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Improved definition of leachate source term from landfills

Phase 1: review of data from European landfills

Science Report P1-494/SR1

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Professor Mike Depledge Head of Science

Executive Summary

Implementation of the Landfill Directive (Council Directive 1999/31/EC) is resulting in significant changes to the management of waste in the UK. The directive requires a phased reduction in the proportion of biodegradable municipal solid waste (MSW) going to landfill as well as treatment prior to deposit and the prohibition of certain types of hazardous waste. It also defines three classes of landfill for:

- hazardous waste;
- non-hazardous waste; and
- inert waste.

Significant changes to waste management practices in the UK will be needed to satisfy the requirements of the directive. The most significant changes likely to be needed include:

- an increase in incineration capacity for MSW;
- an increase in mechanical and biological pre-treatment of MSW; and
- a ban on the co-disposal of inorganic, hazardous and non-hazardous wastes with MSW.

Currently the UK has little information to help manage risks associated with leachate from post-Landfill Directive landfill sites, that will either have significantly reduced degradable carbon loadings or be dominated by chemical rather than biological processes. However, some EU member states have a long track record of incineration, mechanical and biological pre-treatment (MBP), disposal of mixed hazardous wastes and treatment of hazardous wastes prior to landfill.

The Agency needs to understand the effects that these changes in landfill practice will have on leachate, in terms of its quality and potential impact on the environment. There will be a need to obtain good quality data on leachate to define the leachate source term for risk assessment, assess the long-term efficacy of treatment methods, and predict the time taken for waste stabilisation.

Leachate data is discussed for landfill sites containing:

- municipal solid waste incinerator residues;
- mechanical and biological pre-treatment residues;
- mixed hazardous waste;
- non-hazardous low organic wastes.

The focus of the project was the collection of actual leachate data, where available from full-scale landfill sites. The report therefore does not address leachate quality from residues of innovative pre-treatment technologies (e.g. pyrolysis, gasification, anaerobic digestion) as field-scale landfills that have accepted a significant proportion of such wastes could not be identified.

The incineration of MSW is widely practiced in Europe, resulting in significant proportions of ash waste streams for disposal. These are broadly characterised as bottom ash and fly ash/air pollution control residues and are characterised by high pH and high concentrations of dissolved salts and heavy metals. Leachates derived from landfilling of incinerator residues bear little resemblance to leachate from co-disposal landfill sites. Variations in leachate quality are more likely to derive from site topography and hydrology and from the variations in the different chemical reactions (e.g. hydration, carbonation) than from the intrinsic composition of the ash. Flow characteristics at ash landfills may be complex, with rapid flow along preferential pathways and slower flow through the waste matrix, giving considerable short-term variability in leachate strength.

Mechanical and biological pre-treatment of MSW has been required through national legislation in a number of EU member states for several years, and this project focussed on the experience in Austria and Germany. The desk study identified two broad categories of organic residues from waste pre-treatment: mechanically sorted organic residues (MSOR), and biologically treated (MBP)

MSW or MSW fractions.

This project has confirmed existing knowledge that very high strength organic leachates are generated when MSOR fractions of household wastes are landfilled without further treatment. Such leachates can typically be from two to four times as strong as equivalent leachates from MSW landfills, and high organic strengths can persist for at least several decades.

Efficient MBP of the MSOR fraction can considerably reduce the organic strength of leachates, avoiding the acetogenic phase, and more rapidly producing leachates similar to those from MSW landfills in methanogenic phases of decomposition. Landfills receiving MBP wastes will pose a risk to groundwater similar to conventional MSW landfills that have become methanogenic, and may require a similar period of time before active management and treatment of leachates ceases to be necessary.

Surprisingly little recent information had been published on leachate from hazardous waste landfills in EU states where co-disposal has already been banned. This study included the collection of data from current or recent European landfill sites that have been used solely for the disposal of mixed hazardous wastes meeting national waste acceptance criteria similar to those anticipated under the Landfill Directive.

The data collected suggests that hazardous waste leachates are more varied than UK co-disposal leachates, due to a greater effect of site-specific and waste-specific factors. Biological processes, dominant in landfills receiving bioreactive wastes, are relatively unimportant in landfills for hazardous wastes, and leachate quality is determined largely by inorganic chemical reactions and physical processes. Carbonation is of particular importance, as it may often result in the pH within parts of the waste matrix changing from strongly alkaline to near-neutral, and hence locally affect the solubility of amphoteric metals. The extent to which carbonation occurs depends on local factors such as the lime content and the access of air to the wastes. The hazardous waste landfills encountered in this study were at liquid:solid (L/S) ratios of not more than 0.3 and it is clear that they are nowhere near final storage quality.

This report includes the most comprehensive published data set on leachate quality from landfill sites that are similar to those that will be developed in the UK following implementation of the Landfill Directive. The information included can be used to inform the selection of leachate source term data for groundwater risk assessments, at least until a more comprehensive body of data becomes available for UK landfills. It may also be used to inform a strategic approach to waste management, particularly with regard to achieving waste stabilisation. Further research needs are identified to address existing knowledge gaps, and particularly to meet the challenge of achieving landfill stabilisation within decades rather than centuries.

Acknowledgements

A large number of European landfill operators and researchers generously provided valuable assistance in the form of data, background information on the landfills visited, information on management practices, including waste acceptance criteria, detailed technical discussions and help in organizing site visits. The project could not have been undertaken without this help and the contributions of all these individuals and organizations is gratefully acknowledged.

Contents

1 Introduction

1.1 Background

Implementation of the Landfill Directive (Council Directive 1999/31/EC) is resulting in significant changes to the management of waste in the UK. The directive requires a phased reduction in the proportion of biodegradable municipal waste going to landfill as well as treatment prior to deposit and the prohibition of certain types of hazardous waste. It also defines three classes of landfill for:

- hazardous waste;
- non-hazardous waste; and
- inert waste.

Waste acceptable for deposit in a non-hazardous landfill will include treated municipal solid waste (MSW) (a proportion of which will be classed as biodegradable), non-hazardous waste of any other origin and hazardous waste which has been stabilised, although the latter may not be deposited in the same cells as biodegradable wastes. However, there is a possibility that in the longer-term, waste acceptance criteria will be changed to ban biodegradable waste at non-hazardous sites. Before July 2004 waste should not be accepted at hazardous waste landfills without prior treatment if the concentration of contaminants is high enough to give rise to a short-term occupational or environmental risk. After July 2004 wastes must be treated prior to landfilling and after July 2005 additional criteria will apply. By treating waste to meet the initial criteria, the properties and potential behaviour of the waste, once landfilled, will be changed along with the consequential effect on landfill behaviour. Obvious implications include:

- reduced biological activity caused by the reduction in degradable carbon;
- changes in landfill gas and leachate production and composition;
- leachate quality dominated by chemical rather than biological processes; and
- a change in the mechanisms of physical settlement.

Significant changes to waste management practices in the UK will be needed to satisfy the requirements of the directive. The most significant changes likely to be needed include:

- an increase in incineration capacity for MSW;
- an increase in mechanical and biological pre-treatment of MSW; and
- a ban on the co-disposal of hazardous wastes with MSW.

The leachate quality from MSW and co-disposal landfill sites in the UK has been extensively studied over many years (e.g. Robinson, 1996), providing waste management professionals with a good understanding of leachate characteristics. For co-disposal, waste input controls are designed to ensure that the leachate is similar to MSW landfill leachate, i.e. that it is dominated by the biological processes responsible for the degradation of the organic components of MSW. The leachate quality information is currently used to assess and manage risks that the landfill poses to groundwater and local surface water bodies.

Currently the UK has little information to help manage risks associated with leachate from post-Landfill Directive landfill sites, that will either have significantly reduced degradable carbon loadings or be dominated by chemical rather than biological processes. However, some EU member states have a long track record of incineration, mechanical and biological pre-treatment (MBP), mono-disposal of mixed hazardous wastes and treatment of hazardous wastes prior to landfill.

1.2 Scope and purpose of the project

The Environment Agency (the Agency) has responsibility for authorising and regulating waste management facilities in England and Wales. A key objective of this regulatory responsibility is to ensure that waste facilities do not cause harm to the environment from the escape of leachate.

The regulatory implications for the reduction in the proportion of biodegradable municipal waste going to landfill, the separate disposal of hazardous waste, and pre-treatment of waste prior to deposit are likely to include:

- definition of leachate source term for risk assessment;
- long-term efficacy of treatment methods;
- effect of leachate on landfill liner; and
- time taken for waste stabilisation.

The Agency needs to understand the effects that these changes in landfill practice will have on leachate, both in terms of its quality and potential impact on the environment. There will be a need to obtain good quality data on leachate from hazardous waste monodisposal sites and from waste that has been treated prior to disposal.

This study was commissioned to gather information on actual leachate quality from key research and landfill sites in EU member states that meet national waste acceptance criteria as similar as possible to those required by the Landfill Directive. The study was undertaken in two phases. Phase 1 is a collation and review of existing data, both from published literature and from operating landfills and research studies within other parts of the EU and forms the main part of this report. Phase 2 involved sampling and chemical analyses of leachates from MBP landfills in Europe and laboratory-scale leaching of incinerator residues from UK incinerators (the latter reported separately) to supplement the data from Phase 1.

The focus of the project was the collection of actual leachate data, where available from full-scale landfill sites. The report therefore does not address leachate quality from residues of innovative pre-treatment technologies (e.g. pyrolysis, gasification, anaerobic digestion) as field-scale landfills that have accepted a significant proportion of such wastes could not be identified.

1.3 Implementation of Phase 1 of the project

Data for Phase 1 were sought by literature search, and by identifying suitable examples of sites or research studies, through contact with specialists and operators in other EU member states, that have suitable monitoring data available.

In addition to leachate quality data, supporting information was sought for each landfill, to assist interpretation in the UK context. This information included:

- gas generation and gas quality, to assess the level of biological activity (this is currently an important unknown, e.g. for sites receiving mixed hazardous wastes);
- leachate volumes or relative wetness of the site, to assess the extent to which leachate strength has been affected by dilution;
- waste input data – types and quantities, nature of any pre-treatments;
- details of any waste acceptance criteria used (this will be relevant to the practical work undertaken in Phase 2); and
- impacts on wastes stabilisation and effects on landfill design, e.g. leachate collection and liner systems.

A literature search was undertaken through the Waste Management Information Bureau, in February 2001. Visits to EU experts and operating landfills were undertaken during the period March 2001 to June 2001. Discussions were held with experts from Austria, Denmark, France, Germany, Italy and The Netherlands. Visits were made to seven operating landfills in Denmark, France, Germany and The Netherlands. Several of these landfills were subdivided into cells receiving different waste categories and thus provided data on more than one type of waste.

1.4 General approach

A large quantity of analytical data was obtained, with significant variation in both the range of parameters and in the format of the data. Where raw data were received, an attempt has been made to present them in a common format as time series graphs. However, this has not always been possible, and some data can only be presented as scanned copies of graphs, or in tabular form.

Where time series data are available, an attempt has been made to estimate the liquid/solid (L/S) ratios corresponding to the time covered by the data. This provides a common basis for comparison of data from different landfills and from leaching tests. In practice, for the majority of landfills, the L/S ratios are small, typically less than 0.5, even when data have been obtained for periods of 10 years or more. This is because landfills are usually capped, following which the moisture flux drops to a very low level. Data for larger L/S ratios have usually only been obtained from lysimeter studies and leaching tests, except for a very small number of landfills.

It was expected that few, if any, EU landfills would be able to provide results for all of the 57 List I organic Pollution Inventory substances that have to be reported by landfill operators in the UK for discharges to sewer or surface water (Robinson and Knox, 2001; 2003), or for the List I substance groups cited in the Groundwater Directive (80/68/EEC). This proved to be the case. More surprisingly, important conventional parameters were sometimes not analysed, and several sets of data lack one or more of TOC, COD, TKN, NH_4-N or certain major ions.

In the following sections, only brief details are given of the most important facts about the site or study (e.g. liquid/solid ratio, waste types). More detailed notes on the sites visited are given in Appendix 3. These notes typically include derivation of any values for L/S ratio that are referred to in the main text.

All full-scale landfills visited during the study are referred to in this report by a code letter only. An assurance was given to all operators contributing data, that neither the name of the site, nor the EU country in which it was located would be disclosed in the report.

1.4.1 Landfill source term data

Groundwater risk assessment using LandSim 2.5 requires leachate chemistry source term data to be defined in terms of:

- a probability distribution function (PDF); and
- a declining source term.

Distributions

For many parameters, the leachate data from UK landfills approximate to a log-triangular PDF. This requires three numbers to be specified, namely the maximum, minimum and most likely values. The data presented in this report has been used to generate maximum and minimum values for MBP waste and mixed hazardous waste landfills. Data for bottom ash and APC residue monofill landfills are derived from laboratory data discussed in detail in Environment Agency (2004) as the laboratory results are generally consistent with comprehensive data sets of MSW incinerator residue analyses from Europe (e.g. IAWG, 1997).

The default values built in to LandSim at present are based on statistical examination of data from a large number of UK MSW landfills (Environment Agency, 2001, 2003).

For MSW landfills, much of the variability contributing to these log-triangular PDFs arises from the following causes:

- heterogeneity of the waste composition, given the wide range of waste types deposited in a typical UK landfill cell;
- variations due to the hydraulic regime: these depend on the local rainfall (highly variable across the UK) and the site topography, which will influence the degree of short circuiting that occurs as well as the average liquid to solid (L/S) ratio that the waste experiences; and
- varying extent of establishment of methanogenesis, between young and older areas of waste.

For the single source wastes considered in this study, spatial heterogeneity is likely to be far less than in conventional UK mixed waste landfill cells. For example, ash from a single incinerator is likely to be very consistent in quality. Also the question of acetogenic or methanogenic conditions will not arise, in most cases. However, the hydraulic sources of variation will be similar, because the locations and design features of our landfills are likely to remain similar.

The overall variability of leachate quality is likely to be less than at a mixed waste landfill. It is therefore not appropriate to use the spread in existing LandSim PDFs as a guide to the spread that will occur in the single type wastes considered in this study. The approach suggested to derive source term data for landfill sites described in this report is as follows.

1. From available data for each source, derive whatever information is appropriate for the spread of the concentrations due to variations in the nature of the source waste.

2. Add in a factor to account for seasonal dilution and variations in hydraulic regime that are likely to occur at UK landfills.

3. Derive maximum and minimum values based on hydraulic and waste source variation factors.

4. Derive the most likely value based on the shape of the PDF already in use in LandSim.

Each set of proposed values in this report is derived from a statistically small data set. **The use of these values as default input values to a groundwater risk assessment model such as LandSim 2.5 will not be relevant to all sites and it is strongly recommended that site-specific values are used, where possible**. However, no data are currently available for leachate quality from Landfill Directive-compliant classes of landfill and professional judgement should be used to assess whether the proposed values presented in this report represent the best data available for a particular site.

Hydraulic variation

In the Pollution Inventory study (Robinson and Knox, 2001), sites were allocated one of three descriptors (D1 to D3) for their moisture regime: dry, medium or wet. To derive factors applicable to future sites, it has been assumed that wet (D3) sites would not occur. Only the data for the dry and medium sites were therefore used (classes D1 and D2). This amounted to 57 landfills. Percentiles were calculated for conservative parameters such as chloride, conductivity, Na, K and NH_4-N. The ratios of the 95 percentile values to the 5 percentile values, and 90 percentile to 10 percentile values were examined (Table 1.1).

Based on this analysis, a range in concentrations of between 1:5 and 1:10 may be expected at UK landfills, purely due to hydraulic variations resulting from rainfall and landfill topography. A factor of 5 is proposed here as likely to reflect the minimum range that would occur for identical waste types, although this is uncertain and a factor as high as 10 may be proposed.

Table 1.1 Hydraulic variation in conservative chemical parameters

	Cl	NH_4-N	Na	conductivity
5 %ile	403	100	246	3066
95 %ile	3716	1720	3006	21600
ratio	***9***	***17***	***12***	***7***
10 %ile	477	157	327	3928
90 %ile	3424	1474	2460	19340
ratio	***7***	***9***	***8***	***5***

Declining source term

The change in concentration of non-volatile substances with time has been modified within LandSim 2.5 so that it is species-specific, permitting more accurate modelling of the change in physical and chemical characteristics of individual contaminants through time. The revised method of modelling the decline of non-volatile substances in leachate is similar to that adopted by the EU Technical Adaptation Committee, during the derivation of waste acceptance criteria for Annex 2 of the Landfill Directive. The following text is derived from the LandSim 2.5 Help files.

The change in concentration of each non-volatile species through time is based upon the following equation:

$C_{(t)} = C_{(0)} * \exp(-\kappa * L/S)$

Where:

$C_{(t)}$ is the concentration of the species in leachate at any time t (mg/l);
$C_{(0)}$ is the initial concentration of the species in leachate (mg/l), usually determined when liquid:solid ratio L/S = 0.05 l/kg. The current leachate quality can be used to define the initial concentration for most landfills in the UK;
κ (kappa) is a species and waste-specific constant (kg/l).

Kappa is related to the rate of release of a species from the solid (waste) to the aqueous (leachate) phase within the landfill. The concentration of a species with a high value of kappa (e.g. chloride) will decline more rapidly with time than that of a species with a low value of kappa (e.g. arsenic). Kappa values are experimentally derived from column leaching tests and therefore take into account all the physical and chemical processes occurring during the breakdown and flushing of the waste mass.

The change in leachate concentration is therefore controlled by how rapidly the waste mass is flushed by infiltration (i.e. the liquid solid ratio) and by how readily any non-volatile species will be released from the solid to the aqueous phase. There is a strong relationship between kappa and initial leachate concentration. It is this relationship that LandSim 2.5 uses in determining the value of kappa to be adopted during a simulation. The relationship takes the following general form:

$\kappa = m.\ln(C_{(0)}) + c$

Where:

$C_{(0)}$ is the initial concentration of each individual species in leachate and must have units of µg/l;
m and c are empirically derived species-specific values with units of kg/l;

The value of kappa, once calculated, does not change throughout the simulation and default values are provided in LandSim 2.5. The values are used within the model in the absence of any site-specific information. However, these values may be changed if there is documented evidence from site-specific data that justifies the use of different values.

Kappa values may be calculated from the results of continuous upward flow percolation tests conducted in accordance with NEN 7343, 1995 and the draft CEN percolation simulation test. Seven samples of leachate are collected at liquid solid ratios of approximately 0.1, 0.2, 0.5, 1, 2, 5 and 10, and analysed to determine the concentrations of species of interest. Because the leachate is collected over a period of time, the concentration data is often reported for a range of liquid solid ratio.

Kappa is determined by plotting natural logarithm of the concentration against the value of liquid solid ratio (in l/kg) representing the middle of the range for each increment (e.g. L/S 0-0.1, mid-point = 0.05). The data should fall on a straight line. The gradient of this straight line is the value of kappa in kg/l.

Kappa values are presented for bottom ash and fly ash/APC residues, derived from laboratory work carried out during Phase 2 of this project (Environment Agency, 2004). The above protocol was not used; instead kappa was estimated from the results of a continuous recirculation (until constant conductivity) L/S1 upflow column test and a L/S10 batch test. As only two points are available a straight line is assumed rather than confirmed, and it is not known how comparable the kappa values derived from the longer contact time tests are with those using the draft CEN percolation test.

1.5 Structure of report

Data are presented for different waste categories and sub-categories in the following sections.

Chapter 2 sets out a summary of laboratory and field-scale research into the chemistry of leachates from MSW incinerator residues. This information is supported by time-series data from a bottom ash/fly ash landfill in Denmark and two landfills containing fly ash/air pollution control (APC) residues. Proposed values for bottom ash and fly ash/APC residues are derived from laboratory work carried out in Phase 2 (Environment Agency, 2004) and are summarised in this chapter.

Chapter 3 sets out a summary of laboratory and field-scale research into MBP of MSW and the leachates generated by the treated waste. Data is also presented from a research cell in the Netherlands used for the deposit of mechanically sorted residues, and research into leachate quality from composted MSW and MSOR residues is also presented. Proposed values for a variety of MBP wastes are provided based on the data presented in chapters 3 to 5.

Chapter 4 sets out a summary of research into leachate from MBP wastes in Austria and Germany.

Chapter 5 sets out data from Phase 2 sampling and comprehensive analysis of MBP leachates from full-scale landfill sites in Europe.

Chapter 6 sets out data from a number of European mixed hazardous waste landfill sites, including a landfill with cement stabilised/solidified hazardous wastes, a monofill landfill containing bottom ash from incineration of hazardous waste, and a monofill landfill containing fly ash from hazardous waste incineration, deposited in "big bags". Proposed values for mixed hazardous waste landfil leachate are presented, but with a caveat regarding the variability of such leachate and the significance of waste inputs on leachate quality.

Chapter 7 sets out limited data from a mixed non-hazardous, low organic waste landfill.

Chapter 8 presents a discussion of the study findings and identifies some knowledge gaps apparent from this study.

Additional information on the legislative drivers for MBP in Austria and Germany, and more detailed information on the sites visited during the study is provided in the appendices.

2 Residues from incineration of MSW

2.1 Background

The incineration of MSW is widely practised throughout Europe, yet currently only about 2.7 Mt/a of MSW (<10% of MSW arisings) is incinerated in the UK (Environment Agency, 2002). A number of European countries, including France, Germany, Denmark, The Netherlands and Switzerland incinerate a significantly greater proportion of their waste than the UK, to reduce the proportion of biodegradable waste going to landfill. The perceived benefits of incineration include:

- reduction in the volume of biodegradable waste being landfilled;
- reduction in the quantity of gas produced at landfills;
- a shorter timescale to waste stabilisation and improved waste settlement characteristics;
- combined heat and power recovery; and
- opportunity of recovery and reuse of some waste streams (e.g. bottom ash).

However, the ash waste streams produced by incineration contain a high proportion of soluble salts with potential to release contaminants into the environment, including chloride and heavy metals. The incineration process results in a number of ash residues that, for the purpose of this study, are considered under two broad categories:

- bottom ash – comprising clinker that falls through the grate with unburned fractions; and
- fly ash/APC residues – comprising fine particulates, lime and carbon extracted from the flue gas and/or air pollution control scrubbers. The lime and carbon are added to neutralise acid gases and sorb dioxins and furans respectively.

Incinerator residue arisings in the UK are mainly disposed of to landfill - around 80% of the bottom ash and 88% of the APC residues. Some 50% of the APC residues are deposited at a single ash landfill and the remainder to co-disposal landfills. The UK will need to find alternative strategies to manage about 0.6 Mt of bottom ash and 80,000 tonne of APC residues a year, at current production rates (Environment Agency, 2002), when co-disposal is ultimately prohibited by the Landfill Directive.

2.2 MSW incinerator bottom ash

MSW incinerator bottom ash arises from a high temperature oxidative process acting on a heterogeneous mixture of organic and inorganic components, including metals. The solid residues have been characterised in terms of elemental composition and mineralogy (e.g. Eighmy *et al.*, 1994, Blondin *et al.*, 1997) using a range of techniques, including X-ray powder diffraction (XRPD), scanning electron microscopy/X-ray microanalysis (SEM/XRM), neutron activation analysis (NAA) and X-ray photoelectron spectroscopy (XPS). Bottom ash is a highly porous material, with specific surface area up to 50 m^2/g (dry weight), due to vesiculation of solids when in a molten state (Abbas, 1998).

Bottom ash typically contains (e.g. IAWG, 1994; Lechner and Huber, 1999):

- silicates (50 – 60%);
- calcium oxide (~20%);
- other metal oxides (especially Al and Fe);
- some elemental metals (e.g. Fe, Al);
- mineral salts; and
- unburnt carbon and organic compounds (usually <5%).

The calcium oxide content of the ash depends on the calcium content of the unburnt waste e.g. the presence of plasterboard and the effect of pre-sorting.

The predominantly inorganic nature of the ash means that leachate quality is influenced far more by chemical equilibria than by biological processes, though the latter can have some influence.

Leachable bottom ash components may be broadly divided into: inorganic salts, trace elements, and soluble organic compounds. The trace elements and some of the major ions in leachates are strongly influenced by several chemical, and occasionally biological, reactions that begin as soon as the ash reaches the quench tank and continue, often for many decades, within the landfill. These reactions may result in pH values and redox potentials varying over a wide range, not only between landfills, but also within different parts of the same landfill. Collected leachates may be composites, derived to varying degrees from different zones within the site. While it is beyond the scope of this report to describe these reactions in detail, it is helpful to summarise the principal ones in order to understand data on leachate quality. The three most important are:

- hydration of calcium oxide (CaO) to slaked lime, $Ca(OH)_2$;
- carbonation of slaked lime to calcium carbonate; and
- reaction of slaked lime with silica to form calcium silicate hydrates (C.S.H).

The following additional reactions may be important in some cases.

- Oxidation of elemental metals, especially Al and Fe, producing hydrogen and causing the redox potential to fall. This reaction can be fairly rapid, occurring during the first few weeks following deposition and is highly exothermic.
- Microbial degradation of unburnt organics, producing organic acids, decreasing the pH and lowering the redox potential. This reaction will usually occur slowly, and only in zones where carbonation has already reduced the pH value significantly and only if sufficient unburnt organic matter is present.

The occurrence of significant biological activity is likely to have fallen during the 1980s and 1990s, as progressive tightening of standards lead to more consistently low residual organic matter in bottom ash.

The hydration of calcium oxide begins as soon as the ash is brought into contact with water in the quench tank, and may continue for some time thereafter. It is an exothermic reaction, sometimes leading (along with the heat produced from hydrogen formation) to very high temperatures within landfills. It also leads to very strongly alkaline pH values (>12) which can cause amphoteric metals in the ash to become more mobile (e.g. Zn, Pb). Very high calcium concentrations can occur in solution, which may suppress the concentrations of sulphate. The high pH values may vary (e.g. from 11 to 12.5) depending on the extent of prior removal of high calcium wastes, such as plasterboard.

The calcium hydroxide formed in this way is then itself subject to two competing reactions. Carbonation (also referred to as carbonatisation) occurs when carbon dioxide in the atmosphere (or generated by degradation of residual organic matter) reacts with slaked lime to form calcium carbonate, which precipitates as calcite. This causes the pH locally to fall, typically to around 8, rendering some metals less soluble and others more soluble than at higher pH values. Under these conditions the pH is strongly buffered by the solid phase calcite. The extent to which carbonation occurs depends on local factors that govern the access of air to the ash. It may remain incomplete even after several decades. Covering with a low permeability cap greatly restricts access of atmospheric gases to the upper waste surface, while maintaining a leachate collection system in a saturated condition restricts the access of air to the lower layers. The greater the depth of the landfill the less the opportunity for access by atmospheric carbon dioxide. Some operators allow the ash to undergo carbonation in shallow piles, for a period of weeks or even months prior to landfilling, and processes have been developed to accelerate carbonation by routing CO_2-rich flue gas through the ash. The extent of carbonation may thus be highly varied and the reaction may continue at a slow rate for many years within the landfill.

Calcium hydroxide may also react with hydrated silica and alumina to form calcium aluminium silicate hydrates. This process also results in a decrease in pH values, although the reaction products provide less buffering than calcium carbonate against further pH changes (e.g. from organic acids generated during degradation of residual organic matter).

2.3 Leachate quality from bottom ash

A view expressed by EU experts during this study is that variations in ash leachate quality are more likely to derive from site topography and water regime and from the variations in the different reactions described above, than from variations in the intrinsic composition of the ash. To quote one EU expert: “bottom ash is bottom ash”. Because of this, and because of the largely inorganic nature of the material, these EU experts regard leaching test data as a good guide to actual leachate quality (e.g. van der Sloot *et al.*, 2000). Concentration ranges for the maximum levels observed in leaching tests on bottom ashes, at L/S ratios below 0.5 have been reported by Hjelmar (1996), observed data are reproduced in Table 2.1.

Of particular note are the significant levels of non-volatile organic carbon (NVOC) and ammoniacal nitrogen, and the elevated levels of copper, molybdenum and lead. Most of the data in the table date from the early 1990s. It is possible that tests on ash from the most modern incinerators would show lower levels of these organically-derived components.

Table 2.1 Typical concentrations in eluates at low L/S ratio from MSW incinerator bottom ash (Hjelmar, 1996)

Components	Units	Range
SO_4^{2-}, Cl^-, Na K, Ca	g/l	1 – 10
NVOC, NH_4-N	mg/l	100 – 1000
Cu, Mo, Pb	mg/l	1 – 10
Mn, Zn	µg/l	100 – 1000
As, Cd, Ni, Se	µg/l	10 – 100
Cr, Hg, Sn	µg/l	1 – 10

Results for some inorganic components were reported by Stegemann *et al.* (1995) for a lysimeter study of bottom ash leaching. 336 kg of ash was collected after magnetic separation of iron, and leached with distilled water and rainwater up to a L/S ratio of ~1.4, over a period of two years. The maximum concentrations reported by the authors are summarised in Table 2.2

Table 2.2 Maximum concentrations of leachate components from a lysimeter study (after Stegemann *et al.*, 1995)

Parameter	Units	Value
pH		12.8
total dissolved solids (TDS)	mg/l	21,000
chloride	mg/l	10,000
sulphate	mg/l	550
copper	mg/l	17
lead	mg/l	6.5
zinc	mg/l	2.5
nickel	mg/l	0.64
chromium	mg/l	0.36

Chloride exhibited an exponential washout pattern, falling to ~500 mg/l by L/S ~0.7, then continuing at a few hundred mg/l for the rest of the study. By the end only 45% of the chloride content had been leached out. This suggests that a significant proportion of the ash matrix may have been by-passed by the bulk of the percolating water.

The pH remained high (12 to 13) throughout, although sub-sampling at the end of the study showed that lower pH values had developed at the upper (exposed) surface, indicating partial carbonation. This is relevant to the sulphate concentrations, which were initially low (~400 mg/l) presumably because of very high calcium concentrations. The sulphate concentrations then fell further, to ~100 mg/l before rising during the second half of the study, as carbonation began to remove calcium from solution. By the end of the study, only 2% of the sulphate content had been leached.

The elevated levels of copper and lead are consistent with the ranges reported by Hjelmar (1996). Lead concentrations remained >1 mg/l up to an L/S ratio of greater than 1 in the study of Stegemann *et al.* (1995).

Results for leachate quality and its relationship with flow rates and rainfall have been reported for a bottom ash landfill in Switzerland (Johnson *et al.* 1998; Johnson *et al.* 1999). The ash was stored in piles for 3 to 6 months before landfilling and was screened to remove ferrous materials and bulky unburned organic materials. The exothermic reactions had finished by the time the ash was landfilled and it was close to ambient temperature. The ash was placed in an engineered containment cell with a leachate collection system and remained uncovered. Key data on the site were as follows:

year of infilling	1992
years of study	~1995 – 1996
area of cell	5850 m^2
depth of cell	6 m
estimated bulk density	~1.5 t/m^3
annual leachate flow	~550 mm
L/S ratio at time of study	0.2 – 0.25

Between November 1994 and November 1996, 194 samples of the leachate discharged were analysed. Flow rate and conductivity were recorded continuously. Many of the samples were taken during and after individual rainfall events. In addition to the leachate discharge, 19 liquid samples were taken from a piezometer within the waste mass.

Analytical results are presented in Table 2.3 [no results were reported for nitrogen species, redox potential, Ni or As].

Some leaching may have occurred during storage prior to landfilling, and some washout of chloride and sodium would have been expected to have occurred by the time of this study. Even so, the dry weather concentrations of Cl^-, SO_4 and Na are consistent with the Table 2.1 eluate for low L/S ratios reported by Hjelmar (1996), while DOC, K, Ca and most of the heavy metals are generally at lower concentrations.

The pH values of the collected leachate were high, mostly >10, indicating that carbonation reactions were far from complete. This was confirmed by the high pH values in the piezometer leachate (average pH 11.26) and by the observation of extensive scale formation in the leachate collection system, as leachate came into contact with atmospheric CO_2.

Table 2.3 Leachate analyses at Lostorf bottom ash landfill (Johnson *et al.* 1999)

		Collected leachate discharge				**Piezometer P1**
Parameter	**units**	**average**	**Max**	**min**	**dry weather average**	**average**
pH	pH units	10.28	11.28	8.68	10.48	11.26
DOC	mg/l	10.3	44.8	4.4	8.0	136.8
Alkalinity, as $CaCO_3$	mg/l	139	209	78	162	0
Cl	mg/l	1,672	3,657	391	3,042	2,574
SO_4	mg/l	1,190	2,141	529	1,786	1,027
Na	mg/l	1,024	2,098	225	1,746	1,449
K	mg/l	460	944	181	737	757
Ca	mg/l	329	644	124	536	265
Mg	mg/l	15.2	26.2	2.5	18.2	<1.2
B	mg/l	2.4	3.7	1.5	2.7	no data
Si	mg/l	3.8	5.8	2.6	3.8	8.1
Al	mg/l	0.77	3.97	0.02	0.33	3.08
Mo	µg/l	522	1315	145	602	2419
W	µg/l	113	234	34	63	261
Cu	µg/l	101	479	6.9	33	1321
Sb	µg/l	32	57	11	19	54
V	µg/l	22	48	11	35	35
Cr	µg/l	10.9	16.1	8.5	11.6	7.3
Zn	µg/l	5.7	10.5	2.9	5.6	32.3
Mn	µg/l	4.7	18.0	1.5	no data	6.1
Pb	µg/l	2.7	7.0	0.5	no data	4.6
Cd	µg/l	1.3	5.4	0.38	1.7	3.45
Conductivity	µS/cm		14,900	5,900		
Ionic strength	gmol/l	0.1	0.18	0.03	0.16	

Johnson *et al.* (1998) provided strong evidence of preferential flow paths through the ash. Discharge rates responded rapidly to rainfall events of greater than 10 mm, typically within ~10 hours. During dry weather, conductivity was relatively steady, at its maximum value of 14,900 μgS/cm. When the leachate discharge rate increased rapidly, following rain, the conductivity fell equally rapidly, but never fell below 5,900 μS/cm at any time, even when discharge flows increased more than thirty-fold, from dry weather rates of ~1.5 l/min to peaks of 50 l/min and higher. This indicates that the proportion of flow that short-circuits the waste matrix did not increase without limit, but reached some maximum value, with the remainder passing through the waste matrix, flushing out *in situ* leachate. The ratio of minimum to maximum conductivities would suggest that this proportion of by-pass flow is in the order of 60%. Tracer studies also confirmed that infiltrating moisture percolates via a combination of rapid short-circuit flow and slower flow through the waste matrix.

Johnson *et al.* (1998) also measured the moisture content of the *in-situ* wastes to be ~21% w/w (four samples ranged from 19.9% to 22.2%). From analysis of the ‘leachate’ proportions of the discharge flows, they estimated the average residence time of leachate within the landfill to be ~3 years.

2.3.1 Time series data from Danish landfill

A unique long-term data set, from 1973 to 1998 has been compiled by Hjelmar (pers comm.), for an ash landfill at Vestskoven in Denmark. The landfill was PVC lined and received a total of ~10,000 m^3 of waste. The waste consisted of ~85% bottom ash and ~15% fly ash. As such, it may be expected to contain higher concentrations of leachable mineral salts than a purely bottom ash landfill. Infilling began in 1973 and was complete by June 1976, when a 1 m soil top cover was placed. L/S values have been calculated by Hjelmar from the cumulative leachate volumes and by assuming a bulk density of ~1 t/m^3, based on recorded waste inputs. By March 1998, the L/S ratio had reached 0.93, making this a longer data record than for any other landfill in this study, and an opportunity to study washout effects compared with solubility effects, at a full-scale landfill. Because of the date of the waste placement, it may be expected that the ash would have contained a higher percentage of unburnt organics than would arise at a modern incinerator. It is also unlikely that separation of ferrous metals took place. It is therefore expected that biological and chemical reactions may have led to reducing conditions, which would not necessarily occur to the same extent at a modern ash landfill.

Data have been reported for a comprehensive set of conventional determinands, although no results are reported for Mo, W, V or Sb. The results are presented as time series graphs in Figures 2.1 to 2.3.

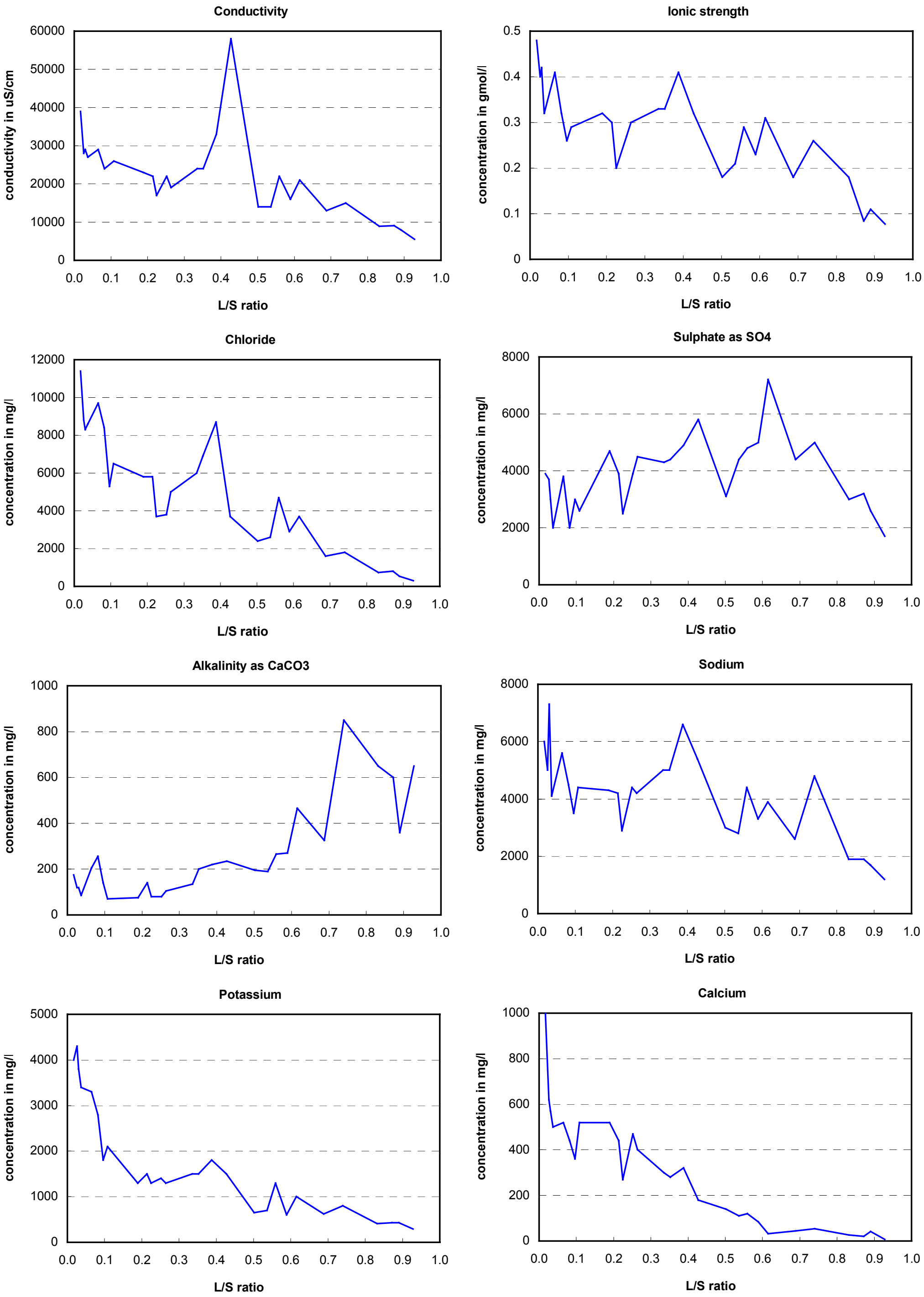


Figure 2.1 **Leachate quality from Vestskoven ash landfill: major ions (after Hjelmar, pers. comm.)**

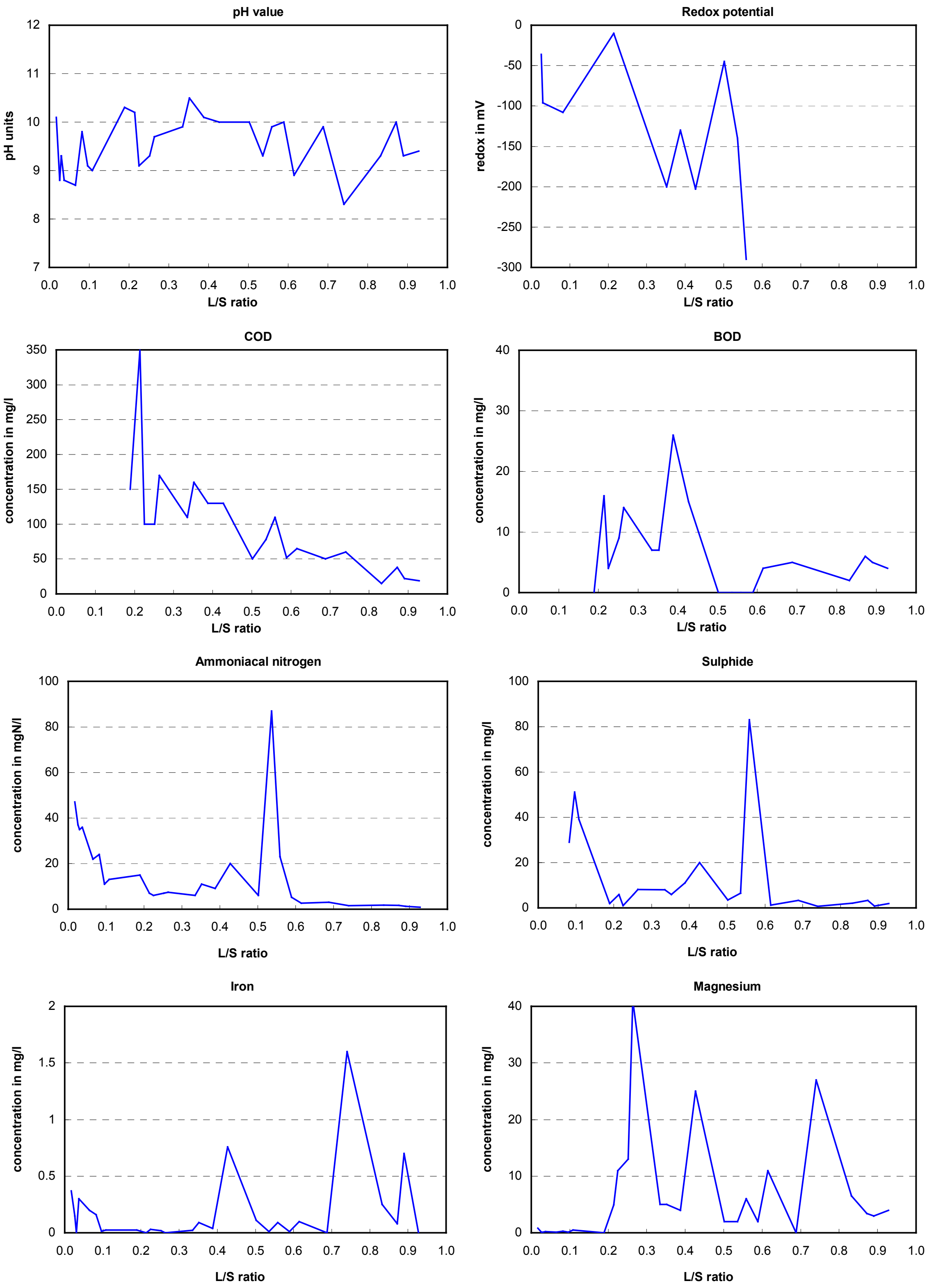


Figure 2.2 Leachate quality from Vestskoven ash landfill: sanitary parameters (after Hjelmar, pers. comm.)

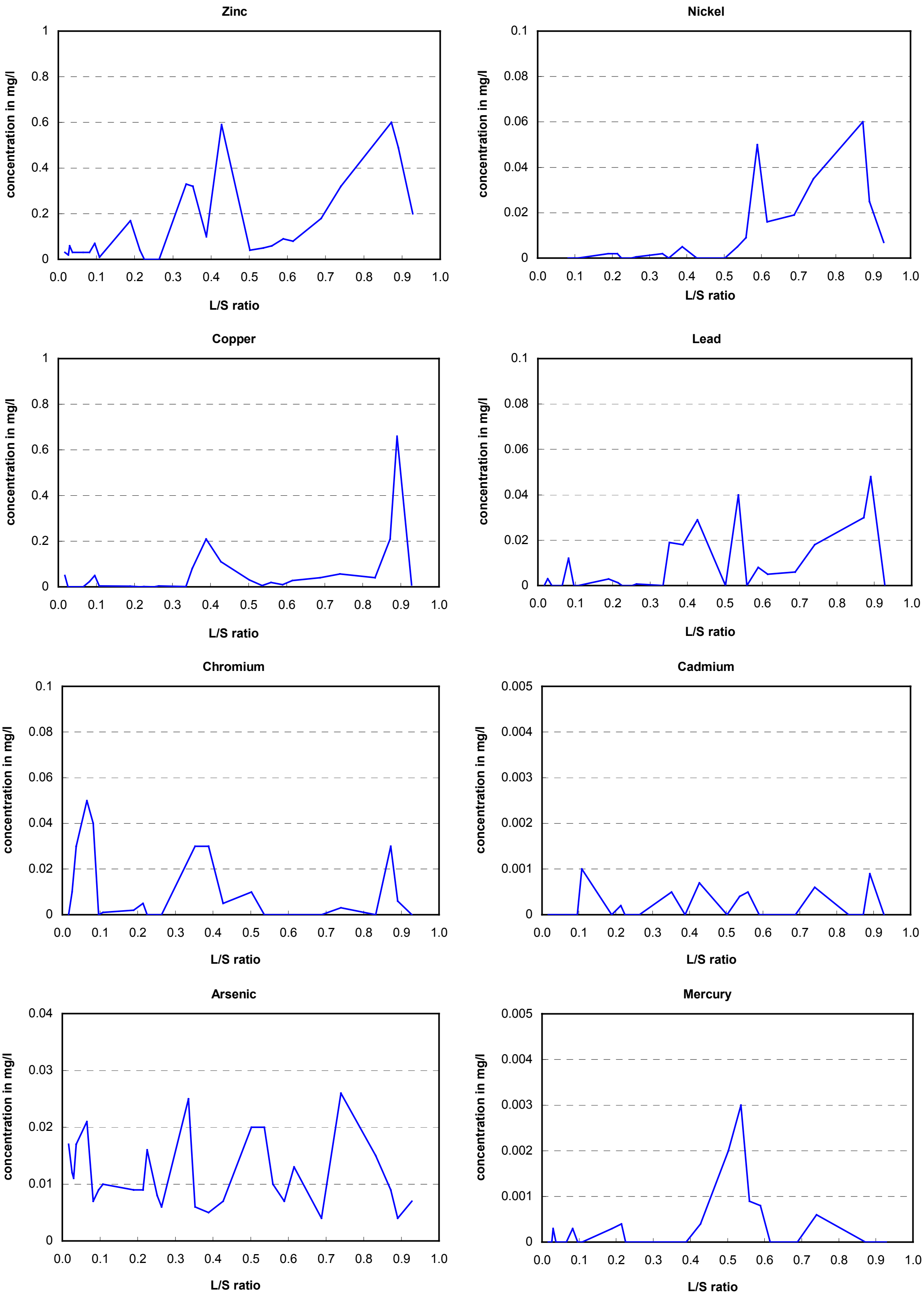


Figure 2.3 Leachate quality from Vestskoven ash landfill: heavy metals (after Hjelmar, pers. comm.)

The following interpretive comments may be made

- The initial total disssolved solids (TDS), at low L/S values, was at the high end of the range reported by Hjelmar (1996), at ~51,000 mg/l, with:

  Cl (~11,000 mg/l) > SO_4 (~3000 mg/l) >> Alkalinity (~200 mg/l)
  Na (6000 mg/l) > K (4000 mg/l) > Ca (1000 mg/l) >> Mg (<1).

- The relatively high initial TDS may be due partly to the presence of 15% fly ash in the landfill.

- Washout of some major ions is evident, epitomised by chloride, decreasing from ~11,000 mg/l to 300 mg/l. At a typical moisture content of 20% for ash landfills, the L/S ratio of 0.93 is equivalent to ~4.6 bed volumes, so dilution of conservative components by ~100 times would be expected in a completely mixed reactor. The chloride results are therefore broadly consistent with this model. They are matched by declines in conductivity and ionic strength.

- Calcium concentrations decreased from 1000 mg/l to 7 mg/l. However, this is unlikely to be due to washout, as the total mass leached is small and calcium is not conservative under the conditions within an ash landfill. It is most likely to be a result of carbonation and this is consistent with the long-term increase in alkalinity from ~100 mg/l to ~600 mg/l.

- Sulphate concentrations increased up to L/S 0.6, from ~3000 mg/l to ~6000 mg/l. This is likely to be also a result of carbonation removing calcium ions from solution. Subsequently, sulphate concentrations declined to just under 2000 mg/l, possibly indicating depletion and dilution of sulphate concentrations that are no longer controlled by solubility. It is also possible that sulphide formation is partly responsible for the decrease in sulphate concentrations.

- Magnesium concentrations are highly variable but are at similarly low levels to those reported at Lostorf (see Table 2.3).

- • pH values remained alkaline throughout the period. Only one result fell as low as 8.3, the level expected when carbonation is complete. Subsequent values were higher. Measurements of pH as well as Ca and alkalinity, in collected leachate samples can be misleading as an indicator of conditions inside the landfill, because absorption of CO_2 and precipitation of calcium carbonate can occur rapidly once leachate contacts the atmosphere.

- Reducing conditions developed within the landfill, becoming more strongly so with time. Unfortunately, no further redox data were reported beyond L/S 0.55. The relative contributions of hydrogen formation and biological processes are not known. However, the formation of sulphide (up to 80 mg/l) and the continued lowering of the redox potential over many years (L/S 0.55 was reached in 1990), suggests that biological processes may have been significant.

- The initial leachate contained significant concentrations of ammoniacal nitrogen (~50 mg/l) and COD. Back extrapolation suggests the initial COD may have been of the order of ~250 mg/l. Both components declined, possibly as a result of simple washout. Modern facilities may be expected to have lower concentrations as a result of more efficient combustion.

- Heavy metal concentrations, even at low L/S ratio, were generally lower than the eluate concentrations reported by Hjelmar (1996). Four metals: Zn, Cu, Ni and Pb, exhibit long-term increases, suggesting significant changes in internal pH and redox conditions (see Figure 2.3).

2.3.2 Overview of MSW incinerator bottom ash leachate

A number of key conclusions can be reached on the quality of leachate from bottom ash landfill, as inferred from research studies or reported from full-scale landfill.

- Concentrations of major ions are typically in the range 1,000 to 10,000 mg/l in initial leachates at low L/S ratios.

- The relative proportions of the major ions are variable, especially those of chloride and sulphate: sulphate concentrations often exhibit an inverse relationship with calcium concentrations due to the limited solubility of calcium sulphate. Calcium concentrations are themselves dependent on the composition of the unburnt waste and the extent of carbonation. Usually, at low L/S ratios, chloride concentrations exceed sulphate.

- Sodium is the dominant cation, followed by potassium, and calcium. Magnesium concentrations are usually very much lower (10 to 100 mg/l).

- Chloride concentrations exhibit washout at high L/S ratios roughly corresponding to the expectation for a completely mixed reactor. At a L/S ratio of ~1, dilution by ~2 orders of magnitude has been demonstrated.

- Alkalinity is lower than in a MSW leachate of comparable TDS, and a significant proportion of the alkalinity may come from hydroxide ions and $Al(OH)_4^-$ ions, rather than from carbonate/bicarbonate.

- Leachate pH values may vary between near neutral and strongly alkaline, depending on the calcium content of the unburnt waste, the extent of carbonation within the landfill and the flow régime.

- Organic carbon and ammoniacal nitrogen concentrations may be similar to those of a very dilute MSW leachate, if the ash contains significant levels of unburnt organic compounds. However, very much lower concentrations can also occur.

- Elevated concentrations (1 to 10 mg/l) of some metals may occur, mainly Mo, W, Cu, Pb and Zn.

- Conditions within the landfill may be oxidising or reducing, depending upon hydrogen generation from metallic Fe and Al, and on the extent of biological activity. Under reducing conditions the leachate may contain several tens of mg/l of sulphide.

- Flow characteristics of water percolating through bottom ash are complex, with some short-circuiting and some matrix flow, giving a considerable amount of short-term variability in leachate strength.

- There is a tendency for extensive scale formation to occur in leachate collection systems if access of air is allowed, particularly while carbonation within the landfill is incomplete. Unless bottom ashes are specifically pre-carbonated, it can take decades for this to occur within the landfill.

- No information has been obtained on trace organics, from the data examined so far in this study.

