics 1981 - 1985.* ISBN 0948540 08 7. Data for Station Number 033002 for the Bedford Ouse at Bedford.

32. Johnston, A G, Edwards, J S, Owen, J & Young, S D, Research Into Methane Emissions From Landfills, IN: Waste 2000, Stratford Upon Avon, United Kingdom, pp. 13-20.

33. Kasali, G. B. & Watson-Craik, I. (1988), Preliminary Investigation of the Influence of pH on the Solid State Refuse Methanogenic Fermentation. Journal of Applied Bacteriology, 65, p 231.

34. Knox, K (1990), The relationship between leachate and gas. In Proceedings of the Third International Landfill Gas Conference, Energy and Environment '90, 16-19 October, Bournemouth. Harwell Laboratories, Oxfordshire. ISBN 0-7058-1628-1.

35. Knox, K & De Rome, L (1998), Practical Benefits for the Waste Industry from the UK's Landfill Test Cell Programme, IN: Wastes Management, November 1998, pp. 18 – 19.

36. Knox, K, De Rome, L, Caine, M & Blakey, N C, (1999) Observations from a Review of the Brogborough and Landfill 2000 Test Cells Data, IN: Sardinia 99, Seventh International Waste Management and Landfill Symposium, 4-8 October 1999, Cagliari, Italy, pp. 45 - 52.

37. Knox, K (2004), Hydraulic Containment of Landfills, Leachate Quality and Waste Stabilisation. Input to Entec project on hydraulically Contained landfills, May, 2004.

38. Micales, J A & Skog, K E (1996), The Decomposition of Forest Products in Landfills, IN: International Biodeterioration & Biodegradation, Vol. 39, No. 2-3 , p/p 145 – 158.

39. McMahon, A M, Carey, M A, Heathcote, J A, & Barker, J A (2001), Guidance on Good Practice for the Development of Conceptual Models and the Selection and Application of Mathematical Models of Contaminant Transport Processes in the Subsurface. National Groundwater and Contaminated Land Centre Report NC/99/38/2.

40. Pacey, J G & DeGier, J P (1986) The Factors Influencing Landfill Gas Production, IN: Energy From Landfill Gas, Proceedings of a Conference Jointly Sponsored by the UK Department of Energy and the US Department of Energy, 28-31 October 1986, United Kingdom, pp. 51-59.

41. Pacey, J G & Dietz, A M (1986) Gas Production Enhancement Techniques, IN: Energy From Landfill Gas, Proceedings of a Conference Jointly Sponsored by the UK Department of Energy and the US Department of Energy, 28-31 October 1986, United Kingdom, pp. 361-374.

42. Powrie, W & Beaven, R P (1999), Hydraulic Properties of Household Waste and Implications for Landfills. Proc. Inst. Civil Engineers, Geotechnical Engineering, October, 1999, pp 235 - 247.

43. Robinson, H D, (1996), A review of the composition of leachates from domestic wastes in landfill sites. Technical Report CWM 072/95, 550pp. Environment Agency, Bristol.

44. Rowe, R K, Quigley, R M & Booker, J R, Clayey Barrier Systems for Waste Disposal Facilities. E & F N Spon, London.

45. Scott, P E, Dent, C G and Baldwin, G (1988), The Composition and Environmental Impact of Household Waste Derived Landfill Gas, Second Report, Department of the Environment Technical Report CWM 041/88.

46. U.S.E.P.A. (2003), Landfills as Bioreactors - *Research at the Outer Loop Landfill, Louisville*, *Kentucky*. National Risk Management Research Laboratory, Interim Report Ref EPA 600/R-03/097, September, 2003.

47. Wardell Armstrong (1998), *Proposed Landfill Construction Works at Astley, near Leigh, Lancashire. Waste Management Licensing Regulations1994 Regulation 15 Report.* Report Ref.KRJM/LKA/LE5009/7A, June 1998.

48. Watson-Craik, I. A. & Sinclair, K. J. (1995), *Co-Disposal of Industrial Wastewaters and Sludges*, IN: Microbiology of Landfill Sites, Second Edition, Edited by Senior, E., p.p. 91 - 130

49. Westlake, K. (1990), *Landfill Microbiology*, IN: Proceedings of International Conference Landfill Gas: Energy and Environment '90, Bournemouth, England, 16-19 October 1990, pp. 271-280.

50. Wyvern Waste (2001), *Pollution Control Action Plan,* prepared for Poole Landfill Site, Wellington, Somerset.

51. Yesiller, N. & Hanson, L. (2003), *Analysis of Temperatures at a Municipal Solid Waste Landfill*, IN: Sardinia 2003, Ninth International Waste Management and Landfill Symposium, 6-10 October 2003, Cagliari, Italy.

52. Zehnder, A. J. B., *Ecology of Methane Formation*, IN: Water Pollution Microbiology, Volume 2, New York, 1978, p. 349.