2.4 MSW incinerator fly ash and air pollution control (APC) residues

A wide variety of processes and process combinations are available for the removal of fine particulates, acid gases and other air pollutants from incinerator flue gases. The solid wastes that result from them may be collected and disposed of separately or combined. Fly ash, the fine particulate matter (1 to 500 μm) carried over from the combustion chamber, is usually collected separately from other air pollution control (APC) residues, for example in an electrostatic precipitator. Scrubbing processes for the subsequent removal of gaseous and trace organic pollutants (e.g. HCl, SO_2) may be wet, dry or semi-dry processes. They produce wastes in the form of sludges or filter cakes. Often, fly ash and other APC residues are landfilled together, at hazardous waste sites, and these days are almost always kept separate from other hazardous wastes. However, in some cases, APC sludges are landfilled separately from the fly ash, in cells receiving mixed hazardous wastes.

The solid matrix of fly ash and APC sludges contains a high proportion of soluble inorganic salts, mainly calcium chloride, sodium chloride, calcium sulphate and calcium hydroxide. Some of the heavy metals (especially zinc and lead) are volatilised in the combustion chamber and captured in the APC residues. Fly ash may also contain a proportion of incompletely burnt organic matter.

2.5 Leachate quality from fly ash/APC residues

Hjelmar (1996) has indicated the typical ranges of contaminant levels in initial leachates from mixed fly ash and APC residues. These are derived primarily from leaching tests and laboratory studies and are shown in Table 2.4. Hjelmar *et al.* (1999) have reported results for total contaminant release during batch leaching tests on semi-dry APC residues admixed with fly ash. For a liquid/solid (L/S) ratio of ~1.92, the average leachate concentrations from these tests are given in Table 2.5.

Table 2.4 Maximum concentrations of contaminants in leachates from mixtures of MSW incinerator fly ash and APC residues (Hjelmar, 1996)

Range	Units	Fly ash and dry or semi dry APC residues	Fly ash and sludge from wet scrubbing
>100	g/l	Cl, Ca	
10 – 100	g/l	Na, K, Pb	Cl, Na, K
1 – 10	g/l	Zn	SO_4, Ca
100 – 1000	mg/l	NVOC, SO_4	
10 – 100	mg/l		
1 – 10	mg/l	Cu, Cd, Cr, Mo	NVOC, Mo
100 – 1000	μg/l	As	
10 – 100	μg/l		As, Cr, Zn
1 – 10	μg/l		Pb
<1	μg/l	Hg	Cd, Cu, Hg

* NVOC = non-volatile organic carbon; roughly equivalent to TOC

Table 2.5 Contaminant release during batch leaching tests (after Hjelmar *et al.*, 1999)

Parameter	Average concentration	Units
chloride	92,000	mg/l
sulphate	1400	mg/l
Ca	37,600	mg/l
Na	9,600	mg/l
K	20,400	mg/l
Pb	800	mg/l
Zn	25	mg/l
Cu	3.76	mg/l
Cd	20	μg/l
As	<80	μg/l
Ni	<40	μg/l
Cr	<12	μg/l

Note that these are averages, based on total mass released up to L/S 1.92, so it may be assumed that initial concentrations could be significantly higher. Chloride was the major anion, and calcium the major cation, while Pb was released at extremely high concentrations.

2.5.1 Leachate quality from landfill sites

Leachate quality data have been obtained from three EU landfills, (sites C, D and E in Appendix 3) with summary details as follows.

- At Site E, mixed APC residues including fly ash have been mono-filled in bulk, in six cells, each with separate leachate collection and sampling chambers. Three of the cells received residues from a semi dry scrubbing process and three received residues from a wet scrubbing process. Time series data have been obtained for all six cells, from their inception (the earliest being 1989) up to December 2000.
- At Site C, fly ash has been mono-filled in 'big bags' in a cell that began operation in 1993. Time series data from 1993 to 2000 have been obtained.
- At Site D, fly ash has also been mono-filled in big bags. Limited time series data from a cell filled during the period 1993-96 have been obtained.

Site E waste cells - semi-dry process residues

Wastes were emplaced in three 6m deep cells, finished to a flat profile, membrane covered, soiled and seeded. The start dates for the cells were 1989, 1996 and 1997. The cells are clay lined and have leachate collection systems. The base of the cells is below the local water table level and some ingress of groundwater is suspected by the operator. Dry weather flow data for the whole landfill suggest this could contribute as much as 50% of the overall flow (see Appendix 3). In capped cells, where rainfall ingress is restricted, it is possible that the contribution of groundwater becomes greater than 50%. Estimated L/S ratios from rainfall ingress are low, at ~0.1 to ~0.23 (see Appendix 3).

Time series graphs of the data received are shown in Figures 2.4a and 2.4b (1989 cell), 2.5a and 2.5b (1996 cell), and 2.6a and 2.6b (1997 cell). The following interpretative comments are made.

- Concentrations of major components are very peaky in all three cells.
- The leaches contain high concentrations of inorganic salts, with TDS typically 100 to 150,000 mg/l at the peaks. The lower TDS at the end of the record, in the 1989 and 1996 cells, may reflect dominance by groundwater ingress, short-circuiting most of the waste mass, after capping.
- The relative abundance of major ions is:

$Cl >> SO_4$ (no data for alkalinity, bromide or NO_x)
$Ca \approx Na > K$ (Na not measured; abundance estimated by comparison of TDS).

- pH values in collected leachate are mostly near neutral but spikes to high pH values occur (up to pH ~11). This is perhaps indicative of variable flow hydraulics as identified in a bottom ash landfill (Johnson *et al.*, 1998).
- Concentrations of most metals are low compared with typical UK MSW leachates, but Pb and Cd are high and extremely variable: Pb varies over four orders of magnitude, with peaks exceeding 1,000 mg/l and many results being several tens of mg/l. Cd varies over a huge range, with most results at several tens or hundreds of µg/l, but occasional peaks up to several mg/l.
- The heavy metals in general fall into the following concentration bands:

1 – 1000	mg/l	Pb, Fe
100 – 1000	µg/l	Zn, Cu, Cr, Cd
10 – 100	µg/l	Ni, Cd
1 – 10	µg/l	-
<1	µg/l	Hg

- The two most prevalent heavy metals, Pb and Zn, are nevertheless at considerably lower concentrations than reported for eluates by Hjelmar (1996).

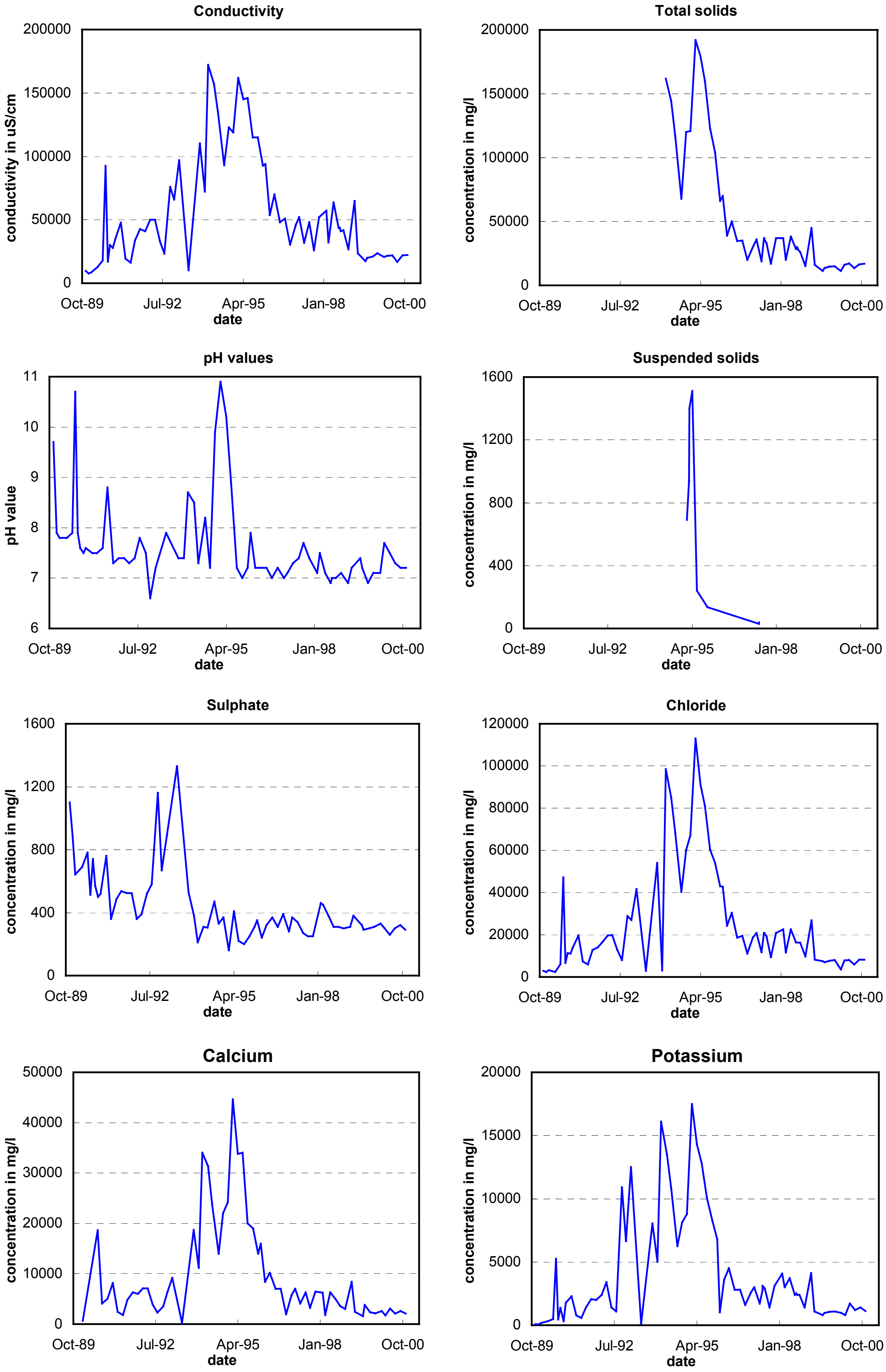


Figure 2.4a **Site E leachate quality data from fly ash and semi dry APC residues, 1989 cell: general parameters**

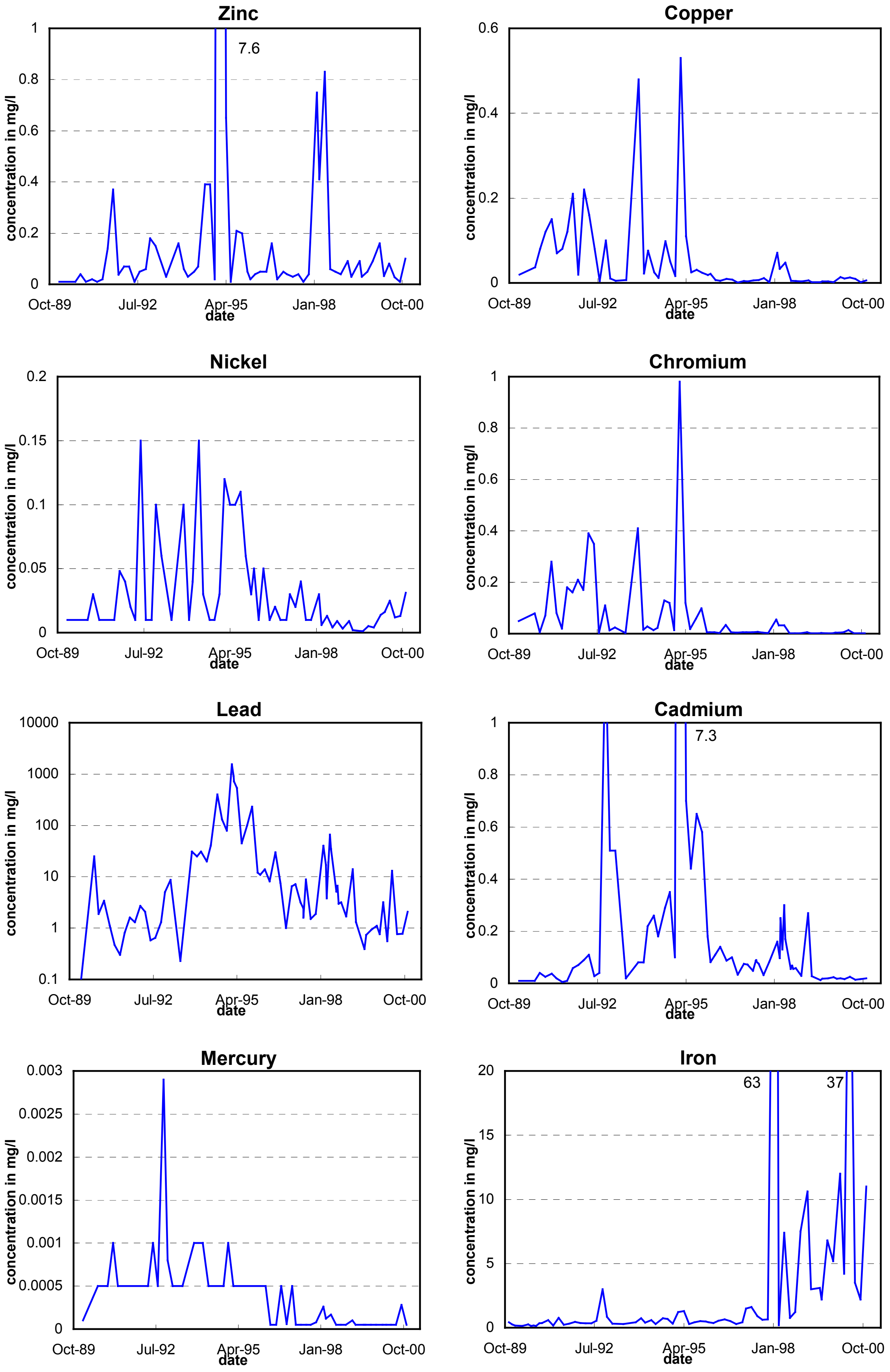


Figure 2.4b **Site E leachate quality data from fly ash and semi dry APC residues, 1989 cell: heavy metals**

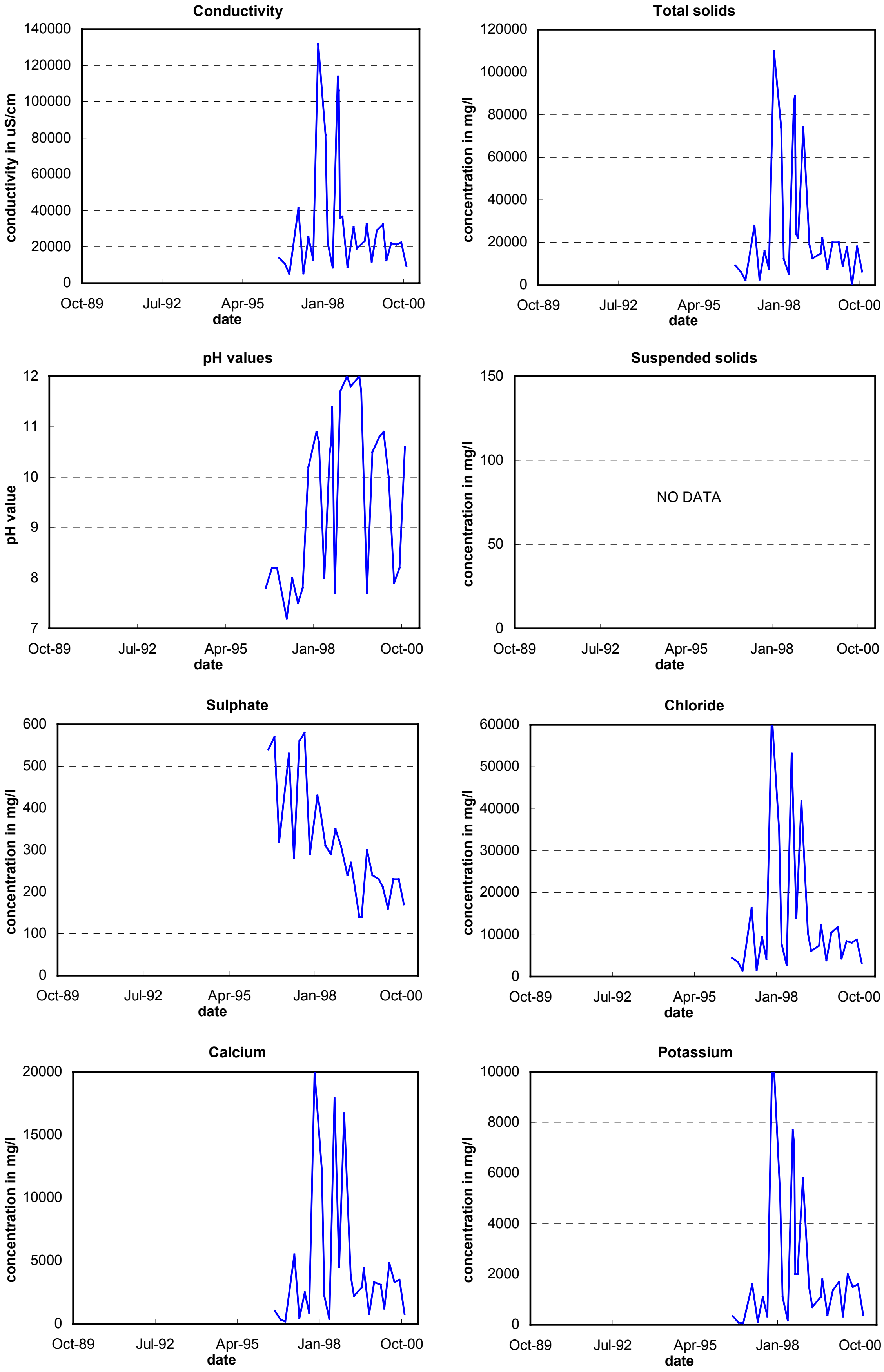


Figure 2.5a **Site E leachate quality data from fly ash and semi dry APC residues, 1996 cell: general parameters**

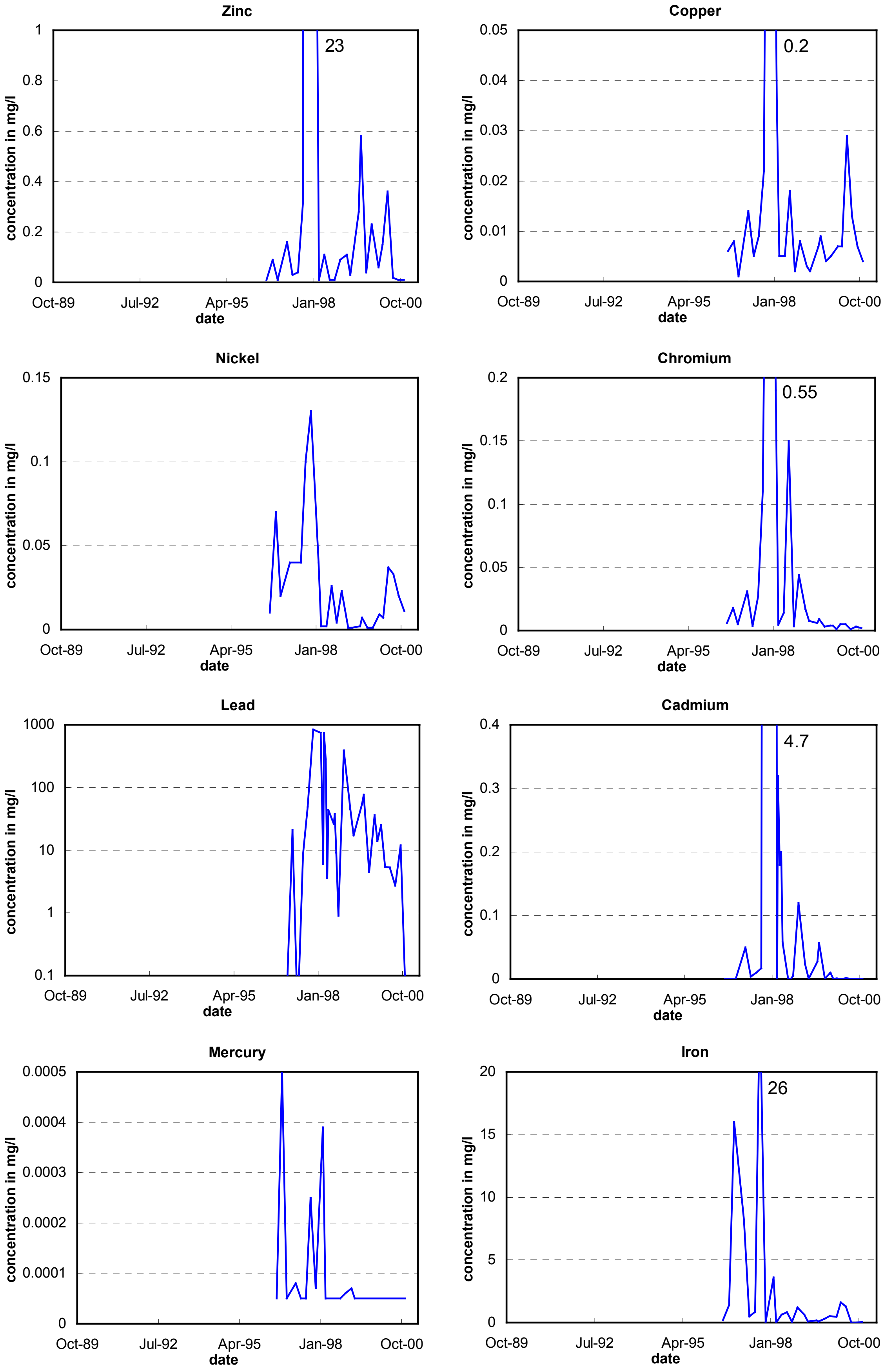


Figure 2.5b **Site E leachate quality data from fly ash and semi dry APC residues, 1996 cell: heavy metals**

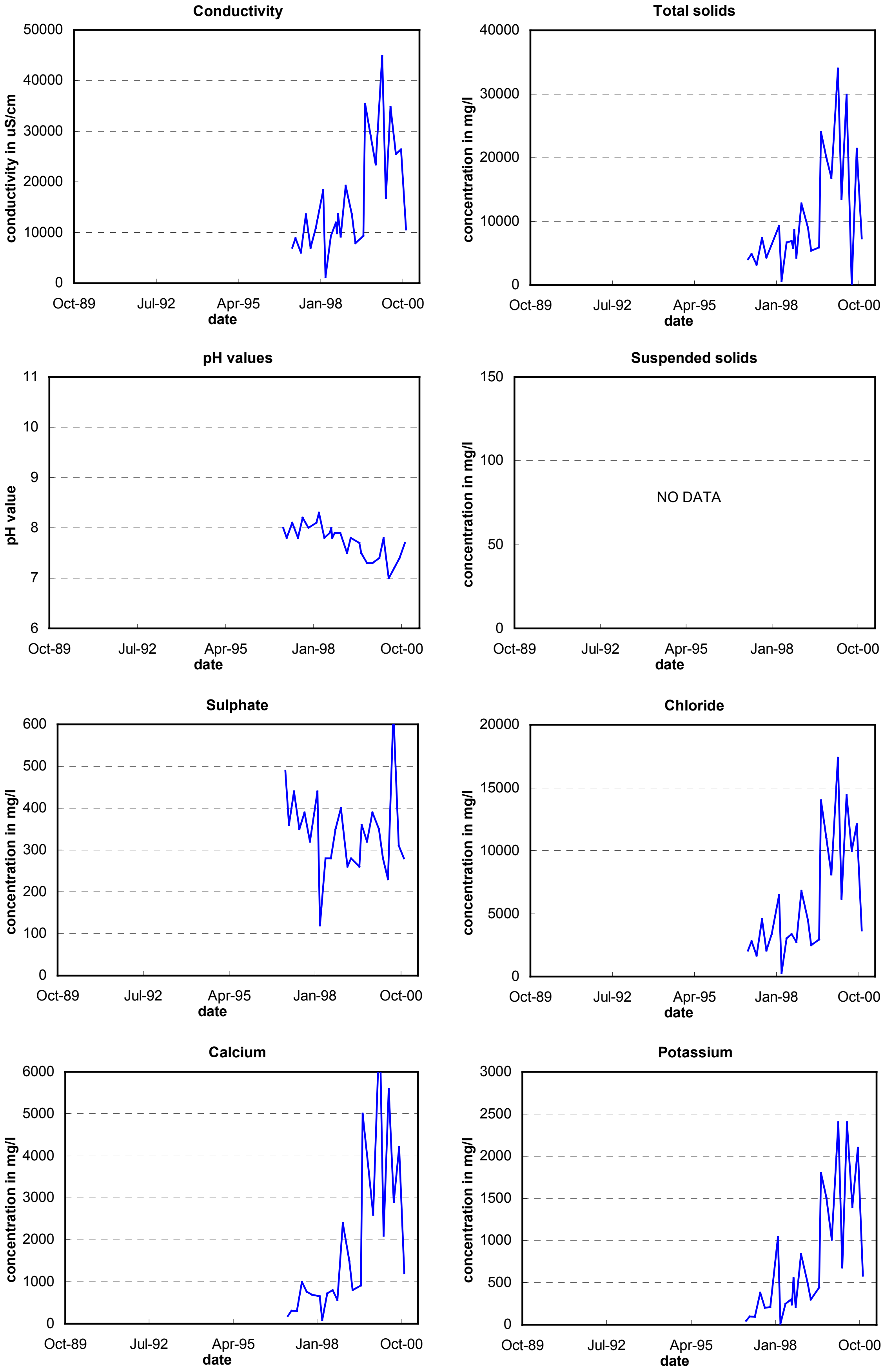


Figure 2.6a Site E leachate quality data from fly ash and semi dry APC residues, 1997 cell: general parameters

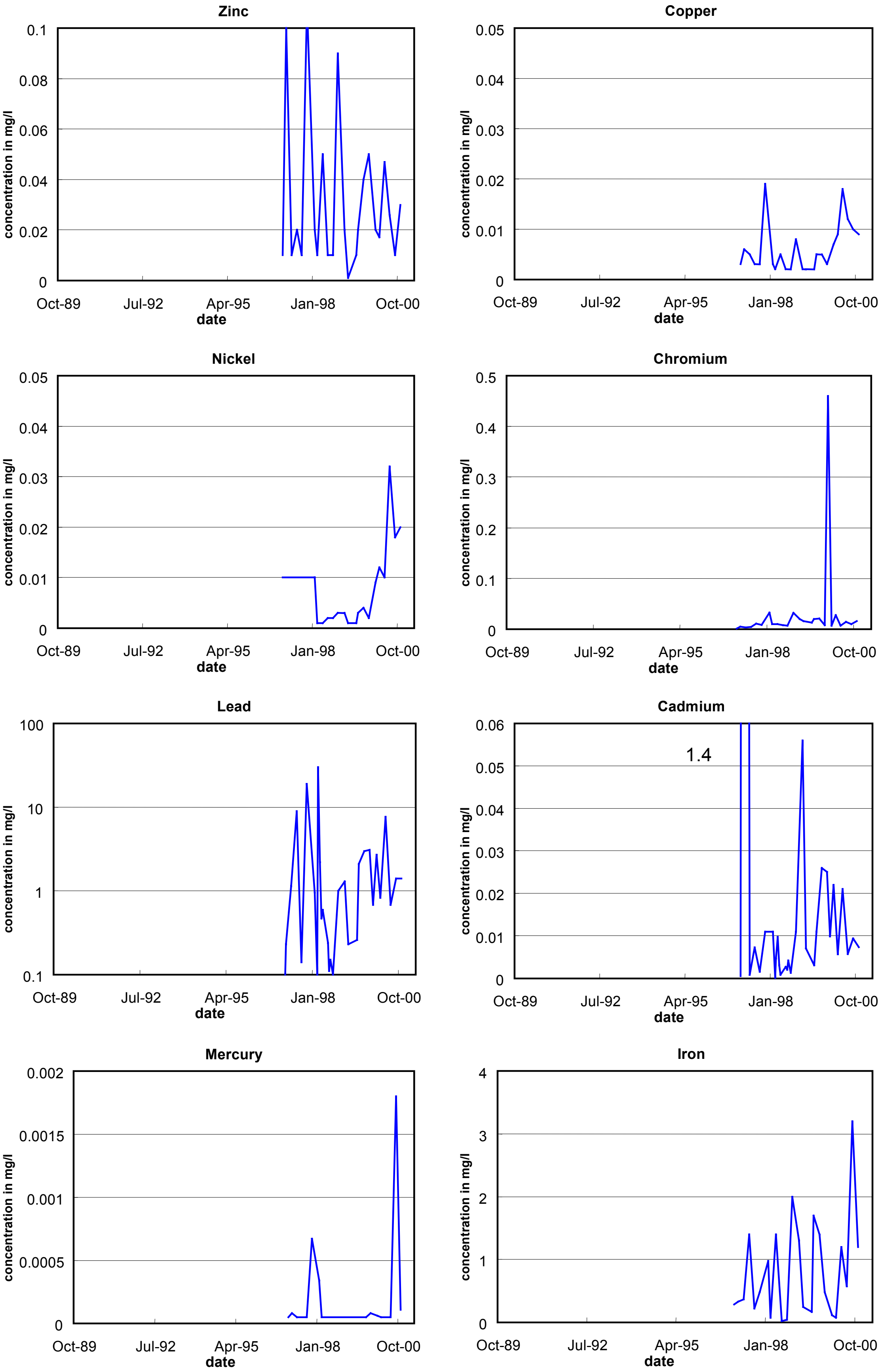


Figure 2.6b Site E leachate quality data from fly ash and semi dry APC residues, 1997 cell: heavy metals

Site E waste cells - wet scrubbing process residues

Wastes were emplaced in three 6m deep cells, finished to a flat profile, membrane covered, soiled and seeded. The start dates for the cells were 1993, 1997 and 1998. The cells are clay lined and have leachate collection systems. The base of the cells is below the local water table level and the operator suspects some ingress of groundwater. Dry weather flow data for the whole landfill suggest this could contribute as much as 50% of the overall flow (see Appendix 3). In capped cells, where rainfall ingress is restricted, it is possible that the contribution of groundwater becomes greater than 50%. Estimated L/S ratios from rainfall ingress are low, at ~0.1 to ~0.23 (see Appendix 3).

Data on filtered (0.45 μm) and unfiltered samples, provided by the operator, indicated that a significant proportion of Pb, and to a lesser extent Cd, may be associated with particulates, but that soluble concentrations were nevertheless high (Pb up to several hundred mg/l; Cd several tens of μg/l). The results contain no data for redox potential, organic indicators, nitrogen compounds or As.

Time series graphs of the data received are shown in Figures 2.7a and 2.7b (1993 cell), 2.8a and 2.8b (1997 cell) and 2.9a and 2.96b (1998 cell). The following interpretive comments are made.

- Concentrations of major components are very peaky in all three cells.
- TDS in the 1997 cell is anomalously low, at around one sixth of those in the other cells. This may reflect the effects of groundwater ingress or may indicate short-circuiting of surface water. The results from the 1997 cell are therefore regarded as being unrepresentative of this waste type.
- The leachate in the 1993 and 1998 cells contains high concentrations of inorganic salts, with TDS typically in the range 50 to 150,000 mg/l. The highest concentrations occur before capping. This would be consistent with post-capping flows being derived to a greater extent from groundwater ingress, that by-passes much of the waste mass.
- The relative abundance of major ions is:
  Cl >> SO_4 (no data for alkalinity or NO_x)
  K ≈ Na >> Ca (Na not measured; abundance estimated by difference, from TDS data);
- pH values are close to neutral, with very few spikes to higher pH values (max 9.5).
- Heavy metal concentrations are mostly low, except for Pb and Cd. Pb concentrations again span several orders of magnitude but are lower than in the leachate from dry scrub residues, while Cd concentrations are similar.

1 – 1000	mg/l	Fe, (Pb)
100 – 1000	μg/l	Pb, Zn, (Cd), Ni)
10 – 100	μg/l	Cu, Cd, Ni, Cr, (Zn), (Pb)
1 – 10	μg/l	(Cu), (Ni), (Hg)
<1	μg/l	Hg

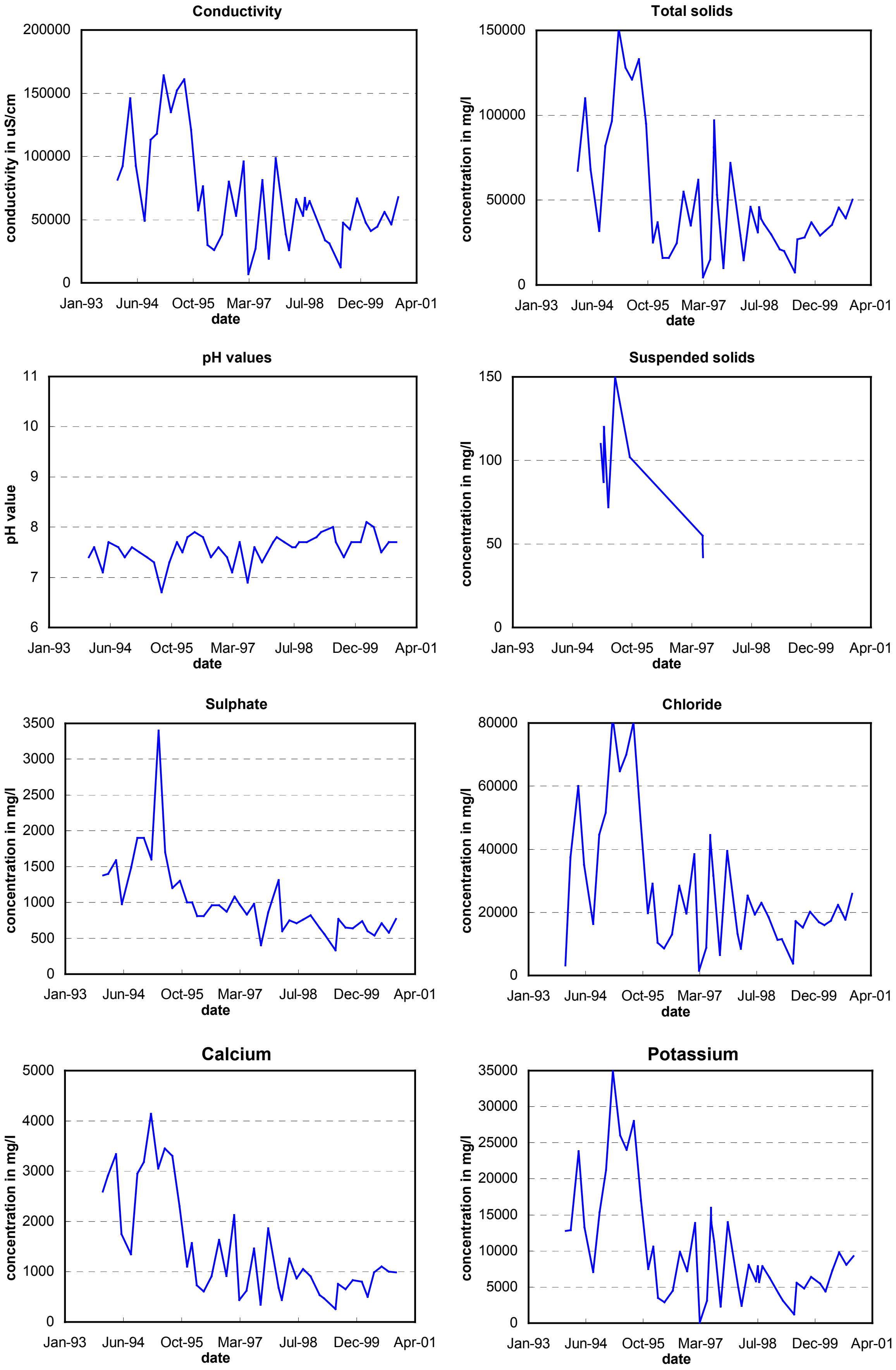


Figure 2.7a **Site E leachate quality data from fly ash and wet scrub APC residues, 1993 cell: general parameters**

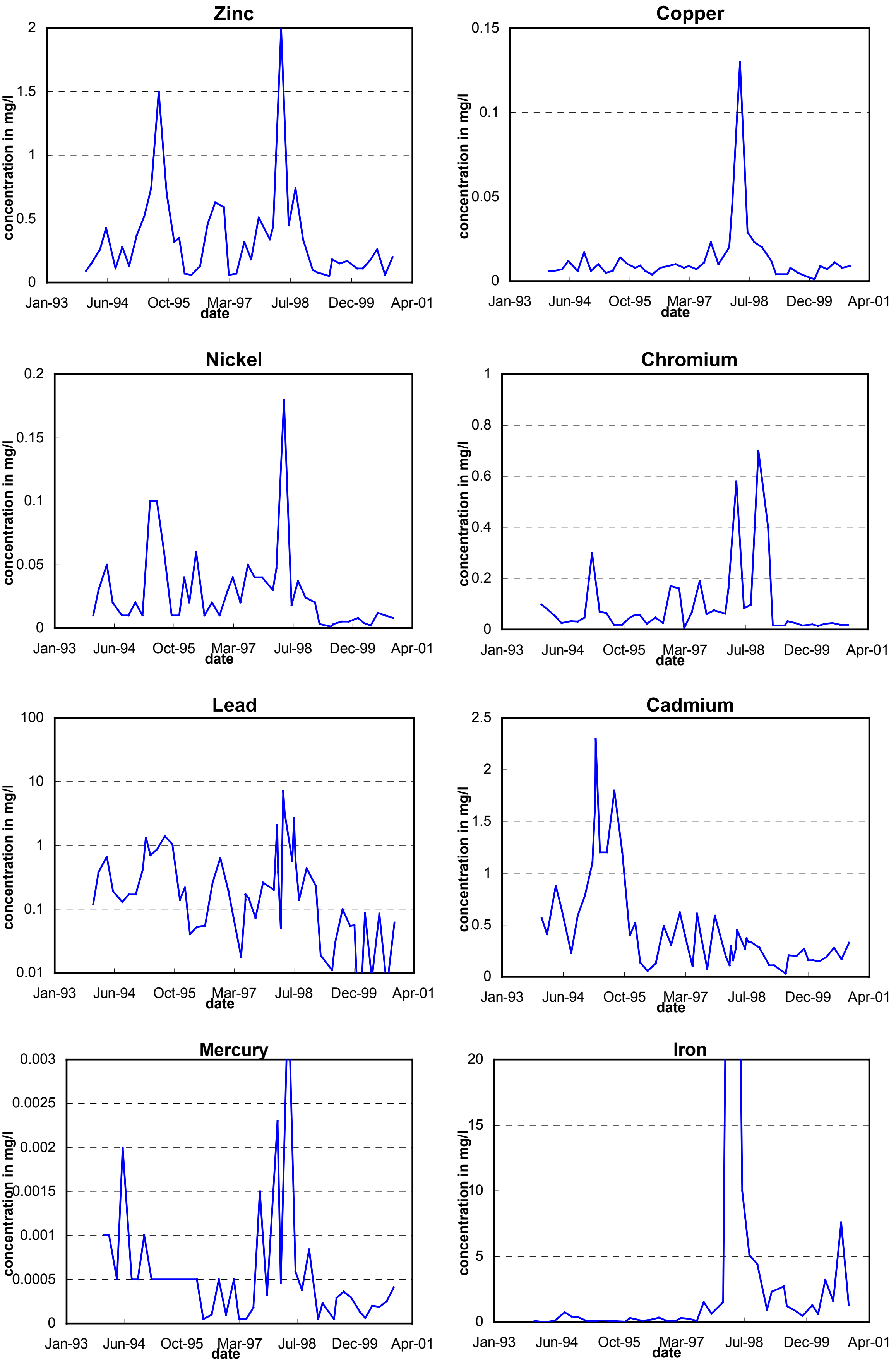


Figure 2.7b **Site E leachate quality data from fly ash and wet scrub APC residues, 1993 cell: heavy metals**

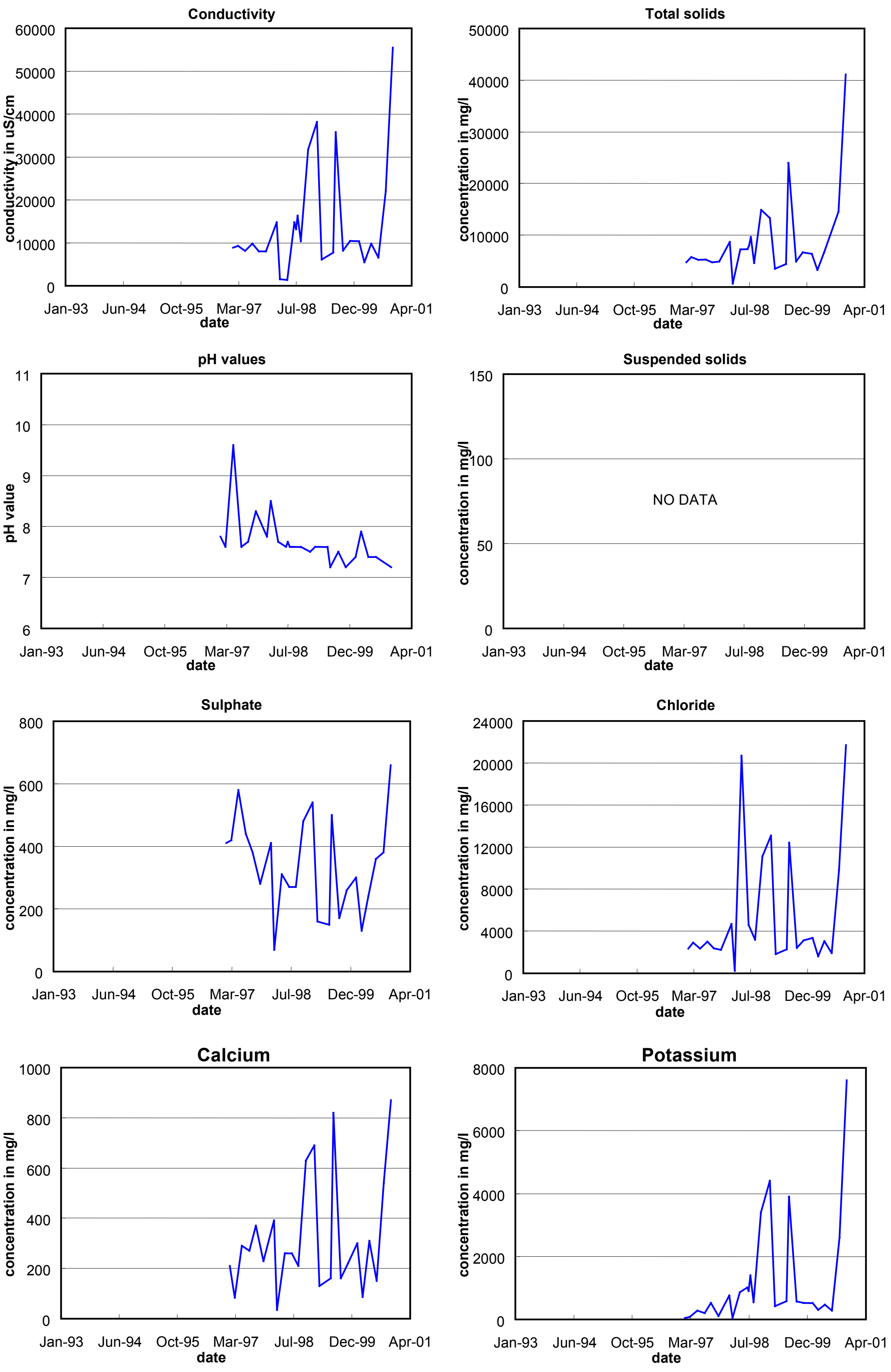


Figure 2.8a **Site E leachate quality data from fly ash and wet scrub APC residues, 1997 cell: general parameters**

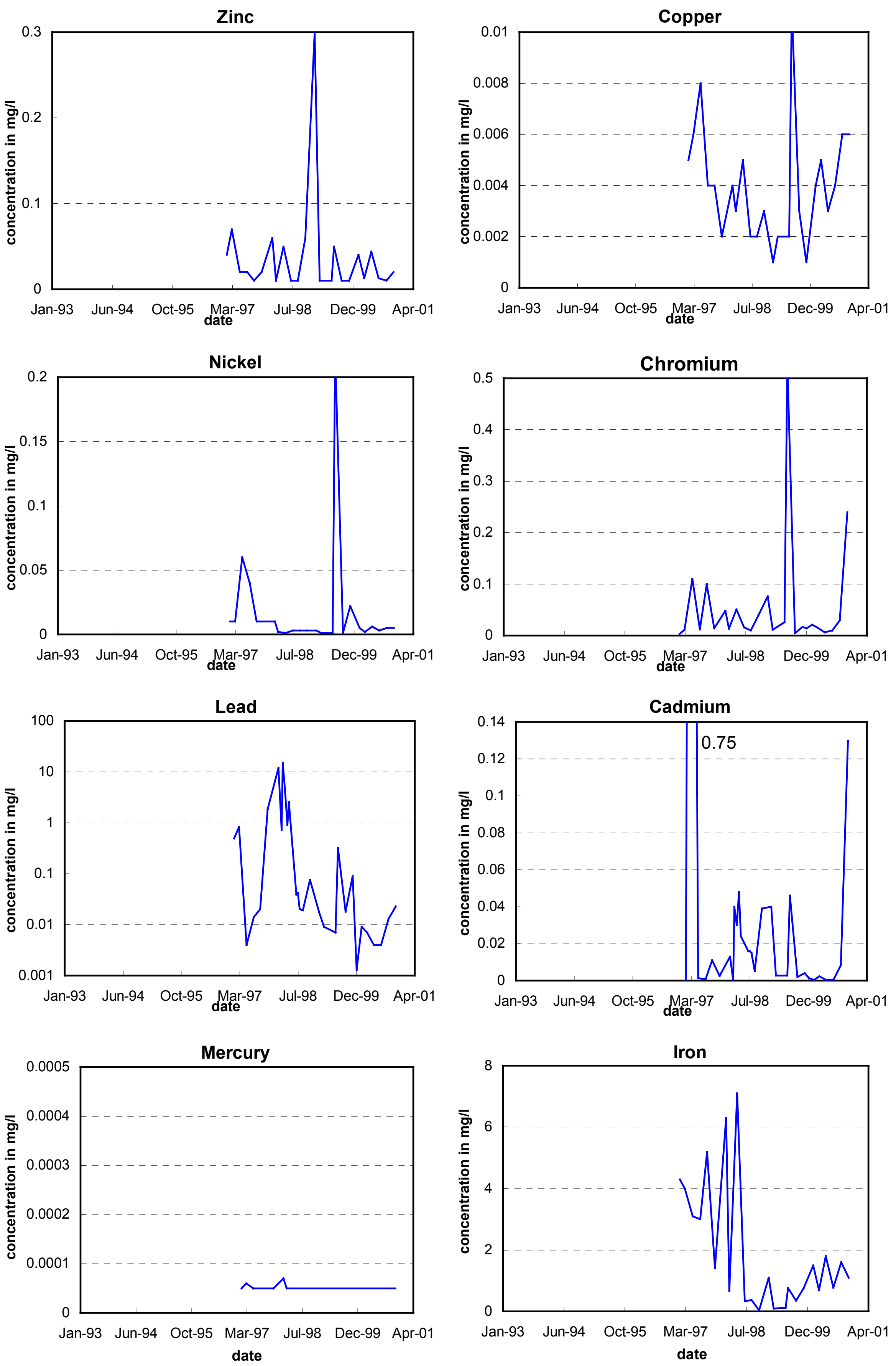


Figure 2.8b **Site E leachate quality data from fly ash and wet scrub APC residues, 1997 cell: heavy metals**

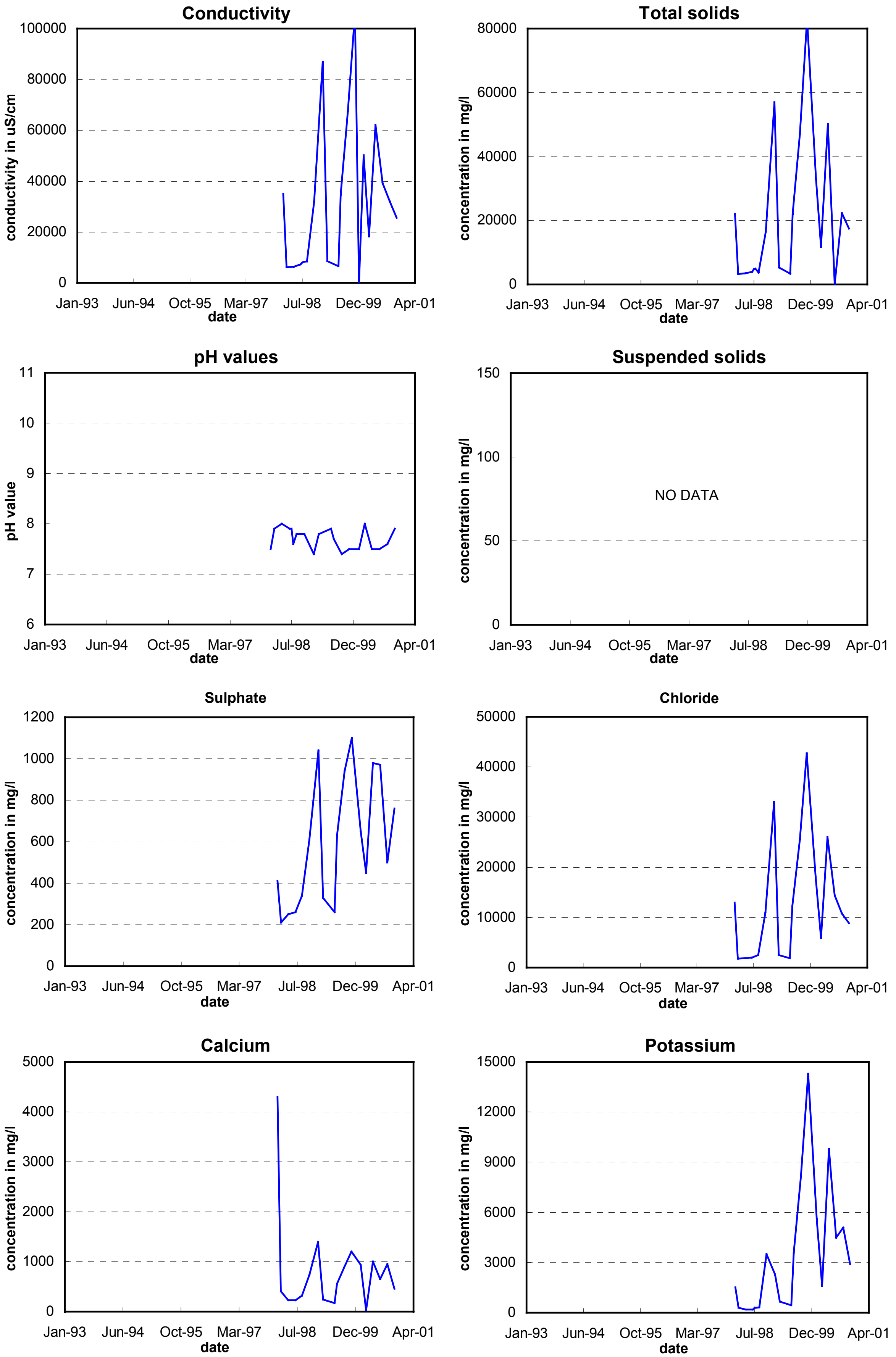


Figure 2.9a Site E leachate quality data from fly ash and wet scrub APC residues, 1998 cell: general parameters

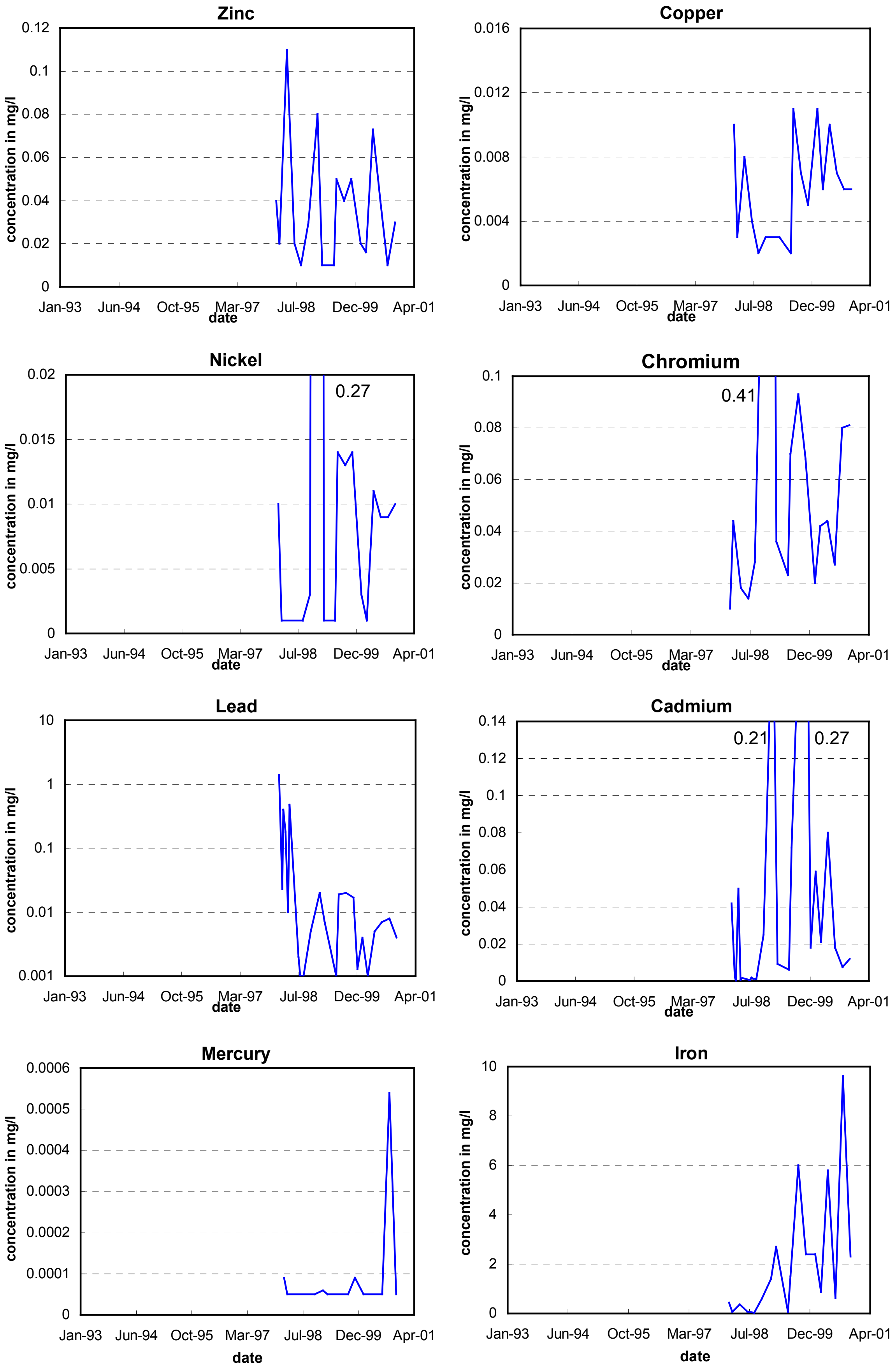


Figure 2.9b Site E leachate quality data from fly ash and wet scrub APC residues, 1998 cell: heavy metals

Site C - fly ash in "big bags"

Site C is an engineered containment site, located above the water table. Big bags containing fly ash are mono-filled and the spaces between them filled with clean sand to assist physical stability of the mass. It is not known whether the fly ash also contains other APC residues. When final levels are reached, the cells are capped with a geomembrane and bentonite and seeded with grass. The cell for which data have been obtained is thought to be partly up to level, and hence partly restored. The L/S ratio for the cell is not known but is expected to be low, given its young age and the partial capping. The relevance of L/S ratio is minor, for wastes emplaced in big bags, because most infiltration will by-pass the bags, passing preferentially through sand-filled spaces.

Analytical data for a restricted number of parameters have been obtained for a cell begun in 1993. The data received are annual averages of quarterly samples. The data for most parameters are shown as time series graphs in Figure 2.10. In addition, analysis for minor organics showed: PAH <1 µg/l, oil ≤1 mg/l and EOX ~5 mg/l

The following interpretive comments may be made.

- Despite the use of big bags, significant concentrations of contaminants occur in the leachate.
- As expected, with annual averages, the data do not exhibit the peakiness that was observed in individual measurements at Site E. Daily leachate flows may exhibit greater variation of quality than the data in Figure 2.10.
- The concentration of chloride, at 1,000 to 5,000 mg/l, is one to two orders of magnitude lower than at Site E. This is not unexpected, given the use of big bags. However, sulphate concentrations (1000 to 2000 mg/l) are comparable with those from bulk deposits at Site E. Other major ions were not analysed.
- Despite its apparently dilute nature, the leachate contains significant concentrations of organic matter and nitrogen: COD concentrations are in the region of 50 to 300 mg/l and TKN typically 5 to 25 mg/l. Given the dilute nature of the chloride concentrations, the COD levels are consistent with TOC concentrations in the range 100 to 1000 mg/l quoted by Hjelmar (1996) for initial leachates from bulk deposits.
- pH values are consistently near neutral.
- Concentrations of heavy metals are low:

100 – 1000	µg/l	Zn, (Pb)
10 – 100	µg/l	Pb, Ni, Cr, Cu

- No results were received for redox potential, major cations, alkalinity, NO_x, As, Cd, Hg, Mo.

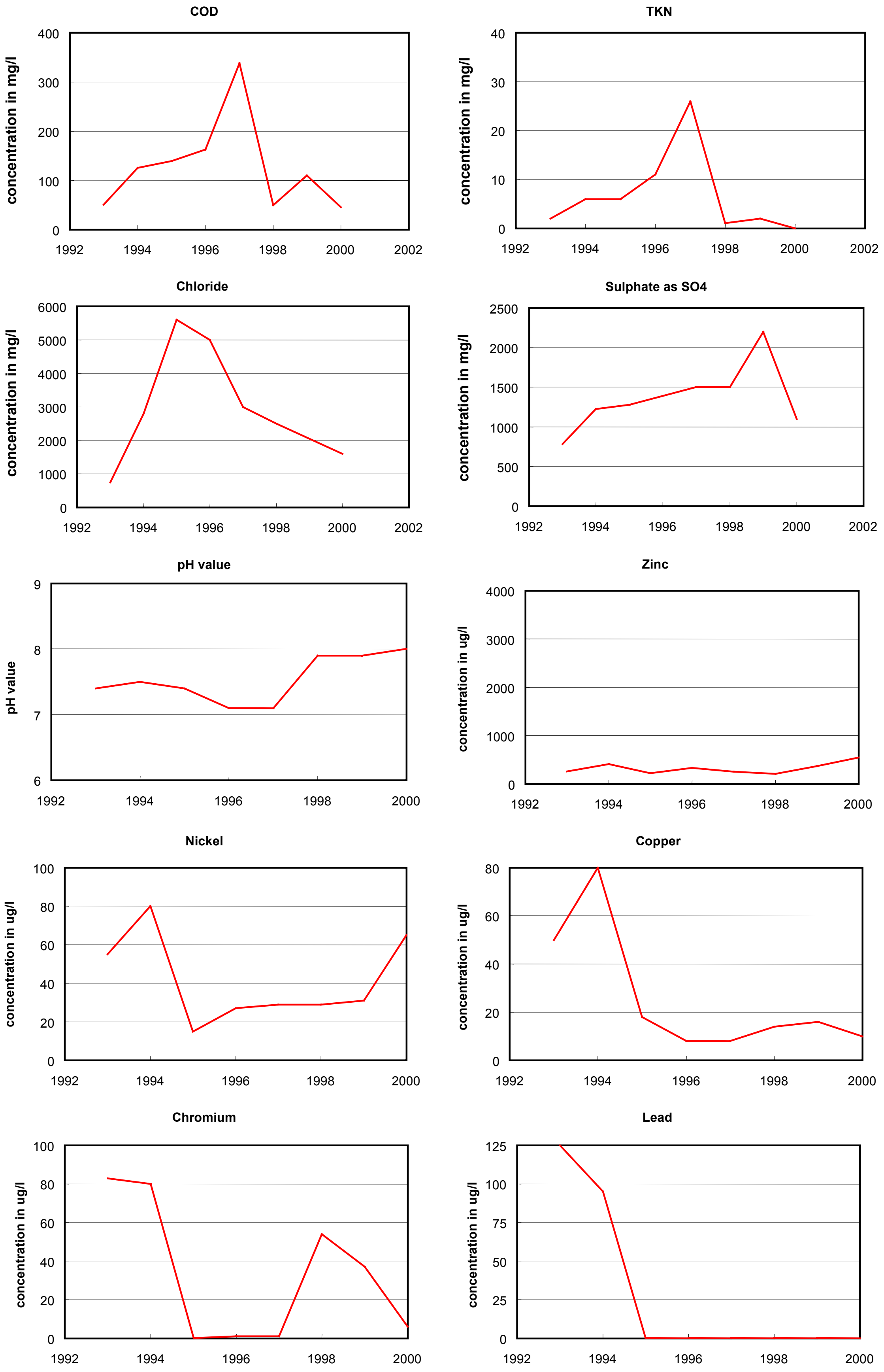


Figure 2.10 **Site C leachate quality data from cell containing MSW incinerator fly ash in big bags**

Site D - fly ash and APC residues in big bags

Data have been obtained for a cell filled during the period 1993 to 1996. A mixture of fly ash (30%) and APC residues (70%) was landfilled in big bags, in a containment cell with leachate collection system. Clean sand was poured into the spaces between the bags, to improve physical stability. The cell was membrane capped when up to final level. The operator's view was that the bags are supposed to be impermeable, so that any contamination in the leachate indicated a leaking bag. The L/S ratio for the cell is estimated to be low, at ≤0.1. It is expected that most percolating water would pass through the sand-filled channels between the bags.

The analytical results received for the cell are shown as time series graphs in Figure 2.11. The following comments may be made.

- Despite the use of big bags, significant concentrations of contaminants occur in the leachate.
- Concentrations of major parameters are extremely spikey, fluctuating over a wide range.
- TDS levels are similar to those at Site C, where big bags were also used: chloride concentrations are mostly in the range 5,000 to 20,000 mg/l. Other major ions were not analysed.
- pH values in the collected leachate are near neutral.
- There is a significant organic content, with COD typically 100 to 200 mg/l, spiking up to >700 mg/l. These are similar to the concentrations at Site C.
- There is a low but significant nitrogen content, with TKN typically ~5 mg/l, spiking up to ~25 mg/l, similar to Site C.
- Pb concentrations are much lower than from the bulk fly ashes and APC residues, but Cd concentrations are still typically several tens or hundreds of μg/l, and Zn is typically at mg/l levels, despite the wastes being in bags.
- Analyses for trace organic groups gave the following:

PAH	typically sub-μg/l; maximum 4 μg/l;
BTEX	all sub-μg/l;
EOX	typically sub-μg/l, maximum 2.4μg/l
oil	typically <50 μg/l, maximum 140 μg/l

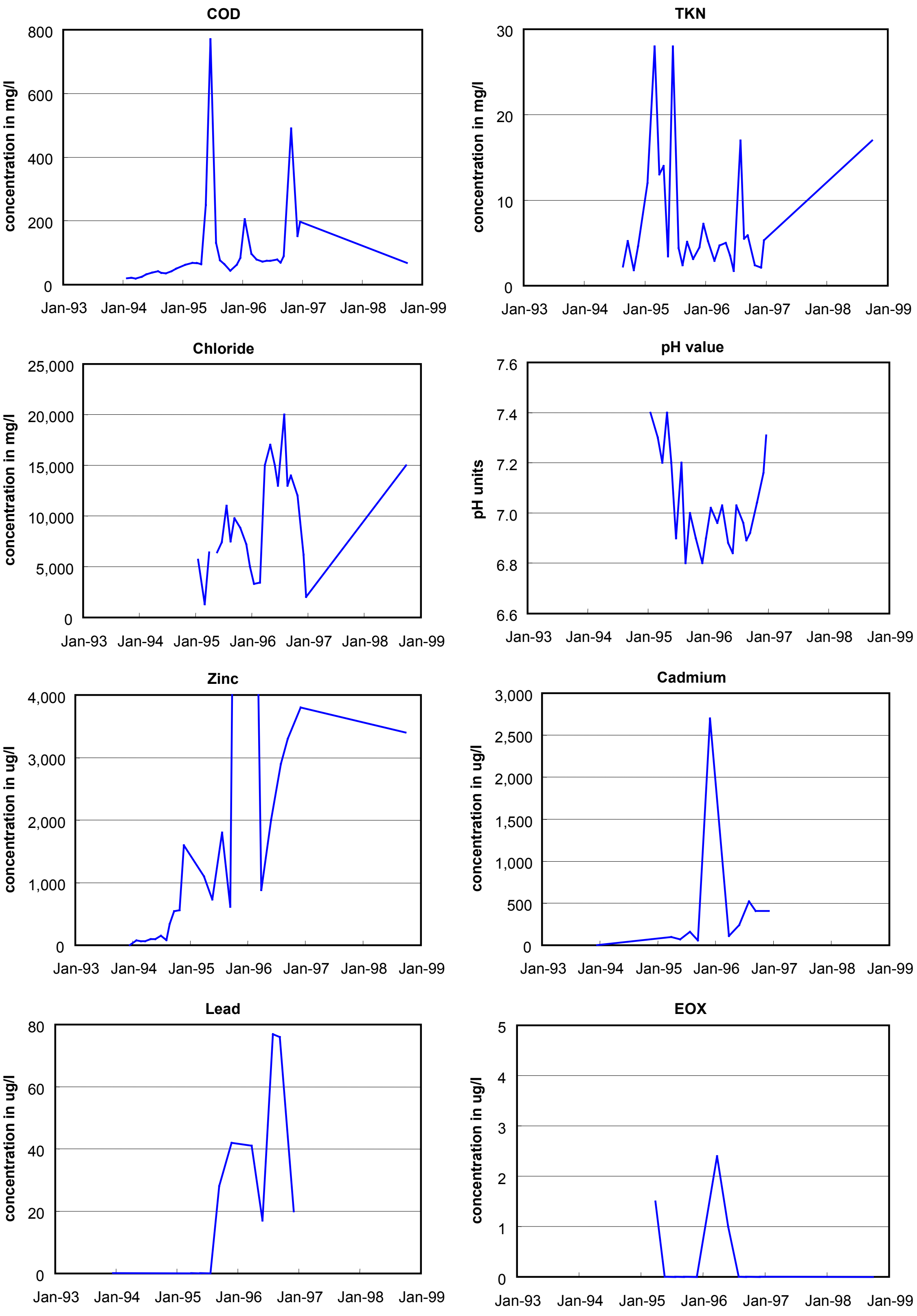


Figure 2.11 Site D leachate quality data from cell containing MSW incinerator fly ash and APC residuesin big bags

2.5.2 Overview of leachates from MSW incinerator fly ash/APC residues

- The leachates contain high concentrations of dissolved mineral salts, with TDS typically in the range 50,000 to 150,000 mg/l.
- Chloride is the dominant anion, with sulphate much lower, often as low as ~1000 mg/l or less.
- The dominance of the cations depends upon the type of scrubbing process: calcium is the major cation in the leachate if the residues are from a dry or semi-dry process. Sodium and potassium greatly exceed calcium in leachates from wet scrubbing process residues.
- Concentrations of major components are often spikey, showing extreme fluctuations from one sample to the next. The reasons for this are not known but may be due to dual phase hydraulic characteristics of the materials (as with MSW incinerator bottom ash) and the management of the leachate collection system (saturated or unsaturated).
- pH values are predominantly near neutral, but occasional spikes up to pH 11 suggest that strongly alkaline conditions may persist within the waste matrix. The erratic values in collected leachates and the predominance of neutral values may be due to a combination of carbonation in the leachate collection system (and perhaps along major flow channels in the wastes) and the two phase flow behaviour noted by Johnson *et al.* (1998) for bottom ash.
- Leachate may contain COD concentrations of several hundred mg/l. No corroboration by TOC analysis has been found, and interference from the high chloride concentrations cannot be completely ruled out. However, the levels are consistent with those reported by Hjelmar (1996).
- No information (e.g. BOD results) has been obtained on the nature of any organics present. Some trace organic analysis has shown that PAH, BTEX and EOX are all very low, typically sub-µg/l, while oil is <1 mg/l.
- The leachate TKN is typically several mg/l, peaking as high as ~25 mg/l. One operator found that the TKN is mainly ammoniacal nitrogen.
- High concentrations of Pb and Cd are present:
  - Pb highly variable, greater in semi-dry process (~1000 mg/l), lower in wet process residues (≤10 mg/l);
  - Cd typically several tens of µg/l, up to several hundred µg/l, and occasionally at mg/l levels. The field data from Site E show similar concentrations from semi-dry and wet-scrubbed residues. This is in contrast to the ranges reported by Hjelmar (1996) for leaching test eluates, which indicated sub-µg/l concentrations from wet-scrub residues. This may be due, in part, to the absence of carbonation, and consequent high pH values, in laboratory leaching tests;
    heavy metals in general may be very variable in collected leachates, due to the contrasting redox and pH conditions that may develop in the leachate collection system, flow channels within the waste mass and in interstices between the waste.
- None of the sites visited showed any evidence of gas generation or other evidence of biological activity in cells receiving fly ash and APC residues.
- No field data have been obtained for L/S ratios >0.25.
- No analyses have been found for some parameters of interest to this project, including oxidised nitrogen and trace organics.

2.6 Proposed leachate source term values

The data set from the Phase 2 study (Environment Agency, 2004) is generally consistent with the chemical analyses reported from both UK and European incinerator residues. Kappa values were estimated for raw (untreated) and carbonated bottom ash and APC residues, using the comparison of leaching test results at LS1 (column) and LS10 (batch tests). The kappa value is derived from:

Kappa value = $\ln [LS_1] - \ln [LS_{10}]/10 - 1$

For some parameters where data are missing, kappa values have been estimated. For the acid neutralised APC residue, no column tests were done so the kappa values for carbonated waste have been used.

The maximum concentration, C_o, has been estimated from the kappa values and either C_{LS10} or C_{LS1}. The spread of the data from the three incinerators provides a high end and low end estimate of the maximum value of C_o. The high end estimate is taken as the maximum point of the PDF for each parameter. The low end estimate has been divided by the hydraulic factor of 5, to give the minimum point of the PDF. The most likely value still has to be evaluated, by reference to the shape of the PDF in the existing LandSim model.

For most APC residue parameters, the average C_o values were very similar to the maximum values obtained from lumped Site E data (see Section 2.5.1). This provides some justification for the use of the kappa values derived from the experimental results. The proposed source term data for raw bottom ash and APC residues are summarised below and full details are provided in Environment Agency (2004).

2.6.1 Bottom ash (untreated)

The proposed source term data presented in Table 2.6 below is derived from a limited number of laboratory tests using L/S10 batch leach tests and a recirculating upflow column test at L/S1. This information should be used to inform professional judgement when carrying out a groundwater risk assessment for a particular landfill site.

Sources of variation

Inputs of some non-municipal waste to the incinerator (e.g. plasterboard) could affect the lime content of the ash, and hence the leachate pH and natural carbonation processes. Bottom ash from modern incinerators is likely to meet a higher standard in terms of screening and burn-out efficiency in comparison to the ash that gave rise to the leachate quality reported by Johnson *et al*. (1999) and Hjelmar (pers. comm.). The extent of pre- and post-separation could affect the metal and unburned organic content of the waste. The degree of burn-out (unburned organic content) and the development of biological processes within the landfill could affect pH and redox potential, as well as the TOC and nitrogen content of the leachate. This, in addition to carbonation and hydrogen generation could affect metal concentrations.

Co-disposal with other wastes, e.g. fly ash (see Section 2.3.1) or non-hazardous wastes with low organic content, may influence both the landfill processes and leachate quality.

The hydrological setting of the landfill can also give rise to significant variation in leachate quality, thought to be due to the significance of flow along preferential pathways or through the waste matrix (e.g. Johnson *et al*., 1998).

Table 2.6 Proposed contaminant range and Kappa values from Phase 2 laboratory studies (from Environment Agency, 2004)

Parameter	Units	Kappa[*1]	PDF max		PDF min[*2]
			High	Low	
Conductivity	µS/cm	0.078	17,000	8,000	1,600
TOC	mg/l	0.255	800	500	100
COD	mg/l	0.240	2,000	1,250	250
BOD	mg/l	0.220	1,100	750	150
NH_4-N	mg/l	0.275	20	15	3
TKjN	mg/l	0.200	45		10
Na	mg/l	0.242	1,300	1,000	200
K	mg/l	0.238	1,300	800	160
Ca	mg/l	-0.010	1,000	200	40
Mg	mg/l		15		3
Cl	mg/l	0.229	2,700	1,800	360
SO_4	mg/l	0.021	500	40	8
NO_x-N	mg/l	-0.050	0.5	0.5	0.5
Alkalinity pH 4.5 as $CaCO_3$	mg/l	0.051	3,500	900	180
As	ug/l		1	1	0.2
Hg	ug/l	-0.426	0.02	0.01	0.002
Cd	ug/l		1	1	0.2
Cr	ug/l	-0.100	25	7	2
Cu	ug/l	0.261	17000	6000	1200
Pb	ug/l	-0.074	5000	300	60
Ni	ug/l	0.173	180	80	16
Zn	ug/l	-0.061	1300	50	10
Sb	ug/l	0.060	30		5
Mo	ug/l	0.180	440	320	64
V	ug/l		5	5	5
Ba	ug/l	0.070	1000	400	80
Tl	ug/l		<10		
Se	ug/l		<1		
Sn	ug/l		1	1	0.2

[*1] Kappa value derived from an average of two values

[*2] derived from the low PDF max divided by hydraulic factor of 5

Long-term aftercare liabilities

Many of the major components are at or below their concentrations in MSW leachates, including NH_4-N, which is at levels close to typical discharge consent levels. These components would therefore lead to an expectation of a shorter flushing time than for an MSW landfill.

The default source term concentration proposed for copper could lead to its being the component that determines the flushing time. Typical consent levels for discharges to surface waters are in the range 20 to 100 µg/l Cu. Since the copper appears to be controlled by organic complexation, its behaviour may be more or less conservative. Dilution by 50 times to 250 times could therefore be necessary. This equals or exceeds that needed for MSW landfills. To the extent that biological activity modifies the nature and concentration of the soluble organic components, this could reduce the copper concentration and hence the flushing time. However, the nature and fate of the DOC remains unknown at present.

Treatment and disposal

Most of the major ion components could be discharged to sewer without pre-treatment or to surface water after simple biological treatment to remove degradable organics.

Several of the heavy metals (especially Cu, Pb, Zn) would require treatment. It is not known whether these would be effectively removed during conventional biological treatment. If not, additional methods would be needed, including methods of breaking down strong complexes or of removing the metal in its complexed form.

2.6.2 APC residues (untreated)

The proposed source term data presented in Table 2.7 below is derived from a limited number of laboratory tests using L/S10 batch leach tests and a recirculating upflow column test at L/S1. This information should be used to inform professional judgement when carrying out a groundwater risk assessment for a particular landfill site.

Table 2.7 Proposed contaminant range and Kappa values from Phase 2 laboratory studies (from Environment Agency, 2004)

Parameter	**Units**	**Kappa**[*1]	**PDF max**		**PDF min**[*2]
			High	Low	
Conductivity	µS/cm	0.123	210000	150000	30000
TOC	mg/l	0.2E	200	50	10
COD	mg/l	0.2E	600	200	40
BOD	mg/l	0.2E	120	20	4
NH_4-N	mg/l	0.290	35	3	1
TKjN	mg/l	0.290	100	4	1
Na	mg/l	0.178	12000	6000	1200
K	mg/l	0.171	19000	7000	1400
Ca	mg/l	0.036	55000	10000	2000
Mg	mg/l	0.040	1	1	1
Cl	mg/l	0.178	120000	80000	16000
SO_4	mg/l	-0.036	1200	500	100
NO_x-N	mg/l	0.2E	14	0	0
Alkalinity pH 4.5 as $CaCO_3$	mg/l	-0.142	700	600	120
As	ug/l		1	1	0.2
Hg	ug/l	-0.430	1	0.001	0.0002
Cd	ug/l	0.05E	400	5	1
Cr	ug/l	0.05E	500		100
Cu	ug/l	0.058	1200	70	15
Pb	ug/l	-0.068	600000	4000	800
Ni	ug/l	0.05E	150		30
Zn	ug/l	0.007	3000		600
Sb	ug/l	0.05E	20		4
Mo	ug/l	0.019	600	200	40
V	ug/l	0.05E	100		20
Ba	ug/l	0.200	44000	30000	6000
Tl	ug/l	0.05E	50		10
Se	ug/l	0.173	110	5	1
Sn	ug/l	0.05E	50		10

[*1] Kappa value derived from an average of two values

[*2] derived from the low PDF max divided by hydraulic factor of 5

E = estimated

Sources of variation

The eluate concentrations of some minor components (e.g. Hg, Se) varied by more than an order of magnitude for raw APC residue from three UK plants sampled for Phase 2 of this study (Environment Agency, 2004), despite having very similar pH values. The causes of this kind of variation between plants are not known but appear to be related to operating conditions.

A significant source of variation is the type of scrubbing process used. The values below are for residues from semi-dry scrubbing processes, the dominant type in the UK. Residues from wet-scrubbing may contain significantly lower concentrations of lead, zinc, calcium and chloride, and higher concentrations of sulphate.

The hydrological setting of the landfill can lead to significant fluctuations of all parameters, up to an order of magnitude. This is thought to be due mainly to flow conditions within the landfill (e.g. Johnson *et al.*, 1998, 1999). It is likely that if heavy rain flushed out high pH matrix leachate from uncarbonated waste, concentrations of some metals, such as Pb and Cd could be temporarily higher than those suggested.

Long-term aftercare

The parameters likely to control the flushing requirement are chloride and lead. Estimates of the dilution needed from raw and carbonated APC residues (from Environment Agency, 2004) are shown in Table 2.8 below.

Table 2.8 Dilution calculations for chloride and lead

Source and components	units	Typical leachate concentration	WAC C_o value for inert waste	Dilution/reduction needed
APC residue from MSW incinerator				
Chloride	mg/l	100,000	460	217
Pb	mg/l	20	0.15	133
Carbonated APC residue from MSW incinerator				
Chloride	mg/l	100,000	460	217
Pb	μg/l	500	150	3.3

These show that dilution by up to 200 times may be needed for conservative parameters. It is unlikely that this would be sufficient for lead, leaching from untreated APC residue. Its concentration is equilibrium controlled rather than availability controlled. Some studies have shown elevated concentrations continuing to be released at very high L/S ratios (20:1 and above).

Carbonation clearly has a dramatic effect on long-term liabilities, and would reduce the long-term leaching requirement for lead, and other heavy metals, but not mobile species such as chloride.

Treatment and disposal

The high salinity may limit the volumes dischargeable into non-saline water bodies without the use of separation techniques such as reverse osmosis and evaporation. Treatment requirements are likely to be physical-chemical in nature and to require the removal of heavy metals and dissolved ions. In certain coastal situations or where large dilution occurs in the receiving water it may be possible to avoid the need to reduce the dissolved major ion concentrations.

Even after carbonation or acid treatment, removal of heavy metals from the leachate is likely to be necessary before discharge to surface waters.

3 Residues from MBP of MSW

3.1 MBP of MSW in the UK and Europe

Around 28 Mt of MSW was produced in England and Wales in 1998/99, and this is estimated to increase by around 3% a year (DETR, 2000). Currently 83% of the MSW is disposed of to landfill without prior treatment. It is estimated that, if the current rate of growth continues, by 2020 around 33 Mt of biodegradable MSW will need to be diverted from landfill in England and Wales alone. The government has acknowledged that recycling and composting of MSW is an essential part of achieving MSW recovery targets (*op cit.*, 2000).

Biological treatment of waste, principally composting, is currently an expanding industry in the UK and, in 1999, >800,000 t of material was composted (92% in England), including >600,000 t of diverted MSW (Slater *et al.*, 2001). However, well over 95% of the MSW composted is green waste, diverted from civic amenity sites (householder-delivered), parks and gardens, and kerbside collection schemes. Composting technology in England is predominantly open-air mechanically turned windrows (88%), compared to 4.5% in-vessel and 4% contained mechanically turned windrows.

Residues from composting of green waste in the UK are successfully recovered, with over 80% used as a product, primarily as mulch or soil conditioner. The re-use of residues from MBP of MSW ("grey" waste) is subject to control under the Animal By-product Regulations 2003 (Statutory Instrument SI No. 1482/2003), and must be composted at facilities that meet national standards for catering waste.

A number of authors have considered the landfilling of residues from materials recovery facilities (MRF), variously termed mechanically sorted organic residues (MSOR), “residual wastes”, or residual municipal solid wastes (RMSW).

The terms MSOR, RMSW, or “residual wastes” are imprecise and poorly defined. They broadly represent those wastes that cannot be re-used or recycled, but in practice often comprises materials that pass through a screen at the MRF of maximum size typically between 40 mm and 100 mm. In general terms, in addition to the specific size of the screen being used, the qualities of the materials will be determined by the extent to which:

- wastes have been subject to separate source separation of, for example, kitchen or garden wastes; and
- wastes have been extracted for recycling at the MRF.

The polluting potential of landfilled residual wastes per tonne of material, is relatively high compared with untreated MSW – in part because of the smaller particle size, but also because of the higher content of readily-degradable organic materials. High concentrations of organic compounds and of ammoniacal-N are typical. Figure 3.1 below presents data from landfill simulation lysimeters operated for 400 days.

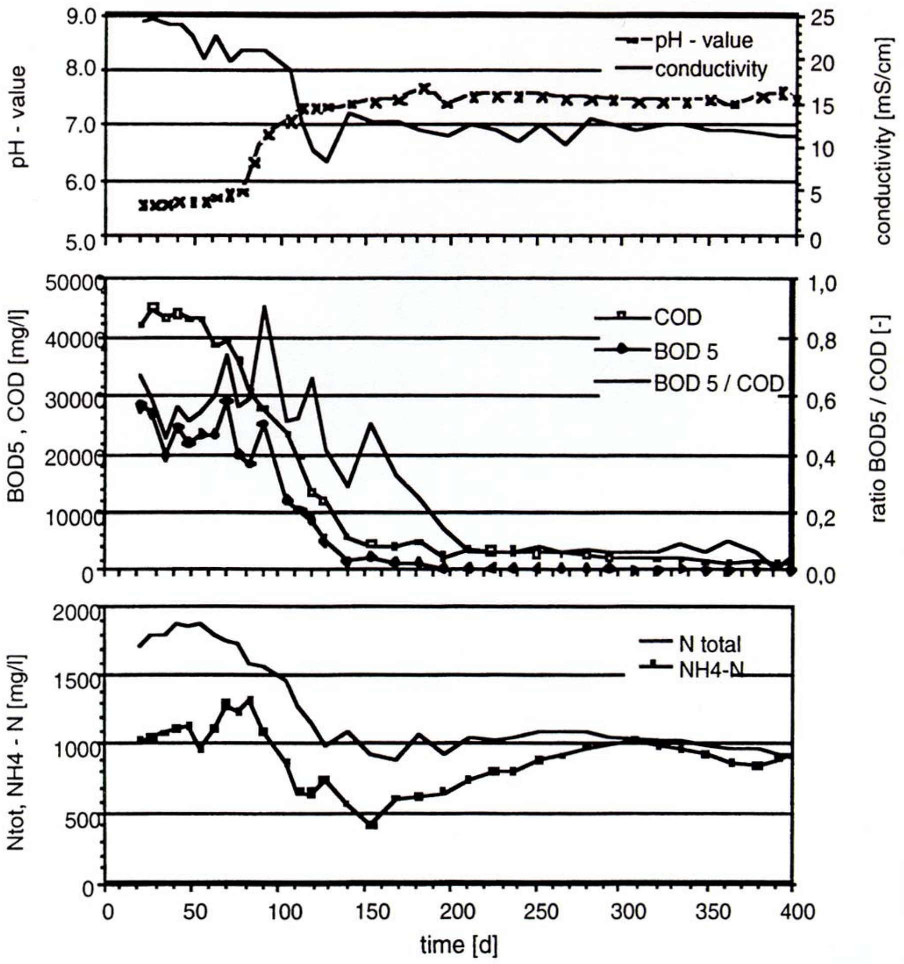


Figure 3.1 pH-value, conductivity, COD, BOD_5 and concentrations of ammoniacal-N in leachates from landfill simulation tests using untreated residual wastes (from Cossu *et al.*, 1998)

The great potential benefits to be gained by biological treatment of these residual wastes, in terms of leachate strength, can be seen clearly in results from equivalent tests also reported by Cossu *et al.* (1998). Figure 3.2 demonstrates this benefit for residual wastes that had been subjected to 4 months of efficient pre-treatment by composting.

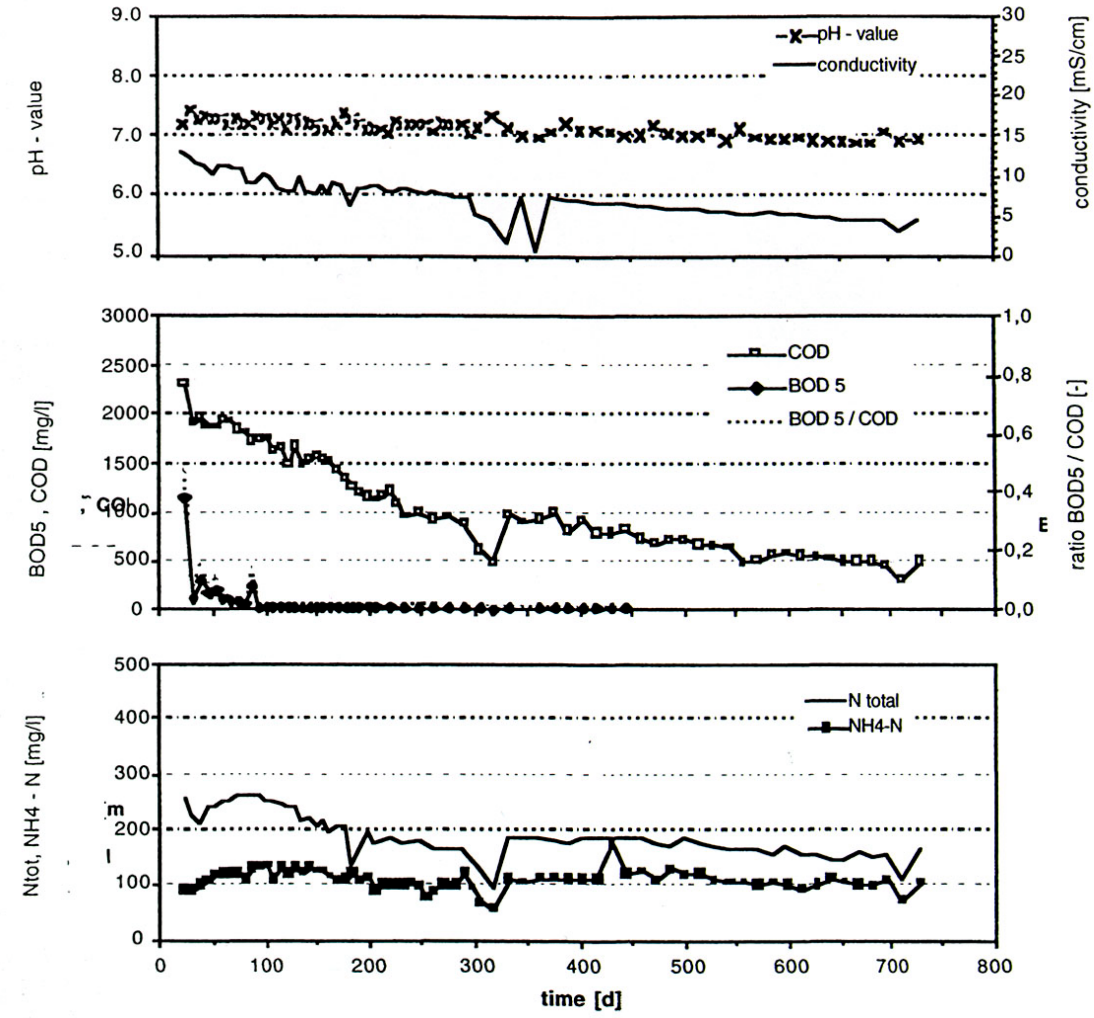


Figure 3.2 **pH-value, conductivity, COD, BOD$_5$ and concentrations of ammoniacal-N in leachates from landfill simulation tests using residual wastes pretreated by composting for 4 months (from Cossu *et al.*, 1998)**

3.2 Mechanical pre-treatment in the Netherlands

Municipal waste management policy in the Netherlands has been based on initial volume reduction through prevention and recycling for more than a decade. Well over half of all glass and paper has been recycled for at least 10 years. Remaining MSW is collected separately in two fractions: biowaste ("green") and residual MSW ("grey"). This selective collection of biowaste has increased rapidly during the 1990s, and in 1994, Government legislation demanded that all municipalities must collect biowaste separately. This waste is generally composted.

The "grey" fraction of MSW must be incinerated with energy recovery (electricity and heat), and, ultimately, residual fractions must be landfilled. Typically, "grey" waste might comprise just over half of the domestic waste stream (biowaste being about one quarter, and reused/recycled paper, glass and plastics together being about one fifth).

Grey waste is often separated mechanically into a fraction of less than 40 mm, termed MSOR (Mechanically Separated Organic Residue), with a larger sized fraction of RDF (Refuse Derived Fuel). The MSOR might typically represent just over one third of the grey wastes.

Historically, MSOR must be landfilled rather than composted, primarily because of contamination with heavy metals, with appropriate monitoring and control of leachate and gas emissions. However, Dutch legislation has banned the landfilling of MSOR after the year 2000, so “flushing bioreactor” technology is being considered for future treatment of this material. Nevertheless, there is some debate about whether this will fall within the definition of “treatment” under the EU Landfill Directive.

Valuable information has been derived from research undertaken by a waste disposal company (VAM) in the Netherlands during the 1990s, and is summarised below.

3.2.1 Characteristics of mechanically separated organic residues (MSOR)

VAM (now ESSENT) is one of the largest waste management companies in the Netherlands, presently handling about 0.8 Mte of domestic and industrial solid waste each year, at its facility in Wijster. Incoming “grey” wastes are mechanically separated into several fractions. Approximately 50% is recovered as RDF, which is subsequently incinerated. About 15% is recovered as reusable paper, plastics, and ferrous metals. The remaining fraction, about 35%, is that material which has passed through the 40mm sieve plates to become MSOR.

MSOR comprises vegetable, fruit and garden wastes not separately collected from households, plus inert fractions of sand, glass and gravel, with small amounts of paper and plastic particles. It is characterised by its relatively small particle size and its homogeneity, compared with normal MSW, and typically contains about 40 – 50% moisture, 25-30% inerts, and 25-35% dry organic material (all percentages by weight). Table 3.1 below shows a typical composition of MSOR (1992 data, after Woelders *et al*. 1993), for an (assumed) separate collection of biowaste, where the MSOR represents 30 percent by weight of total MSW.

Table 3.1 MSOR Composition (in percent by weight)

Fraction	% of wet MSOR
water	40.0
organic fraction	36.0
grit/sand/inorganic	10.8
glass	6.0
stones	3.6
paper	2.4
rigid plastics	0.6
residual	0.4
non-ferrous metals	0.12
ferrous metals	0.06
TOTAL (approximately)	**100.0**

Research at VAM commenced in 1990, with field tests on the landfill site (Woelders *et al*, 1993). A 3 metre layer of MSOR was emplaced, and studies were complemented by container tests that were carried out to determine water storage capacity. The following conclusions were derived:

- landfilling requires specialist equipment, because of the limited initial bearing capacity of the MSOR;
- stability appears to be sufficient, at slopes of 1 in 1;
- permeability is in the same order as MSW (i.e. 10^{-6} to 10^{-8} m/s);
- moisture content of about 40% is almost equal to the field capacity; and
- initial bulk density in the landfill was 0.9–1.0 t/m^3, but this had increased to 1.4-1.5 t/m^3 after several months.

The research was continued, using separate pilot-scale studies, to investigate the emission potential of landfilled MSOR, and the possibility of controlling degradation and leaching processes by means of leachate recirculation etc. Work was carried out in three sealed 300 mm diameter columns, 2 metres long, that were maintained at a temperature of 40ºC.

- Column 1 was basically a control column, irrigated at natural rainfall rates, after rapid initial saturation of the wastes from about day 90 onwards (until leachate production began). After it became clear that significant gas production in this column had not begun by day 229, subsequently the irrigation rate was increased to 5 times natural infiltration rates, and a solution of sodium bicarbonate (100 meq/l) was used to try and assist development of methanogenic activity.

- Column 2 was irrigated with methanogenic leachate from the VAM Landfill at a rate equivalent to twice natural infiltration. From day 75 to day 100 the irrigation rate was increased to 5 times natural infiltration, to saturate the wastes.

- Column 3 was irrigated with methanogenic leachate from VAM at a continuous rate of 5 times natural infiltration.

No data for concentrations of ammoniacal-N have been published, but Figure 3.3 below presents results for COD values (given in g/l), in leachates from the 3 columns.

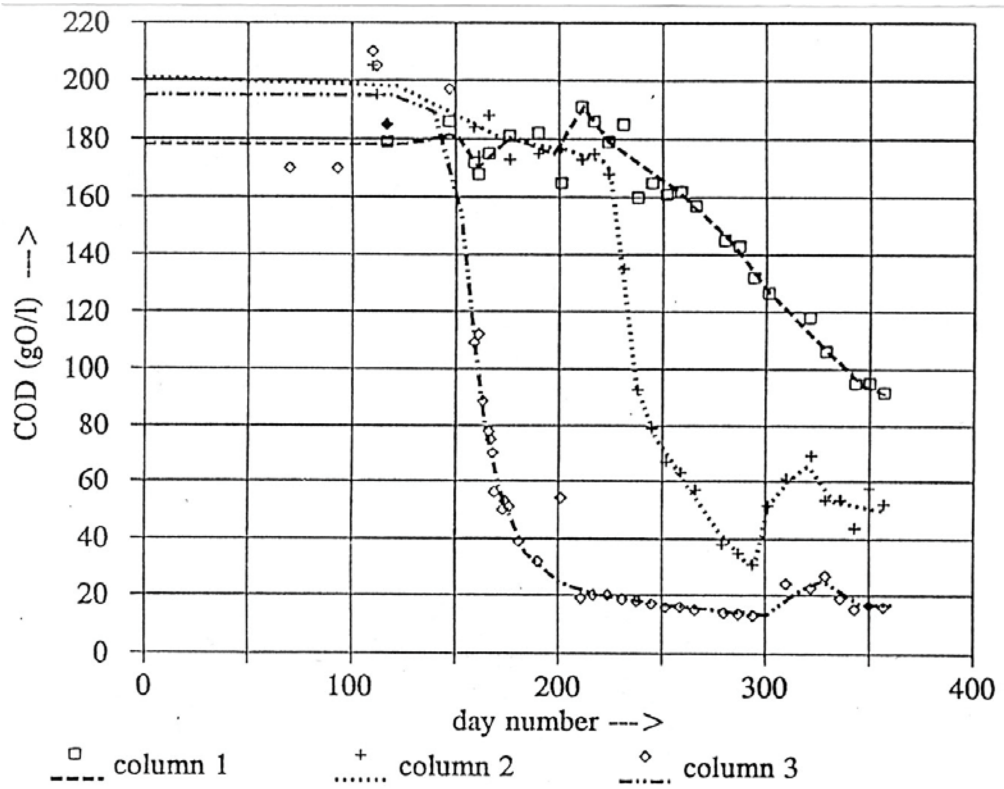


Figure 3.3 **Leachate COD in drainage from the 3 columns, over one year (results in g/l), from Woelders *et al.*, 1993**

The most important observation is that in all three columns, although only 2 m deep, initial COD values approached 200,000 mg/l, and in column 1 these levels continued for 7 months. Sodium bicarbonate was then added to buffer pH-values as low as 6.0, and allow methanogenic conditions to begin to establish by the end of the trial – although even then, COD values of 90,000 mg/l remained.

In the other two columns (2 and 3), where methanogenic leachate was irrigated, methanogenic activity established more quickly. Calculations demonstrated that over half (58 and 59%) of dry organic matter in the columns had been degraded and lost, primarily (>90%) as landfill gas – which was collected and analysed. Nevertheless, at the end of the 12-month study, COD values in leachates from columns 2 and 3 remained at 45,000 and 18,000 mg/l respectively.

3.2.2 Long-term behaviour of MSOR in a flushing bioreactor cell

In 1996 an integrated research programme by VAM began to examine the pre-treatment of MSOR in a flushing bioreactor. The objectives of the research were to investigate whether this would provide an alternative to landfilling of MSOR, which recent changes in Dutch waste policy were likely to prevent after the year 2000.

The concept of "flushing bioreactor landfills", or "enhanced fermentation cells" has been discussed extensively during the last decade. Although technologies adopted vary, a common feature is the recirculation of leachate, to achieve the enhanced and more complete biodegradation of waste materials, coupled with the flushing of pollutants from the waste mass.

The integrated research programme is summarised by Oonk and Woelders (1999a; 1999b), and has three parts:

1. enhanced treatment of MSOR at laboratory scale in 130 litre columns (Vroon *et al.*, 1999);
2. full-scale demonstration of the bioreactor in a 50,000 tonne test cell (Woelders and Oonk, 1999; Oonk *et al.*, 2000); and
3. a desk study of the characteristics of the final bioreactor product, where options are evaluated (van der Sloot *et al.*, 1999).

The laboratory-scale research focussed on whether the objectives of the larger-scale demonstration project can be attained (degradation of organic wastes to gas, improvement in leachate quality, biological stabilisation of MSOR). It also assessed whether additional post-treatment is necessary (e.g. flushing with clean water, post-treatment of MSOR residues), and predicted the final product of MSOR after treatment in a bioreactor.

Four test columns (diameter 300 mm, height 2.1 m) were filled with MSOR at a density of 1.2 t/m^3, and placed in rooms at 38ºC. Three of the columns (1, 2 and 3) were operated in a saturated manner, with upward flow of leachate that discharged at the top. (Initially, the infiltration was started at the top, downwards, but due to low permeability this failed to achieve desired rates, and after 80 days the upward-flow system was adopted). Column 4 remained in a downward flow mode, to function as a reference column.

During the initial methanogenic phase of about one year, leachate was infiltrated at a rate of 3000 mm/yr. After this phase (day 376), column 1 was flushed with clean water at a rate of 7400 mm/yr, to accelerate flushing of contaminants, and improve final leachate quality. Composition of the MSOR used was broadly similar to that shown earlier in Table 3.1, with 40 percent water, 26 percent dry organic matter, and 10 percent glass/stones.

The upwardly leached columns produced leachates containing high levels of volatile fatty acids during the first 40 – 50 days, and these were flushed out of the columns instead of being converted to biogas. Figure 3.4 below shows the cumulative production of biogas, (typically containing 57-63% methane by volume), for the 4 columns, showing that maximum generation rates for column 4, in downflow mode, were typically less than one eighth of those achieved in the upward flow columns.

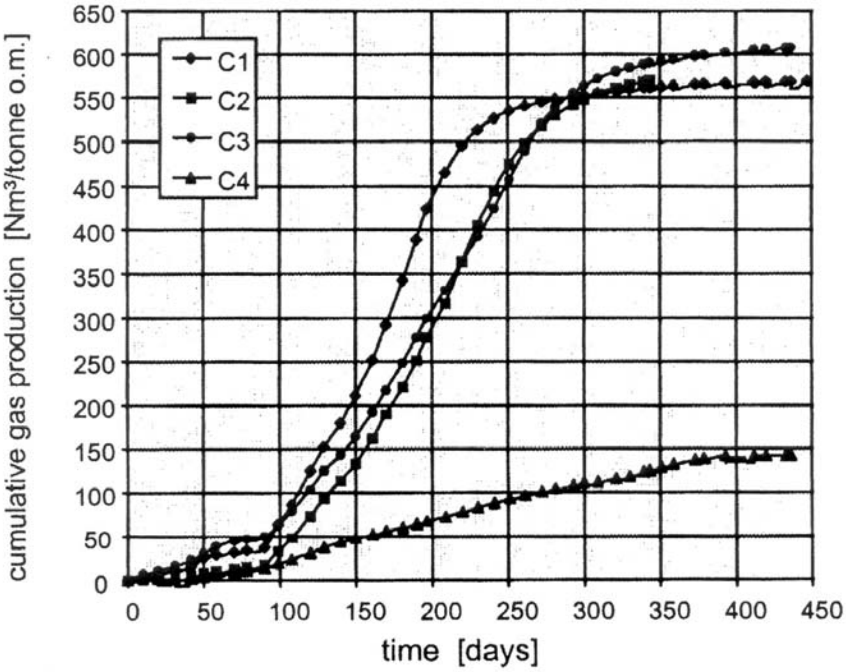


Figure 3.4 **Cumulative gas production, (Nm^3 gas per tonne dry organic material) for the four test columns**

Flushing of column 1 after completion of the methanogenic phase (after day 376) significantly reduced the concentrations of all contaminants by factors of from 2 – 30. This process is shown in Figure 3.5 below, for COD, ammoniacal-N, chloride and conductivity, and in Table 3.2 for a wider range of contaminants during the methanogenic and flushing phases. Results cover a period of about 437 days in total, during which a total of about 218 litres of leachate, plus 87 litres of clean water, had been used to flush a mass of 139 kg of MSOR, with an initial moisture content of about 40 percent. This represents an overall liquid/solid ratio of about 5.5 bed volumes of flushing.

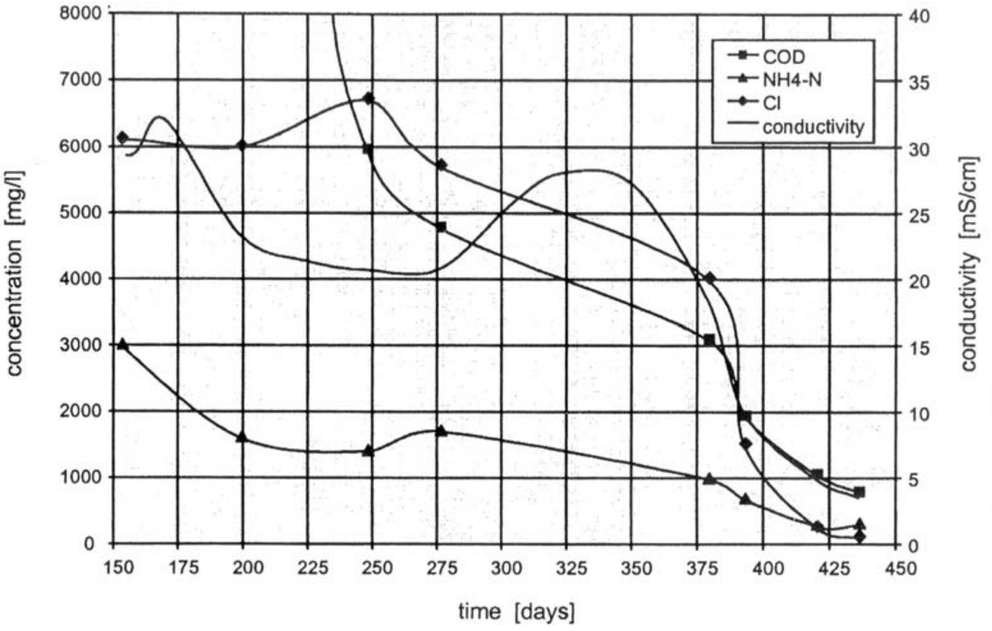


Figure 3.5 **Composition of leachate from column one, during methanogenic phase (to day 376), and during subsequent flushing phase**

Table 3.2 Composition of leachates during methanogenic phase, and after post-methanogenic flushing phase (results in mg/l, heavy metals in µg/l)

Determinand	Methanogenic Mean	Methanogenic Range	After post-methanogenic flushing
COD	4950	(4780-5080)	780
BOD_5	313	(215-490)	40-80
Ammoniacal-N	2000	(1700-2200)	290
Chloride	7470	(5700-8400)	95
Chromium	273	(120-500)	110
Nickel	300	(210-350)	65
Copper	100		16
Zinc	2300	(1000-3200)	110
Cadmium	0.9	(<0.2-<2)	<0.2
Lead	45	(<40-50)	<10
Arsenic	14	(<1-34)	3
Mercury	1.0	(<0.1-2.7)	<0.1

It is clear that a much greater degree of flushing would be necessary, in order to enable concentrations of organic materials, and of ammoniacal-N, to be acceptable for discharge into surface watercourses. In addition, it must be expected that the wastes in the bioreactor will retain a continuing potential to generate and release these contaminants, albeit at a reducing rate, as processes of degradation continue.

Vroon *et al.* (1999) concluded from the lab-scale trials that MSOR contains readily-degradable organic matter with a high gas formation potential, which was higher than initially expected (at 570-610 Nm^3 per tonne dry organic matter). During MSOR treatment in the lab-scale flushing bioreactor, it was concluded that anaerobic biodegradation and stabilisation can be enhanced significantly, and leachate quality can be improved in respect of salts and heavy metals, by flushing the stabilised material with clean water at higher rates. However, for “biological” contaminants (such as BOD and nitrogen), the concentrations cannot be lowered adequately, since these compounds are generated subsequently by degradation processes that will continue over periods of years or decades.

The work, nevertheless, was progressed to a full-scale flushing bioreactor cell, in order to provide much larger scale data over an extended period.

The large-scale research cell to investigate treatment of MSOR wastes has been operated by VAM. Work has been described in detail elsewhere (Oonk *et al.*, 2000; Woelders and Oonk, 1999; Oonk and Woelders, 1999a), and is summarised below.

During June to November 1997 a 49,000 tonne demonstration cell was constructed, filled with MSOR and instrumented, and completely sealed at base and surface using BES and VLDPE liners and cap. Cell area was 70 x 100 m, with a maximum depth of wastes of 8m. During February 1998, leachate started to be fed into the MSOR via a surface infiltration system, beneath the cap, but the design rate of 30 mm per week could not be achieved. This was partly due to engineering problems from irregular settlement, but primarily resulted from the low permeability of the MSOR, that has been emplaced at a density of 1.3 t/m^3. To overcome these problems, water was infiltrated under pressure, through pipes placed lower in the wastes, and rates of up to 15 mm/week could be maintained during the period to July 1999. Over a 2 year period, 7730 m^3 of water was infiltrated, representing 1100 mm (about 11 mm/week), or about one third of a bed volume in total. During this 2 year period, about 2.8 Mm^3 of gas was produced, about half of the estimated gas potential of 5.7 Mm^3, and there were no difficulties experienced in extraction of leachate, although some scaling of recirculation pipework was observed during the first 6 months. At the end of the trial it was clear that moisture distribution remained very inhomogeneous.

Table 3.3 presents detailed composition data for leachate from the VAM test cell, which is complemented by time-series data for a range of contaminants in Figure 3.6.

Table 3.3 Composition of leachate from the VAM test cell containing MSOR, formed during construction and operation of the bioreactor (after Woelders and Oonk, 1999)

Determinand	Construction	6 months	15 months
COD	60,700	39,200	19,400
BOD_5	42,000	26,000	9,400
Kjeldahl-N	4,700	5,400	4,200
Chloride	4,700	5,700	6,500
PH-value	7.1	7.8	8.2
Chromium	450	670	1,300
Nickel	770	350	450
Copper	64	18	330
Zinc	2,500	180	560
Cadmium	<0.5	<0.5	<0.5
Lead	180	28	56
Arsenic	<50	210	190
Mercury	0.63	0.35	0.58

(results in mg/l, heavy metals in µg/l)

Leachate quality is in the same order as that reported earlier from the laboratory-scale trials. However, the greater depth of wastes is reflected to some extent in higher COD and BOD values (COD remained at 10,000 mg/l after 2 years), and in very high values of Kjeldahl-N (to above 5000 mg/l). Ammoniacal-N remained above 3000 mg/l at the end of the trial.

Unlike in the laboratory-scale studies, no attempt was made to complete the test cell study by accelerated flushing with clean water, and it seems likely that this would have been extremely problematical. The study has now ended, and the test cell is no longer accessible (having now been buried under several metres depth of wastes).

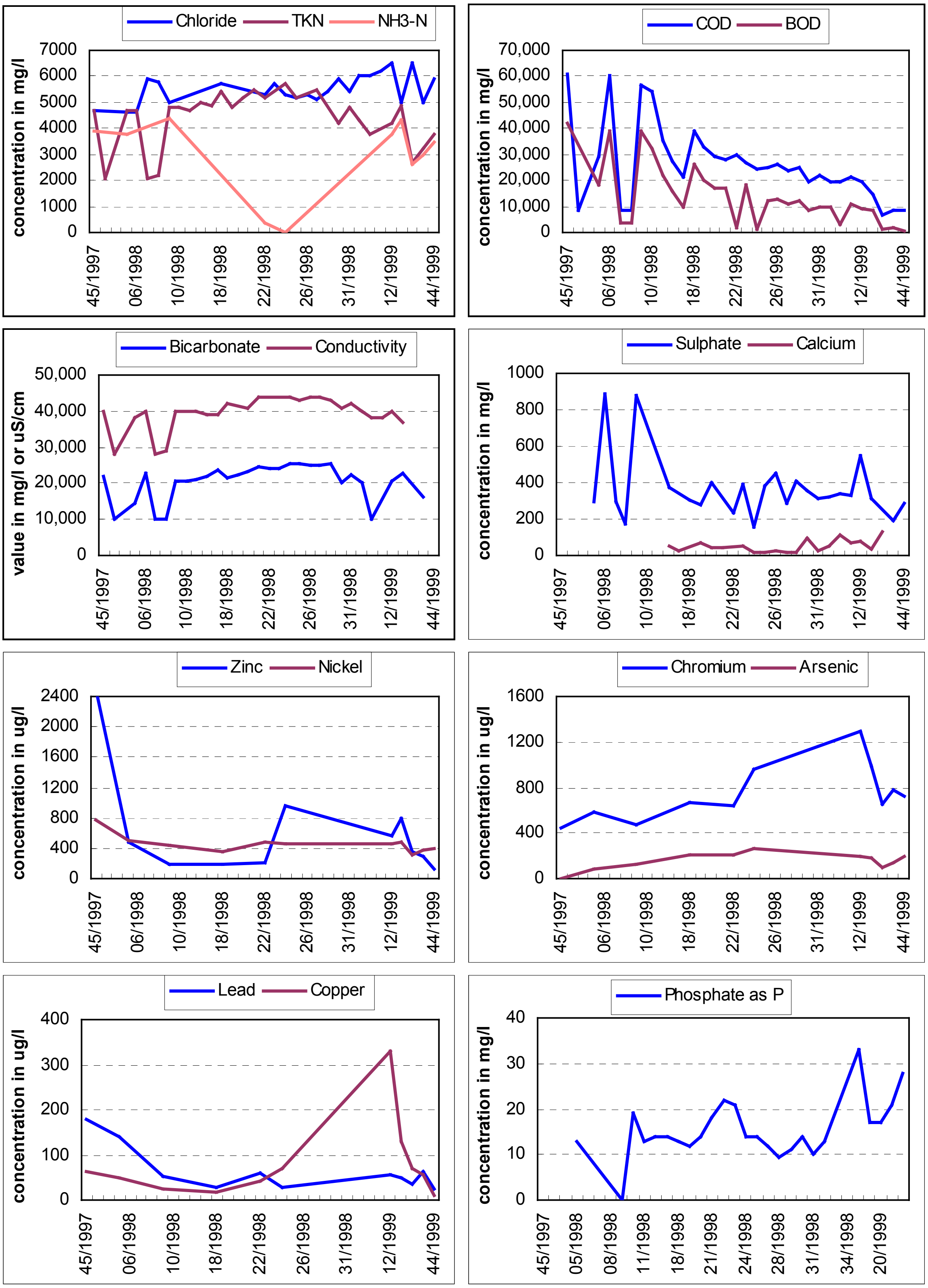


Figure 3.6 **Composition of leachate from the VAM test cell containing MSOR, from November 1997 to July 1999**

3.2.3 Summary of leachate production from landfilled MSOR

It is clear that the MSOR fraction generated by mechanical pre-treatment of MSW, which can represent between 30 and 40% of the MSW mass, has higher polluting potential when landfilled than crude MSW. To some extent, especially during initial stages after emplacement, this may result from smaller particle size (typically <40 mm) and from higher moisture content (typically ~40 percent). However, in the longer term, the high organic content of the MSOR appears to give rise to much higher production of landfill gas, and of contaminants such as COD and ammonium.

The reported gas emission from MSOR landfilled in the VAM bioreactor during the 2 year trial was 2.8 Mm^3 of gas per 49,500 t of fresh MSOR, which had an initial moisture content of 41.9 percent. (This was, nevertheless, only between 10 and 12 percent of the maximum gas production rates achieved in the earlier column tests.) This represents approaching 100 m^3 of gas per dry tonne of MSOR, and was estimated to represent about half of the likely ultimate gas generation of the MSOR – higher than typical observed yields from crude MSW.

Extremely strong leachates would be expected in any potential medium-term scenario for landfilling of MSOR (e.g. to reduce the quantity of biodegradable waste going to landfill), especially if this material is landfilled in isolation from other waste streams. It is predicted that strong leachate would persist for many decades, even if it proves possible to incorporate efficient accelerated degradation and flushing techniques into landfill designs.

3.3 Anaerobic digestion of MSW or MSW fractions

Anaerobic digestion is a well-established technology for treating sewage and industrial sludges and an expanding technology for MSW treatment in Europe. The available capacity in Europe is predicted to rise to 2.5Mt/a by 2004, a 150% increase over 4 years (de Baere, 2003). It is estimated to be used to treat over 5% of potentially digestible MSW in Spain, The Netherlands and Belgium and is attractive in comparison with other biological treatment technologies because of the efficient production of biogas and associated energy recovery.

However, the anaerobic digestion of MSW, or of MSW fractions, does not generally produce an end-product that is suitable for direct landfilling. Although the process can readily be undertaken within completely-contained vessels, and emissions can be effectively controlled, the material produced retains potential to produce high strength leachates, and significant odour, if directly landfilled.

Contaminants released during anaerobic digestion, such as ammoniacal-N, some metals, and chloride, as well as long-chain refractory organic compounds, may be controlled to some extent by dewatering of the digestate, to be treated on-site or passed to sewer for treatment in combination with domestic wastewaters. However, EU experts contacted during this study indicated that solid residues from anaerobic digestion of MSW fractions are invariably subjected to post-treatment by composting processes, before being landfilled.

Section 5.2.5 (sample HR9) provides some data for a landfill site where incoming MBP wastes have been subjected to an initial anaerobic digestion phase of treatment, before composting. However, in the light of the above, it has not been possible to obtain full-scale or pilot-scale leachate quality data for landfilled MSW, which has been subjected to anaerobic digestion alone.

3.4 Composting of MSW or MSW fractions

In view of the very high pollution potential, MSOR wastes have often been subjected to various composting processes, for many years in some European countries (notably Austria, Germany and Holland). Several good papers and PhD theses (often not translated into English) have been published in these countries, and it was recognised that much could be learned directly from researchers with more experience than was available than in the UK. Detailed information on the research and practice of composting MSW and MSOR residues in Austria and Germany is provided in Chapter 4 and summarised below.

The objectives of this study were to provide data and guidance for UK landfill operators and regulators on the impact that the landfilling of MSW fractions, or composted MBP wastes might have on leachate quality at their sites. This information is necessary to allow them to:

- define a leachate source term for groundwater risk assessments;
- assess the implications of waste pre-treatment on the timescales, and long-term liabilities, of the landfilling of pretreated waste materials;
- make appropriate and adequate provisions for leachate management, treatment and off-site disposal; and
- consider the effects of changes in leachate quality on landfill liners and leachate drainage blankets and systems.

At several sites where research has been undertaken into the disposal of MBP wastes, leachate quality data runs of 20 years or more have been obtained. Nevertheless, in spite of these long periods of data, leachate quality results from MBP mono-landfills are limited, because typically MBP wastes are disposed of together with variable proportions of untreated MSW, MSOR or commercial and industrial wastes. The co-disposal of MBP waste and MSW or organic residues (MSOR) is likely to reflect the situation in the UK until significant infrastructure for MBP becomes available, at least locally. The quality of the MBP waste materials themselves will also vary as a consequence of:

- the extent of source-separation, for example of food and garden wastes;
- the waste inputs (e.g. urban or rural source, summer or winter collection);
- type of mechanical pre-treatment; and
- type and duration of biological treatment.

3.4.1 Leachates from MBP landfills

A number of studies have looked at the benefits of composting MSOR for reducing pollutant emissions. Leikam and Stegmann (1999) studied the behaviour of composted MSOR wastes in landfill simulation tests, in comparison to untreated MSOR. For treated MSOR, the acetogenic phase during which strong organic leachate is produced was absent, and after about 250 test days the COD of the leachate was below 1,000 mg/l (BOD_5 <20 mg/l).

A much more significant benefit of pre-treatment becomes apparent when concentrations of total-N (primarily ammoniacal-N) are considered. Whereas the total-N content in leachate from untreated residual waste stabilised at about 1,000 mg/l, this value was below 200 mg/l for pre-treated wastes. Nevertheless, in terms of L/S ratio, a trial period of 250 days corresponds to a period of about 50 years for a 20 m deep landfill, with an annual infiltration rate of 250 mm, or significantly longer if the site was capped.

Table 3.4 contains basic summary data, from published sources, for leachates from landfilled MSOR, and composted MSOR.

Table 3.4 Basic leachate quality for leachates from landfills/test cells containing untreated MSOR, and MSOR subjected to various composting regimes

Waste Inputs	**MSOR**		**Composted MSOR, various sources**				
Composting (weeks)							
intensive	0	0	4	4	2	16	3
secondary	0	0	9	43	1	8	19
COD	172000	19400	2780	1170	540	4000	1900
BOD_5	123000	9400	52	9	158	111	14
NH_4-N	3965	4200	197	11	56	292	340
Chloride	9100	6500	11300	6900	5700	6200	4100
Chromium	0.41	1.3	0.14	0.04	0.03	0.21	0.09
Nickel	2.10	0.45	0.23	0.71	0.16	0.40	0.09
Copper	1.41	0.33	0.71	0.80	0.28	0.52	0.18
Zinc	102	0.56	3.4	1.0	0.22	1.6	1.2
Notes:	all results in mg/l table based on data from Danhamer and Jager (1999) and others						

The results are typical of many other data, showing the very high strength leachates generated by untreated MSOR when landfilled. They also demonstrate the improvement in leachate quality achieved by various degrees of composting pre-treatment – widely observed to be capable of removing the initial strong organic leachates generated during the acetogenic stage of degradation, leading to a more rapid onset of methanogenic conditions.

Results for Kjeldahl nitrogen in leachates sampled during the present studies were inconsistent, sometimes (especially at higher concentrations) being determined at values less than those obtained for ammoniacal-N. Any calculation of a figure for “organic-N”, as the difference between the two values was therefore unhelpful. Future research must seek to confirm the relative importance of nitrification/denitrification, generation of stable, organically-bound forms of nitrogen, and other routes during MBP of waste fractions, in reducing long-term emissions of ammoniacal-N from landfilled wastes, in order that appropriate processes can be encouraged and optimised during waste pre-treatment.

Concentrations of ammoniacal-N and Kjeldahl-N in MBP leachates may be significantly lower than those from conventional MSW landfills, but the extent to which this occurs is variable, and not easily related to the design of specific composting and pre-treatment processes. The biochemical transformations involved are not well understood, and the influence of nitrification/ denitrification, incorporation within stable organic fractions (possibly related to the elevated “hard COD” values), and other processes, require further research.

It is clear that the degree of composting achieved, and the efficiency of individual composting processes, cannot be determined simply based on the duration of intensive and secondary composting being carried out at each location. For example, at one site only two weeks of intensive, and a further week of secondary composting are applied. This short-lived process achieves substantial improvements in leachate quality. Key findings from these and other published studies on leachates from landfilled MBP wastes are summarised below.

- Organic residues from mechanical sorting (MSORs) can produce leachates with higher pollution potential than both acetogenic and methanogenic leachates from conventional landfills.
- Composting such residues can reduce the organic pollution potential from both leachate and landfill gas, through the avoidance of the peak acetogenic phase of decomposition.
- Concentrations of ammoniacal-N in MBP leachates can be either similar to, or much lower than, methanogenic leachates from conventional landfills. This raises the possibility that a nitrogen removal or attenuating process may operate, to varying extents, during composting.
- Landfills receiving MBP wastes will pose a risk to groundwater similar to conventional MSW landfills that have become methanogenic, and are therefore likely to require a similar period of time before active management and treatment of leachates ceases to be necessary.

Of particular importance in the future, when MBP processes are being designed to effect composting of residual wastes, will be considerations related to the fate of nitrogen within the composting process itself. In the past, when “green wastes” have been composted, or when there has been an assumption that composted products will be of value as fertilisers or soil improvers, the presence or removal of ammoniacal-N has not been a concern to composters and has generally provided a benefit to users.

Few research data have been published regarding the fate of nitrogen during composting of MSW fractions, and mechanisms involved are not clearly understood. However, the effects of composting processes on subsequent emissions of ammoniacal-N in leachates from the landfilled product are evident from other published data and from this study. The extent to which removal occurs during the composting process itself, or whether composting may mineralise nitrogen compounds to nitrates, which are subsequently reduced to nitrogen gas within an anoxic/anaerobic landfill, have not yet been well-established.

Leikam *et al*. (1997) investigated injection of air as a remedial technique for old landfills, and in pilot-scale trials demonstrated dramatic reductions in TKN concentrations, although no oxidised nitrogen was found in the leachate from the two aerated lysimeters. Work by Heiss-Ziegler and Lechner (1999) has demonstrated the stability of nitrogen-containing organic substances, such as humic acids formed during composting processes, which may also play a key role in minimising emissions of ammoniacal-N from MBP wastes in landfills.

Efficient MBP of MSW or the MSOR fraction can considerably reduce the organic strength of leachates, avoiding the acetogenic phase, and more rapidly producing leachates similar to those from MSW landfills in methanogenic phases of decomposition. Nevertheless, levels of “hard” COD in these leachates, not readily biodegradable by aerobic or anaerobic processes, are typically at least as strong as those found in methanogenic leachates – often in the range 1000 to 4000 mg/l, in spite of BOD values which are often less than 100 mg/l.

Few data are available on the presence of potentially harmful trace organic substances in leachates from MBP waste landfills, to allow these to be compared with data from conventional MSW sites. This study has obtained preliminary results, based on a programme of sampling at EU landfills (see Chapter 5). As at MSW sites, few trace organic compounds are present, but for those which have been measured at significant concentrations (e.g. mecoprop), evidence from this study is that effective composting processes are able to reduce concentrations significantly.

In the case of mecoprop, this is generally present at significant levels in leachates from MSW, and was present at up to 120 µg/l in leachates from landfilled MSOR. However, it was absent or present at very much lower concentrations in leachates from composted residues. The extent of removal of mecoprop from leachate may well be a good measure of the efficiency of the composting process itself.

3.5 Overview of leachates from MBP residues

The main residue from MRFs, which is typically landfilled either untreated or after pre-treatment by composting, is termed MSOR (mechanically separated organic residue), but also variously referred to as “residual waste” or “RMSW” (residual MSW). Although the extent of MSW source separation which takes place, and the precise means by which the waste stream arises at the MRF may vary, generally the material represents the fraction of MSW which falls through a grid – variously sized in the range 40-100 mm in most cases. Larger waste fractions, generally of higher calorific value, are often incinerated either directly, or as RDF.

Having higher overall organic content and greater surface area than MSW, MSOR produces a material with potential to generate leachate of much greater strength than those from MSW when landfilled. Various studies have indicated COD values well in excess of 50,000 mg/l in early stages of decomposition, and extremely high concentrations of ammoniacal-N in the range 2,000-6,000 mg/l, which may persist for many years.

In the longer term, the high organic content of the MSOR also appears to give rise to a much higher production of landfill gas, and increased release of contaminants, such as NH_4-N.

There are, therefore, great potential benefits in terms of leachate strength and long-term pollutant emissions, to be gained by pre-treatment of these residual waste materials, and composting processes of varying intensity and duration have been widely used.

Reductions in gas emission potential of between 80 and 90% have often been reported from various trials, involving periods of composting from 16 to 30 weeks. Extensive studies are underway within a coordinated research programme in Germany (see Section 4.2), using an agreed protocol and standardised and calibrated 200-day lysimeter tests, to indicate how various pre-treatment and composting options can reduce this gas emission potential.

Nevertheless, although such trials widely and generally report that composting is able to remove an initial acetogenic decomposition phase when the material is finally landfilled, most studies and researchers acknowledge that over the longer term, leachates can be very similar to dilute methanogenic leachates derived from unpretreated MSW landfills.

The efficiency of composting processes is often assessed by 24-hour agitated batch leaching tests. Increasingly there is evidence that such tests can seriously underestimate likely future emissions of contaminants because they take no account of biodegradation potential of the wastes.

There is no long-term evidence to show convincingly, at full-depth anaerobic landfills containing pretreated residual wastes (as opposed to shallower laboratory pilot-scale trials), that significant residual concentrations of COD, ammoniacal-N, and other contaminants may not persist for decades or centuries, as at normal MSW landfill sites. The German research programme continues to provide results which demonstrate the extremely long timescales over which high residual concentrations of TOC (>200 mg/l) and of ammoniacal-N (>100 mg/l) will persist in leachates, even from wastes that have been subjected to advanced and current composting processes.

In particular, specific processes which may remove or treat organic forms of nitrogen during pre-treatment of MSOR, and so significantly reduce the release of ammoniacal-N during subsequent landfilling, have not been identified or demonstrated adequately. Such work is key to demonstration of the long-term benefits of pre-treatment of MSW residual wastes. Until relatively recently, the objectives of composting MSOR fractions were primarily to reduce odour nuisance, and to minimise their emission potential in terms of landfill gas (if to be landilled). In many instances, where final composted materials were to be applied to land or used in horticulture, the presence of ammoniacal-N, or organic-N, brought considerable benefits. The fate of nitrogenous compounds was therefore not generally a key concern within the composting process.

During the last 3 or 4 years, concerns about transmissible livestock diseases such as Foot and Mouth Disease (FMD), and Bovine Spongiform Encephalopathy (BSE), has very much reduced the potential for composted MSOR outputs to be applied to land. The Animal By-products Amendment Order 1999 (as amended) prohibits the re-use of compost where livestock (including wild birds) may have access to the compost produced from some catering wastes containing meat or products of animal origin, including household kitchen waste. The recent EU Animal By-products Regulation (EC No. 1774/2002) enforced in England since July 2003, allows composting to be used for catering waste and other low risk animal by-products, subject to national treatment standards being met (as set out in Defra, 2003). Further work will be needed to assess the pollution potential from composts derived from approved composting facilities.

Where such composts are landfilled, it becomes more important that research should focus on the behaviour of nitrogenous compounds during the composting process, with the intention of maximising overall removal efficiency (e.g. by nitrification/denitrification processes), in order to minimise release of ammoniacal-N when the compost residues are landfilled.

It is nevertheless clear from data obtained, that significant removal of nitrogen can be achieved in some processes and in some full-scale composting plants. The extent to which total nitrogen removal occurs during the composting process itself, or whether composting may mineralise organic nitrogen compounds to nitrates, which are subsequently reduced to nitrogen gas within an anoxic/anaerobic landfill, has not been established.

In practice, in Germany and recently in Austria, research is increasingly being focussed on minimisation of volumes of air used during the composting process. This is driven not by the objective of better or more cost-effective treatment, but instead by political pressures that demand exceptionally high standards of treatment for off-gases from composting processes, which must be enclosed.

Work to look at the effects of composting of MSOR on detailed composition of leachates produced has been reported. Previous studies (Robinson and Knox, 2001; 2003) have demonstrated that relatively few trace organic substances from the UK Pollution Inventory List (www.environment-agency.gov.uk/pi) are typically present in leachates from MSW landfills, and many of those found are effectively removed by biological leachate treatment processes.

Evidence from this study suggests that effective composting processes are able to reduce concentrations of several trace organic substances present in leachates (see Chapter 5). For example, the herbicide mecoprop is generally present at significant levels in leachates from landfilled MSW, and was present at up to 120 µg/l in leachates from landfilled MSOR. However, it was absent, or present at much lower levels, in leachates from composted residues.

Of more potential concern is the presence of heavy metals in leachates from untreated MSOR fractions in landfills. Chromium is of particular concern at levels of up to 13.1 mg/l in samples tested, and elevated levels of nickel and copper were also observed. It may well be the case that these concentrations are related to co-disposal with sewage sludges at the landfills sampled. The heavy metal concentrations were significantly lower, but still elevated, in leachates from landfills containing composted residues (where sewage sludge inputs were much more restricted), and at the VAM test cell containing crude MSOR.

In general terms, many landfills receiving MBP wastes will continue to pose risks to groundwater, and require aftercare periods similar to conventional MSW landfills that have become methanogenic. There is no doubt that MBP processes have potential to reduce both organic strength, and concentrations of ammoniacal-N in leachates from such landfills, as well as the total mass release of these and other contaminants. However, even at such landfills, the extent to which leachate management timescales will require management can be reduced significantly, remains to be determined.

One objective of this study was to try to investigate the effects that waste pre-treatment processes may have on the impacts of leachates on landfill liners, and on leachate drainage blankets. No conclusive data have been obtained. In practice, for leachates from landfilled MBP residues, whether treated or not, no researchers or landfill operators have raised this as an issue of concern. At some MSOR landfills, and other composted waste landfills, drainage systems are occasionally jetted, but at others they are not, and any differences between sites receiving untreated and treated waste fractions do not appear to be significant in operational terms.

3.5.1 Long-term aftercare liabilities

In the long-term the component most likely to influence leachate management is ammoniacal nitrogen, as it does for most leachates from methanogenic MSW landfill sites. It is possible that very efficient composting of MSOR materials may significantly reduce emissions of ammoniacal-N, in which case other components such as high concentrations of poorly-biodegradable COD, or inorganic components such as chloride (which is not significantly affected by the composting process), may also become important. It seems unlikely that the presence of trace organic compounds in leachates will comprise a significant long-term concern, regardless of whether wastes are composted or not. Nevertheless, mecoprop, for example, will remain ubiquitous in leachates from untreated MSOR for long periods of time (as it does at MSW landfills), whereas it has been shown to be readily degradable during efficient composting processes.

Flushing and dilution requirements for untreated MSOR are likely to be significantly greater than for untreated MSW, while the requirements for well-composted MSOR could be an order of magnitude less than for MSW.

3.5.2 Management of leachate collection systems

Discussion with European landfill operators and researchers has not indicated that there is any significant difference in maintenance requirements for leachate drainage and collection systems, between landfills that have received treated or untreated MSOR inputs, and those that have received untreated MSW. At some sites leachates drains are jetted occasionally, at others this does not prove necessary. It does not therefore appear that this will be a significant issue at such sites.

3.5.3 Treatment and disposal of leachates

The strong organic leachates from untreated MSOR and high concentrations of ammoniacal-N from some low intensity composted MSW are likely to be more expensive to treat than MSW leachate. However, this is not likely to preclude treatment to very high standards. A particular issue may be the presence of relatively high COD values which resist normal biological treatment processes, may remain in effluents, and require additional physical and chemical treatment processes (such as activated carbon, ozonation, dissolved air flotation with flocculants) in order to remove them. These refractive COD compounds may arise either during composting, or as by-products of microbial metabolism during leachate treatment, as nitrifiers treat increased concentrations of ammoniacal nitrogen within a biological leachate treatment plant (Carville *et al.*, 2003). Elevated concentrations of some metals (e.g. chromium), may have potential to inhibit biological leachate treatment processes, if they result from combined disposal of sewage sludges (see Section 6.4.1 above). However, successful on-site treatment of leachates has been observed during this study at a range of landfills, receiving both treated and untreated MSOR wastes.

3.6 Proposed leachate source term values

A range of contaminant concentrations is proposed in Table 3.5. Maximum values are derived from the European leachate collected during this study and minima derived by dividing maxima by a hydraulic factor of 5. Kappa values have not been derived for MBP wastes, and one of the methods described in the LandSim 2.5 Help files should be used to derive kappa from m and c values or from column tests.

For some parameters the range encountered in the available data was greater than 5:1. This occurred mostly with parameters that are likely to be affected by the process, e.g. BOD, nitrogen, phosphorus and metals. Where there were no TKN data, the contribution of organic nitrogen has been estimated on the basis of 4% of hard COD. This value is based on experience of effluents from biological treatment of leachate.

3.6.1 Sources of variation from default values

The most important factors that will determine actual values for contaminants at any specific site, will be the source materials that are being subjected to MBP, the MBP processes themselves, and primarily the efficiency of any composting stages. Composting efficiency cannot be predicted precisely, solely from desk studies of particular facilities, and wide variations have been demonstrated during this and other projects. In particular, specific processes which remove or treat reduced forms of nitrogen during composting of MSOR, and thus significantly reduce release of ammoniacal-N during subsequent landfilling, have been clearly shown to exist, but have not been investigated adequately. Work focussing on the behaviour of nitrogenous compounds during the composting process is therefore fundamental to the derivation of more reliable source term values.

A further issue that may be important at some landfills receiving treated or untreated MSOR waste materials, is the historic or continuing addition of sewage sludges to wastes, during or after the composting process. Evidence suggests that concentrations of some heavy metals, subsequently found in leachates, may be increased by up to an order of magnitude by such additions.

Table 3.5 Proposed contaminant distributions for MBP residues

Parameter		**MSOR, methanogenic**		**Low-Medium intensity composted MSOR**		**High intensity composted MSOR**	
	units	max	min	max	min	max	min
pH		8		7.5 – 8.5		8	
Conductivity	µS/cm	40,000	8,000	20,000	5,000	10,000	2,000
TOC	mg/l	4,000	800	2,000	400	500	100
COD	mg/l	10,000	2,000	5,000	1,000	1,500	300
BOD	mg/l	4,000	800	200	20	30	6
NH_4-N	mg/l	4,000	800	1,000	50	200	40
TKjN	mg/l	4,200	840	1,300	100	260	50
Na	mg/l	4,000	800	4,000	800	1,200	240
K	mg/l	2,000	400	2,000	400	800	160
Ca	mg/l	50	10	800	100	300	60
Mg	mg/l	100	20	400	80	100	20
Cl	mg/l	6,000	1,200	8,000	1,600	2,000	400
SO_4	mg/l	400	80	5,000	1,000	500	100
P	mg/l	20	4	15	1	3	1
NO_x-N	mg/l	<1	<1	<1	<1	<1	<1
Alkalinity pH 4.5 as $CaCO_3$	mg/l	18,000	3,600	6,000	1,500	2,000	400
Fe	mg/l	5	1.0	20	4	10	2
Mn	mg/l	0.5	0.1	2.0	0.5	3	0.6
As	ug/l	100	20	100	10	6	1.2
Hg	ug/l	0.1	0.02000	10	0.1	0.1	0.02
Cd	ug/l	<1	<1	100	5	3	0.6
Cr	mg/l	5	1.0	0.5	0.1	0.1	0.02
Cu	mg/l	0.5	0.1	0.5	0.1	0.2	0.04
Pb	mg/l	<0.1	<0.1	0.4	0.08	0.04	0.008
Ni	mg/l	0.5	0.1	0.7	0.14	0.1	0.02
Zn	mg/l	0.5	0.1	3	0.5	0.2	0.04
AOX	ug/l	6000	1,200	1000	200	300	60
phenols	ug/l	200	40	absent	absent	absent	absent
mecoprop	ug/l	100	20	5	1	absent	absent
BTEX	ug/l	absent	absent	absent	absent	absent	absent
PAH	ug/l	absent	absent	absent	absent	absent	absent
VOCs	ug/l	absent	absent	absent	absent	absent	absent
other List 1 herbicides and pesticides	ug/l	absent	absent	absent	absent	absent	absent

4 MBP in Austria and Germany

4.1 MBP research and practice in Austria

MBP wastes have been landfilled for more than 20 years in Austria, although in most instances this has been in combination with untreated residual and commercial wastes. From 2004 (2009 in some Austrian provinces), this will no longer be allowed. Instead, MBP wastes must be disposed of alone at so-called "Massenabfalldeponien", or "Mass waste landfills", in separate landfill compartments, and will only be accepted if they are less than a maximum gross calorific value of 6000 kJ/kg TS (total solids). Background information on the important role of MBP in Austrian waste management is given in Appendix 1.

In recent years, several Austrian research programmes have been started to optimise MBP technology, in order to achieve this limit cost-effectively. It appears likely (Raninger *et al.*, 1999), that up to half of Austria's residual waste will continue to be treated in MBP plants, and much of the outputs from these plants will continue to be landfilled. Practical data on the consequences of mono-disposal of MBP waste outputs nevertheless remain extremely limited. Accordingly, a large landfill test cell project has been started, entitled "Model Landfill Allerheiligen", in order to investigate the long-term behaviour of MBP wastes under such conditions (Raninger *et al.*, 1999).

The input material to be landfilled was taken from the MBP plant at Allerheiligen, in Styria, some 50 km south-south-west of Vienna. MBP technology had been updated to become state-of-the-art in 1996, and comprises mechanical treatment (grinding, metal extraction, admixture with sewage sludge, separation of high calorific fractions >80 mm and >12 mm) followed by a biological phase of treatment. This second stage includes intensive composting for 2 weeks within 6 process-controlled tunnel composting reactors, a 3 week aerated maturation period, and an additional post-maturation phase - often of more than 9 weeks. The plant treats 10,500 tonnes per annum (tpa) of residual waste and 4,500 tpa of sewage sludge. The output fine fraction (<24 mm) achieves the future requirements for landfilling of MBP wastes, within a total treatment period of 14 to 22 weeks.

Four test cells, each of dimensions 5 x 5 x 3 m (L x W x D) were constructed within the nearby landfill site, lined with 2 mm HDPE, and equipped with leachate drains, gas extraction equipment, thermocouples, and leachate recirculation facilities. About 50 t of MBP waste was placed into each cell, at a density of 1 t/m^3, and was irrigated to achieve an initial moisture content of 45% dry matter.

Inputs were calculated to meet the new Landfill Ordnance (see Table 4.1 below). It is interesting to note that the average GPR$_{90}$ (gas production rate) was 35.2 litres/kg TS, which is higher than the suggested limit value of 20 litres/kg TS.

Table 4.1 Analysis of MBP waste inputs to Allerheiligen Model Landfill

Parameter	Unit	Average	Limit value
Maturation time	weeks	16 (1)	-
AT4 (3)	mg O_2/g TS	6.6	5 (5)
AT7 (3)	mg O_2/g TS	11.5	9 (5)
Calorific Value	kJ/kg TS	6,400	6,000
Ignition Loss	% TS	33	-
GPR_{90} (2)	litres/kg TS	35	20 (5)
TOC	g/kg TS	189	-
Arsenic	mg/kg TS	7.5	500
Barium	mg/kg TS	550	10,000
Lead	mg/kg TS	120	3,000
Cadmium	mg/kg TS	3.2	30
Chromium	mg/kg TS	372	5,000
Cobalt	mg/kg TS	15	500
Copper	mg/kg TS	250	5,000
Zinc	mg/kg TS	1,030	5,000
Silver	mg/kg TS	8.7	50
Mercury	mg/kg TS	8.2	20
Nickel	mg/kg TS	330	2,000
Ammoniacal-N	mg/kg TS	2,130	10,000

Notes:
(1) Range 14 - 22 weeks; 14 week material did not fulfil requirements for calorific value or AT_4
(2) GPR_{90} = total gas production rate in 90 days
(3) Respiration activity (AT_4 = 4-day test etc)
(4) TS = total dry solids
(5) Recommended limit values (Raningen *et al.*, 1998)

The cells were filled and capped by January 1998, and two test cells (Nos. 2 and 4) operated with leachate recirculation, while Nos. 1 and 3 were operated without irrigation. Within 4 months the internal temperatures had risen to a typical range of between 20 and 26ºC (maximum 31ºC), which was maintained to November 1998. Samples of the landfilled compost material were taken one year after filling, and tested for respiration activity (SOUR = Specific Oxygen Uptake Rate). The MBP input material had a value of 55.8 mg O_2/g TS in a 4-day test, which had been reduced by MBP to a value of 6.6 mg O_2/g TS in the landfilled compost. After one year in the test cell, this had reduced by almost half to 3.4 mg O_2/g TS. Equivalent reductions for the 7-day SOUR test were from 11.5 to 5.2 mg O_2/g TS during one year in the landfill.

Leachate production was measured in test cells where it was recirculated, and in those where it was not. In each case, by the end of the first year of the trials, all limit values of the Landfill Ordnance (1996) were met by the exhumed compost, but leachate strength remained high (see Table 4.2 below).

Table 4.2 Composition of leachates from test cells of MBP waste at Allerheiligen (results in mg/l except conductivity, in µS/cm; n.d.=not detected)

Determinand	Starting point, first sampling	After first year	
		Leachate recirculation	
		without	with
pH-value	7.3	7.6	7.2
COD	11,427	9,984	8,112
BOD_5	748	522	250
BOD/COD	0.07	0.05	0.03
TOC	3,755	2,437	2,953
Ammoniacal-N	2,217	2,212	2,867
Chloride	5,393	-	-
Sulphate	0.05	31	169
Nitrate-N	0.59	-	-
Conductivity (µS/cm)	26,730	30,000	27,300
Metals			
Chromium	0.39	0.31	0.55
Nickel	1.08	1.14	1.25
Copper	1.94	-	-
Zinc	0.96	-	-
Cadmium	0.006	n.d.	n.d.
Lead	0.24	0.11	0.05
Mercury	0.41	-	-
Trace organics			
PAX	n.d.	n.d.	n.d.
BTX	0.012	-	-
Hydrocarbons	7.46	-	-

Gas production rate from the landfilled compost was much reduced compared to wastes that do not receive MBP, and was typical of data from other studies on similar materials, (see Figure 4.1 below). However, the leachates from both cells remained strong, and were typical of leachate from the stable methanogenic phase of a controlled sanitary landfill (see Figures 4.2 and 4.3).

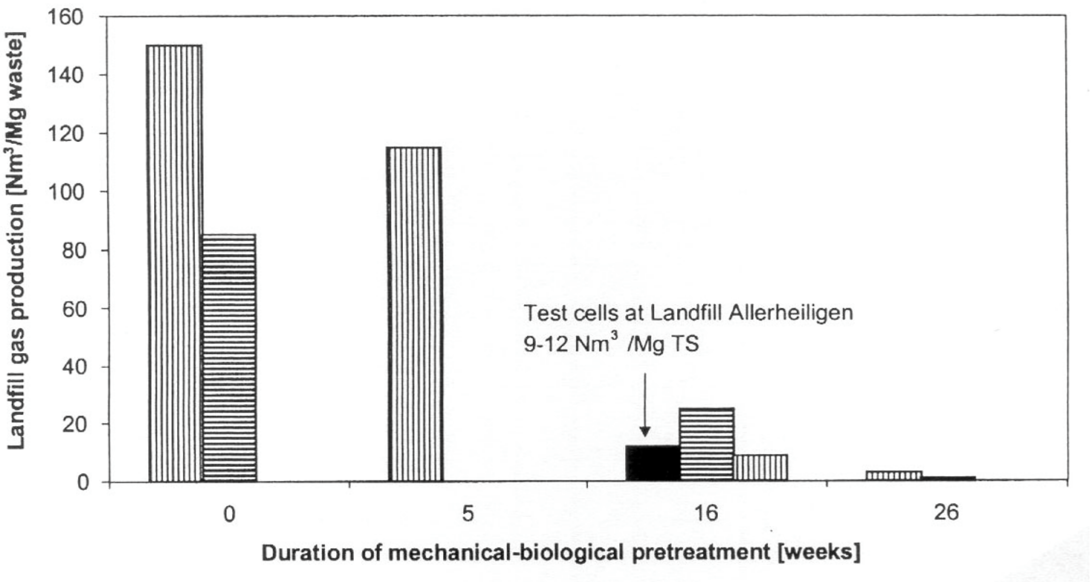


Figure 4.1 Gas production rate over 500 days, from the test cells at Allerheiligen, compared with untreated wastes and data for MBP material after 5, 16 and 26 weeks (after Brinkmann *et al.*, 1996; Bidlingmaier *et al.*, 1998; and Stegmann *et al.*, 1998b)

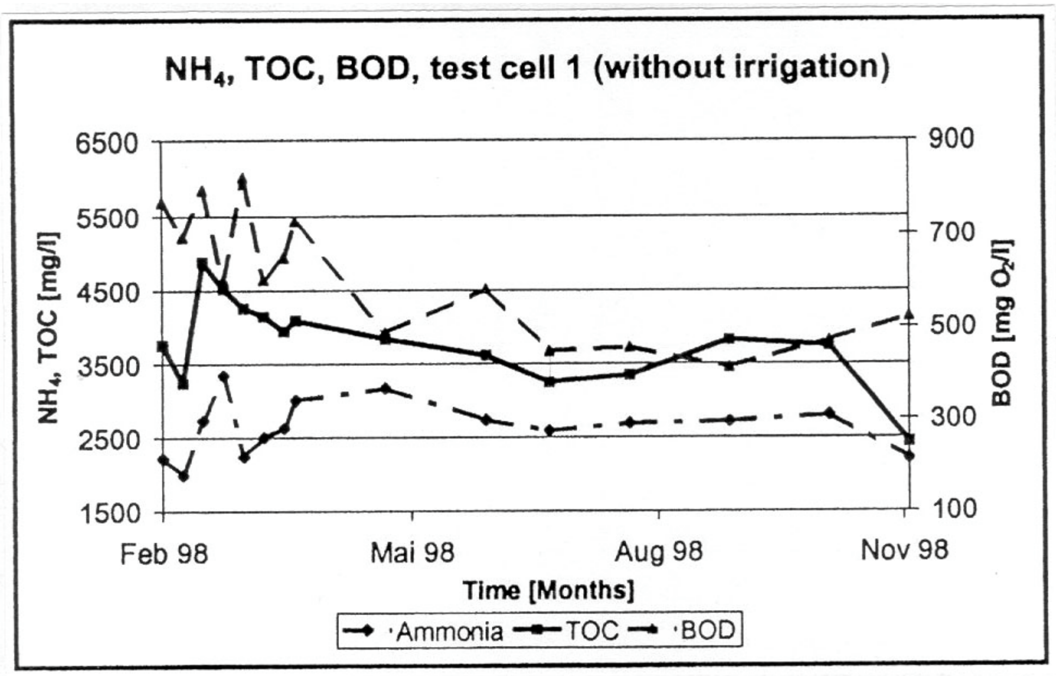


Figure 4.2 Concentrations of contaminants during the first 12 months following filling at Allerheiligen, test cell 1 (no irrigation)

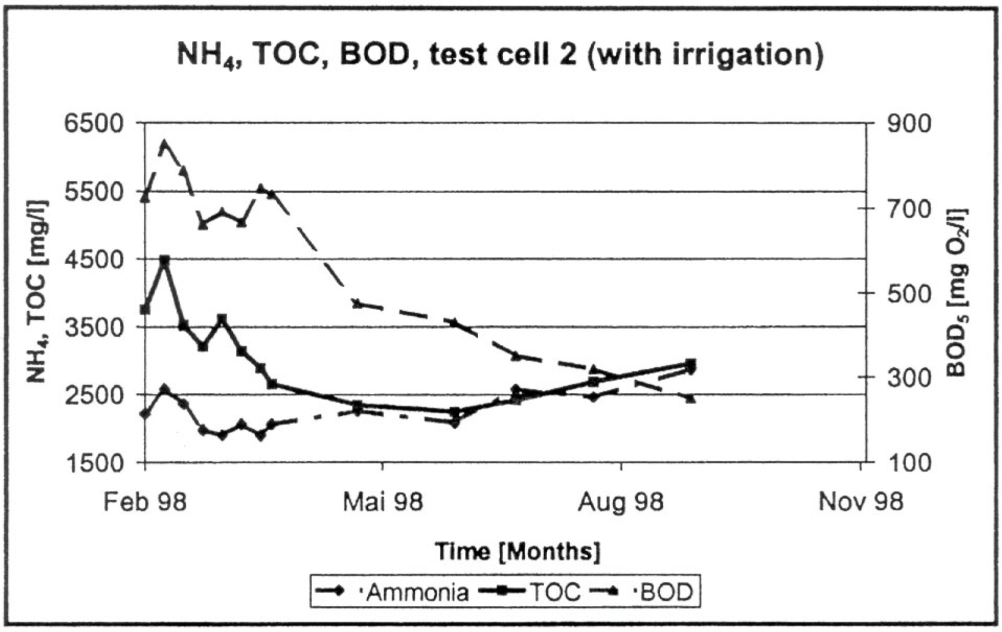


Figure 4.3 **Concentrations of contaminants during the first 12 months following filling at Allerheiligen, test cell 2 (with irrigation)**

Figures 4.2 and 4.3 present time-series data for BOD, TOC and ammonium, over a period from February 1998 (month 3) to November 1998 (month 12), in leachate from test cell 1 (no irrigation) and from test cell 2 (with irrigation). In test cell 1, a total of 19 litres of leachate per tonne TS was measured (i.e. L/S = 0.019), and in cell 2 all leachate was recirculated.

Although relatively low volumes of leachate have been generated from the test cells, and stabilisation of the emplaced MBP wastes is proceeding rapidly, it is clear that very strong leachates that would require considerable treatment, are still being generated. In a full-depth operating landfill, this leachate would require treatment for many years or even decades.

The intention at Allerheiligen is to exhume 2 of the 4 test cells after a period of 3 years and to maintain operation of the other 2 identical cells for a minimum of 5 years.

4.1.1 Data from full-scale MBP landfills in Austria

Although large quantities of MBP material are produced, there are few, if any, sites where only MBP wastes are disposed. Some MBP composts are used in agriculture. Other composts are subjected to different degree of treatment – for example, the fraction less than 25 or 30 mm is typically composted for about 20 weeks, but larger fractions up to 60 or 70 mm often have much shorter pre-treatment of only about 3 days. In some landfills the two fractions may ultimately be disposed of together, often in combination with other waste fractions such as demolition wastes or sewage sludges, in various proportions.

A student at the Universität für Bodenkultur in Vienna, Helmut Wurz (Wurz, 1999) has undertaken a diploma thesis in this field, to look at a number of Austrian landfills that have received significant inputs of MBP wastes. Some of these sites were still operational, while other had closed. Inevitably, different proportions of MBP wastes had been landfilled at each site, having been pretreated to different extents. He tried to look at differences in leachate quality from each site, and then to relate these to received waste inputs. Wurz looked for both “old” and “young” landfill sites in 3 categories:

- no MBP wastes received;
- about 20 percent MBP wastes received; and
- 50 percent or greater input of MBP wastes.

Wurz was unable to find any young sites containing about 20 percent MBP wastes, but obtained data for each of the other five site combinations, and published these in detail in his thesis (Wurz, 1999 – in German). Data was obtained from 12 landfills, covering various periods and of variable quality. For the purposes of this review, we have concentrated on data from 4 landfills, selected as 2 pairs of sites – one pair of young sites and one pair of old sites. For each pair, one site provides leachate data for non-pretreated wastes, and the second site provides comparative data for leachate from wastes including at least 50 percent MBP domestic waste. The 4 sites are as follows.

Breitenau comprises a large experimental landfill that was established during 1987-88, about 50 km south-south-west of Vienna, to investigate the release of contaminants from landfilled household wastes, and their potential impacts on groundwater quality. The site was constructed and lined as 3 separate cells, termed "Fields 1, 2 and 3". Fields 1 and 2 are of most interest to the present study, having received 35,000 t and 25,600 t respectively (wet weight) of non-processed MSW from Vienna (subject to separate collection of biowaste), emplaced to a depth of about 8 m. Wastes were emplaced during a 12-month period to September/October 1988 (Binner and Lechner, 1995; Binner 1996b; Rank *et al.*, 1992; Binner *et al.*, 1997b; Riehl-Herwirsch and Lechner, 1995; Wurz, 1999). The main difference between the two cells was that Field 1 was provided with a low permeability cap, whereas Field 2 had a relatively high permeability top cover. Leachate data have been obtained for the period from early 1988 to mid-1995.

Attnang-Redlham landfill site is located about 120 km south west of Vienna, and has received about 20,000 t of waste each year since 1975. Before 1995, just residual wastes were being deposited, but subsequently composted waste has been landfilled, and has comprised 80% of the incoming waste streams. Other inputs include 10% commercial wastes, 5% bulky wastes, and 5% demolition wastes. Total depth of the site is about 18 m. Incoming pulverised residual MSW is wetted with landfill leachate (no sewage sludge enters the site), to be placed in windrows and naturally aerated for about 6 months. It is turned about 2 or 3 times and disposed of without further treatment. Leachate data exist for the period 1980-1998.

Halbenrain is located about 40 km south of Graz, in southern Austria, and received 70,000 t/a of residual wastes, with no pre-treatment, between 1979-1991. The waste comprised 70% commercial wastes, 8% bulky wastes and residual wastes, with 22% of other wastes (including sewage sludge with 30% dry solids, plus primary sewage screenings. Data exist for leachate quality from 2 areas of the site, for the period 1993-1998.

Ort/Innkreis is about 25 km from the town of Linz, and received MSW from 1974-1980, before composting of residual wastes began at the site. About 20,000 t/a of incoming wastes have received mechanical and biological pre-treatment, and these MBP residues comprise about 50% of the total landfill inputs. No sewage sludge or screenings are accepted. The whole incoming MSW spends 36 hours in a Dano drum, being sieved to 120 mm afterwards, and subsequently to 10 mm. The fraction greater than 10 mm goes directly for disposal, and the smaller fraction receives "natural aeration" in unturned windrows for a few months. 4000 t/a of this fraction is used for agriculture, and the rest is landfilled – achieving a density of about 0.9 t/m^3.

Breitenau and Attnang-Redlham

Results from the first pair of sites represent leachate quality data from two landfills with both "young" wastes and "older" wastes, including results for the period from 2 years before and up to 6 years after closure of the landfills.

Figure 4.4 compares BOD$_5$ values in leachates from the two landfills, data being normalised with respect to the dates on which the sites stopped receiving waste inputs. Leachate from the non-pretreated wastes contained much higher BOD values (to 30,000 mg/l) than that from the 80% MBP wastes, but within 2 years of site closure, BOD values in both leachates had fallen to similar low values. Figure 4.5 presents equivalent data for COD values, confirming both this conclusion, and also the fact that untreated MSW residual wastes can give rise to very high organic strength leachates.

Figure 4.6 provides comparative data for pH-values at the two sites, and results at both sites reflect the onsets of methanogenic conditions. Results for concentrations of ammonium, given in Figure 4.7 are both more variable and less conclusive, but values typically remain above 500 mg/l for both sites, at least up until 6 years after closure of each site.

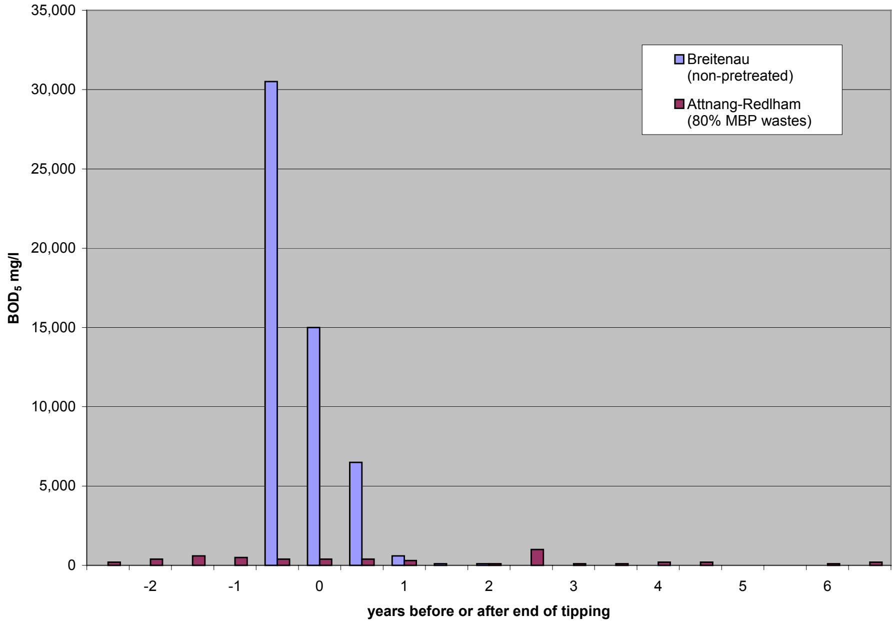


Figure 4.4 Comparison of BOD_5 values in leachates from Breitenau (field 1) containing untreated residual wastes, with leachates from Attnang-Redlham that has received 80 percent MBP wastes (Wurz, 1999)

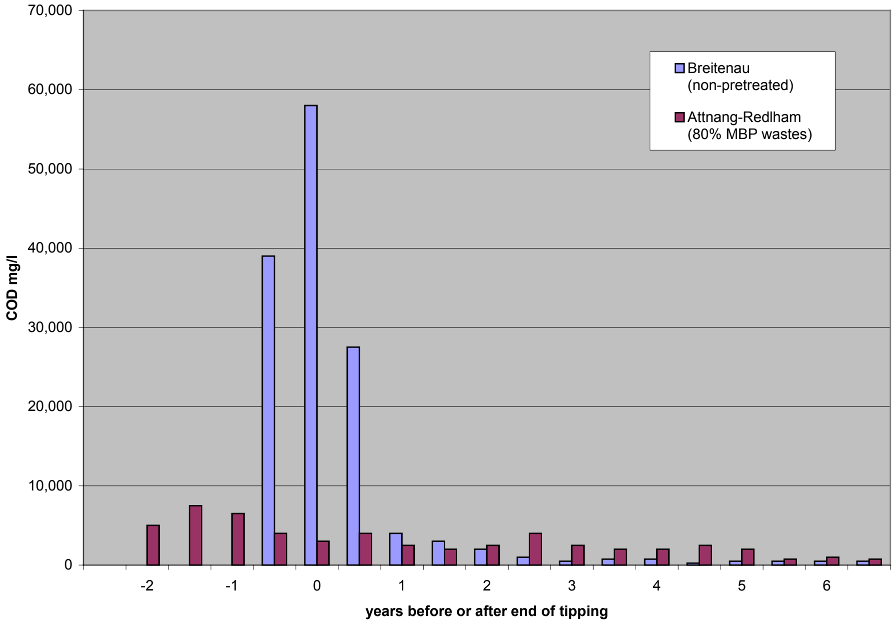


Figure 4.5 Comparison of COD values in leachates from Breitenau (field 1) containing untreated residual wastes, with leachates from Attnang-Redlham that has received 80 percent MBP wastes (Wurz, 1999)

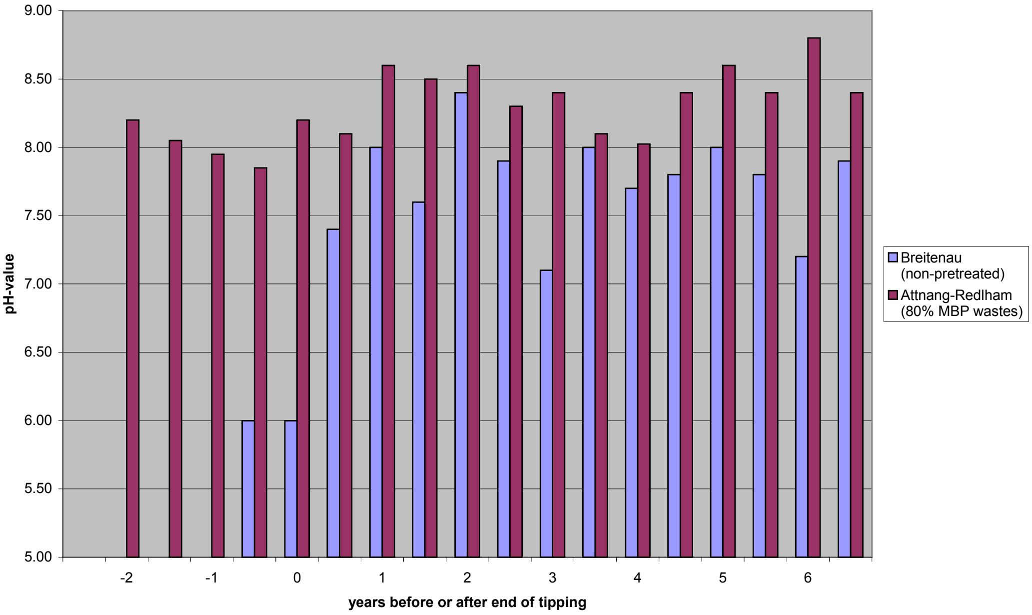


Figure 4.6 Comparison of pH-value in leachates from Breitenau (field 1) containing untreated residual wastes, with leachates from Attnang-Redlham that has received 80 percent MBP wastes (Wurz, 1999)

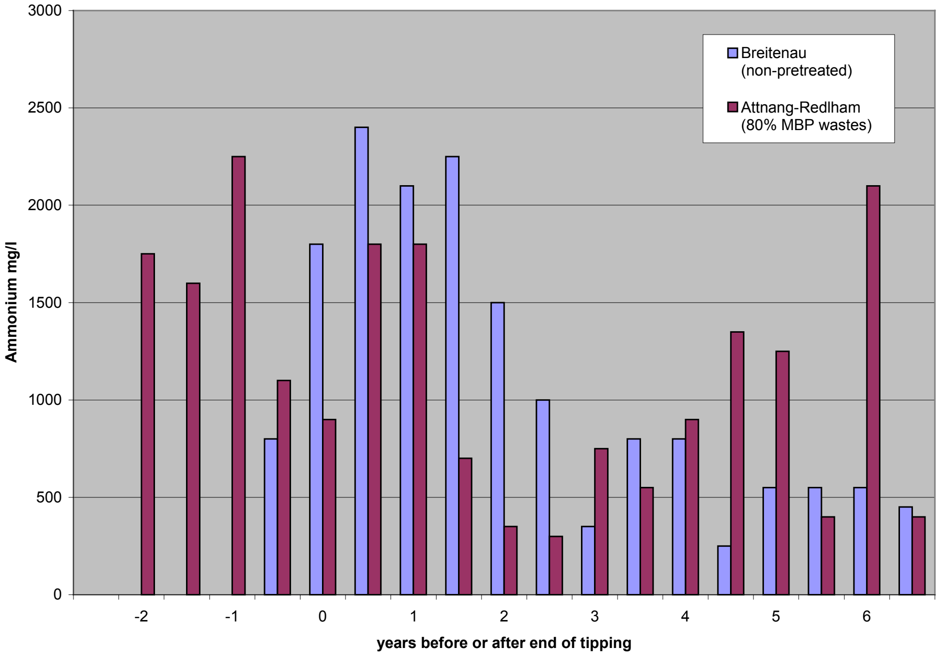


Figure 4.7 Comparison of concentrations of ammonium in leachates from Breitenau (field 1) containing untreated residual wastes, with leachates from Attnang-Redlham that has received 80 percent MBP wastes (Wurz, 1999)

Halbenrain and Ort/Innkreis
Data from the second pair of sites provide a much clearer comparison. Both sites are old sites, the former having received non-pretreated wastes, the latter an input of 50% MBP wastes since 1980. Figures 4.8 and 4.9 provide comparative data for BOD_5 and COD respectively, and demonstrate the much higher organic strength of leachates from the non-pretreated wastes, where COD continues to exceed 12000 mg/l, of which measured BOD_5 comprises nearly half.

This is surprising, since pH values in excess of 8.0 at each site (see Figure 4.10), would normally indicate establishment of methanogenic conditions. The much higher polluting potential of the unpretreated wastes at Halbenrain (which include sewage sludges) are apparent in continuing concentrations of ammonium above 4000 mg/l (see Figure 4.11), which have previously exceeded 8000 mg/l during 1993, two years after closure of the site.

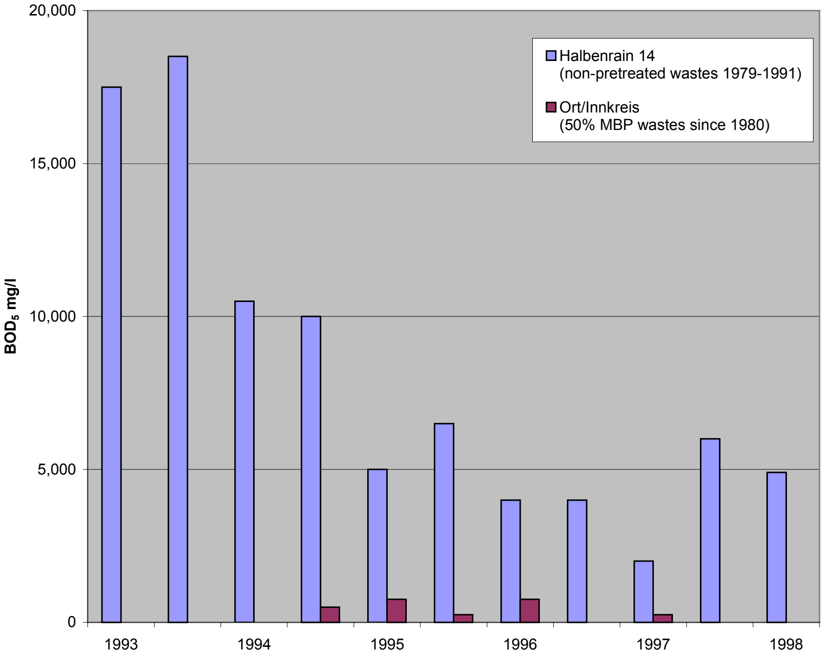


Figure 4.8 **Comparison of BOD_5 values in leachates from Halbenrain 14 containing untreated residual wastes, with leachates from Ort/Innkreis that has received 50 percent MBP wastes since 1980 (Wurz, 1999)**

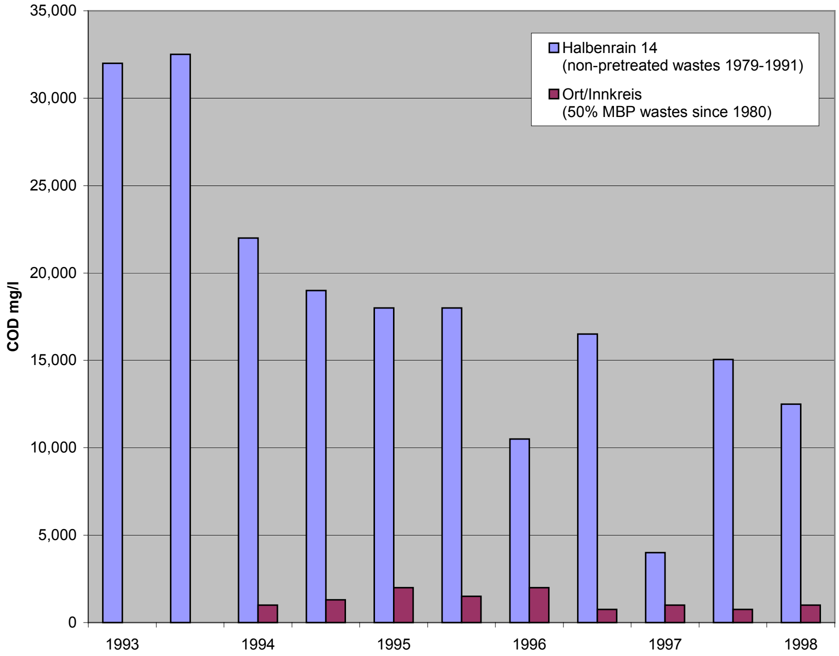


Figure 4.9 Comparison of COD values in leachates from Halbenrain 14 containing untreated residual wastes, with leachates from Ort/Innkreis that has received 50 percent MBP wastes since 1980 (Wurz, 1999)

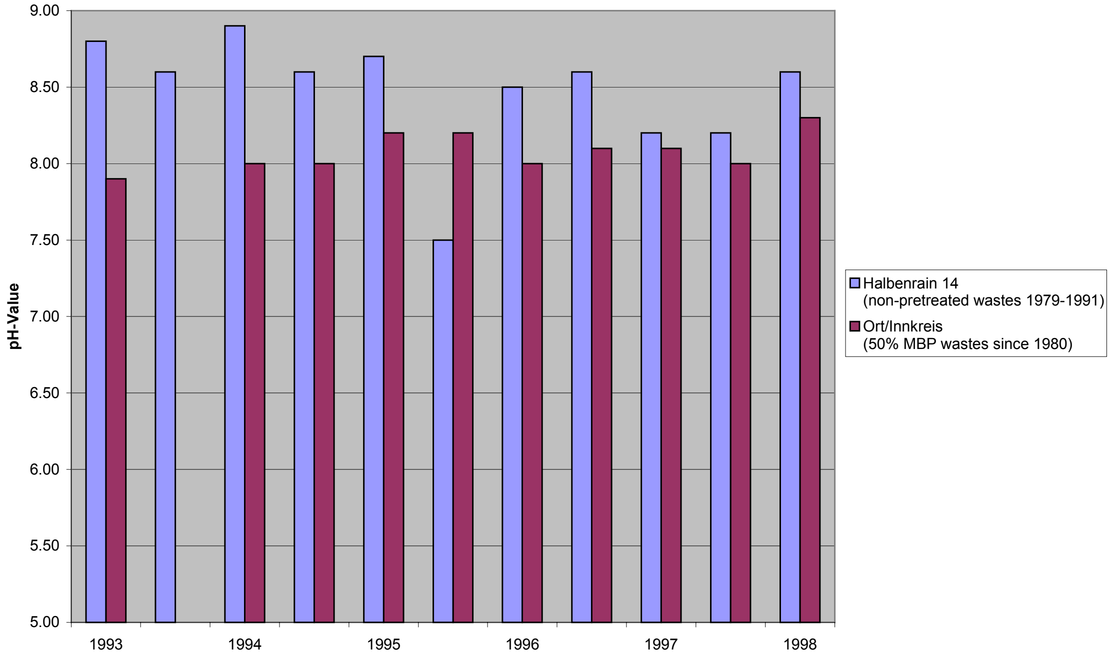


Figure 4.10 Comparison of pH-value in leachates from Halbenrain 14 containing untreated residual wastes, with leachates from Ort/Innkreis that has received 50 percent MBP wastes since 1980 (Wurz, 1999)

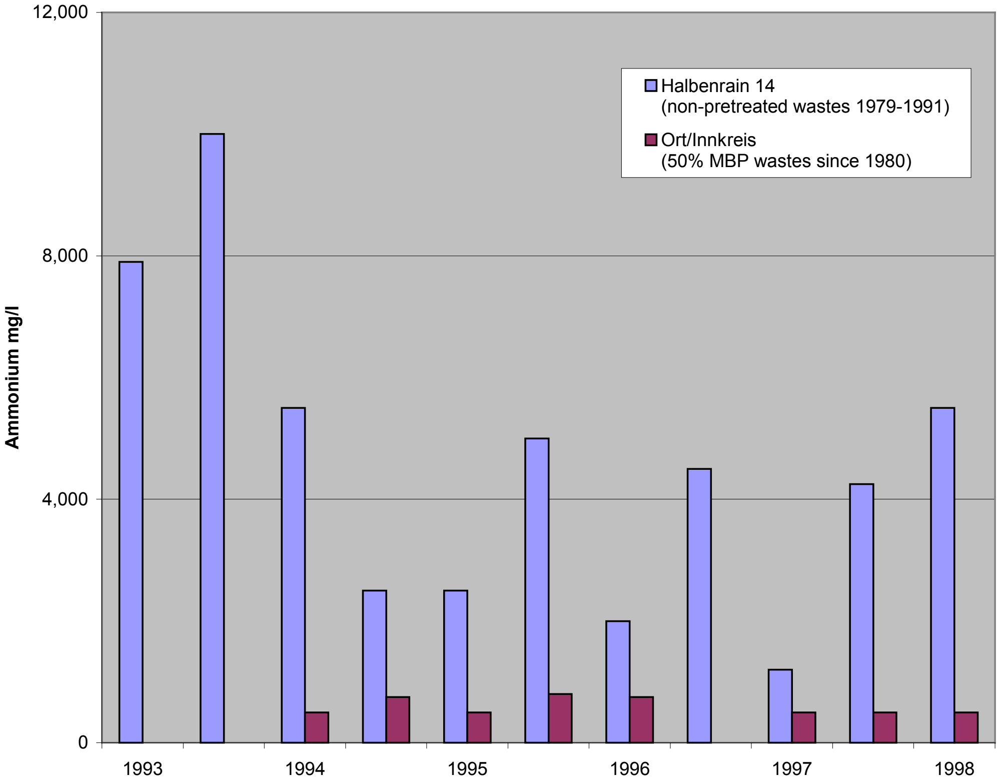


Figure 4.11 Comparison of ammonium values in leachates from Halbenrain 14 containing untreated residual wastes, with leachates from Ort/Innkreis that has received 50 percent MBP wastes since 1980 (Wurz, 1999)

Wurz (1999) provided overall summaries of the results from all of the sites that he examined, expressed in terms of maximum, minimum and mean values for each contaminant in leachates from each specific waste input landfill. Figures 4.12 to 4.16 below provide results for COD, BOD_5, pH-value, ammoniacal-N and Zn.

Results presented confirm both the potential high organic strengths of landfilled residual wastes that have not received MBP, and the more rapid onset of methanogenic conditions that can be achieved in wastes that have. However, the data shown, and also the more detailed results that Wurz includes within his thesis, indicate that less significant reduction in concentrations of ammoniacal-N and zinc (and also for nickel and copper) is achieved in the early years following landfill of MBP wastes. The data do indicate that concentrations of these contaminants in leachates reduce more rapidly in the period after 3 years following deposit.

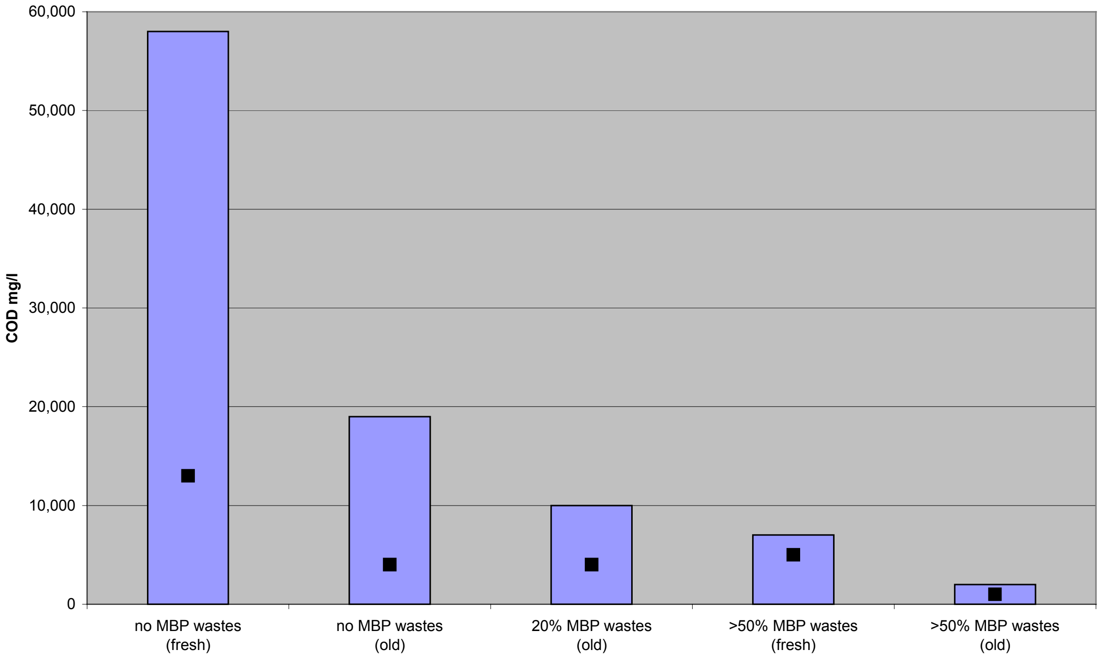


Figure 4.12 Ranges of COD values in leachates from various landfill ages receiving wastes containing different proportions of MBP wastes (Wurz, 1999) (mean value shown as black square)

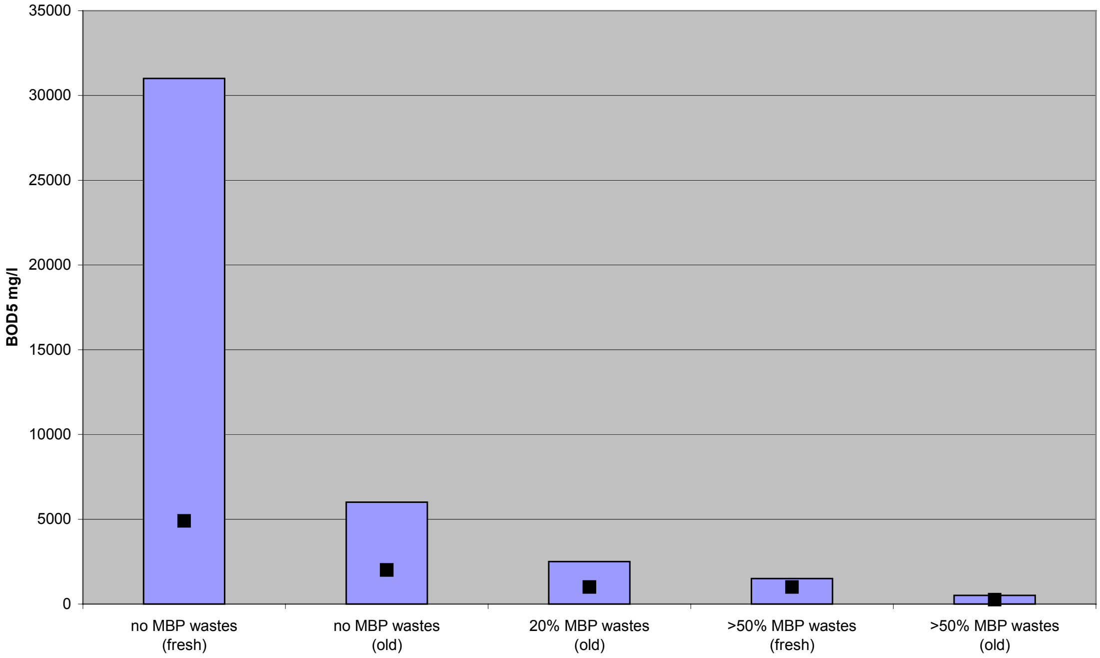


Figure 4.13 Ranges of BOD_5 values in leachates from various landfill ages receiving wastes containing different proportions of MBP wastes (Wurz, 1999) (mean value shown as black square)

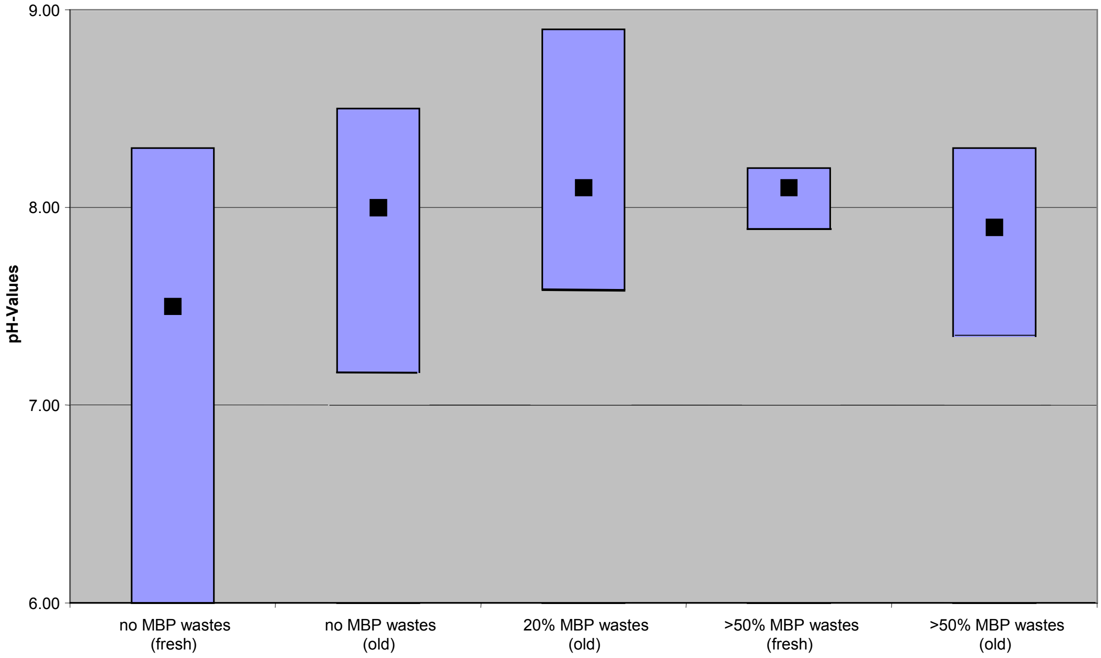


Figure 4.14 **Ranges of pH-values in leachates from various landfill ages receiving wastes containing different proportions of MBP wastes (Wurz, 1999)** (mean value shown as black square)

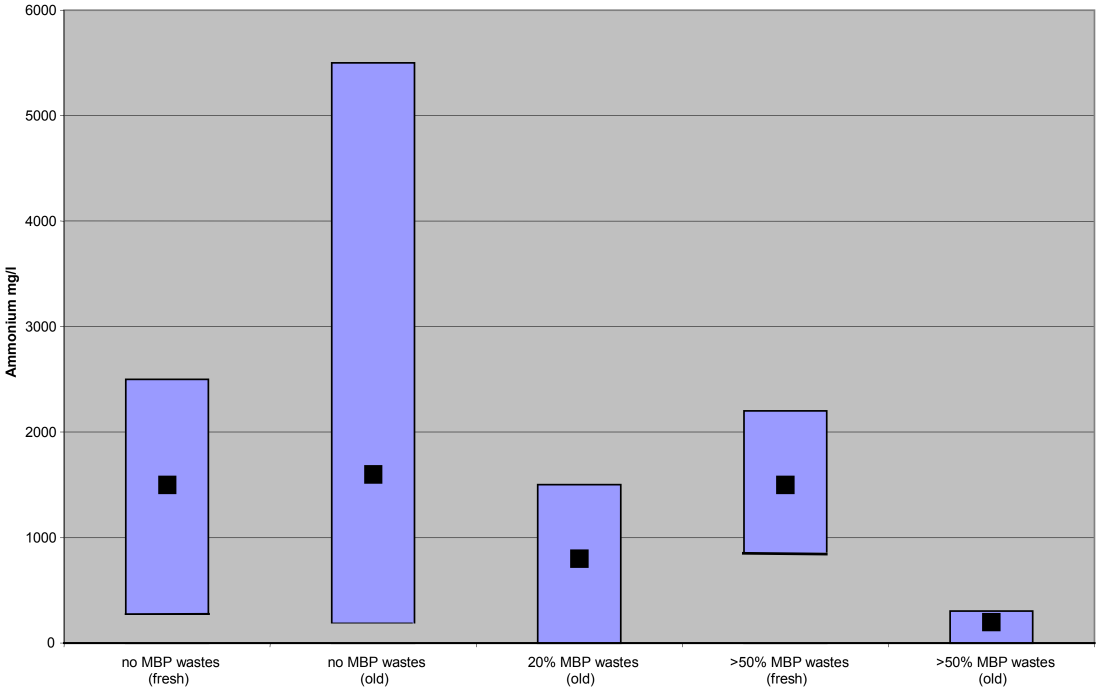


Figure 4.15 **Ranges of concentrations of ammonium in leachates from various landfill ages receiving wastes containing different proportions of MBP wastes (Wurz, 1999)** (mean value shown as black square)

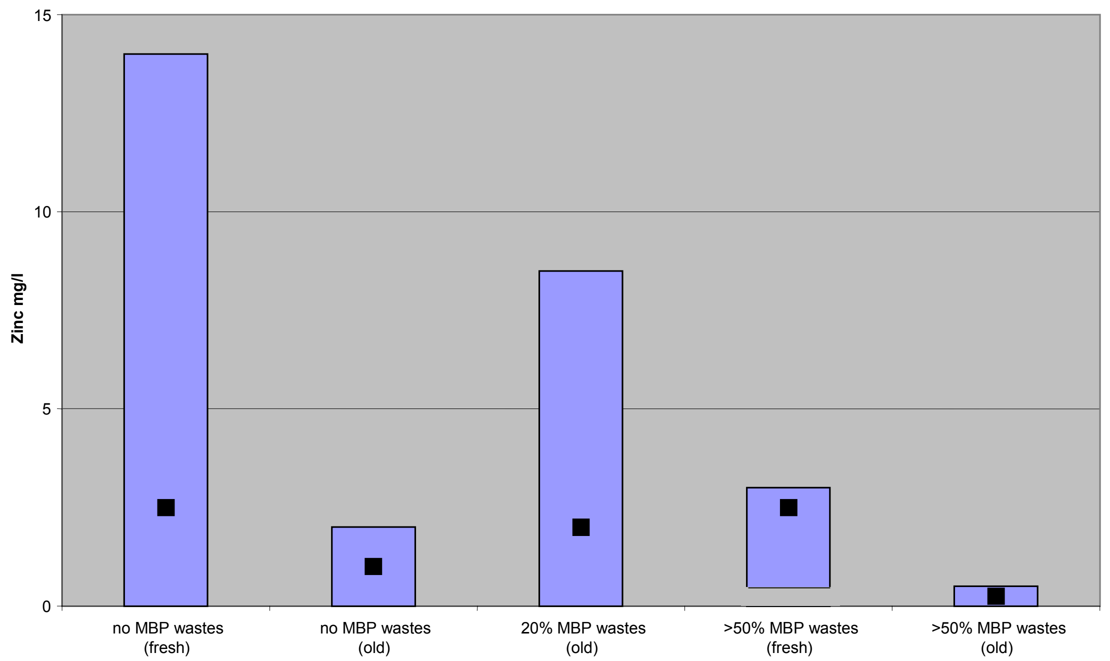


Figure 4.16 Ranges of concentrations of zinc in leachates from various landfill ages receiving wastes containing different proportions of MBP wastes (Wurz, 1999) (mean value shown as black square)

4.2 MBP research and practice in Germany

As in Austria, MBP has been widely developed and used in Germany. More than 1 million tonnes per annum of residual wastes are presently pretreated by MBP in approximately 20 plants of various sizes. More detailed background information on the role and development of MBP in Germany is given in Appendix 2.

In 1995 the German Federal Ministry of Education and Research (BMBF) launched a major research programme, which has involved 18 research projects in this field (Soyez *et al.*, 1997).
One area of research has focussed on the physical behaviour of MBP wastes in landfill sites. It has been widely found that MBP landfills can achieve landfilled densities as high as 1.5 tonnes per cubic metre (fresh weight), and extremely low hydraulic conductivity. Hydraulic conductivity values around 1 x 10^{-8} m/sec (Scheelhaase *et al.*, 2000; Dach and Eiebel-Pahlke, 2000; Scheelhaase and Bidlingmaier, 1999; Von Felde and Doedens, 1997), or 5 x 10^{-9} to 1 x 10^{-10} m/sec (Horing *et al.*, 1999) have been reported. This is despite the beneficial properties in terms of much reduced settlement of the pretreated wastes (Bidlingmaier and Scheelhaase, 1997). Other work has concentrated on attempts to correlate emission potential of landfilled MBP wastes with the results of short-term leaching and fermentation tests, with some success (Bidlingmaier and Scheelhaase, 1999). A large body of research has considered emissions from MBP wastes, including contaminants in leachates.

Much work has been undertaken on the quality of runoff, or “process water” that is generated during composting itself (e.g.. Loll, 1998; 2000). This can be extremely variable in strength, depending on process and weather conditions, and on dilution prior to sampling, but extensive data have been reported for both “leachates” and “condensates” by Loll (1998). Mean values from 26 samples of “leachate” and from 8 samples of condensate are presented in Table 4.3 below.

Table 4.3 Mean contaminant concentrations in run-off water and condensates from the MBP process (after Loll, 1998)

Determinand	Compost run-off	Condensate
COD	32700	1070
BOD_5	17700	980
Ammoniacal-N	410	~450
pH-value (unitless)	~6.5	~7.9
Conductivity (μS/cm)	14250	2490
AOX	2.43	0.77

(units mg/l unless otherwise stated)

Most of these liquors are generated in relatively small amounts, and are recirculated back into the composting process, although some are treated on-site in small plants with effluents discharged to sewer.

The great majority of MBP of residual wastes that arise in Germany, does so by means of composting. At present, anaerobic digestion (AD) is used only for source-separated waste streams, which are subsequently composted and used as soil conditioners/fertilisers. It is not possible to have an AD process alone – secondary composting is always necessary. Nevertheless, AD pre-treatment will undoubtedly increase in future, primarily as a means of complying with MBP air emission regulations, because a closed system with little gaseous emission is involved. At present there are few data to indicate the extent of pre-AD treatment that will be required to enable secondary windrow composting to be acceptable, but work is already underway.

Three major centres of research on MBP were visited during this study. These were:

- the Institute of Sanitary Engineering and Waste Management at The University of Hannover;
- the Technical University of Hamburg-Harburg; and
- the University of Wuppertal.

Research at these institutions is described in the following sections.

4.2.1 MBP research at the University of Hannover

The Institute of Sanitary Engineering and Waste Management (ISAH) at the University of Hannover, has been working at three government-funded, full-scale MBP plants in Lower Saxony. This work has looked specifically at the landfilling properties and emissions of pretreated residual wastes that have been subjected to varying degrees of composting, and are landfilled in separate landfill cells. Various papers have been published, some in English (Von Felde and Doedens, 1999a; 1999b; 1997), but most in German (e.g. Doedens and Kühle-Weidemeier, 2000; Doedens *et al.*, 2000; Doedens, 2000; Von Felde, 2000) to describe these studies.

The three Lower Saxony plants are located as follows.

Lüneburg (near Hamburg) - Mechanical processing with separation of fractions consisting of bulky materials, iron and RDF (material >100 mm) followed by 16 weeks of composting in a closed system, with outgoing air treatment through an air washer and biological filter.

Bassum (10km south of Bremen) - Mechanical processing with separation of fractions consisting of bulky materials, iron and RDF (material >80 mm) and anaerobic treatment of fractions <40 mm followed by 8 weeks of composting in a closed system.

Wiefels (Friesland/Wittmund) (in NE Lower Saxony) - Mechanical processing with separation of fractions consisting of bulky materials, iron and RDF. This is followed by 2 weeks of composting in a closed system with outgoing air treatment in biological filters, and 7 months of after treatment by open-air composting on the landfill, with different processing alternatives.

The plants incorporate a wide range of mechanical and biological processes for treatment of residual MSW prior to landfilling. They are described in detail in Table 4.4 below.

Table 4.4 Characteristics of the three mechanical-biological pre-treatment plants in Lower Saxony

	MBP Lüneburg	**MBP Bassum**	**MBP Wiefels Friesland/ Wittmund**
Capacity	29,000 t/a	65,000 t/a	55,000 t/a
Components of mechanical treatment	• Sorting • Milling • Magnet • Homogenisation • Sieving • Addition of water		
Sorted out fractions	• RDF • Compounds • Ferrous metals • Hazardous compounds • Bulky		
Type of biological treatment	Aerobic, active aeration	• Main stream aerobic, active aeration • Partial stream (fraction <40mm) 20 days one-stage anaerobic treatment	Aerobic, 1. intensive composting, active aeration 2. open composting with different processing variants, partly passive aeration
Encapsulation	Whole process encapsulated (112 days)	Whole process encapsulated (56 days)	• intensive composting encapsulated (14 days) • open composting on landfill (210 days)
Outgoing air treatment	Air washer + biofilter	Identical	Identical
Start of operation	April 1996	April 1996	Second half of 1997

The most extensive studies have been carried out on MBP wastes from the Lüneburg plant, in terms of both eluate testing and of landfill monitoring. Gladstone (1997) has described operation of the plant in detail. Table 4.5 below summarises results from eluate testing using the German DEV-S4 (DIN: 38414-S4) test – [this test is conducted at L/S 10, and involves 100 grammes of solid material mixed with 1 litre of distilled water, agitated for 24 hours without pH correction, and filtered].

These data provide a general indication of reduction in pollutant emission potential, within the constraints of a 24-hour agitated leaching test that takes no account of biodegradation potential.

Table 4.5 Results of eluates (DIN: 38414-S4) from residual MSW composted for various periods at MBP Lüneburg (after Von Felde and Doedens, 1997)

Pre-treatment (days)	Size fraction (mm)	pH	COD (mg/l)	TOC (mg/l)	TKjN (mg/l)	NH_4-N (mg/l)	NO_3-N (mg/l)
0	<100	7.2	3685	1374	88	35	5.3
1	<100	6.6	4363	1806		29.7	
14	<100	7.0	3443	1304	204	83	11
15	<100	7.5	1422	467		37.3	
22	<100	7.5	1762	519		40.4	
36	<100	7.5	1163	399		25.1	
50	<100	7.5	789	230		30.9	
56	<100	7.5	2241	857	163	80	11
78	<100	7.9	456	153		18.1	
84	<100	7.5	1381	496	81	36	7.6
106	<100	7.5	887	279		9.8	
119	<100	7.6	378	172	20	0	2.1
120	<100	7.6	756	258		54.9	
133	<100	7.3	372			1.3	1.1
140	<100	7.3	495	180	25	3	2.3
148	<100	7.6	779	291		14.8	
162	<100	7.6	505	193		18.9	

In Figure 4.17, eluate data for COD and TOC are plotted against time. COD values decrease during treatment, from a maximum of 4,300 mg/l to values below 500 mg/l, with corresponding values for TOC from 1,800 to about 150 mg/l. The final TOC concentration is close to the German criteria for Class II landfills, which require an eluate TOC of 100 mg/l or less, plus a tolerance of 50 mg/l. In several instances, eluate TOC values of below 200 mg/l can be reached by treatment times of less than 80 days.

Figure 4.18 shows the very close correlation between COD and TOC values in eluates, with COD values typically 2.6 times the TOC concentration (theoretical value for carbon compounds completely oxidised to CO_2 in the COD test would be 2.67).

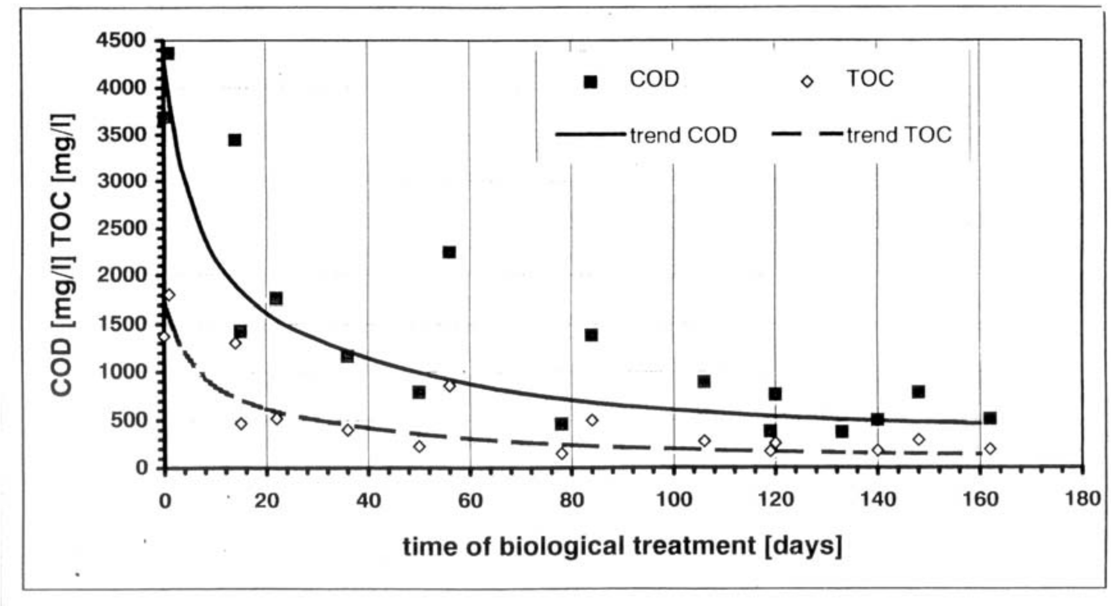


Figure 4.17 COD and TOC in eluates at L/S 10 (DIN:38414-S4) related to treatment time in MBP Lüneburg (after Von Felde and Doedens, 1997)

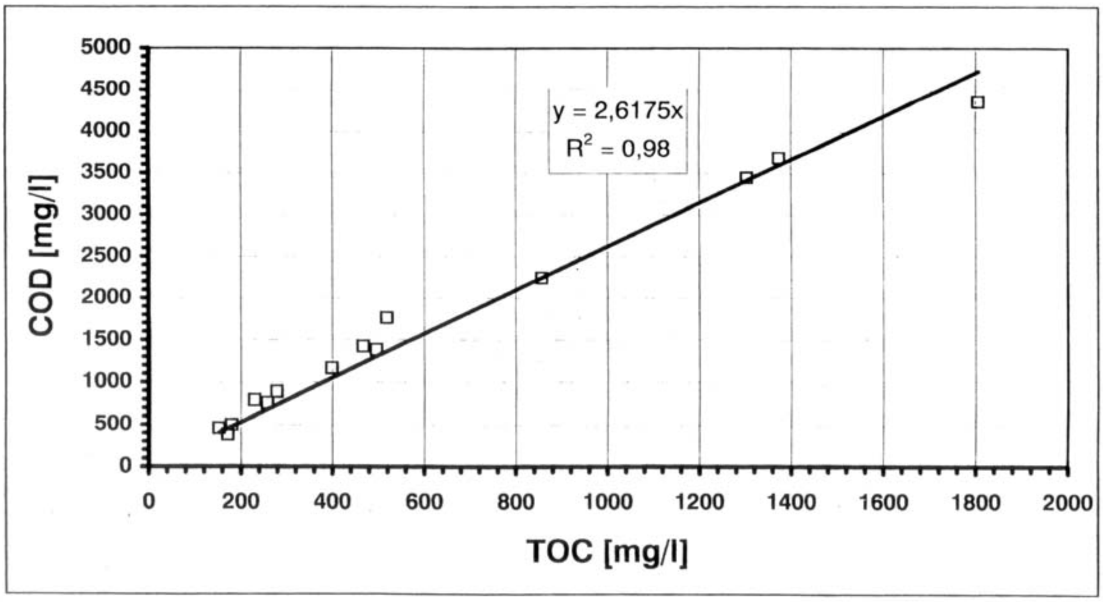


Figure 4.18 Relationship between COD and TOC in eluates (L/S 10) of composted material (various treatment times) from MBP Lüneburg (after Von Felde and Doedens, 1997)

It is valuable to compare these data, which use a high L/S ratio of 10:1, with actual data from pilot-scale and full-scale landfilling of the MBP Lüneburg wastes. The waste has typically been subjected to a total of about 112 days contained composting (corresponding eluate COD about 400-900 mg/l and TOC about 170-280 mg/l). Electricity use to achieve this degree of composting in the plant is about 48 kWh per tonne, of which 20-25 kWh is consumed in the mechanical section, and 15-20 kWh in the composting process (Gladstone, 1997).

Table 4.6 compares initial leachate quality from the full-scale landfill cell receiving MBP wastes from Lüneburg, with laboratory-scale landfill reactor trials using similar wastes, with typical acetogenic and methanogenic leachates from untreated MSW, and with German limit values for direct discharge (under German ‘51, Anhang Rahmen Abwasser Verwaltungsvorschrift’).

Table 4.6 Comparison of leachate from landfill of Lüneburg MBP with typical untreated MSW landfill and German limit values (results in mg/l except pH-values) (after Von Felde, 1999; Ehrig, 1980; ATV, 1988; Theilen, 1995; Leikam and Stegmann, 1997)

Parameter	MBP Lüneburg, initial phase of landfilling	Laboratory scale landfilling of MBP waste, initial phase	Untreated MSW acid start phase	Untreated MSW methane phase	German limit values
pH	7.5	7.0-7.5	4.5-7.5	7.5-9.0	-
COD	700-2500	2,000-2,400	6,000-60,000	500-4,500	200
TOC	300-950	-	2,000-30,000	200-2,000	67
BOD_5	1-55	100-1100	4,000-40,000	20-550	20
COD/BOD_5	20-150	2-20	2	15-20	-
TKN	10-37	-	1350	1350	-
NH_4-N	0-27	90-130	750	750	-
NO_3-N	15-66	-	-	-	-
NO_2-N	0.1-1.7	-	-	-	2
$N_{inorganic}$	16-75	-	750	750	70
total N	35-140	200-250	1,350	1350	-
COD/total N	4.6-8.7	10	8-12	2-3	-
AOX	0.1-0.9	-	0.3-3.4	0.3-3.4	0.5

The first landfill cell at Lüneburg has an area of 0.5 ha, and began to receive MBP waste inputs in summer 1997. The first 2 m of fill was uncompacted to protect the basal liner, and wastes have been tipped to a very shallow total depth of between 2.5 and 4.0 m. Conditions within the landfill remain aerobic, with concentrations of methane below 1% v/v. The dominant form of nitrogen is as nitrate-N.

Data in Table 4.6, and in Figure 4.19 for leachate composition during the first 2 years in the landfill cell at Lüneburg, show COD values in the range 700-2500 mg/l, with lowest levels during summer months. The acid phase typical of landfilled but unpretreated MSW does not occur, but nevertheless extensive treatment of leachate would still be required to achieve discharge standards. Treatment would primarily need to focus on ammoniacal-N, and on the relatively non-biodegradable COD in leachate. Although BOD_5 comprises a small proportion (only 1 to 5%) of COD values, it remains to be seen what reductions could be achieved by an acclimatised biological population in a leachate treatment plant. This is likely to be up to 30 or 40%.

There is also some uncertainty about the longer-term behaviour of such landfilled MBP wastes, and whether anaerobic conditions may subsequently develop – especially where much greater compaction efforts and landfill depths increase waste densities, and reduce pore spaces and hydraulic permeability.

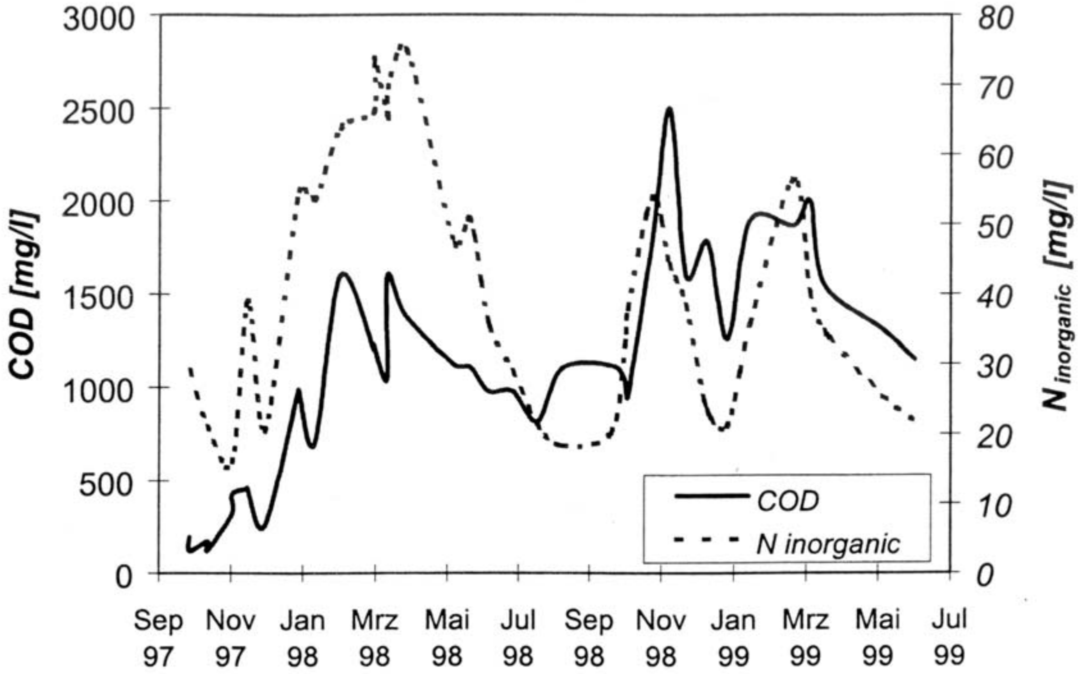


Figure 4.19 Concentrations of COD and inorganic-N in leachate from full-scale landfilling of MBP at Lüneburg (Von Felde and Doedens, 1999a)

4.2.2 MBP research at the University of Hamburg-Harburg

Over many years, Professor Rainer Stegmann and co-workers at the Technical University of Hamburg-Harburg have been working on the use of MBP of MSW as a means of improving control over their subsequent degradation in landfills (e.g. Heerenklage *et al.*, 1994). The MBP is intended to reduce production of strong leachates and extensive gas formation, reduce landfill settling (Stegmann, 1992), and lessen the clogging of leachate drainage systems that has commonly been observed (e.g. Ramke, 1991; Brune *et al.*, 1991).

Easily-degradable fractions of the MSW are degraded during biological pre-treatment, resulting in lower emissions of leachate contaminants, landfill gas, and odours from the landfill. Nevertheless, after biological pre-treatment the remaining wastes still contain slowly-degradable organic substances such as lignin, waxes, humic acids etc., and so biological processes will continue to be the dominant feature of their behaviour when landfilled. Stegmann (1991) calculated that only 10% of the biodegradable organic fraction might remain after effective pre-treatment, which corresponds to a reduced gas production in the order of 15-20 m^3/tonne of biologically pretreated MSW (see Figure 4.20).

Much work has focussed on achieving specified eluate concentrations in leachate from the standard German DIN leaching test, which uses a L/S ratio of 10:1 during a 24-hour test. In Germany, regulation under TA Siedlungsabfall (see Appendix 2 and Stegmann *et al.*, 1998a) demands that a TOC value of 100 mg/l or below must be achieved if MBP wastes are to be landfilled.

The reliability of the 24 hour leaching test has long been considered to be doubtful (e.g. Krogmann, 1993). Nevertheless, many early trials demonstrated successful reductions in the measure it provided. After a biological pre-treatment of 12 weeks, the TOC of an MSW eluate could be reduced from 800 mg/l to 230 mg/l (average of several tests, Damiecki, 1992), although the MSW investigated had only a 5% organic fraction, due to a high content of demolition wastes. In other trials, the TOC of an eluate of MSW (from which recyclables and garden waste had been excluded) could be as low as 79 mg/l (Spillman, 1993).

At an early stage in studies, Stegmann (1992) had proposed that leachates from the landfilled MBP wastes should be similar to leachate from a landfill in a methanogenic phase of decomposition.

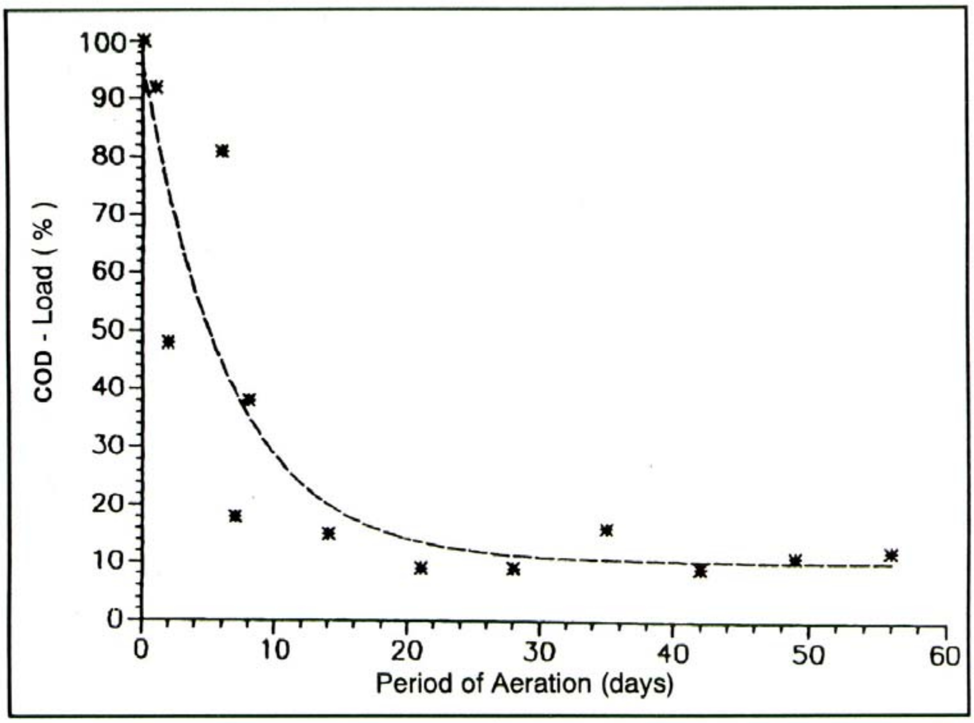


Figure 4.20 **Percentage of COD load remaining in MSW after up to 100 days of anaerobic fermentation, as a function of period of pre-aeration/composting (Control without pre-aeration = 100%) (after Spendlin, 1991)**

Kettern (1993) confirmed this to some extent, with pilot-scale anaerobic lysimeters in a large-scale laboratory experiment. From day 70 to day 330 of the trials, the COD of leachate from the MBP wastes decreased from 15,000 to about 1,500 mg/l – similar to methanogenic leachates from untreated MSW landfills (e.g. Robinson, 1996).

Other relatively early work looked at the benefits of in-situ pre-composting of the bottom layers of waste within a landfill, in order to prevent a too-vigorous acid phase of degradation, even when untreated MSW is subsequently emplaced above this. Figure 4.21 shows results from full-scale landfills, where such techniques were tested.

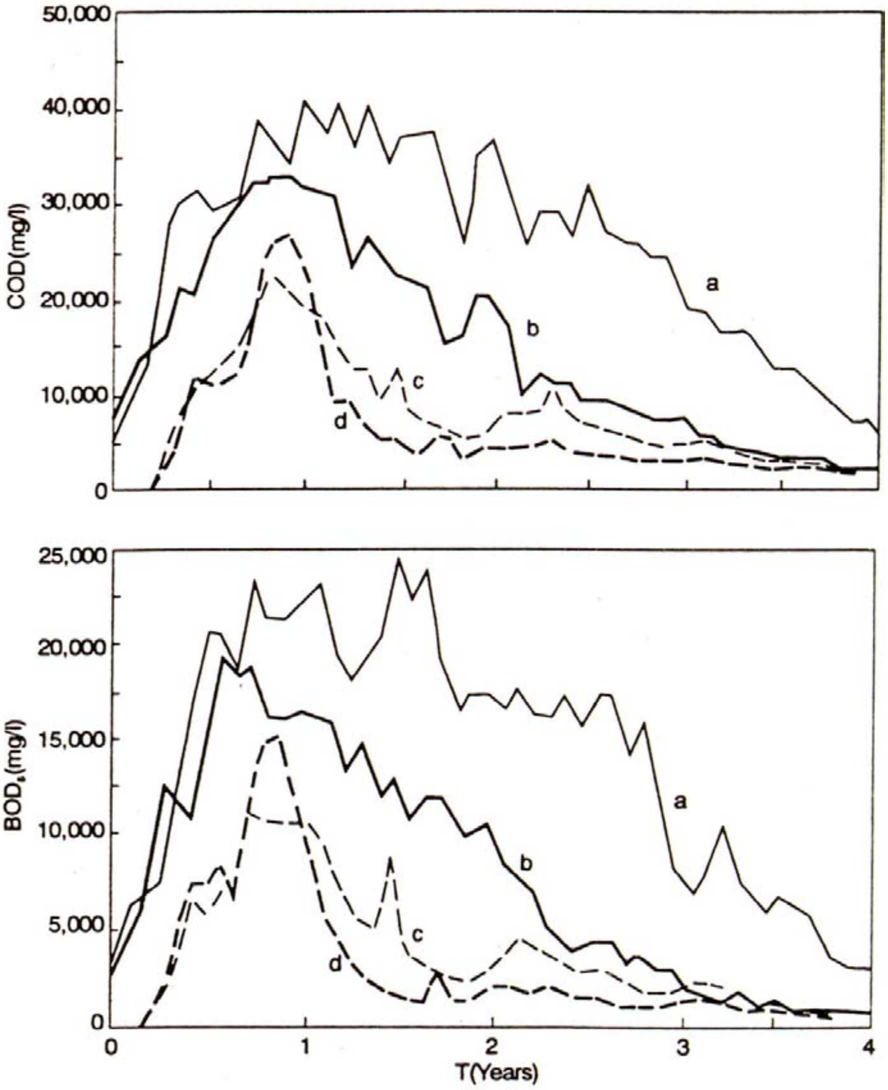


Figure 4.21 COD and BOD values in leachate from full-scale landfills employing compaction of the waste (a) and recirculation (b) and as alternatives pre-composting (c) and pre-composting combined with leachate recirculation (d) (after Stegmann and Spendlin, 1986; 1989)

More recent work in which Stegmann has been involved has provided a firmer basis of understanding of the effects of MBP on residual waste characteristics. Recent work by Scheelhaase (Scheelhaase, 2000; Scheelhaase and Bidlingmaier, 2000) included detailed study of the effects of period of composting on TOC and BOD_5 values in eluates from the DEV-54 (DIN) test. Figures 4.22 and 4.23 below present these results.

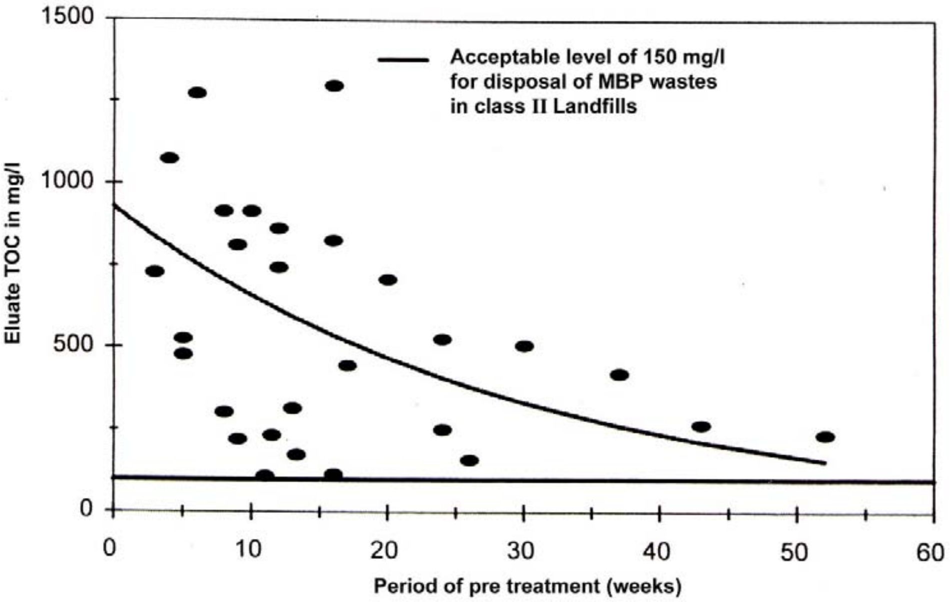


Figure 4.22 **Effect of period of composting pre-treatment of residual wastes (wastes) on TOC in eluate from the DEV-54 (DIN) test (Scheelhaase, 2000)**

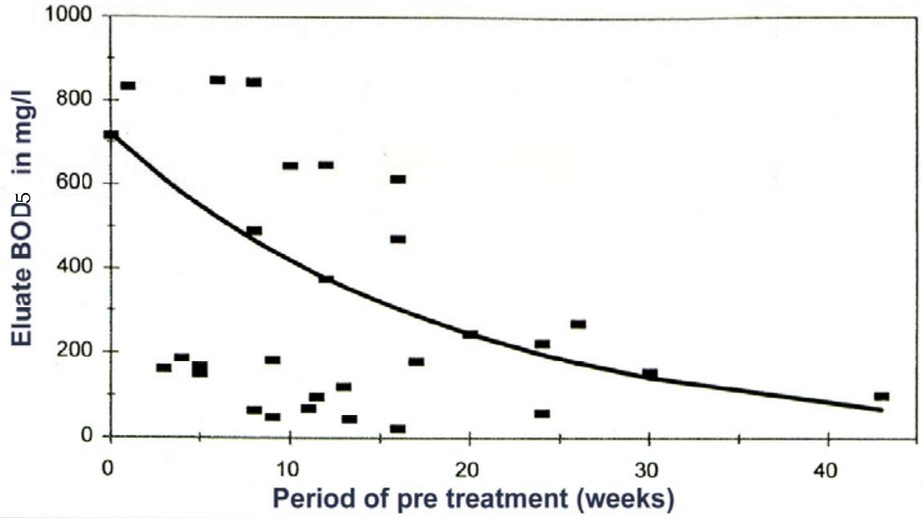


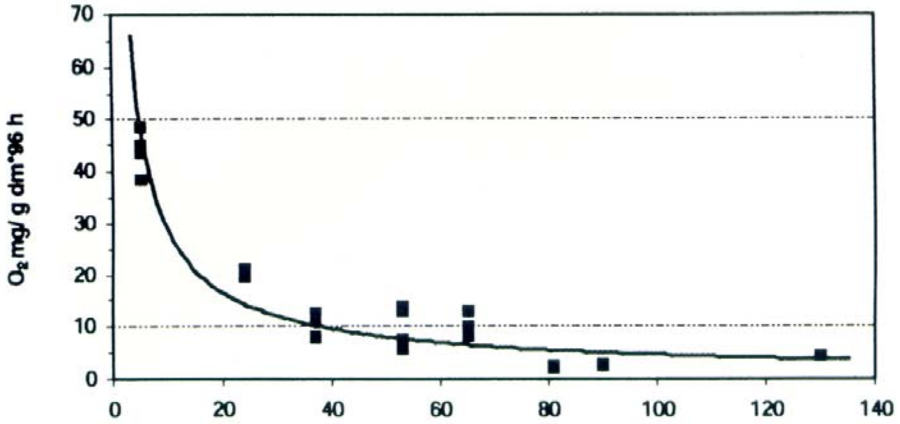
Figure 4.23: **Effect of period of composting pre-treatment of residual wastes (wastes) on BOD$_5$ in eluate from the DEV-54 (DIN) test (Scheelhaase, 2000)**

German legislation demands that TOC values in eluate from MBP wastes in the DEV-54 (DIN) test should not be greater than 150 mg/l, if these materials are to be disposed of into a Class II Landfill. Extensive treatment is clearly necessary to meet such standards, and doubt remains about how appropriate such 24-hour leaching tests are to check compliance.

It is exceptionally difficult to obtain high rates of flushing at any MBP waste landfill. Experience at Hamburg-Harburg is that emplaced density of composted wastes can achieve about 1.5 tonne per cubic metre, resulting in permeabilities, in the order of 10^{-7} to 10^{-9} m/sec. Although the high COD and BOD values of early acetogenic phases of decomposition can be avoided by MBP, the long-term COD “tail” in leachates is little reduced compared to that of residual wastes not pretreated biologically. This demonstrates that although MBP processes can degrade readily-degradable organic waste components, many of these materials which only degrade over longer periods remain essentially unaffected.

A “Sapromat” test can be used to determine the reduction of respiration rate of residual wastes during the composting process. Work has demonstrated that there is no significant difference between composting in windrows or in closed containers (Leikam and Stegmann, 1999; 1997; 1996; Fricke *et al.*, 1995) (see Figure 4.24).

(a) windrow composting



(b) container composting

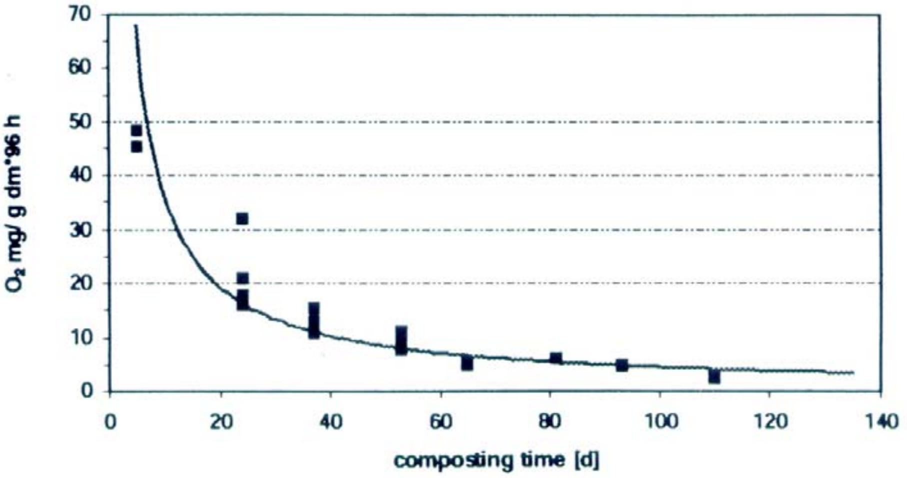


Figure 4.24 Reduction in waste respiration rates during (a) windrow composting, and (b) container composting, of residual wastes in full-scale MBP plant (Leikam and Stegmann, 1996)

In both cases the respiration rate at the end of composting amounted to 5 mg O_2 per gramme of dry matter over 96 hours, which was recommended by the authors to define a biological stable material.

The behaviour of these composted residual wastes when landfilled was modelled in landfill simulation tests, where results could be compared with behaviour of residual wastes not subjected to such pre-treatment (see Figure 4.25 below).

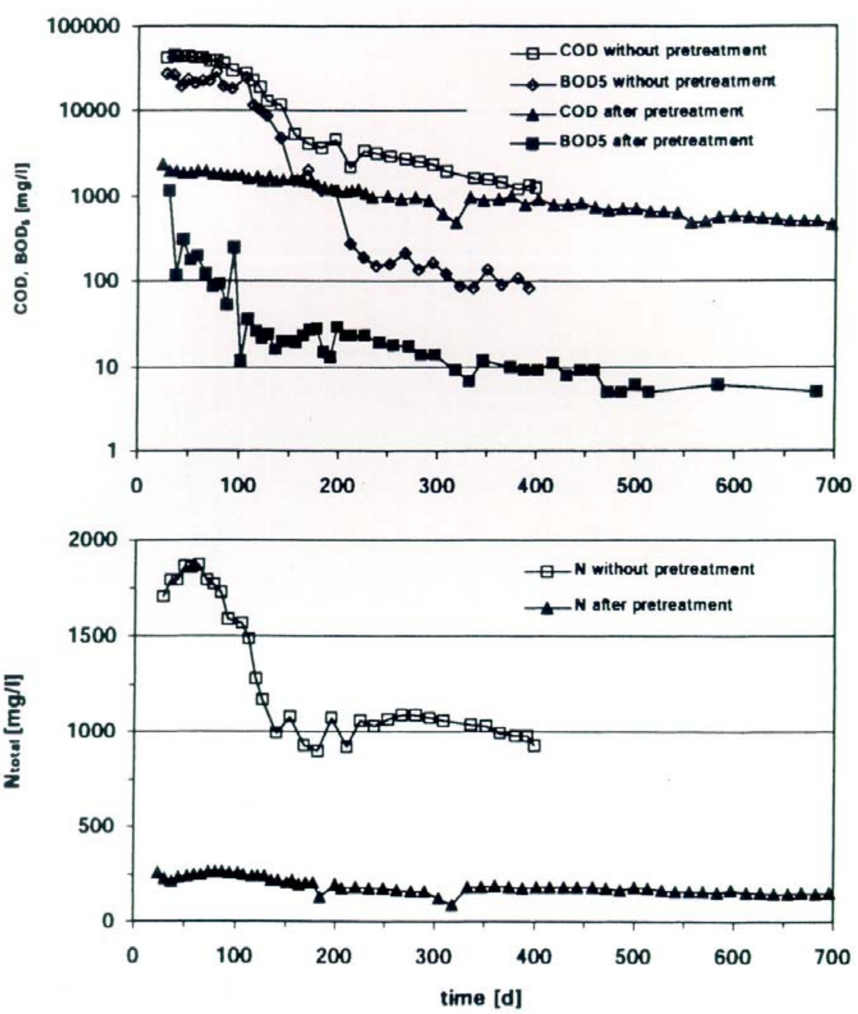


Figure 4.25 COD, BOD_5 and total-N in leachates from untreated and treated residual wastes during landfill simulation tests (Leikam and Stegmann, 1999) (NB: liquid:solid ratio at 400 days = 1.6:1, at 700 days = 2.6:1)

For pretreated residual wastes, the acetogenic phase is absent, and after about 250 test days the COD of the leachate is below 1000 mg/l (BOD_5 <20 mg/l). A trial period of 250 days corresponds to a flushing period of about 50 years for a 20 m deep landfill, with an annual infiltration rate of 250 mm, or 250 years for a similar site where capping reduces annual infiltration to only 50mm.

A much more significant benefit of pre-treatment becomes apparent when concentrations of total-N (primarily ammoniacal-N) are considered. Whereas the total-N content in leachate from untreated residual waste stabilises at about 1000 mg/l, this value is below 200 mg/l for pretreated wastes. This will result in substantially reduced aftercare costs in terms of leachate treatment, and possibly also in terms of reduced periods of aftercare, although no data have been obtained that can confirm this.

Further studies compared the effects of MBP in terms of total load of contaminants emitted by treated and untreated wastes, and results are presented for COD and total nitrogen blow, in Figure 4.26.

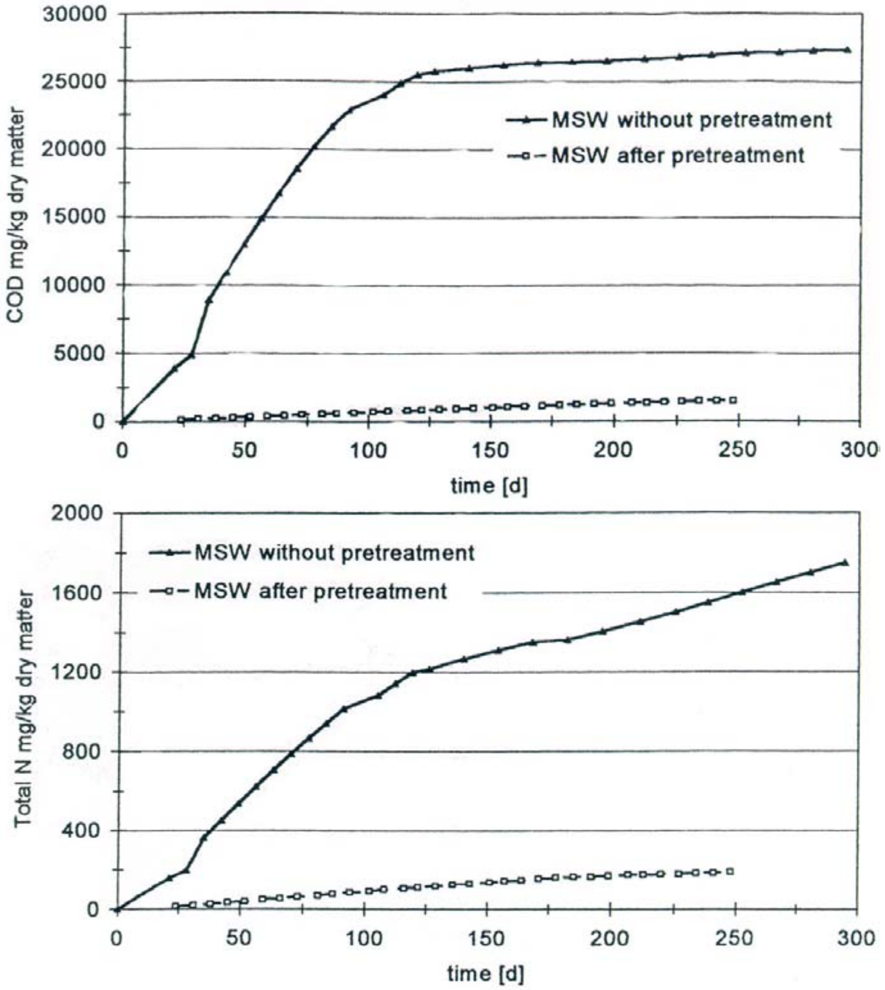


Figure 4.26 Total COD and ammoniacal-N load for untreated, and biologically pretreated (4 months) residual wastes in landfill simulation test. (Leikam and Stegmann, 1999; Stegmann *et al.*, 1996)

For a liquid:solid ratio of 1:1 in the simulation trials (representing periods of typically from 50 – 250 years at a full scale landfill), the mass transfer of contaminants was about 90 percent less for biologically pretreated wastes than for untreated residual wastes. This is similar to the reduction in gas production noted in similar tests, and corresponds well with the reduction in waste respiration rates noted earlier (Figure 4.24).

Danhamer and Jager (1999) undertook similar landfill lysimeter studies using a range of different MBP wastes from full-scale plants, and compared these with results from an equivalent lysimeter using untreated residual wastes. They provided much more detailed composition data for leachates in the methanogenic phase, which compare well with the results above. Again, the reduction in nitrogen is most pronounced and important (see Table 4.7).

Table 4.7 Data for detailed leachate composition from landfill lysimeters operated by Danhamer and Jager (1999), containing untreated residual waste, and MBP wastes from a range of German full-scale plants

Location (1)	DA0	DL1	DL2	DL3	QB1	WS1
No. of Samples	10	3	5	7	13	6
Volatile solids (%)	45.9	30.6	31.8	4.2	37.7	21.2
Determinand:						
COD	172,000	2,780	1,170	540	4,000	1,900
BOD_5	123,000	52	9	158	111	14
TOC	57,000	1,260	450	250	2,100	750
pH-value	6.1	7.5	7.7	9.5	7.5	7.7
EC (µS/cm)	41,000	23,000	21,000	12,000	20,000	16,000
Ammoniacal-N	3,965	197	11	56	292	340
Nitrate-N	<0.5	<0.5	<0.5	<0.5	<0.5	<1.0
Nitrite-N	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5
Sulphate	4,100	9,500	4,200	2,300	700	5,300
Fluoride	<0.5	<0.5	<0.5	0.8	<0.5	<0.5
Chloride	9,100	11,300	6,900	5,700	6,200	4,100
Cyanide (free)	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Sodium	6,100	4,300	4,500	1,900	3,900	1,600
Magnesium	1,164	268	96	4	226	412
Potassium	3,400	1,500	1,400	900	2,100	1,600
Calcium	6,500	910	320	100	390	380
Chromium	0.41	0.14	0.04	0.03	0.21	0.09
Iron	752	14	4	5	17	16
Nickel	2.10	0.23	0.71	0.16	0.40	0.09
Copper	1.41	0.71	0.80	0.28	0.52	0.18
Zinc	102	3.4	1.0	0.22	1.6	1.2
Cadmium	1.860	0.013	0.016	0.010	0.130	0.022
Lead	0.63	0.31	0.17	0.13	0.46	0.11
Arsenic	0.035	0.013	0.008	0.008	0.010	0.005
Mercury	0.005	0.003	0.003	nd	0.013	0.001
AOX	11.1	1.33	0.73	0.18	1.28	0.97
Weeks composting:						
• Intensive	nil	4	4	2	16	3
• Secondary	nil	9	43	1	8	19

Notes: (1)

- DAO = Darmstadt (untreated residual waste)
- DL1, DL2 and DL3 = Darmstadt and Lohfelden
- QB1 = Quarzbichl
- WS1 = Wittstock

(2) All results in mg/l except pH-value and EC (µS/cm)

Kabbe (2000) published a detailed thesis (in German) at the University of Aachen, that contains much complementary data from pilot-scale outdoor lysimeters containing 1.2 m^3 of untreated MSW, untreated residual wastes, MBP residual wastes, and of MSW incinerator bottom ash.

The lysimeters were operated (without temperature control – typically 5 to 15ºC) for a period of nearly 4 years, and produced results similar to those reported above. Table 4.8 contains summary data for a range of heavy metals in the respective leachates.

Table 4.8 Concentration of metals in leachates from test lysimeters containing various untreated and treated wastes (Kabbe, 2000)

Metal	units	Untreated MSW		Untreated MSOR		MBP wastes		Incinerator bottom ash	
		min.	max.	min.	max.	min.	max.	min.	max.
Chromium	mg/l	0.014	4.18	0.042	0.7	0.008	0.06	<0.0001	0.014
Manganese	mg/l	<0.01	0.83	<0.01	0.82	0.01	0.69	<0.001	0.09
Iron	mg/l	27	513	18	294	31	58	<0.005	8.6
Nickel	µg/l	50	1200	60	1200	<20	420	<20	440
Copper	µg/l	2	120	<1	66	<1	170	24	220
Zinc	µg/l	30	1300	20	4400	10	360	<10	260
Cadmium	µg/l	<2	358	<1	203	<2	11	<2	70
Lead	µg/l	<5	360	5	110	<5	600	28	310
Arsenic	µg/l	3	42	4	39	<1	34	<2	15
Mercury	µg/l	<0.5	14	<0.5	71	<0.5	<4	<0.5	4.3

Results from all three lysimeters which contained non-incinerated MSW are similar in terms of heavy metal content. Concentrations of ammoniacal-N (Figure 4.27) are similar for MSW and MSOR, stabilising at about 500 mg/l, but lower for the MBP wastes – at about 150 mg/l, similar to results reported earlier in Figure 4.25.

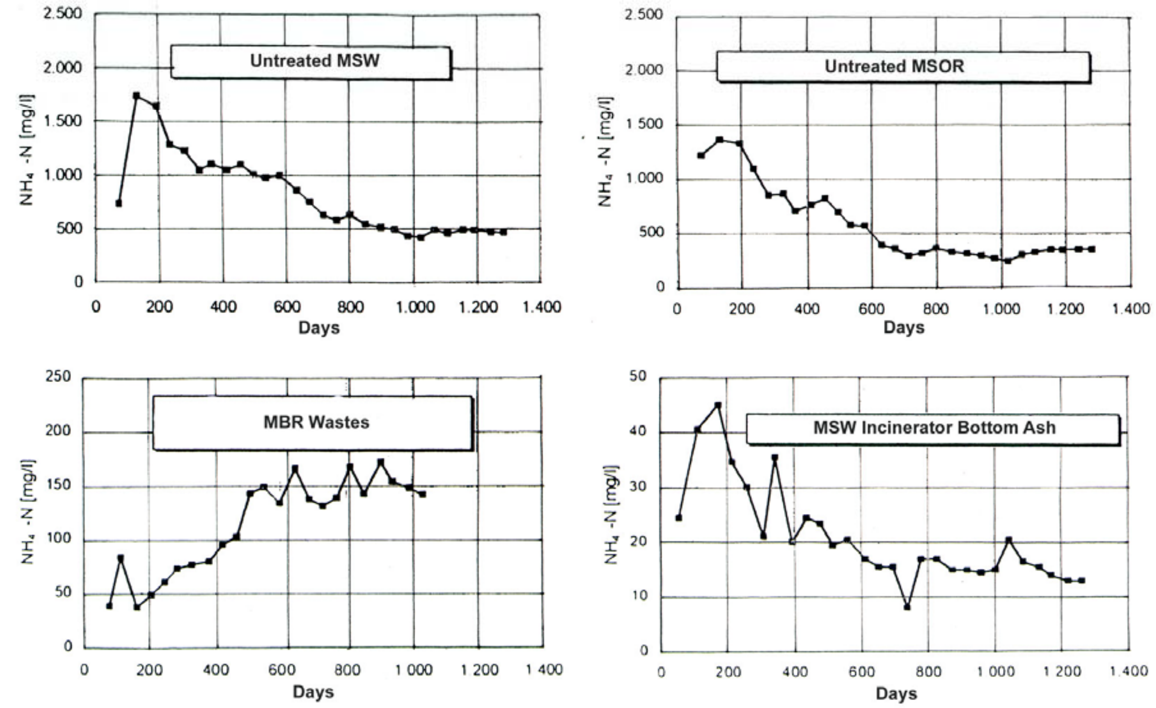


Figure 4.27 Concentrations of ammoniacal-N in leachates from landfill lysimeters containing various wastes. (after Kabbe, 2000) (N.B. MBR = MBP)

4.2.3 MBP research at the University of Wuppertal

For over two decades Professor Hans-Jürgen Ehrig, currently at the University of Wuppertal, has been one of very few international researchers to be at the leading edge of work to try and understand in detail the composition of leachates from landfilled MSW. As long as 20 years ago, he published some of the definitive papers on this topic (e.g. Ehrig 1978, 1983).

With various co-workers, he has progressed understanding of leachate composition at landfills during active and aftercare phases (Ehrig and Krümpelbeck, 1999) in a major study. Various papers in English have summarised the research data (e.g. Krümpelbeck and Ehrig, 1999, Ehrig and Krümpelbeck, 2001). The study is based on analysis of data from more than 50 German landfills, many of which cover periods of up to 30 years, and include leachate quality results for extended periods of aftercare. Data cover an extensive range of determinands, and Table 4.9 indicates the type of information available, for more important parameters.

Workers at the University of Wuppertal, together with collaborators at other regional universities (e.g. T. Scheelhaase and W. Bidlingmaier at the University of Essen), have generated valuable data which compare this extensive database with equally detailed data from many pilot-scale landfill simulation reactors. Figure 4.28 below summarises results for COD values in leachate from both full-scale landfills (in blue) with equivalent data from pilot-scale landfill simulation reactors (in red). Beyond the L/S ratio of 0.5, no data are available from full-scale sites, although pilot-scale reactors have been operated up to a value of ten times this.

Table 4.9 Minimum, median and maximum concentrations of contaminants in leachates from untreated MSW in German landfills of varying age (after Ehrig and Krümpelbeck, 1999)

Landfill Age	(years)	1-5	6-10	11-20	21-30
Determinand:					
COD	max.	22,700	22,500	29,150	6,997
	median	3,810	3,255	1,830	1,225
	min.	303	194	120	123
BOD_5	max.	16,000	64,880	25,800	1,100
	median	2,285	1,210	465	290
	min.	106	20	10	12
Ammoniacal-N	max.	7,000	2,360	2,870	1,571
	median	405	600	555	445
	min.	18	71	33	66
Chloride	max.	11,950	28,000	13,000	2,880
	median	1,300	2,135	1,760	1,025
	min.	140	172	154	157

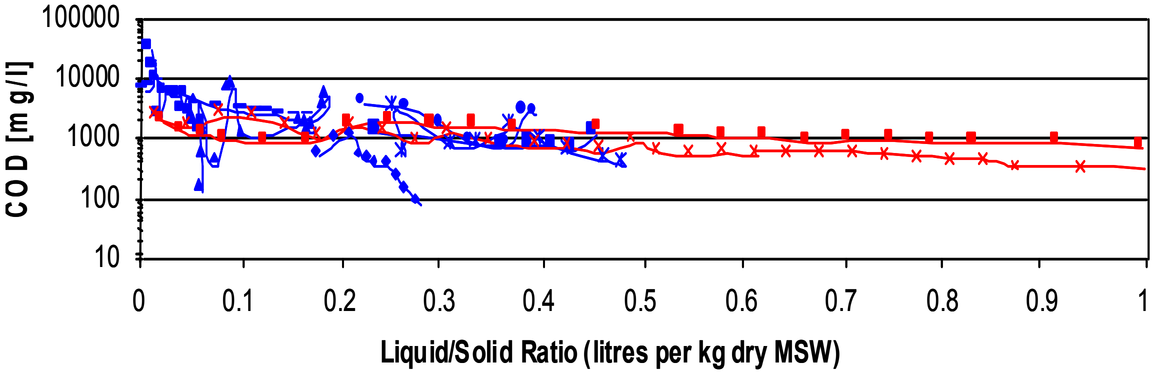


Figure 4.28 COD values in leachates from full-scale MSW landfills (in blue), and from MSW landfill simulation reactors (in red) at different L/S ratios (after Ehrig, personal communication, 2001)

This work has provided confidence that the cumulative L/S ratio can be used as a reasonably accurate timescale for leachate development under actual landfill conditions, if the rate of percolation or flow through the landfilled wastes is known, or can be estimated. Leachate percolation rates used in pilot-scale landfill reactor studies are typically up to 100 times as great as those experienced in full-scale landfill sites. Consequently, although data may be available for leachate quality at landfills containing untreated MSW only up to L/S Ratios of about 0.5, reliable data for leachate amounts and quality at landfills containing MBP wastes are far more limited.

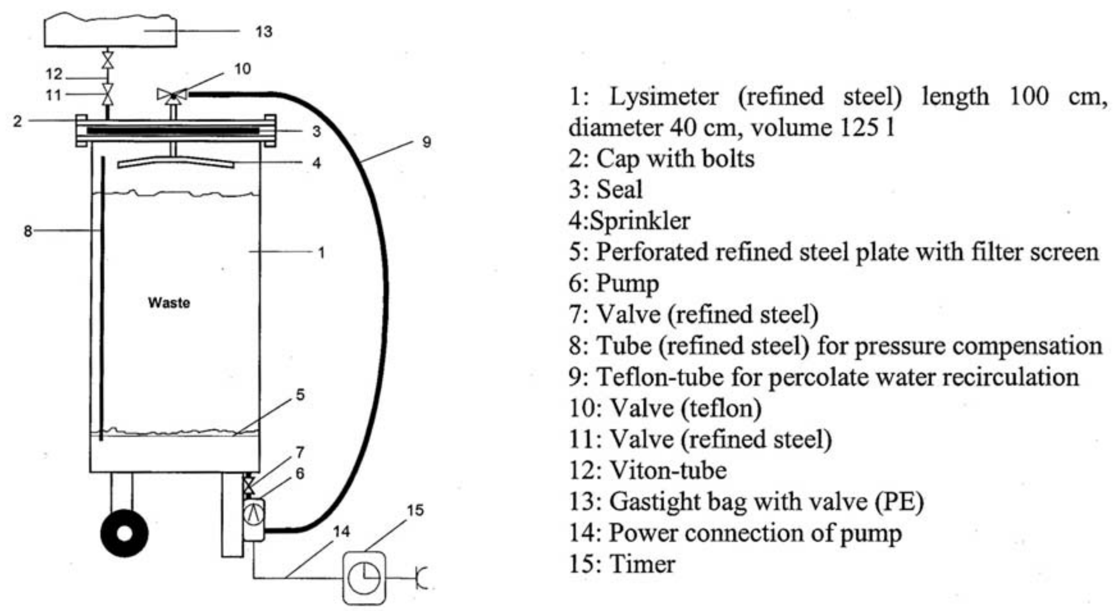
Additionally, it is widely acknowledged in Germany that MBP wastes can be compacted to very high densities and low hydraulic conductivity (5×10^{-9} to 1×10^{-10} m/s) (e.g. Höring *et al.*, 1999). Thus, physical properties of the MBP material will further constrain flushing rates at full-scale landfills, and retard the leaching of contaminants. This makes it extremely difficult to use full-scale landfill data to predict the timescales involved, before concentrations of contaminants in leachate will reach acceptable qualities for direct discharge.

This time period, as in landfills containing untreated MSW, will primarily be determined by concentrations of nitrogen in leachate, which can be determined from landfill simulation studies at the University of Wuppertal (see Figure 4.29). In contrast to carbon emissions, which primarily take place via landfill gas, nitrogen transfer depends on the far slower processes of leaching and transport in leachates.

Therefore, in order to advance understanding of the benefits of MBP of household wastes prior to landfilling, in terms of timescales for leaching of contaminants, the University of Wuppertal has co-ordinated many studies that have used similar landfill simulation reactors to provide extensive data. These pilot-scale data have then been translated into timescales for full-scale landfills, using understanding as described above.

The arrangement and use of landfill simulation reactors for these studies remains the same as has been widely used in Germany since first described by Stegmann in 1981, with uncompacted wastes, leachate recycling, and a controlled temperature (Stegmann, 1981; Heyer and Stegmann, 1995).

The MSW fractions, or MBP wastes (40 kg of dry solid matter) are incubated at 30ºC in the reactor. Water is added until field capacity is reached, and a further 3 litres is added for recirculation. Water is recirculated several times per day, and every week one litre of leachate is replaced by one litre of fresh water. This corresponds to a rate of 1.7 litres of water per kg dry MSW per year, whereas in full-scale landfills typical rates are 0.013-0.038 litres water per kg dry MSW per year, or between 45 and 130 times slower (Brinkmann *et al.*, 1995).



1: Lysimeter (refined steel) length 100 cm, diameter 40 cm, volume 125 l
2: Cap with bolts
3: Seal
4:Sprinkler
5: Perforated refined steel plate with filter screen
6: Pump
7: Valve (refined steel)
8: Tube (refined steel) for pressure compensation
9: Teflon-tube for percolate water recirculation
10: Valve (teflon)
11: Valve (refined steel)
12: Viton-tube
13: Gastight bag with valve (PE)
14: Power connection of pump
15: Timer

Figure 4.29 Design of a landfill simulation reactor, as used at the University of Wuppertal (Horing *et al.*, 1999)

It is now recognised throughout Germany that loss on ignition and TOC cannot describe the emission behaviour of either landfilled MSW or MBP wastes. These parameters take account of non-biodegradable organic waste components such as plastics, lignin, leather etc., which make an insignificant contribution to landfill emissions. The value of landfill simulation reactors in this area has also been demonstrated.

Table 4.10 below shows simulation reactor results demonstrating that state-of-the-art MBP systems are able to reduce the landfill gas generation potential of MSW by about 90-95%. When compared with untreated MSW, stabilised waste decomposes very slowly, because of the amount of readily-degradable organic matter that has been degraded during the pre-treatment process. Consequently, because no acid phase of decomposition occurs when MBP wastes are landfilled, the organic carbon potentially released via leachate can also be reduced by up to 95%.

Table 4.10 Typical ranges of emission potential for organic carbon, nitrogen and chloride, by gas and leachate, in MSW and MBP wastes (minimum values represent degree of stabilisation presently achievable by state of the art pre-treatment processes)

Emission Potential		**Unit**	**Untreated MSW**	**Mechanical-biological pretreated MSW**
carbon transfer by gas		[litres/kg]	134-233	12-50
		[g C_{org}/kg]	71.7-124.7	6.4-26.8
transfer by leachate	**TOC**	[g/kg]	8-16	0.3-3.3
	TN	[g/kg]	4-6	0.6-2.4
	Cl^-	[g/kg]	4-5	4-6
Notes: All results expressed per dry kg of waste material				

Landfill simulation reactor studies have consequently demonstrated the low gas formation activity of landfilled MBP MSW. These studies extend over 600 days and represent L/S ratios of 2 to 3 (equivalent to landfill timescales well in excess of a century). Figure 4.30 demonstrates that reductions in gas formation of up to 90% can be achieved. However, the figure also clearly shows that in spite of the removal of this rapid initial gas generation phase which is found in untreated MSW, subsequent long-term rates of gas generation (and probably soluble contaminant release in leachate), remain similar for pretreated and untreated wastes.

In addition, emissions of nitrogen in leachate may also be decreased by means of efficient MBP, by up to about 80 or 90%, although the processes responsible for this are not well described or understood.

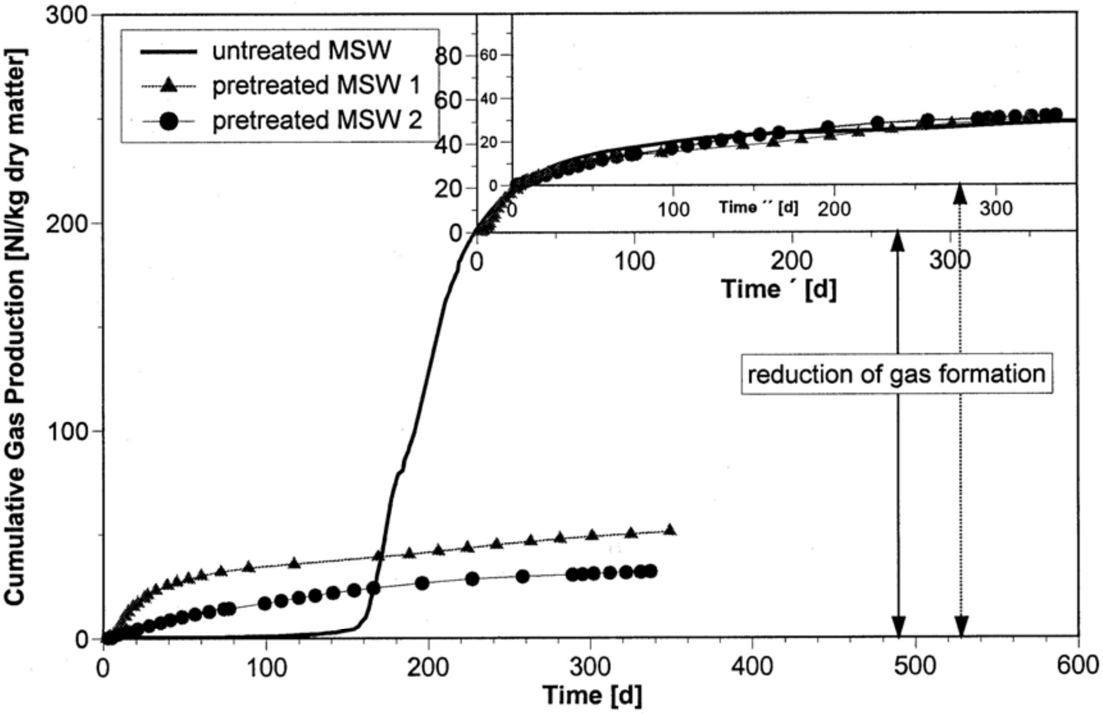


Figure 4.30 **Gas formation in landfill simulation reactors from:**

- **untreated MSW,**
- **pretreated MSW1 (16 weeks of windrow composting), and**
- **pretreated MSW2 (14 days of anaerobic pre-treatment, plus a post-treatment period of 12 weeks in-container composting)**

A key issue is whether MBP of MSW will reduce the timescale and extent of long-term, slow, release and leaching of contaminants, and generation of landfill gases. Many of the waste components which contribute to very long-term degradation processes and contaminant release, when landfilled in crude MSW (wood, fabrics etc.) may not be greatly affected by a composting process, and so may take part in the long-term decomposition processes of landfilled MBP wastes. Little information exists to assess this as yet, but it does appear likely that the long-term “tail” of contaminant release may remain for landfills receiving MBP. Estimates in landfill simulation reactors receiving MBP waste fractions, by Ehrig and co-workers, nonetheless indicate that at full-scale landfills, leachate treatment might still be required for periods between 150 and 200 years.

“Mechanical-biological pre-treatment of municipal solid waste before landfilling" project being co-ordinated at Wuppertal

A major research project “*Mechanical-biological pre-treatment of municipal solid waste before landfilling*”, promoted by the German Ministry for Eduction, Science, Research and Technology, has investigated the emissions and landfill behaviour of MBP wastes, by testing wastes extensively from many full-scale pre-treatment plants, in long-term landfill simulation reactors.

In total, 18 waste samples from different pre-treatment plants, which use different biological treatment procedures, have been investigated (aerobic treatment in windrows, containers and tunnels, anaerobic pre-treatment in a one-stop dry thermophilic process with post-composting). The substantial report of these studies (in German) comprises probably the most detailed such document available (Ehrig *et al*., 1998), and only a summary of the results can be included here.

Initial research (Brinkmann *et al.*, 1995)
Source material was obtained from a landfill site in Germany, where MSW was collected separately from the collection of glass, paper, metal and textiles. MSW as collected contained 10 percent “bulky waste”, which was passed through a hammer mill. The MSW was then passed through a sieve, with an RDF fraction (>150mm) removed, to provide a basic residual MSW fraction collected for use in each of the trials.

Four separate MSW fractions were generated and tested, as follows:

- A-O = original MSW as collected above
- A-M = material A-O above, milled by a hammer mill with magnetic separation of ferrous metals, and separation of a second RDF fraction (>60mm), to leave a mechanically pretreated MSW residue
- A-MB1 = material A-M above (<60mm), submitted to a 35-day intensive composting process
- A-MB2 = material A-M above (<60mm), composted for an overall period of 6 months.

Table 4.11 below presents values for weight loss on ignition (%), and for BOD of the solid material in a 10-day test.

Table 4.11 Loss on ignition (%) and BOD (mg O_2/g dry solids material, and per g organic dry solid material, in a 10-day test)

MSW-fraction	Loss on Ignition (%)	BOD_{10} (mgO_2/g ODSM)	BOD_{10} (mgO_2/g DSM)
A-O	60 (111)	nd	nd
A-M	54 (100)	143 (100)	103 (100)
A-MB1	<54 (<100)*	76 (53)	41 (40)
A-MB2	38 (70)	53 (37)	20 (20)

Notes:
(-) = relative values to source material for composting
* = inaccurate data

Concentrations of polychlorinated biphenyls (PCBs) and of polycyclic aromatic hydrocarbons (PAHs) were determined in the various MSW solid phases (see Figure 4.31 below). In view of the heterogeneity of the waste materials, the low concentrations determined, and the limited precision of these determinations, the observed variations are not considered to be significant. These compounds are sparingly volatile and soluble, only poorly available, and degradable to a limited extent by micro-organisms. There is no evidence that pre-treatment processes influenced them.

In addition, concentrations of BTEX (benzene, toluene, ethylbenzene and xylenes) were measured in leachates from the lysimeter tests (see Figure 4.32 below). In contrast to PCBs and PAHs, the BTEX compounds were efficiently eliminated during the pre-treatment processes, initially to some extent by volatilisation, but ultimately by aerobic degradation during the composting process. After thorough MBP processes had been completed, no BTEX compounds were detectable in lysimeter leachates.

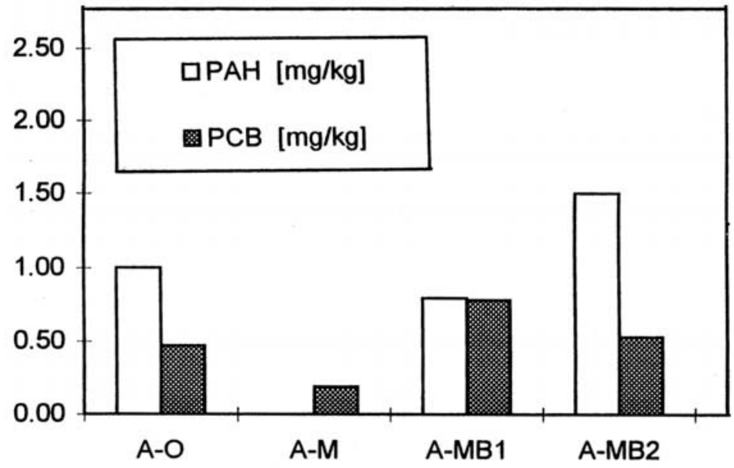


Figure 4.31 **Concentrations of PCBs and PAHs in different solid waste fractions tested (units as shown)**

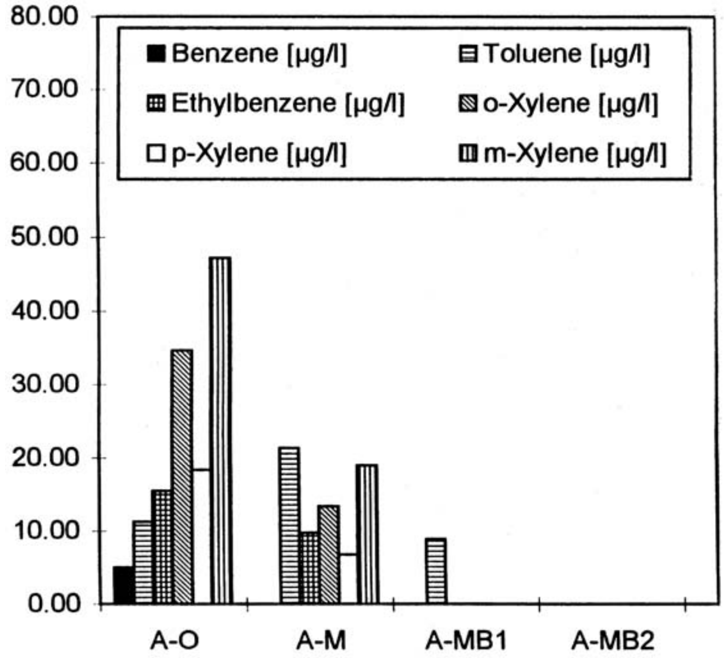


Figure 4.32 **Concentrations of BTEX compounds in lysimeter percolates from the different solid waste fractions tested (units as shown)**

Results from the incubation of the various waste fractions within the landfill simulation reactors are presented below. Each simulation lysimeter was operated for a period of up to 300 days, representing an ultimate L/S ratio of about 1.8 to 2.2, equivalent to about 45 to 150 years of leaching within a full-scale landfill site. Results from lysimeters A-O and A-M were very similar, so only data from A-M have been shown, in Figure 4.33.

Untreated MSW A-O, and mechanically pretreated MSW A-M, each demonstrated typical behaviour of residual MSW under anaerobic conditions. After an initial acid phase of just below 100 days, methanogenic gas production began and continued. Total amounts of gas generation were extrapolated, to total about 165 litres of gas per kg of dry solid material, and are typical of those for untreated MSW internationally. High concentrations of organic compounds (COD to >30,000 mg/l), and of ammoniacal-N (to >1500 mg/l), were observed, typical of many leachates from landfills containing crude MSW, or MSOR wastes. After this period, active methanogenic decomposition became established, and progressed in a typical manner. Concentration of ammoniacal-N remained greater than 600mg/l at the end of the trial period.

Results from the lysimeter trial that received pretreated material A-MB1, are presented in Figure 4.34 below. This material represented MSW A-M, which had then been subjected to 35 days of intensive, forced-air composting. In this trial no significant acetogenic phase was observed, and production of methane began almost immediately, although at only about 110 litres of gas per kg of dry material, total gas production was about two thirds of that for MSW A-M. COD values reduced much more rapidly, but concentrations of ammoniacal-N were high – exceeding 1000 mg/l for over 200 days, and remaining at 700 mg/l at the end of the trial.

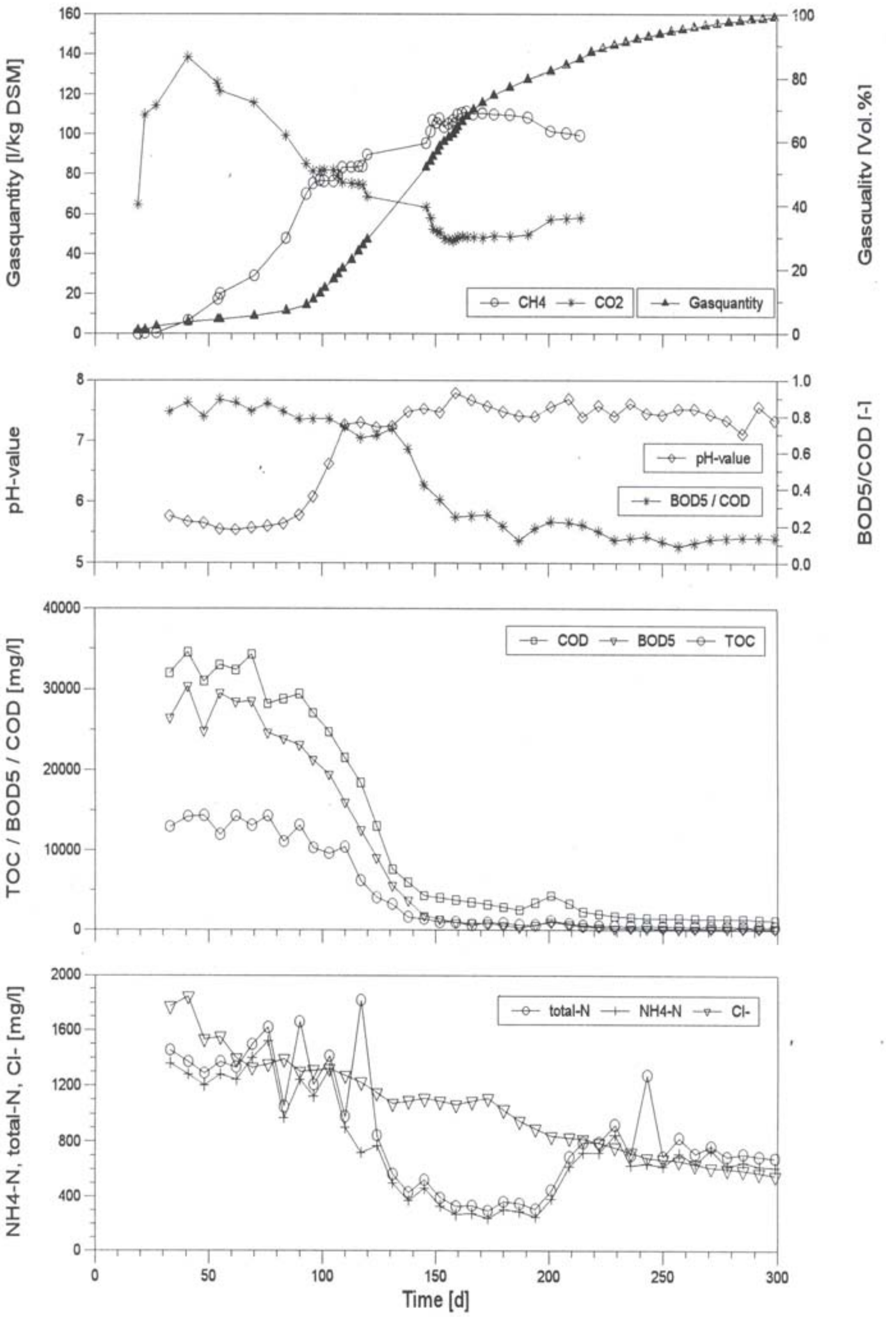


Figure 4.33 **Results from lysimeter studies on MSW A-M, containing residual wastes subjected <u>only to mechanical pre-treatment</u> (units as shown). Trial period represents an overall L/S ratio of about 1.4.**

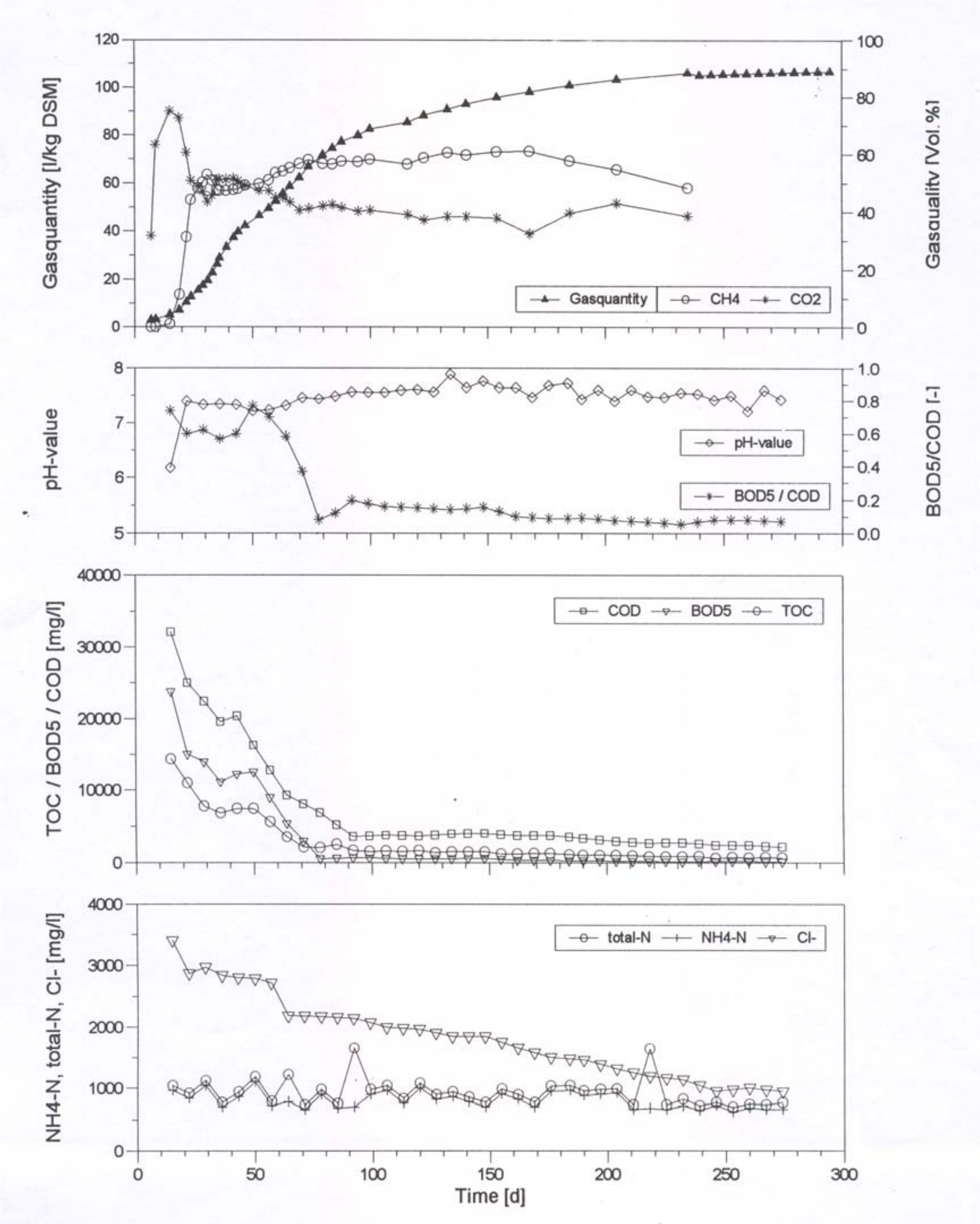


Figure 4.34 **Results from lysimeter studies on pretreated material A-MB1, containing residual wastes subjected to mechanical pre-treatment followed by 35 days of intensive composting (units as shown). Trial period represents an overall L/S ratio of about 1.4.**

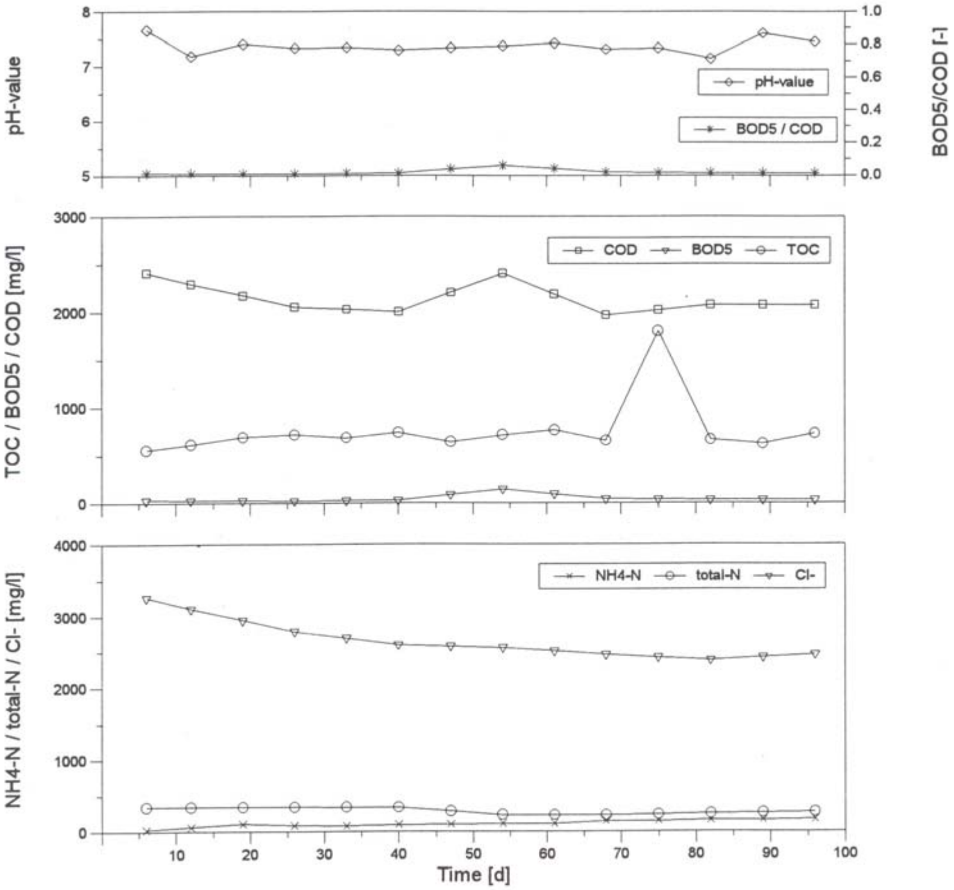


Figure 4.35 **Results from lysimeter studies on pretreated material A-MB2, containing residual wastes subjected to mechanical pre-treatment, 35 days of intensive composting, and to a further period of composting of 6 months (units as shown). Trial period represents an overall L/S ratio of about 0.46.**

Figure 4.35 presents results from the lysimeter containing pretreated material A-MB2, that had received as A-MB1, followed by a further 6 month period of composting. In this trial no significant quantities of gas were generated, although low concentrations of methane were detected within the lysimeter, showing that some methanogenic conditions were present.

BOD_5 values in leachate remained low (below 150 mg/l throughout, and 30 mg/l or less at the end of the trial), although COD values remained above 2000 mg/l throughout the trial (TOC typically 600-700 mg/l). Concentrations of ammoniacal-N rose steadily through the 96 days of this trial, ending it at just above 150 mg/l, and chloride fell from 3200 to about 2400 mg/l, purely by flushing.

In Figure 4.36, the mass transfer of contaminants into leachate, at different levels of MSW pre-treatment, is compared. This is plotted against L/S ratio. For organic contaminants, (represented by TOC), and for ammoniacal-N, pre-treatment reduced emissions, but only an extended composting period minimised these. Release of chloride was essentially unaffected.

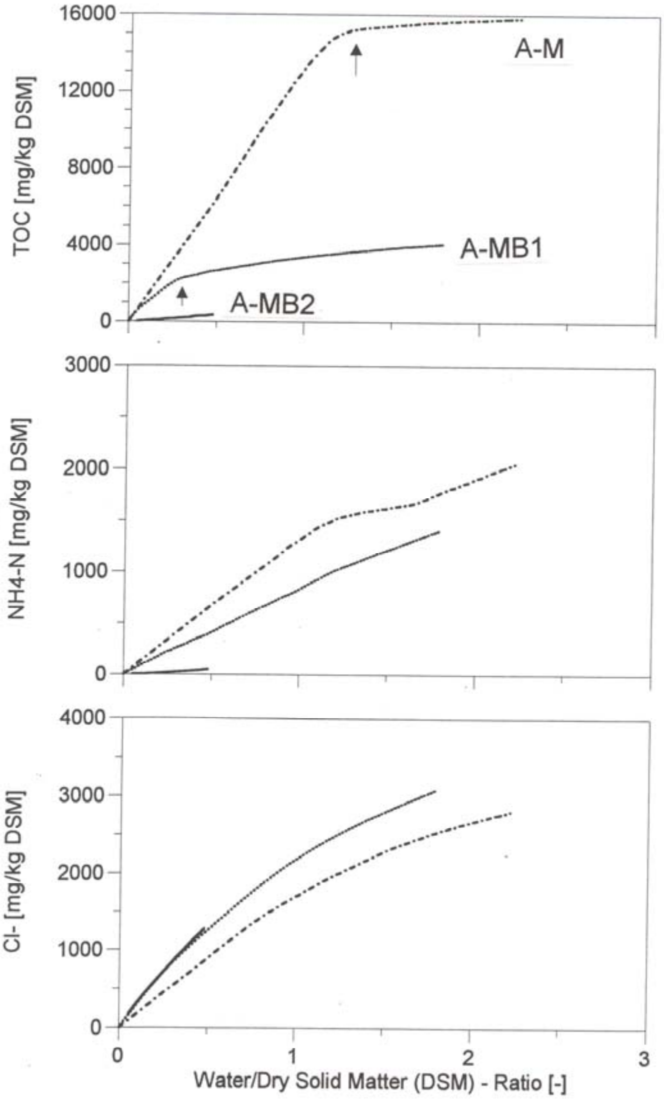


Figure 4.36 Mass transfer of contaminants into leachate, for different degrees of pre-treatment as discussed, plotted against cumulative L/S ratio. Arrows indicate the start of gas generation (if any)

The study demonstrated that a MBP process can be an efficient method of reducing emissions from landfilling of residual wastes, although mechanical pre-treatment by itself has no significant effect. It was also shown that the biological composting process must be performed adequately, if it is to be effective in changing the options for the landfilling of the end products. Ehrig (pers comm.) suggests that the aim of the pre-treatment process should be that no landfill gas will be formed, or that it might be formed in such small amounts that a gas collection system may be omitted from the landfill design.

Few composting systems currently achieve such standards, and none are likely to achieve the limiting values of 5 percent maximum loss on ignition, which must be observed for landfilling of wastes in Germany in future. However, the loss on ignition value provides no information of direct relevance regarding the behaviour of pretreated MSW fractions when landfilled. Since only degradable compounds will cause emissions during the landfill process, it is the view of Ehrig and many other German researchers, that biological test systems such as methane potential or respiratory activity will provide more valuable results. It is recognised that further study is necessary to refine such tests.

Further research (Ehrig et al., 1998)
A continuing programme of research coordinated between several research groups is presently underway. Only initial results have been published to date, and only in German (see Ehrig *et al*., 1998), but these are described in outline below.

The work is based on an extension of the standardised pilot scale lysimeter studies (as above) on residual wastes that have been subjected to a wide variety of biological pre-treatment processes, at a number of universities and research institutes. Test protocols are compared and checked by each institute testing a “standardised compost residue”, derived from an MBP plant at Horm, to provide “calibration” between researchers.

Basic data for the gas potential of the various composted materials is one outcome of the study, to indicate how the extent of composting undertaken reduces this, and Table 4.12 below has been translated to show the way in which such work is progressing. A 200-day period for each lysimeter trial and standardised leaching rates and experimental protocols, enable maximum benefit to be obtained from the work.

Table 4.12 Example of the type of data for gas production potential of composted residual wastes, being obtained by the co-ordinated German research programme, using standardised lysimeter tests (Ehrig *et al*., 1998)

Waste source (MBP plant)	Container composting (weeks)	Open compost heap		Anaerobic digestion (weeks)	Total treatment (weeks)	Total gas production(1) (l/kg dry mat.)
		Aerated (weeks)	Unaerated (weeks)			
Horm (std) (2)		9			9	87.9
Quarzbichel			0		0	212.7
			8		8	209.3
			24		24	145.6
		8			8	89.4
		16			16	37.3
		16	8		24	16.5
		16	16		32	13.4
		16	36		52	4.7
	8			2	10	23.6
	12			2	14	14.8
Lohfelden	8				8	9.0
	12				12	6.3
	12		78		90	2.0
Wilhelmshaven			20		20	89.4
/Meisenheim			44		44	12.5
			56		56	6.4
Scharfenberg		4	16		20	62.5

Notes:
(1) Total gas production in litres at stp, per kg dry solids, over a 200 day lysimeter test.
(2) Horm MBP waste is used as a “standard material” to enable the various research groups to compare the results that they obtain from MBP wastes derived from other sites.

Far more detailed information is also being derived from the coordinated research programme, regarding the potential of composted residues to release contaminants in leachates, and timescales for this to take place within the lysimeter tests. Results can be expressed in terms of timescales, but more usefully as a function of L/S ratio of leaching achieved.

Figures 4.37 and 4.38 provide typical examples of the type of data becoming available, for release of TOC and ammoniacal-N respectively.

As well as issues referred to earlier, it is hoped that the research programme will provide more information on the extent to which composting of residual wastes is able to reduce the timescales for contaminant release from landfills. At present, Ehrig and co-workers, and other researchers (e.g. Scharff *et al.*; 1995), have much evidence to demonstrate that the composting process has taken away the acetogenic phase of decomposition at many existing full-scale sites, substantially reducing the total contaminant emission from the landfilled wastes. However, after 10 years or so there may be little difference between the actual landfill situation with or without pre-treatment of the solid wastes.

Clearly, these data relate to composting processes that were being carried out ten or more years ago. It remains to be determined and proven, whether more advanced and intensive composting systems such as those presently being constructed and operated, can affect this situation. The German research programme is likely to provide the best understanding of this, by means of the co-ordinated and accelerated lysimeter trials.

Results from the research, shown in Figures 4.37 and 4.38, continue to indicate the extremely long timescales over which high residual concentrations of TOC (>200 mg/l), and of ammoniacal-N (>100 mg/l) persist in leachates, from even advanced, current composting processes.

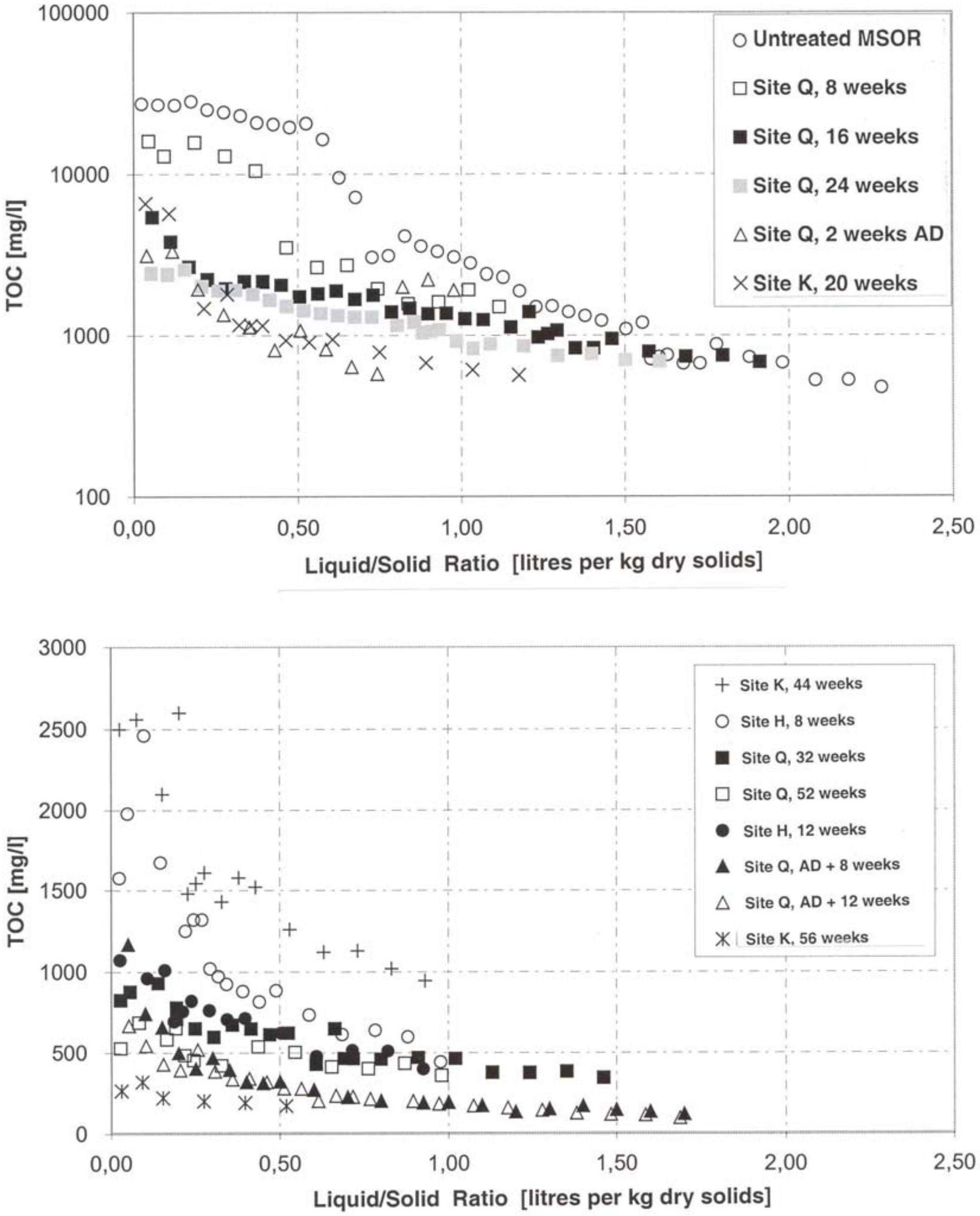


Figure 4.37 Examples of data from the coordinated German research programme, for long-term release of TOC from landfilled wastes that have been subjected to various types and intensity of biological pre-treatment

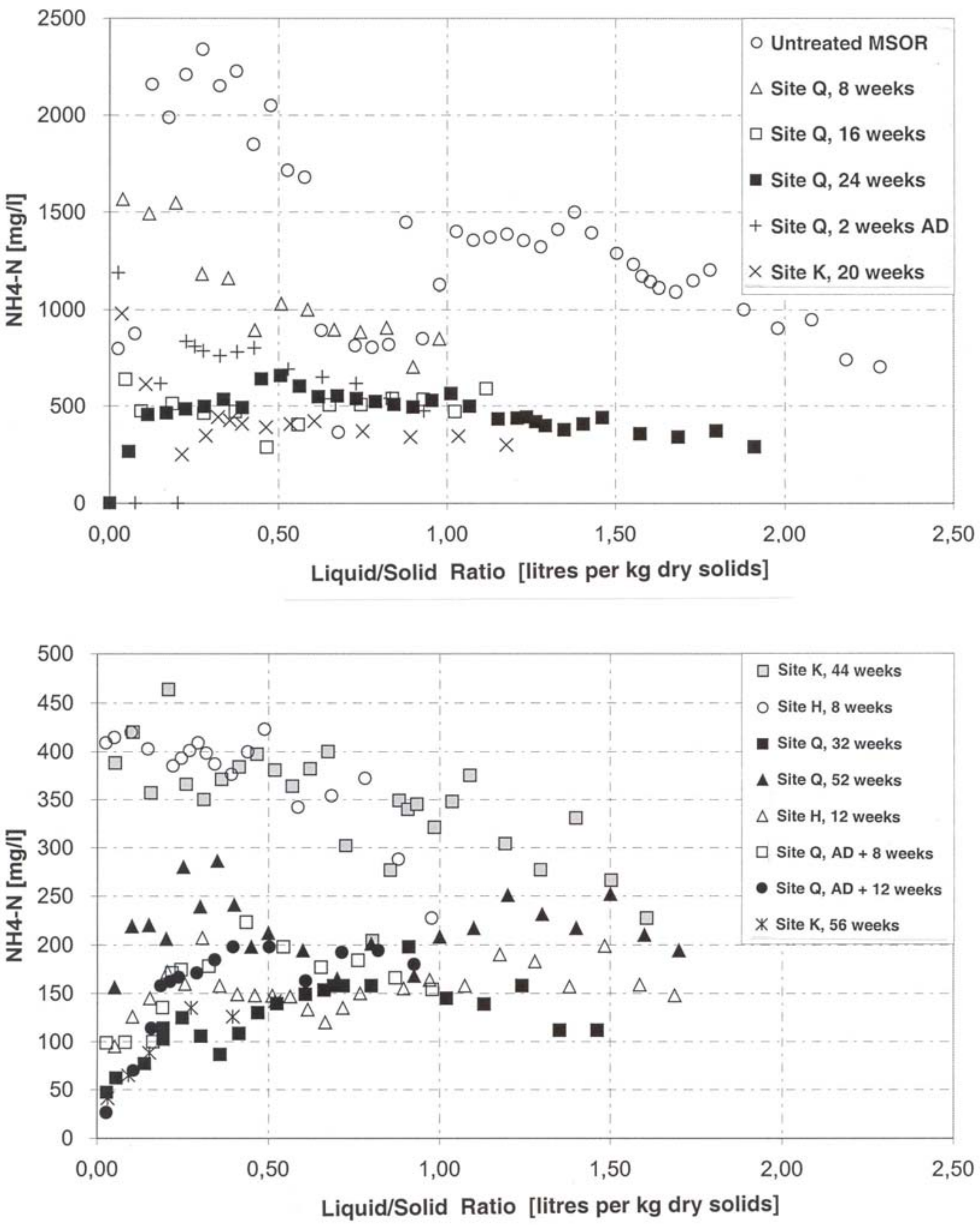


Figure 4.38 Examples of data from the coordinated German research programme, for long-term release of ammoniacal-N from landfilled wastes that have been subjected to various types and intensity of biological pre-treatment

5 Sampling of leachate at European MBP waste landfill sites

5.1 Background

In order to complement and extend published data obtained from EU landfill sites and experimental studies reported above, a second phase of practical work was commissioned by the Agency. This has included obtaining and testing samples of leachates from full-scale landfill sites across the EU, which have received large inputs of MBP wastes. A field sampling exercise was undertaken in two visits, and nine samples of leachate were obtained during February and July 2002. Results from this work are reported below.

The objectives of this phase of work were to provide data on trace constituents of leachates from the pretreated wastes, which could be compared with results for leachates from conventional untreated MSW landfills. The intention was to obtain leachate samples from each of the following categories of landfill site outlined in Table 5.1.

Table 5.1 Types of landfill from which leachate was sampled

	Age of wastes	
Type of waste inputs	**"Young"**	**"Old"**
Crude MSW	not needed	✓
Non-pretreated residual wastes (MSOR)	✓	✓
>50% MBP wastes	✓	✓
>90% MBP wastes	✓	✓

Sampling concentrated on landfills which had received one of the three categories of pretreated wastes, and aimed to obtain samples from both recently-deposited, and from older wastes. The terms "Young" and "Old" were difficult to define precisely, but were taken to represent wastes which had been landfilled for less than, or more than about 4 or 5 years. It was recognised that landfill drainage system arrangements might mean that leachates actually being sampled would represent composite samples from wastes of various ages, but best efforts were made to obtain the most valuable samples.

It was hoped that historic data for broad indicators of leachate quality (taken to be COD, BOD_5, ammoniacal-N, chloride, nitrate-N and pH-value) might also be available at the sites sampled, to assist in placing samples, and analytical results for trace substances, in a well-informed historic context.

Results already obtained and published for trace organic substances in leachates, as part of extensive sampling studies at UK MSW landfills (see Robinson and Knox, 2001; 2003), would provide baseline information against which data from the MBP landfills could be assessed.

5.2 Sampling and analysis

Nine samples of untreated leachate (labelled HR1 to HR10, excluding HR4) were obtained from seven EU landfill sites. Sample HR4 could not be obtained on the day of sampling, as the leachate drainage system at the site was flooded, and it was not possible to isolate leachate flow from the area of "young" MBP wastes (>50%).

In all instances samples of leachate were obtained from boreholes, sumps, or pipelines which were frequently and routinely pumped, to avoid sampling of stagnant leachates. No samples were filtered or specifically settled in any way before being taken, the intention being to represent, as fairly as possible, the overall quality of leachate that might realistically be discharged from such a site. In practice, no sample taken contained high levels of suspended solids.

At each site, various sub-samples were taken. Severn Trent Laboratories (STL) in Coventry determined a range of sanitary parameters and metals, on samples taken in two one-litre PET bottles. Although not refrigerated throughout transport to the UK, samples were kept cool and delivered to the laboratory between 36 and 48 hours after sampling.

For determination of trace compounds and elements, the UK Pollution Inventory List was used as a basis for an analytical suite to be determined by the laboratories of SAC Scientific (SAC) in Biggleswade, Bedfordshire. SAC had previously been used to undertake determination of Red List Substances, as part of a large sampling and review exercise into leachate quality at UK Landfill Sites, undertaken during the period 1990-1996 on behalf of the DoE/Agency (Robinson, 1996). They had also undertaken specialised analyses as part of a major study to look at Pollution Inventory discharges to sewer or surface waters from landfill leachates (Robinson and Knox, 2001).

Samples for trace analyses were taken in various prepared sample bottles, and despatched by courier to SAC, to arrive within 48 hours of sampling. Sub-samples were also taken for the Environment Agency laboratories, to assist in the development of a rapid screening methodology for trace organics in water.

Landfills selected were inevitably constrained by available sites, which could also be included within the two short and intense sampling visits. Nevertheless, a great deal was achieved, with help from many landfill operators and researchers from different countries. Sites used for this stage of work have not been identified, as part of agreements made with the operators.

5.2.1 Completeness of data

The categories of site from which leachate samples could be obtained are summarised below in Table 5.2. The table also shows the time period over which data for basic leachate chemistry was available from the site operator, at the location where a leachate sample was obtained.

The full sets of results obtained are presented at the end of this section in Tables 5.10 and 5.11. For each sample, results are separated into extensive sanitary analyses (by STL), and detailed trace analyses (by SAC). Summarised results are shown below in Table 5.3.

Results for Kjeldahl nitrogen were not always consistent, sometimes (especially at higher concentrations) being determined by STL at values less than those obtained for ammoniacal-N. Any calculation of a figure for “organic-N”, as the difference between the two values, is likely to be imprecise.

Table 5.2 Descriptions of the categories of landfill site from which leachate samples were obtained

Type of Waste Inputs	Young Wastes (<5 years)	Old Wastes (>5 years)	Period of historic data (1)
Crude MSW (sometimes + sludge)	not needed (2)	HR1	NA
		HR2 (+ sludge)	15 – 23 years
Non-pretreated Residual Wastes (sometimes + sludge)	no samples obtainable	HR3 (+ sludge)	14 – 27 years
>50% MBP Wastes	(sample HR4 not supplied) (3)	HR5	5 – 27 years
		HR6	10 – 18 years
>90% MBP Wastes	HR7		NA
	HR8		0 – 3 years
	HR9		NA
		HR10	NA

Notes:
(1) Extent of historic leachate quality data available at sampling location used. Not always covering a period to date. Years represent period since waste disposal began in that cell. NA = not available.
(2) Data available for crude MSW landfills, in Pollution Inventory studies, reported elsewhere (Robinson and Knox; 2001, 2003).
(3) Sample containers left with landfill operator to allow taking of a sample, but sample never received.

Table 5.3 Results for sanitary parameters and metals in leachates sampled during this study

Waste Inputs	**MSOR**	**Composted MSOR, various sources**					
Composting processes	**None**	**Passive windrows**	**Passive windrows**	**Turned windrows**	**Turned windrows**	**Container**	**Container + windrows**
Sample number	**HR3**	**HR6**	**HR10**	**HR5**	**HR9**	**HR8**	**HR7**
Period (weeks)							
Intensive	0	0	0	0	0	16	2
Secondary	0	12	30	25	8	0	30
COD	15590	582	4670	228	1620	869	1020
BOD_{20}	7840	>157	843	82	130	59	24
BOD_5	4240	46	202	3	35	6	3
TOC	4694	180	1480	78	543	308	340
fatty acids (as C)	707	<20	<10	<20	<10	<10	<10
ammoniacal-N	4024	195	1130	286	197	34.2	1.8
oxidised-N	<1	10.3	<1	16.1	<1	7.3	5.0
phosphate (P)	8.2	1.1	12.4	0.3	2.8	0.4	0.3
sulphate (SO_4)	423	433	117	18	449	414	878
chloride	6000	612	2270	384	2290	901	1090
NH_4-N/cl ratio	***0.671***	***0.319***	***0.498***	***0.745***	***0.086***	***0.038***	***0.002***
conductivity	39400	4960	14000	3210	9540	4860	5900
alkalinity	1740	879	6120	1100	2010	1670	895
Ph-value	8.3	8.1	8.3	7.9	7.9	8.4	8.5
sodium	4080	509	1520	419	1250	622	789
magnesium	77	91	88	47	104	64	67
potassium	1310	328	728	211	777	393	387
calcium	27	122	176	84	329	232	255
chromium	13100	110	870	<50	<250	<250	<250
manganese	380	460	1380	320	2940	1450	1610
iron	4310	960	19500	1050	13900	2590	1310
nickel	<100	30	210	<10	<50	<50	<50
copper	325	22	374	6	89	55	152
zinc	174	115	1032	<5	232	225	705
cadmium	<60	<6	<30	<6	<30	<30	<30
lead	<500	<50	<250	<150	<250	<250	<250
arsenic	<50	18	61	<10	<10	<1	<10
mercury	<1	<1	<1	<1	<1	<10	<1
Mecoprop, mcpp	**120**	**13**	**7.9**	**<1**	**0.47**	**<1**	**<1**

Notes:

- Heavy metals, mecoprop and iron in µg/l; Alkalinity expressed as mg/l of $CaCO_3$
- All other results in mg/l, except pH-value and conductivity (µS/cm), and where shown
- Alkalinity expressed as mg/l of $CaCO_3$

5.2.2 Landfills containing untreated MSW

Young wastes
No attempts were made to obtain leachate samples from MSW landfills where wastes had been emplaced recently, since extensive recent studies by the Agency (Robinson and Knox; 2001; 2003) provide detailed characterisation of trace organic substances in leachates from such landfills in the UK.

Older wastes
Sample HR1 was obtained from an area of a landfill in continental Europe, which received untreated MSW during the period 1974 – 1980, tipped to a depth of 20-25m. This sample (see Table 5.4 below) was used to benchmark old untreated MSW leachate data against similar landfill leachates in the UK, but collected from a country where composting is now well-established (and therefore more directly comparable with the old MBP leachates collected - samples HR3 & HR5)

Table 5.4 Leachate quality (key parameters) of sample HR1

Determinand:	Leachate sample taken on 5.2.02
COD	1099
BOD_5	111
Ammoniacal-N	495
nitrate-N	<0.5
Chloride	1400
pH-value (unitless)	7.5

(all units mg/l unless otherwise stated)

Sample HR2 was obtained from a landfill site which received a mixture of untreated MSW and sewage sludges between about 1978-1991, tipped to a depth of about 20 m, and providing a good comparison with the site above, where no sludge was disposed. Extensive leachate quality data exist for the period June 1993 to October 2001 (83 samples), and these are presented in Figure 5.1 below. Recent results are very similar in composition to sample HR2 (see Table 5.5 below), giving confidence that this is representative.

Table 5.5 Leachate quality (key parameters) of sample HR2

Determinand:	Leachate sample taken on 6.2.2002	Mean of values during 2001 (2 samples)
COD	4406	3623
BOD_5	207	136
Ammoniacal-N	2790	1992
nitrate-N	<0.5	<0.5
Chloride	7400	4668
pH-value (unitless)	8.0	7.95

(all units mg/l unless otherwise stated)

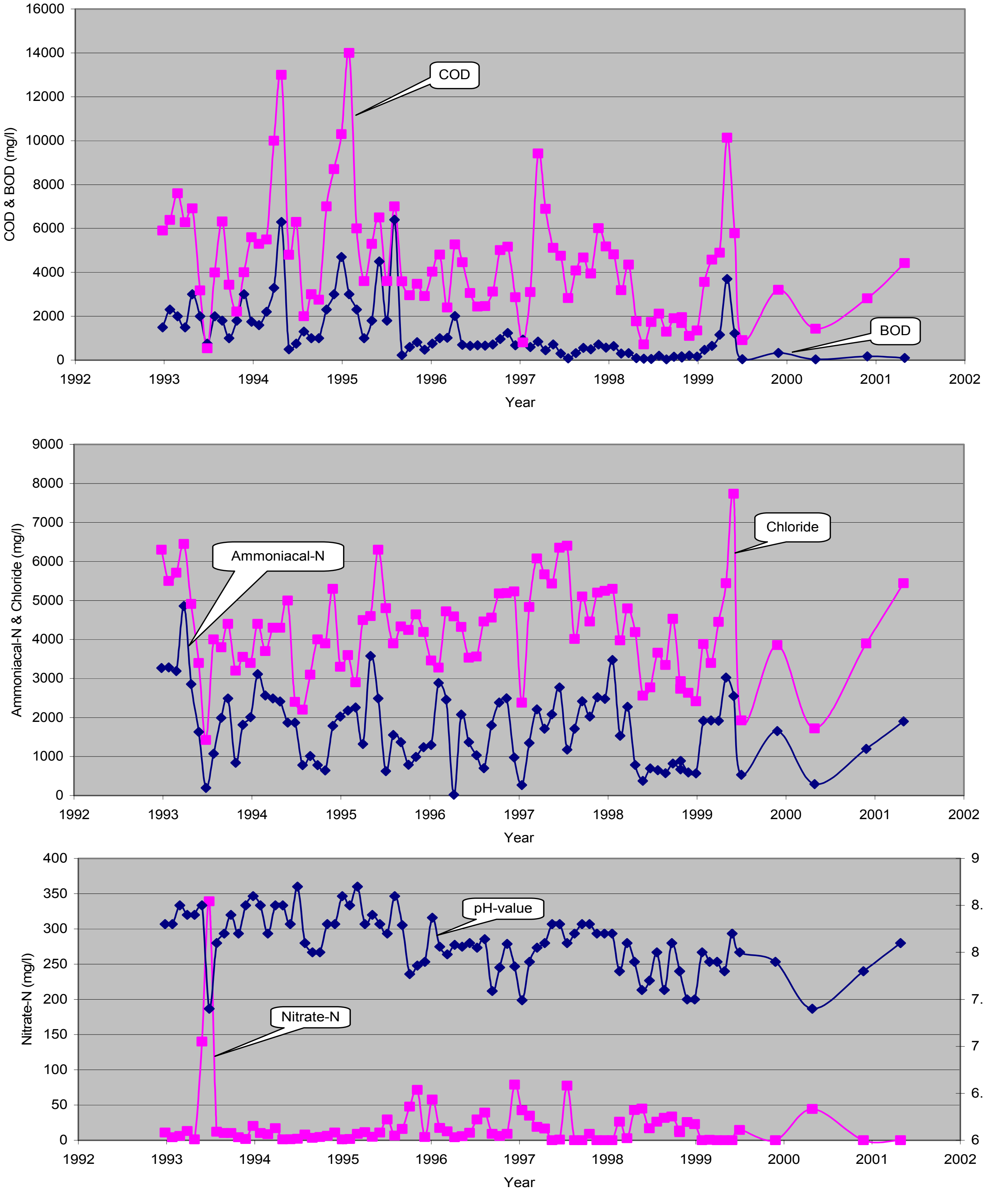


Figure 5.1 Historic leachate quality at HR2 sampling location

Sample HR1 was entirely typical of many leachates from aged MSW landfill sites. A COD value of 1100 mg/l, of which about 10 percent represented BOD_5, and about 500 mg/l of ammoniacal-N, indicate a site where methanogenic processes remain active. Chloride concentration (1400 mg/l) indicates relatively low dilution of leachate. The only significant trace substances are mecoprop at 38 µg/l (found in 98% of UK MSW landfills, mean value 22 µg/l, see Robinson and Knox, 2001), and xylenols at 17 µg/l (detected in 20% of UK landfill leachates, see Robinson, 1996).

Sample HR2, from a very similar landfill, except waste inputs included significant quantities of sewage sludge, was of a very different leachate. At about 4000 mg/l, COD values were higher, even if a BOD_5 result of 200 mg/l indicated relatively low biodegradability of the organic materials. The main difference was a concentration of nearly 2800 mg/l of ammoniacal-N; typical of results over the previous decade, and indicative of much higher organic content, and fermentation activity, within the degrading wastes. The influence of the sewage sludge is also apparent in high concentrations of chloride (7400 mg/l), conductivity (37000 µS/cm), sodium and potassium. Mecoprop (47 µg/l), traces of atrazine (2.1 µg/l) and phenol and cresols (~40 µg/l in total) are not especially significant. However, the leachate contained high concentrations of chromium (2320 µg/l), and relatively high concentrations of nickel (180 µg/l) – more likely to be the result of the co-disposed sewage sludge. A high AOX value of 3416 µg/l was measured.

5.2.3 Landfills containing non-pretreated residual wastes (MSOR)

Young wastes
It was not possible to identify sites, and to obtain leachate samples, from newly-developed sites where untreated residual wastes had recently been landfilled. This is primarily because such sites are now rare, and in recent years the highly organic residues from MRFs have been widely treated in one way or another, before landfill disposal. Nevertheless, good historic data for general leachate chemistry could be obtained from two older sites in this category, discussed below.

In view of the very high organic content of residual wastes and the well-documented high strength of leachates it seems likely that few such sites will be developed in the future.

Older wastes
A good leachate sample, and good historic leachate quality data, were obtained from a landfill that during the 1970s and 1980s began to receive substantial inputs of non-pretreated residual wastes, together with significant inputs of sewage sludge.

Leachate sample HR3 was obtained from a large landfill site, which received about 70,000 tonnes of residual wastes (not biologically pretreated) each year during the period 1979-1991. The site also received large quantities of commercial wastes, plus sewage sludge with 30% dry solids, and also primary sewage screenings, as well as some bulky waste inputs.

A leachate sample HR3, was obtained from an area of the site which received a mixture of non-pretreated residual wastes (from a municipal area having separate biowaste and paper collection at source, these wastes then passing a 100 mm sieve), and of sewage sludges, during the period 1979-1991. Extensive leachate quality data exist between June 1993 and October 2001 (a total of 80 samples), and recent data compare well with the composition of sample HR3 (see Table 5.6 below).

Table 5.6 Leachate quality (key parameters) of sample HR3

Determinand	Mean of values during 2001 (2 samples)	Leachate sample taken on 6.2.2002 HR3
COD	12802	15590
BOD_5	2422	4240
Ammoniacal-N	3611	4025
Nitrate-N	<0.5	<0.5
Chloride	5296	6000
pH-value	8.1	8.3

Figure 5.2 presents extensive historic leachate chemistry data for the location where sample HR3 was taken, over the previous decade. Quality closely mirrors that at HR2 (discussed above) where the landfill had also received inputs of sewage sludge, albeit together with crude household wastes. In HR3, COD values remained even higher, in excess of 15,000 mg/l, even after more than a decade since waste inputs ended, with more than 25 percent of the COD represented by the BOD_5 result, and more than 50 percent by the BOD_{20} value. Substantial quantities of volatile fatty acids (>700 mg/l as C) remained in the leachate.

Concentrations of chloride, at 6000 mg/l, typical of historic data, are similar to the HR2 data, but values for ammoniacal-N in excess of 4000 mg/l are even higher. A very high concentration of chromium (13100 µg/l), and a raised level of copper (325 µg/l), may well have resulted from the co-disposed sewage sludge.

As in HR2, mecoprop (120 µg/l) and various phenols were detected, and a high AOX value of 6064 µg/l was measured. No other trace organic compounds were detected.

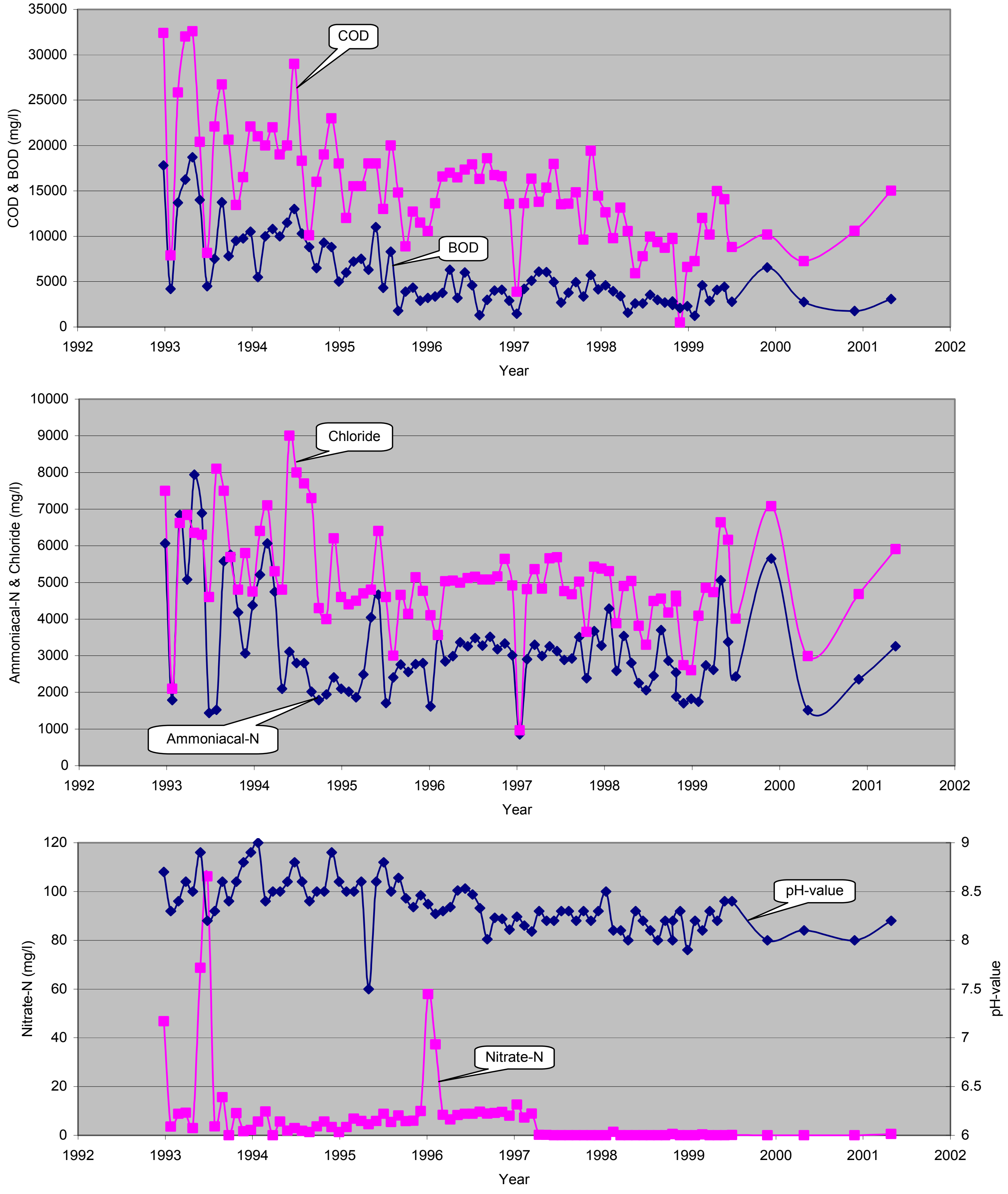


Figure 5.2 **Historic leachate quality at HR3 sampling location**

5.2.4 Landfills receiving >50 percent MBP waste inputs

Young wastes
It was not possible to obtain a sample of leachate from recently-emplaced wastes comprising >50 percent (but less than 90 percent) MBP wastes.

This omission is not a serious problem, because:

(a) good historic data were obtained from leachate quality at two older sites within this category (see below); and

(b) excellent data were obtained from four sites (two young, two old) that have received >90 percent MBP waste inputs.

Older wastes
Good samples, and good historic data for leachate quality, were obtained for two landfills in this category. Leachate sample HR5 was obtained from a medium-sized landfill, which has received about 20,000 t of waste each year since 1975. The site covers an area of just over 10 hectares, and was tipped in four phases, each lined and incorporate a leachate drainage system. Before 1995, only untreated residual wastes were being deposited, but subsequently composted waste has been landfilled, and has comprised 80% of the incoming waste streams. Other inputs include 10 percent commercial wastes, 5% bulky wastes, and 5% demolition wastes. Total depth of wastes is between 18 and 23 metres.

Incoming pulverised residual MSW is wetted with landfill leachate (no sewage sludge enters the site), to be placed in windrows and naturally aerated for about 6 months. During this period the wastes are turned about 2 or 3 times, and then disposed of without further treatment. Leachate quality data exist for the period 1980-present, and leachate was sampled as part of this project on 5 February 2002, as sample HR5.

The leachate sample obtained as HR5 was closely representative of previous monitoring results from the site (see Table 5.7 below).

Table 5.7 Leachate quality (key parameters) of sample HR5

Determinand	Leachate sample taken on 5.2.2002 HR5	Mean of values from 1999-2001 (6 samples)
COD	228	399
BOD_5	3	115
ammoniacal-N	286	268
nitrate-N	15.1	8.8
chloride	384	540
pH-value	7.9	7.88

Historic leachate quality data, over a period of more than twenty years, at the location where HR5 was sampled, are presented below in Figure 5.3.

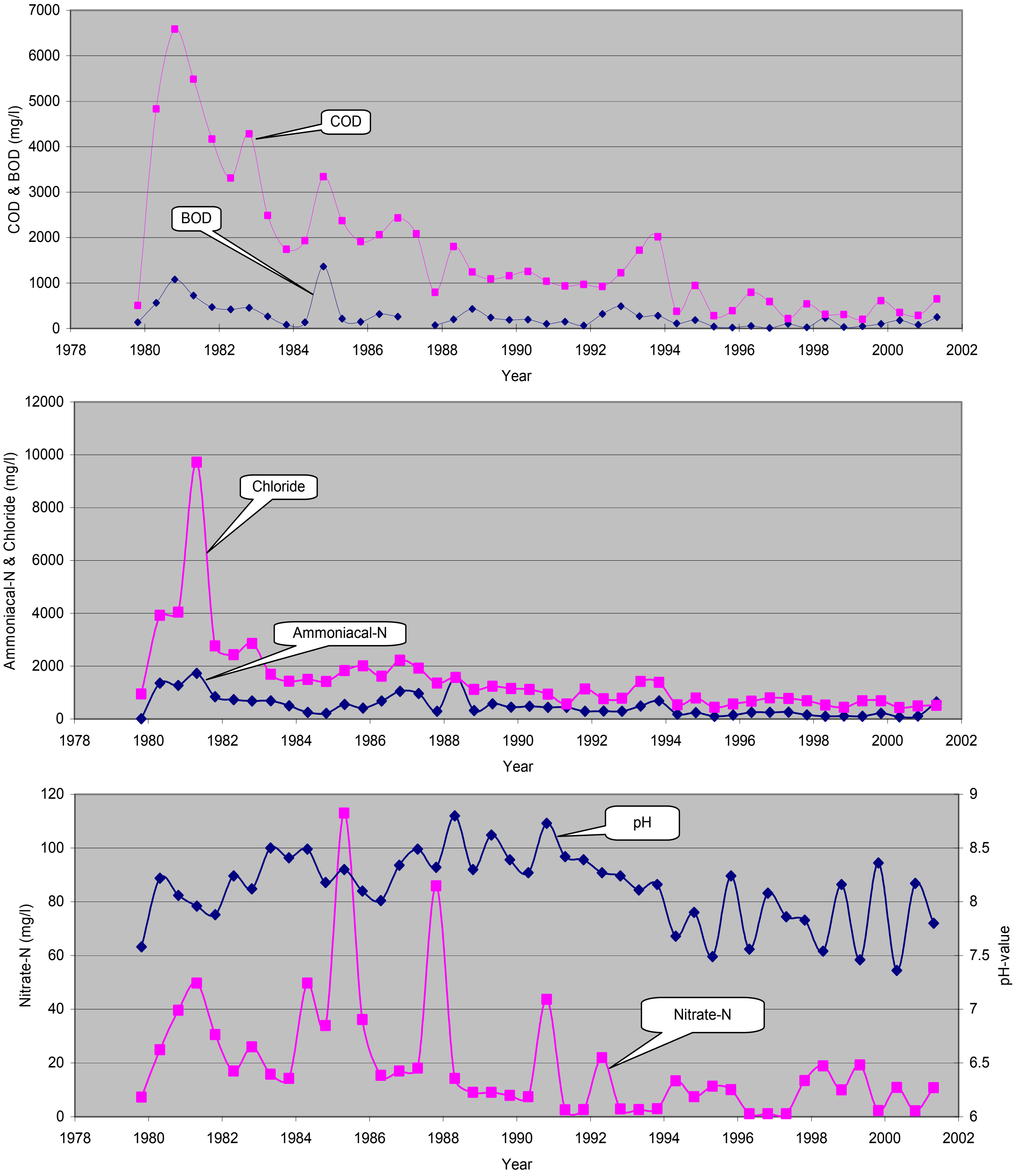


Figure 5.3 Historic leachate quality at HR5 sampling location

Leachate sample HR6 was obtained from a similar-sized landfill, which received inputs of untreated MSW from 1974-1980, before composting of incoming pulverised residual MSW began. No sewage sludge additions are made to the composting process, the whole incoming MSW spending 36 hours in a Dano drum to be pulverised, with the fraction >10mm going directly to the landfill. The fraction smaller than 10mm is composted in unturned windrows for several months. Some of this treated material is used as compost in agriculture, and the rest is landfilled – representing about half of the 20,000 t of wastes disposed of each year at the site.

Leachate is pumped from the site, for treatment in combination with domestic wastewater at a nearby sewage treatment works, and leachate quality data for flows from the entire site are available from early 1990 to early 1998. The leachate sample obtained as HR6 on 5 February 2002 was closely representative of previous monitoring results from the site (see Table 5.8 below).

Table 5.8 Leachate quality (key parameters) of sample HR6

Determinand	Leachate sample taken on 5.2.2002 HR6	Mean of values from 1996-1998 (4 samples)
COD	582	642
BOD_5	46	51
ammoniacal-N	195	144
nitrate-N	2.0	24.0
chloride	612	696
pH-value	8.1	8.15

Figure 5.4 presents historic leachate quality data for this location for the period 1991-1998 (10-18 years since filling of composted MSW began).

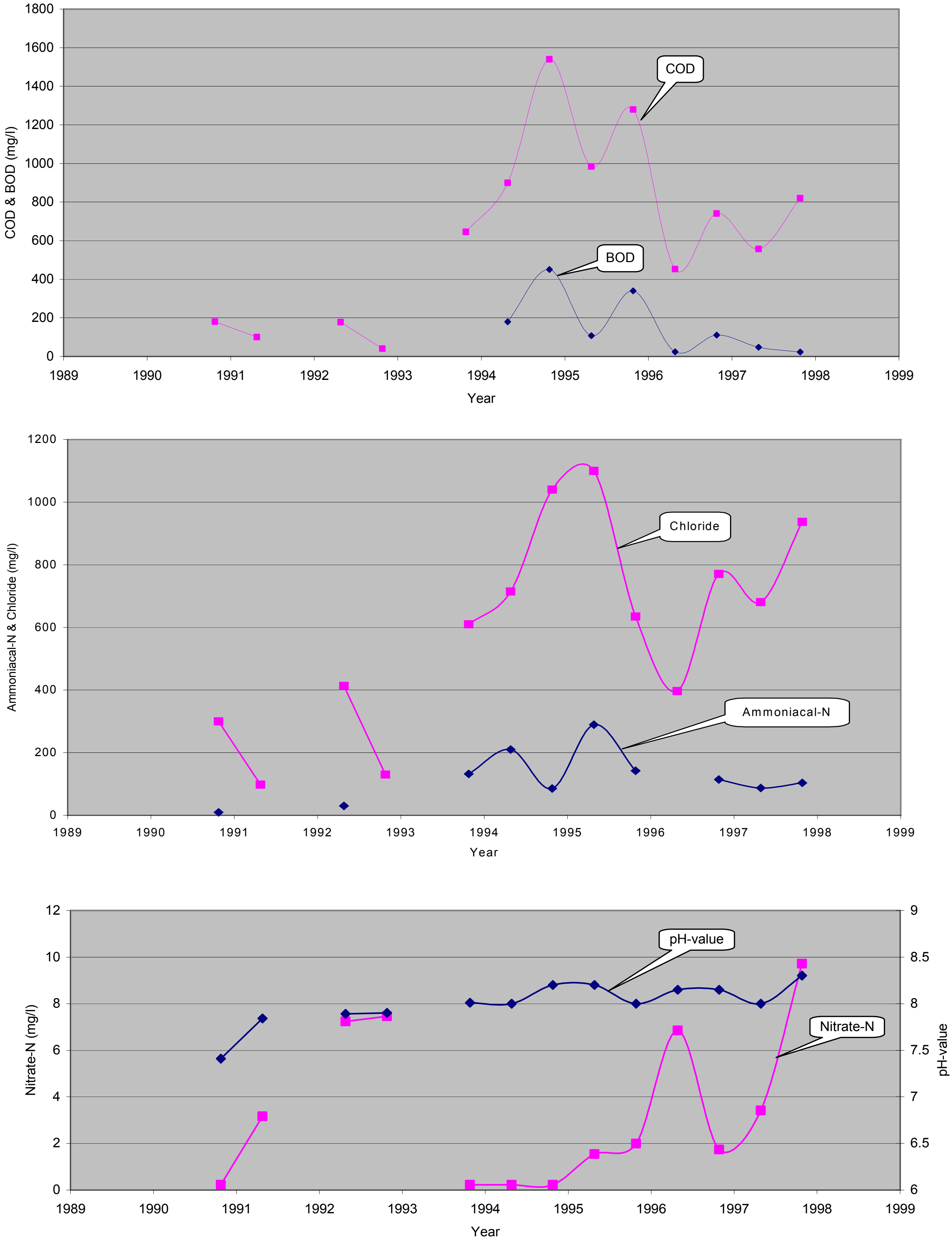


Figure 5.4 **Historic leachate quality at HR6 sampling location**

The two leachate samples HR5 and HR6 are very similar in composition, both from historic data and from samples taken as part of this project. They both represent leachates in essentially later methanogenic stages of decomposition, although unusually for such a sample, more than 400 mg/l of sulphate was present in HR6. In each sample, COD values are relatively stable, at 228 and 582 mg/l in HR5 and HR6 respectively. Concentrations of ammoniacal-N are typically 100 to 300 mg/l at each location.

In the historic data for location HR5, which only begin in 1980, 5 years after waste disposal began at the site, an initial more-acetogenic phase of leachate quality is evident during early years, with COD values to more than 5000 mg/l, and ammoniacal-N in excess of 1000 mg/l. However, following the onset of composting of residual MSW at the site, leachate quality rapidly changed to become more typical of the sample HR6.

At the second site, where incoming residual MSW has been composted for more than 20 years, leachate quality has been typical of sample HR2 over a much longer period, in excess of 10 years.

AOX results in both leachates were relatively low (213 and 366 µg/l), and in sample HR5 the only trace organics to be detected, at very low concentrations (<0.5 µg/l), were some PAHs on the EPA 16 list. Mecoprop was not detected in HR5 (<1 µg/l), but was present at 13 µg/l in HR6. Trace levels of atrazine (1.1 µg/l) and of isoproturon (0.69 µg/l) were also measured, Concentrations of heavy metals were also higher (e.g. chromium 110 µg/l; zinc 115 µg/l). Both samples were far less contaminated than samples discussed earlier above.

5.2.5 Landfills receiving >90 percent MBP waste inputs

It was considered to be of highest importance to try and obtain reliable samples from sites within this category, representing landfills where inputs primarily comprise well-composted MBP wastes. Four locations were identified and samples obtained, although it has been possible to obtain only few historical leachate quality data from these sites. Three samples (HR7, HR8 and HR9) represent wastes emplaced within the last 4 or 5 years, and a further sample (HR10) was from older wastes.

Young wastes
Three samples of leachate were obtained from sites in this category. The first was from a large landfill that receives 100 percent MBP waste, treated in a state-of-the-art plant, at a rate of 55,000 t/a. Larger fractions (>120 mm) are processed to become RDF, with residual wastes treated by containerised and intensive composting for 14 days (with treatment of all off-gases). The composted product is then subjected to further composting in open windrows, for a period of 6-8 months, before being landfilled in lined cells.

Leachate sample HR7 was obtained on 16 July 2002, from a cell which had received wastes since 1999. No historic leachate quality data were available from this cell.

A second sample of leachate, HR8, was obtained from a large landfill cell at a state-of-the-art MBP plant, which began to receive inputs of MBP wastes during mid-1997. MBP of wastes comprises extraction of metals and bulky items, followed by milling and removal of materials by a 100 mm sieve, for conversion to an RDF fraction. The residual waste materials are subjected to 16 weeks of intensive composting, with active aeration in a sealed static pile system, incorporating advanced treatment of all off-gases in an air washer and bio filter. Water, but no sewage sludge, is added during the composting process, and the resulting compost is landfilled with high rates of compaction, although only to a shallow depth (<5 m).

Leachate sample HR8 was obtained from the main drainage system of the cell, on 17th July 2002. Detailed historic records for leachate quality from the cell exist, for the period from initial generation (soon after filling, in autumn 1997) to date, although it has only been possible to obtain these to the end of 1999. It is clear that the sample obtained during this study is representative of these results (see Table 5.9 below).

Table 5.9 Leachate quality (key parameters) of sample HR8 (in mg/l)

Year	Half	COD	Ammoniacal-N
1997	I	-	-
	II	500	25
1998	I	1200	60
	II	1400	35
1999	I	1500	35
	II	1200	25
HR8 (mid 2002)		869	34

Again, the two leachate samples, HR7 and HR8, were very similar in nature. Although each contained about 1000 mg/l of chloride and COD values were similar, BOD_5 results were as low as 3 and 6 mg/l, and even BOD_{20} values were only 24 and 59 mg/l respectively. This indicates relatively low rates of biodegradation within the landfilled wastes, which is confirmed by concentrations of ammoniacal-N of only 2 mg/l and 34 mg/l respectively, in spite of relatively recently emplaced MBP wastes.

AOX values are also relatively low (370 and 180 µg/l), and no significant trace organic compounds were detected in either leachate, although concentrations of zinc (705 and 225 µg/l), and copper (152 and 55 µg/l) were relatively high.

Mecoprop was not detected in either sample, possibly indicating that effective aerobic biological degradation is being achieved during the composting process at each site.

The third site is a large landfill test cell, which exclusively received MBP wastes from March 2000 to January 2002, at a rate of about 25,000 t/a, representing a total input approaching 50,000 t. Wastes arise primarily from local municipal collections, although a small proportion is of “domestic type” from businesses. The waste is nominally pre-sorted by the householder, to remove glass, paper, organics and packaging (although visual inspection on-site demonstrated that this was not necessarily happening consistently.

The MBP plant has operated since 1997, receiving an annual input of about 60,000 t of these pre-sorted wastes, which are initially reduced in size by milling and divided into different fractions. Materials greater than 80 mm go to be incinerated and small organic fractions (smaller than 40 mm) are digested anaerobically in a Dranco Fermenter. Stabilised residues from the digester are then mixed with the 40 to 80 mm fraction, and the mixture is composted aerobically for eight weeks in windrows, being turned weekly. This produces a relatively dry and stable end product which is landfilled, and amounts to about one third of the volume of incoming wastes to the MBP plant.

Sample HR9 was collected on 16 July 2002, and sent for analysis – unfortunately it was not possible to obtain any historic leachate quality data from this location.

Older Wastes

A single sample of leachate was obtained from a site containing older wastes in this category, although again, the availability of historic leachate quality data was restricted.

Sample HR10 was obtained from a large site where shredded residual wastes are composted in windrows on the top of the landfill, with passive aeration and no turning, for a period of 6-8 months. Wastes remain exposed and are kept moist by leachate recirculation, ultimately being spread out across the site, to be replaced by the next batch of waste to be treated. The waste inputs began in 1993, and leachate sample HR10 was obtained on 16th July 2002, from the main site drainage system.

It was clear that the intensity of composting being achieved at sites (from where leachate samples HR9 and HR10 were obtained) was not so high as at the previous two sites containing recently composted wastes (HR7 and HR8). In particular, leachate sample HR10 was derived from wastes that had been subjected to only basic outdoor windrow treatment, with only passive aeration and no turning – albeit for a relatively long period.

These differences in the extent of composting achieved were evident in the leachate quality results obtained. Sample HR9 indicated a reasonably efficient composting process, with a COD of 1620 mg/l and BOD_5 of 35 mg/l (BOD_{20} 130 mg/l), and a chloride concentration of 2290 mg/l demonstrated minimal dilution of the sample. However, a concentration of ammoniacal-N of just below 200 mg/l in HR9 was much higher than for the MBP wastes derived from more intensive composting (HR7 and HR8). An AOX result of 1500 µg/l and trace of mecoprop (0.47 µg/l) in HR9 confirmed this, although no other significant trace organic compounds were detected. Zinc and copper were present at 232 µg/l and 89 µg/l respectively.

Sample HR10 was clearly derived from MBP wastes which had been subjected to composting processes that were of relatively low intensity, and this was evident in the nature of the leachate. A COD value of 4670 mg/l (BOD_5 = 202 mg/l, BOD_{20} = 843 mg/l) and ammoniacal-N of 1130 mg/l demonstrated a bioreactive landfill, albeit in a methanogenic state. Chloride is relatively high and similar to HR9 at 2270 mg/l.

The AOX result was medium at 790 µg/l, and a concentration of mecoprop of 7.9 µg/l demonstrated that this herbicide had not been completely degraded during the aerobic composting process. Nevertheless, no other trace organic compounds were detected in leachate HR10.

Concentration of some heavy metals were elevated in HR10 including chromium at 870 µg/l, copper at 374 µg/l, zinc at 1032 µg/l and nickel at 210 µg/l, compared with the more intensively composted residual wastes above.

5.3 Findings from the MBP leachate sampling exercise

The sampling of leachates from EU landfills that have received inputs of MBP wastes, has generated valuable data. In the case of data for trace substances in such leachates, results are among the first to be published.

In terms of basic leachate chemistry, data obtained strongly confirm those that have been compiled from work by other researchers earlier in this report. Comparison of leachate data obtained, with historic and existing data for sites from which they were obtained, demonstrates that they are certainly representative of results from those sites.

Data confirm that landfilling of untreated MSOR wastes, which generally comprise the most putrescible fractions of household wastes, has potential to generate very strong leachates, within very bioreactive landfill sites. Concentrations of ammoniacal-N in the range 4000-6000 mg/l are typical in leachates from such sites, and were sampled in this study. In many ways, the co-disposal of sewage sludges will produce similar leachates, although evidence from this study demonstrated that the sewage sludge could lead to high concentrations of some heavy metals in leachates. In particular, chromium (to 13,100 µg/l), nickel (to 180 µg/l) and copper (to 325 µg/l) would need to be considered particularly when assessing leachate treatment options.

Leachate data demonstrates that when effective state-of-the-art composting processes are undertaken on residual wastes, the landfilling of such products can produce a substantial reduction in bioreactivity, and consequently the generation of more stable and weaker leachates. BOD_5 values can be as low as <10 mg/l, even in leachates from wastes deposited only a few months or years ago. Concentrations of ammoniacal-N can be in the order of those found in domestic sewage (<40 mg/l), and even <5 mg/l at very successful composting plants.

Nevertheless, significant COD values (to 1000 mg/l or above) were found in leachates from even the best MBP composting processes examined and sampled. It remains to be determined over how many years, or decades, such leachates would continue to arise from landfills containing MBP wastes that have been subjected to state-of-the-art composting processes.

In spite of the extensive suites of trace organic substances that have been determined in leachate samples, relatively few were detected at any significant levels in any leachate. Mecoprop (MCPP), an acid herbicide which is relatively degradable by aerobic biological processes, but resistant to equivalent anaerobic processes, has been found to be present extensively in leachates from MSW landfills in the UK (see Robinson and Knox, 2001). It was also commonly found in leachates from equivalent EU landfills sampled as part of this study.

Of particular interest, however, was the fact that that mecoprop was not detected in leachates at landfills where MBP wastes had been subjected to efficient composting processes. Indeed, the extent of removal of mecoprop from leachates appeared to be a good measure of the efficiency of the composting process itself. Unlike BOD and ammoniacal nitrogen, mecoprop cannot be generated in the landfill subsequent to composting and may therefore be a less ambiguous indicator.

The fate of nitrogen within the composting process itself is of particular importance in designing MBP to accelerate stabilisation of the waste. In the past, when “green waste” fractions have been composted to produce fertilisers or soil improvers/conditioners, the presence of ammoniacal-N, or of organic nitrogen which may ultimately give rise to it, has not been a concern to composters and has generally provided a benefit to users.

During coming years, it is likely that increasing proportions of household wastes will be subjected to MBP processes to meet pre-treatment requirements of the Landfill Directive, reduce inputs of organic wastes into landfills, and minimise greenhouse gas and other emissions from those landfills. In such circumstances, removal of nitrogen during the composting process will be a key element in minimising emissions of ammoniacal-N in leachates, over the long-term.

Limited research data have been published regarding the fate of nitrogen during composting of MSW fractions, and the mechanisms involved are not clearly understood. However, this project has highlighted a range of research data that identifies the effects of composting processes on subsequent emissions of ammoniacal-N in leachate. The extent to which total removal occurs during the composting process itself, or whether composting may mineralise nitrogen compounds to nitrates, which are subsequently reduced to nitrogen gas within an anoxic/anaerobic landfill, has not, to our knowledge, been established. This provides a key area on which research into composting of MSW fractions should be concentrated.

The presence of sulphate within landfilled MSW compost leachates is also an interesting issue. Within the strongly anaerobic and methanogenic environment of a MSW landfill, sulphates are generally present at only low concentrations, being reduced to sulphides - which typically precipitate as sulphides of iron or other metals. In the landfills investigated here, concentrations of several hundred mg/l of sulphate were commonplace in leachates from landfilled composted MSW. This may well reflect the heterogeneous nature of such landfills and waste materials, with more and less anaerobic zones of the fill.

Finally, there remain major gaps in knowledge about the timescales involved for complete degradation and flushing of composted MSW fractions within landfill sites. There is little doubt that such pre-treatment has the potential to reduce the organic strength and concentrations of ammoniacal-N within leachates, as well as mass release, from such sites. However, the extent to which leachate management timescales can be reduced remains to be determined.

Table 5.10 (1 of 3) Detailed analytical results from samples HR1 to HR6, for samples obtained during February 2002

Determinand:	HR1	HR2	HR3	HR4	HR5	HR6
COD	1099	4406	15590	NO SAMPLE OBTAINED: LOCATION SUBMERGED	228	582
BOD$_{20}$	129	1060	7840		82	>157
BOD$_5$	111	207	4240		3	46
TOC	313	1404	4694		78.5	180
volatile fatty acids (as C) (2)	<20	<20	707		<20	<20
Acetic	ND	ND	563		ND	ND
Propionic	ND	ND	62		ND	ND
n-butyric	ND	ND	ND		ND	ND
iso-butyric	ND	ND	23.8		ND	ND
n-valeric	ND	ND	ND		ND	ND
iso-valeric	ND	ND	57.9		ND	ND
ammoniacal-N	495	2790	4025		286	195
Kjeldahl-N	540	2620	3820		91.8 (?)	186
organic-N (1)	45	ND	ND		ND	ND
nitrite-N	<0.05	<0.05	<0.05		1.0	8.3
nitrate-N	<0.5	<0.5	<0.5		15.1	2.0
phosphate (ortho, as P)	2.8	17.8	8.2		0.3	1.1
sulphate (as SO$_4$)	28	29	423		18	433
Chloride	1400	7400	6000		384	612
conductivity (µS/cm)	9610	36900	39400		3210	4960
alkalinity (as CaCO$_3$)	2530	14500	17400		1100	879
pH-value	7.5	8.0	8.3		7.9	8.1
Sodium	819	2380	4080		419	509
Magnesium	99	77	77		47	91
Potassium	585	1370	1310		211	328
Calcium	205	31	27		84	122
Chromium	0.248	1.289	8.71		0.095	0.221
Manganese	1.69	0.98	0.38		0.32	0.46
Iron	8.45	2.77	4.31		1.05	0.96
Nickel	0.073	0.286	0.448		0.025	0.086
Copper	<0.02	0.06	0.37		<0.02	0.06
Zinc	0.04	0.08	0.35		<0.03	0.12
Cadmium	<0.001	<0.001	<0.001		<0.001	<0.001
Lead	0.02	0.04	0.10		0.01	0.02
Arsenic	0.075	0.282	0.001		0.009	0.017
Mercury	<0.0001	<0.0001	<0.0001		<0.0001	<0.0001

Notes:

Results in mg/l except pH-value and conductivity (µS/cm)

(1) "organic-N" calculated as Kjeldahl-N minus ammoniacal-N

(2) detection limit is <10 mg/l for individual fatty acids

Table 5.10 (2 of 3) Detailed analytical results from samples HR1 to HR6

Substance	Detection limit (µg/l)	HR1	HR2	HR3	HR5	HR6
Organochlorine Pesticides						
1,2,3-Trichlorobenzene	0.2	<0.2	**0.21**	<0.2	<0.2	<0.2
1,2,4-Trichlorobenzene	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
1,3,5-Trichlorobenzene	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Aldrin	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
alpha-HCH	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
beta-HCH	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
delta-HCH	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Dieldrin	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
alpha-Endosulphan	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
beta-endosulphan	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Endosulphan Sulphate	0.2	<0.2	**0.52**	**0.33**	<0.2	<0.2
Endrin	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
gamma HCH	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Hexachlorobenzene	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Hexachlorobutadiene	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Isodrin	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
o,p-DDE	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
o,p-DDT	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
o,p-TDE	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
p,p-DDE	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
p,p-DDT	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
p,p-TDE	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Permethrin	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Trifluralin	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Organophosphorus Pesticides						
Azinphos-ethyl	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Azinphos-methyl	0.4	<0.4	<0.4	<0.4	<0.4	<0.4
Chlorfenvinphos	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Diazinon	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Dichlorvos	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Dimethoate	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Fenitrothion	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Fenthion	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Malathion	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Mevinphos	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Parathion	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Parathion-methyl	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Triazophos	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Acid Herbicides						
2,4-D (ester and non-ester)	1	<1	<1	<1	<1	<1
Bentazone	1	1.1	<1	<1	<1	<1
MCPP (Mecoprop)	1	**38**	**47**	**120**	<1	**13**
Pentachlorophenol	1	<1	<1	<1	<1	<1
Triazine Herbicides						
Atrazine	0.2	<0.2	**2.1**	<0.4	<0.2	**1.1**
Simazine	0.2	<0.2	<0.2	<0.4	<0.2	<0.2
Uron Herbicides						
Isoproturon	0.2	<0.2	<0.2	<0.8	<0.2	**0.69**
Linuron	0.2	<0.2	<0.2	<0.8	<0.2	<0.2
Phenols						
2,4-Dichlorophenol	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
2-Chlorophenol	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Cresols (sum)	0.2	<0.2	**11**	**65**	<0.2	<0.2
Dichlorophenols (sum)	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Monochlorophenols (sum)	0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Phenol	0.2	**0.80**	**30**	**130**	<0.2	**0.65**
Trichlorophenols (sum)	0.2	<0.2	<0.2	**1.3**	<0.2	<0.2
Xylenols (sum)	0.2	**17**	<0.2	**13**	<0.2	<0.2
Total of analysed phenols	0.16-1.6	**18**	**41**	**210**	<1.6	**0.65**

Table 5.10 (3 of 3) Detailed analytical results from samples HR1 to HR6

Substance	**Detection limit (µg/l)**	**HR1**	**HR2**	**HR3**	**HR5**	**HR6**
AOX	*~100*	*967*	*3416*	*6064*	*213*	*366*
PCBs as Congeners						
PCB as Congener 101	0.1	<0.1	<0.1	<0.1	<0.1	<0.1
PCB as Congener 118	0.1	<0.1	<0.1	<0.1	<0.1	<0.1
PCB as Congener 138	0.1	<0.1	<0.1	<0.1	<0.1	<0.1
PCB as Congener 153	0.1	<0.1	<0.1	<0.1	<0.1	<0.1
PCB as Congener 180	0.1	<0.1	<0.1	<0.1	<0.1	<0.1
PCB as Congener 28	0.1	<0.1	<0.1	<0.1	<0.1	<0.1
PCB as Congener 52	0.1	<0.1	<0.1	<0.1	<0.1	<0.1
BTEX						
Benzene	30	<30	<30	<30	<30	<30
Ethyl benzene	30	<30	<30	<30	<30	<30
m-Xylene	30	<30	<30	<30	<30	<30
o-Xylene	30	<30	<30	<30	<30	<30
p-Xylene	30	<30	<30	<30	<30	<30
Toluene	30	<30	**31**	<30	<30	<30
PAH EPA 16						
Benzo(a) pyrene	0.1-1.0	<0.10	<1.0	<10	**0.41**	<1.0
Benzo(b) fluoranthene	0.1-1.0	<0.10	<1.0	<10	**0.33**	<1.0
Benzo (ghi) perylene	0.1-1.0	<0.10	<1.0	<10	**0.18**	<1.0
Benzo(k) fluoranthene	0.1-1.0	<0.10	<1.0	<10	**0.18**	<1.0
Fluoranthene	0.1-1.0	<0.10	<1.0	<10	**0.37**	<1.0
Indeno (1,2,3-cd) pyrene	0.1-1.0	<0.10	<1.0	<10	**0.30**	<1.0
Naphthalene	0.1-1.0	<0.1	<1.0	<10	<0.1	<0.1
Haloforms and DCE						
1,1,1-Trichloroethane	1	<1	<1	<1	<1	<1
1,1,2-Trichloroethane	1	<1	<1	<1	<1	<1
1,2-Dichloroethane	1	<1	<1	<1	<1	<1
Carbon Tetrachloride	1	<1	<1	<1	<1	<1
Chloroform	1	<1	<1	<1	<1	<1
Tetrachloroethene	1	<1	<1	<1	<1	<1
Trichloroethene	1	<1	<1	<1	<1	<1
US EPA Method 8240/624						
Dichloromethane	1	<1	<1	<1	<1	<1
Chloronitrotoluenes						
Chloronitrotoluenes	0.1-10.0	<0.1	<1	<10	<0.1	<0.1
Biphenyl						
Biphenyl	20-40	<0.1	<1	<10	<0.1	<0.1
Cyanide						
Cyanide, complex (as CN)	20-40	<20	<40	<40	<20	<20
Cyanide, free (as CN)	20-40	<20	<40	<40	<20	<20
Cyanide, total (as CN)	20-40	<20	<40	<40	<20	<20
Metals						
Total Organic Tin	0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Dibutyl tin	0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Tributyl tin	0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Triphenyl Tin	0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Chromium	50	**220**	**2320**	**13100**	<50	**110**
Nickel	10	**40**	**180**	<100	<10	**30**
Copper	5	<5	**43**	**325**	**6**	**22**
Zinc	5	<5	<25	**174**	<5	**115**
Cadmium	6	<6	<30	<60	<6	<6
Lead	50-500	<50	<250	<500	<50	<50
Mercury	1	<1	<1	<1	<1	<1
Arsenic	10-50	**44**	**53**	<50	<10	**18**

Table 5.11 (1 of 3) Detailed analytical results from samples HR7 to HR10, obtained during July 2002

Determinand:	HR7	HR8	HR9	HR10
COD	1020	869	1620	4670
BOD_{20}	24	59	130	843
BOD_5	3	6	35	202
TOC	340	308	543	1480
volatile fatty acids (as C) (2)	<10	<10	<10	<10
Acetic	ND	ND	ND	ND
Propionic	ND	ND	ND	ND
n-butyric	ND	ND	ND	ND
iso-butyric	ND	ND	ND	ND
n-valeric	ND	ND	ND	ND
iso-valeric	ND	ND	ND	ND
Ammoniacal-N	1.8	34.2	197	1130
Kjeldahl-N	43.3	55.9	-	1320
organic-N (1)	41.5	21.7	-	190
nitrite-N	0.3	1.0	0.1	<0.1
nitrate-N	4.7	6.3	<0.3	<0.3
phosphate (ortho, as P)	0.3	0.4	2.8	12.4
sulphate (as SO_4)	878	414	449	117
Chloride	1090	901	2290	2270
conductivity (µS/cm)	5900	4860	9540	14000
alkalinity (as $CaCO_3$)	895	1670	2010	6120
pH-value	8.5	8.4	7.9	8.3
Sodium	789	622	1250	1520
Magnesium	67	64	104	88
Potassium	387	393	777	728
Calcium	255	232	329	176
Chromium	0.090	0.042	0.096	0.494
Manganese	1.61	1.45	2.94	1.38
Iron	1.31	2.59	13.9	19.5
Nickel	0.097	0.051	0.082	0.260
Copper	0.25	0.08	0.15	0.18
Zinc	0.53	0.19	0.20	0.67
Cadmium	0.003	0.002	0.003	0.006
Lead	0.02	0.01	0.04	0.13
Arsenic	0.004	<0.001	0.006	<0.001
Mercury	<0.0001	<0.0001	<0.0001	<0.0001

Notes: Results in mg/l except pH-value and conductivity (µS/cm)
(1) "organic-N" calculated as Kjeldahl-N minus ammoniacal-N
(2) detection limit is <5 mg/l for individual fatty acids

Table 5.11 (2 of 3) Detailed analytical results from samples HR7 to HR10

Substance	Detection limit (µg/l)	HR7	HR8	HR9	HR10
Organochlorine Pesticides					
1,2,3-Trichlorobenzene	0.16	<0.16	<0.16	<0.16	<0.16
1,2,4-Trichlorobenzene	0.16	<0.16	<0.16	<0.16	<0.16
1,3,5-Trichlorobenzene	0.16	<0.16	<0.16	<0.16	<0.16
Aldrin	0.16	<0.16	<0.16	<0.16	<0.16
alpha-HCH	0.16	<0.16	<0.16	<0.16	<0.16
beta-HCH	0.16	<0.16	<0.16	<0.16	<0.16
delta-HCH	0.16	<0.16	<0.16	**0.29**	<0.16
Dieldrin	0.16	<0.16	<0.16	<0.16	<0.16
alpha-Endosulphan	0.16	<0.16	<0.16	<0.16	<0.16
beta-endosulphan	0.16	<0.16	<0.16	<0.16	<0.16
Endosulphan Sulphate	0.16	<0.16	<0.16	<0.16	<0.16
Endrin	0.16	<0.16	<0.16	<0.16	<0.16
gamma HCH	0.16	<0.16	<0.16	<0.16	<0.16
Hexachlorobenzene	0.16	<0.16	<0.16	<0.16	<0.16
Hexachlorobutadiene	0.16	<0.16	<0.16	<0.16	<0.16
Isodrin	0.16	<0.16	<0.16	<0.16	<0.16
o,p-DDE	0.16	<0.16	<0.16	<0.16	<0.16
o,p-DDT	0.16	<0.16	<0.16	<0.16	<0.16
o,p-TDE	0.16	<0.16	<0.16	<0.16	<0.16
p,p-DDE	0.16	<0.16	<0.16	<0.16	<0.16
p,p-DDT	0.16	<0.16	<0.16	<0.16	<0.16
p,p-TDE	0.16	<0.16	<0.16	<0.16	<0.16
Permethrin	0.16	<0.16	<0.16	<0.16	<0.16
Trifluralin	0.16	<0.16	<0.16	<0.16	<0.16
Organophosphorus Pesticides					
Azinphos-ethyl	0.2	<0.2	<0.2	<0.2	<0.2
Azinphos-methyl	0.4	<0.4	<0.4	<0.4	<0.4
Chlorfenvinphos	0.2	<0.2	<0.2	<0.2	<0.2
Diazinon	0.2	<0.2	<0.2	<0.2	<0.2
Dichlorvos	0.2	<0.2	<0.2	<0.2	<0.2
Dimethoate	0.2	**0.27**	**0.34**	<0.2	<0.2
Fenitrothion	0.2	<0.2	<0.2	<0.2	<0.2
Fenthion	0.2	<0.2	<0.2	<0.2	<0.2
Malathion	0.2	<0.2	<0.2	<0.2	<0.2
Mevinphos	0.2	<0.2	<0.2	<0.2	<0.2
Parathion	0.2	<0.2	<0.2	<0.2	<0.2
Parathion-methyl	0.2	<0.2	<0.2	<0.2	<0.2
Triazophos	0.2	<0.2	<0.2	<0.2	<0.2
Acid Herbicides					
2,4-D (ester and non-ester)	0.1-1.0	<1	<1	<0.1	<1
Bentazone	0.1-1.0	<1	<1	<0.1	<1
MCPP (Mecoprop)	0.1-1.0	<1.0	<1	**0.47**	**7.9**
Pentachlorophenol	0.1-1.0	<0.1	<1	<0.1	<1
Triazine Herbicides					
Atrazine	0.2	<0.2	<0.2	<0.2	<0.2
Simazine	0.2	<0.2	<0.2	<0.2	<0.2
Uron Herbicides					
Isoproturon	0.2	<0.2	<0.2	<0.2	<0.2
Linuron	0.2	<0.2	<0.2	<0.2	<0.2
Phenols					
2,4-Dichlorophenol	0.2-2.0	<0.2	<0.2	<2	<0.2
2-Chlorophenol	0.2-2.0	<0.2	<0.2	<2	<0.2
Cresols (sum)	0.2-2.0	<0.2	<0.2	<2	<0.2
Dichlorophenols (sum)	0.2-2.0	<0.2	<0.2	<2	<0.2
Monochlorophenols (sum)	0.2-2.0	<0.2	<0.2	<2	<0.2
Phenol	0.2-2.0	<0.2	<0.2	<2	<0.2
Trichlorophenols (sum)	0.2-2.0	<0.2	<0.2	<2	<0.2
Xylenols (sum)	0.2-2.0	<0.2	**0.65**	<2	<0.2
Total of analysed phenols	1.6-16.0	<1.6	**0.65**	<16	<1.6

Table 5.11 (3 of 3) Detailed analytical results from samples HR7 to HR10

Substance	Detection limit (µg/l)	HR7	HR8	HR9	HR10
AOX	***~100***	***370***	***180***	***1500***	***790***
PCBs as Congeners					
PCB as Congener 101	0.1	<0.1	<0.1	<0.1	<0.1
PCB as Congener 118	0.1	<0.1	<0.1	<0.1	<0.1
PCB as Congener 138	0.1	<0.1	<0.1	<0.1	<0.1
PCB as Congener 153	0.1	<0.1	<0.1	<0.1	<0.1
PCB as Congener 180	0.1	<0.1	<0.1	<0.1	<0.1
PCB as Congener 28	0.1	<0.1	<0.1	<0.1	<0.1
PCB as Congener 52	0.1	<0.1	<0.1	<0.1	<0.1
BTEX					
Benzene	30	<30	<30	<30	<30
Ethyl benzene	30	<30	<30	<30	<30
m-Xylene	30	<30	<30	<30	<30
o-Xylene	30	<30	<30	<30	<30
p-Xylene	30	<30	<30	<30	<30
Toluene	30	<30	<30	<30	<30
PAH EPA 16					
Benzo(a) pyrene	0.25-2.5	<0.25	<0.25	<0.25	<2.5
Benzo(b) fluoranthene	0.25-2.5	<0.25	<0.25	<0.25	<2.5
Benzo (ghi) perylene	0.25-2.5	<0.25	<0.25	<0.25	<2.5
Benzo(k) fluoranthene	0.25-2.5	<0.25	<0.25	<0.25	<2.5
Fluoranthene	0.25-2.5	<0.25	<0.25	<0.25	**2.6**
Indeno (1,2,3-cd) pyrene	0.25-2.5	<0.25	<0.25	<0.25	<2.5
Naphthalene	0.25-2.5	<0.25	<0.25	<0.25	<2.5
Haloforms and DCE					
1,1,1-Trichloroethane	1	<1	<1	<1	<1
1,1,2-Trichloroethane	1	<1	<1	<1	<1
1,2-Dichloroethane	1	<1	<1	<1	<1
Carbon Tetrachloride	1	<1	<1	<1	<1
Chloroform	1	<1	<1	<1	<1
Tetrachloroethene	1	<1	<1	<1	<1
Trichloroethene	1	<1	<1	<1	<1
US EPA Method 8240/624					
Dichloromethane	1	<1	<1	<1	<1
Chloronitrotoluenes					
Chloronitrotoluenes	0.25-2.5	<0.25	<0.25	<0.25	<2.5
Biphenyl					
Biphenyl	0.25-2.5	<0.25	<0.25	<0.25	<2.5
Cyanide					
Cyanide, complex	20	<20	<20	<20	<20
Cyanide, free	20	<20	<20	<20	<20
Cyanide, total	20	<20	<20	<20	<20
Metals					
Total Organic Tin	0.1	<0.1	<0.1	<0.1	<0.1
Dibutyl tin	0.1	<0.1	<0.1	<0.1	<0.1
Tributyl tin	0.1	<0.1	<0.1	<0.1	<0.1
Triphenyl Tin	0.1	<0.1	<0.1	<0.1	<0.1
Chromium	250	<250	<250	<250	870
Nickel	50	<50	<50	<50	**210**
Copper	5	**152**	**55**	**89**	**374**
Zinc	5	**705**	**225**	**232**	**1032**
Cadmium	30	<30	<30	<30	<30
Lead	250	<250	<250	<250	<250
Mercury	1	<1	<1	<1	<1
Arsenic	10	<10	<10	<10	**61**

6 Residues from hazardous waste pre-treatment

6.1 Background

The Landfill Directive will prohibit the co-disposal of hazardous wastes with biodegradable municipal wastes and require such wastes to be deposited in a hazardous waste landfill or in a non-hazardous waste landfill as long as acceptance criteria are satisfied and biodegradable wastes are not deposited in the same cell. A significant proportion of the solid hazardous waste in the UK is currently co-disposed with MSW. Biological processes dominate leachate quality and the UK has a great deal of information on leachate generated from co-disposal landfill. However, leachate chemistry in landfills for hazardous wastes is likely to be dominated by chemical processes, and the UK has very little information on what to expect from such sites.

Surprisingly little recent information has been published from hazardous waste landfills in other EU states where co-disposal is already banned. The only wide-ranging review of leachate quality from European hazardous waste landfills (Albers, 1991) dated from a period prior to the low organic limits currently in force in several Member States. Detailed research has been undertaken at two more recent sites, but only a limited amount of leachate data have been published (e.g. Gade *et al*., 1997 and 1999). Little information is available for leachate quality from hazardous waste landfills that accepted pre-treated waste.

Data from landfills accepting hazardous and pretreated hazardous wastes were obtained during this study. This chapter presents some of the few available datasets for leachate quality from sites where:

- mixed hazardous wastes have met waste acceptance criteria close to those set by the Landfill Directive, particularly with regard to organic limits and heavy metals;
- mixed hazardous wastes have been deposited with less restrictive controls; and
- hazardous waste have been pretreated by cement stabilisation or incineration.

6.2 Mixed hazardous wastes

The PhD thesis of Albers (1991) presented data on leachate quality at a number of older sites within Europe, mostly dating from the 1980s and earlier. These revealed COD, BOD and NH_4-N concentrations similar to those found in MSW landfills, in addition to elevated concentrations of mineral salts and of some specific organics. For example, one analysis from a 1988 reference (Table 2.6 of Albers, 1991) gave the following results (mg/l) for two landfills (see Table 6.1).

VOCs were typically several hundred μg/l in these leachates. At other landfills of the period, chlorobenzenes and chlorophenols were detected at concentrations of up to several mg/l (Albers, 1991).

Table 6.1 Leachate quality from two hazardous waste landfills

Parameter (mg/l)	1	2
Cl	13,300	28,600
SO_4	2,460	6,180
COD	5,750	7,870
BOD	2,750	4,790
NH_4-N	921	1,250
Zn	2.9	6.0
other metals	<1	<1
phenols	26	49
hydrocarbons	30	12
AOX	32	7.3

For this study, data have been obtained from five currently operating hazardous waste landfills in EU countries (Table 6.2). These are sites A, B, C, F and H in Appendix 3. All impose much stricter limits on organic content than was applied in the past. This is achieved in a variety of ways, including limits on leachable COD or TOC, or a limit for loss on ignition. Typically the average organic content of the wastes that are landfilled now is in the order of 4 to 5%. The effect of these restrictions is that a much higher proportion of these countries' hazardous wastes is incinerated than is currently the case in the UK. The range of wastes being landfilled in a typical hazardous waste site is therefore much narrower than the range of solid wastes currently co-disposed in the UK, despite these countries having broadly similar industrial bases to the UK. At three of the sites (A, B and F), data are also available from older cells, providing some basis for assessing the impact of the present organic limits.

Leachate quality from these landfills is summarised below. The landfills, and the derivation of L/S estimates for them, are described in Appendix 3. The appendix also contains many of the time series graphs and raw data for these sites that are too voluminous to include in the main text.

Table 6.2 Summary of waste inputs to landfill sites in this study

Site	Main waste inputs
A	Filter cakes and treatment plant sludges, bottom ash from hazardous waste incineration, contaminated soil and foundry wastes. Low organic content (average LoI ~4%).
B	Filter cakes and treatment plant sludges, bottom ash from hazardous waste incineration, contaminated soil and demolition wastes. Low organic content (average LoI ~4%).
C	Contaminated soils, filter cakes, metal processing sludges, bottom ash from hazardous waste incineration, wastes from paint manufacture and recycling, grit blasting wastes. Low organic content.
F	Filter cakes from centralised liquid waste treatment plant, and from treatment of APC wash waters from hazardous waste incineration, plus filter cakes from treatment of metal finishing wastes. Organic content low.
G	Cement-stabilised wastes from treatment of, *inter alia,* fly ash and APC residues from municipal and hazardous waste incineration, sludges from liquid waste treatment, metallurgical wastes, foundry sands etc.
H	Cement-stabilised wastes from a plant receiving, *inter alia,* contaminated soils, fly ash, inorganic wastewater treatment plant sludges and filter cakes, paint wastes and metallurgical residues. The site also receives untreated fly ash and APC residues in 'big bags'. Low organic content.

6.2.1 Site A

Information on the site and leachate quality for the current operational area (1989 to present) and old area (1985 to 1989) is summarised below.

Current area (1989 to present)

Site

Main waste inputs: filter cakes and treatment plant sludges, bottom ash from hazardous waste incineration, contaminated soil and foundry wastes. Low organic content (average LoI ~4%).

Engineered containment, leachate collection system maintained empty, waste depth ~8 m, L/S 0.25 – 0.35

Leachate quality [Data and graphs are given in Appendix 3]

- Weekly analyses exhibit extreme spikiness. [Figures 6.1a and 6.1b]
- TDS ~15 – 30,000 mg/l
    - Cl (5 – 11,000) > SO_4 (2 – 6,000) >> NO_x (~500) ~ CO_3 (~400)
    - Na (3 – 7,000) >>Ca (~1,000) > K (~500) > Mg (300)
- pH near neutral (8.2 to 8.4) but spikes to ~10.
- High redox (+200 mV) and level of NO_x indicate non-reducing conditions.
- Significant organic content:
  TOC 20 – 50 mg/l;
  COD 50 – 450 mg/l.
- Ammoniacal nitrogen content several hundred mg/l
  NH_4-N 150-390 mg/l
- Elevated levels of some metals:

V	14 – 55	mg/l	
Zn	2 – 6	mg/l	
Ni	2 – 4	mg/l	
Mo	1 – 2	mg/l	
Cu	0.5 – 4	mg/l	declines with time
Cr	0.6 – 4	mg/l	declines with time
Cd	several tens of μg/l		
Se	a few tens of μg/l		

- Trace organics low or no different to UK MSW or co-disposal landfills:

AOX	< 0.5	mg/l
oil	< 1	mg/l
phenols	<0.1	mg/l
PCB	< 1	μg/l
PAH	<1	μg/l
VOC	<1	μg/l

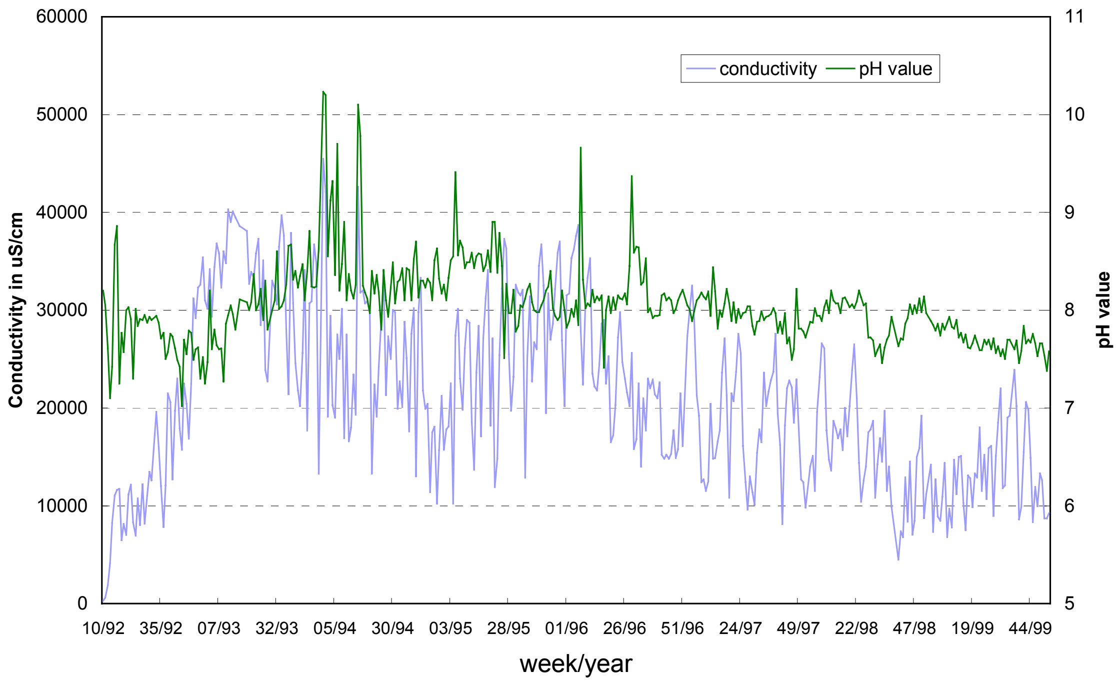


Figure 6.1a Site A, mixed hazardous wastes, sub-area A1: weekly leachate analysis for pH value and conductivity

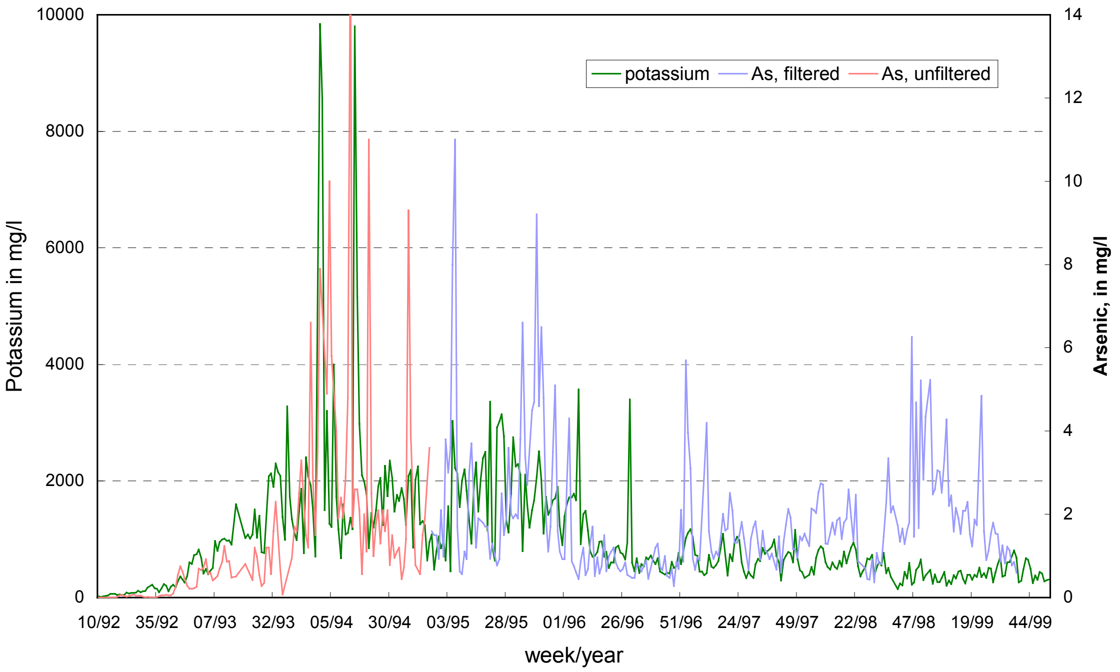


Figure 6.1b Site A, mixed hazardous wastes, sub-area A1: weekly leachate analysis for potassium and arsenic

Old area (1985 to 1989)

Site

Main waste inputs similar range to current area but with less restrictive limits on organic content and leachable metals.

Engineered containment, leachate collection system maintained empty, waste depth ~21 m, L/S 0.25 to 0.35; now restored.

Leachate quality [data and graphs in Appendix 3]

- TDS ~11 to 35,000 mg/l
  Cl (2 – 8,000) ≈ SO_4 (1 – 10,000) > NO_3 (300 – 1000) ?CO_3
  Na (3 – 10,000) >> K (~900) > Ca (150) ~ Mg (150)
- pH alkaline (9.3)
- Significant organic content, similar to current area:
  COD 30 – 300 mg/l
- Significant ammoniacal nitrogen content:
  NH_4-N 200 – 600 mg/l
- Elevated levels of some metals:

V	24 – 161 mg/l	
Ni	~2.5 mg/l	
Mo	1 – 2.3 mg/l	
Cr	0.2 - 5.3 mg/l	declines with time
As	0.4 – 1.1 mg/l	
Se	40 – 800 µg/l	

- Significant fluoride content:

F	30 – 60 mg/l

- Trace organics slightly higher than current area:

AOX	<0.5 mg/l
oil	~3 mg/l
phenols	<0.1 mg/l
PCB	<1 µg/l
PAH	<1 µg/l
VOCS	12 – 60 µg/l

6.2.2 Site B

Current area (1990 to present)

Site

Main waste inputs: filter cakes and treatment plant sludges, bottom ash from hazardous waste incineration, contaminated soil and demolition wastes. Low organic content (average LoI ~4%)

Engineered containment, leachate collection system maintained empty, waste depth ~17-36 m, L/S 0.15 – 0.2

Leachate quality [data are given as time series graphs in Appendix 3]

- Quarterly data not particularly spiky. Not known whether data are means or spot results.
- TDS ~30 – 50,000 mg/l, little decrease with time
  Cl (15 – 25,000) > SO_4 (2 – 6,000) > CO_3 (1 – 3,000) > NO_x (0 - 250)
  Na (8 – 15,000) > K (3 – 6,000) >Ca (400 – 1,400) > Mg (200)
- pH near neutral (7 - 8), no evidence of spikes.
- Redox potential in range –150 to –300mV, so reducing conditions; sulphide initially present at up to 35 mg/l but absent since 1994; NO_x initially absent, then present mainly in winter.
- Significant organic content, declining over period, TOC falling from ~2000 mg/l to ~250 mg/l in 8 years; BOD shows significant degradable content. Data suggest some biological activity; operator confirms some methane produced but very low rates and concentrations.

- Significant ammoniacal nitrogen content up to 800 mg/l, declining to 200 mg/l; this decline could also be the result of biological activity.
- Elevated levels of some metals:

Zn	initially 1 – 2 mg/l, increasing to 6 – 8 mg/l
Ni	initially 2 – 4 mg/l, increasing to ~8 mg/l
Cu	initially 0 –1 mg/l, increasing to ~2 mg/l
As	several tens or hundreds of µg/l
Cd	several tens of µg/l, occasional peaks to ~1 mg/l

- Hg not detected in any sample
- Trace organics:

AOX	3 - 4	mg/l
oil	0 - 10	mg/l
phenol	initially 5 – 10 mg/l then absent	

- Cyanide and halogenated VOC virtually absent

Older area (1976 to 1990)

Site

Main waste inputs as in current area but with less restrictive limits on organic content and leachable metals. Also it is thought that some fly ash and APC residues were deposited.

Engineered containment, leachate collection system maintained empty, waste depth 10 to 15 m, L/S ~0.2.

Leachate quality [quarterly data are shown as time series graphs in Appendix 3]

- Quarterly data not particularly spikey for leachate strength and major ions but NH_4-N and organics show large variability.
- TDS very high, rising gradually over time to ~110,000 mg/l
  Cl (60,000) > SO_4 (1 – 4,000) ~ CO_3 (~2,500)
  Na (30,000) > K (12,000) > Ca (~400) > Mg (~200)
- pH near neutral (7-8), no high pH spikes
- No redox data but generally reducing conditions: NO_x usually absent, but with occasional spikes, up to 150 mg/l; sulphide often absent but with frequent spikes, up to ~100 mg/l; significant BOD, declining slowly with time; observed emission of methane, at very low rates.
- Significant organic content:

TOC	~2,000 mg/l	steady
COD	4-6,000 mg/l	~steady
BOD	2-4,000 mg/l	declining slowly

- Significant ammoniacal nitrogen; rising with time, possibly as a result of capping:

NH_4-N	1,500 mg/l

- Some elevated metals, most declining with time, but some increasing:

Ni	8 mg/l declining to ~0.5 mg/l
Pb	1 mg/l declining to <0.1 mg/l
Cr	0.5-1.5 mg/l declining to <0.1 mg/l
Cd	100-200 µg/l declining to <10µg/l
As	initially <10 µg/l, now spiking to several hundred µg/l

- Several minor organics at significant concentrations:

phenol	10-40 mg/l
hydrocarbons	10-60 mg/l very spikey
AOX	~5 mg/l
VOCs	100-1,000 µg/l

- Cyanide mostly absent, occasionally detected at up to 0.2 mg/l.

6.2.3 Site C: various cells from 1988-present

Site
Main waste inputs: contaminated soils, filter cakes, metal processing sludges, bottom ash from hazardous waste incineration, wastes from paint manufacture and recycling, grit blasting wastes. Low organic content, achieved by leaching test and limit on LoI.

Engineered containment, leachate collection system often maintained in flooded condition, average waste depth ~9 m, L/S estimated at ≤ 0.3 on oldest cells. Completed areas are membrane capped.

Leachate quality [data are shown as time series graphs in Appendix 3]

- Large temporal and spatial variations in leachate strength
- TDS appears lower than at other sites in category, though major ions incomplete.
  Cl (2 – 12,000) > SO_4 (200 – 1,000)
- pH near neutral (6 - 8), no evidence of spikes.
- Redox potential not known
- Significant organic content, COD 100 to 800 mg/l.
- Significant (?ammoniacal) nitrogen content, TKN 20 to 100 mg/l.
- Metals (Ni, Cu, Cr, Pb) mostly low, <1 mg/l, but zinc results of several mg/l are common, and zinc concentrations are generally rising with time.
- No data for minor organics, cyanide, As.

6.2.4 Site F

Current area (1994-present)

Site
Main waste inputs are filter cakes from centralised liquid waste treatment plant, and from treatment of APC wash waters from hazardous waste incineration, plus filter cakes from treatment of metal finishing wastes. Organic content low and controlled by limit on leachable TOC.

Engineered containment with leachate collection system; waste depth ~8m, L/S 0.26

Leachate quality [data are shown as time series graphs in Appendix 3]

- Twice yearly data quite variable.
- High TDS ~20 – 70,000 mg/l, profile (low-high-low) not explained at low L/S ratio
  Cl (10–40,000) >> SO_4 (600–1,000) ~ Br (400–1600) ~ NO_x (200–1,000)> CO_3
  Ca (3–12,000) ~ Na (2–11,000) >K (200–3,000) >> Mg (30-120)
- pH near neutral (6.5 to 7.5), no evidence of spikes.
- Redox potential high, range +200 to +500mV (no gas detected, very little settlement)
- Organic content low, TOC ~30 mg/l.
- Significant ammoniacal nitrogen content 20 – 50 mg/l
- Most metals low, except:
  Zn 1 – 4 mg/l,
  Cd 20 - 80 μg/l
- No analysis for trace organics, V, Mo or Se
- Cyanide <0.1 mg/l
- Fluoride < 1 mg/l

Old area (late 1980s to 1994)

Site

Main waste inputs were to present area: filter cakes from centralised liquid waste treatment plant, and from treatment of APC wash waters from hazardous waste incineration, plus filter cakes from treatment of metal finishing wastes. The inputs also included fly ash APC residue from incineration of hazardous wastes (not deposited in the current area). Organic content low and controlled by limit on leachable TOC.

Engineered containment with leachate collection system; waste depth ~8 m, L/S estimated at 0.14.

Leachate quality

- Twice yearly data less variable than many other sites, possibly a result of maintaining leachate collection system in a part flooded condition.
- Very high TDS ~150 – 200,000 mg/l
  Cl (60 – 100,000) >> Br (3 – 5,000) > SO_4 (~1000) ~ NO_x (~800)> CO_3 (500)
  Na (30– 35,000) > K (10 – 18,000) ~ Ca (6 – 14,000) >> Mg (200)
- pH initially high (~10) then fell sharply to neutral (~7).
- Redox potential high, several hundred mV, non-reducing conditions.
- Organic content significant, TOC 40 – 90 mg/l.
- Ammoniacal nitrogen fairly high 100 – 200 mg/l despite low organic content of wastes.
- Most metals low, except:
  Zn initially low, then rose to ~5 mg/l when pH fell,
  Pb initially low, then rose to 14 mg/l when pH fell,
  Cd 100 – 200 μg/l, unaffected by the pH change.
- No analysis for trace organics, V, Mo or Se
- Cyanide <0.1 mg/l except for a single result of 0.22 mg/l.
- Fluoride < 1 mg/l
- Differences compared with current area (higher TDS, Pb, Cd) most likely due to inclusion of fly ash/APC residues.

6.2.5 Site H (1993 to present)

Site

Main waste inputs: cement-stabilised wastes from a plant receiving, *inter alia,* contaminated soils, fly ash, inorganic wastewater treatment plant sludges and filter cakes, paint wastes and metallurgical residues. The site also receives fly ash and APC residues in 'big bags'. Low organic content, achieved by a limit on leachable COD. The relative dominance of the two categories of waste input cannot be assessed.

Engineered containment with leachate collection system; L/S ratio estimated to be ≤ 0.2.

Leachate quality [data are shown as time series graphs in Appendix 3]

- Leachate strength moderately variable, but no overall change during the period of 3 years
- TDS high, estimated to be at least 40–50,000 mg/l based on chloride and conductivity (other major ions not analysed):
  Cl (20 – 30,000 mg/l)
  conductivity (40,000–80,000μS/cm)
- pH near neutral (7–8), but with occasional high pH spikes (maximum 9.5).
- Redox potential not known.

- Significant organic content, TOC at several tens of mg/l (50-100mg/l), COD much higher (350 to 1,000 mg/l), leading to suspicion of interference by chloride.
- Fatty acid (VFA) content fairly consistent, averaging ~30mg/l as HOAC (≡ 12mg/l as C), therefore comprising ~ 20% of the TOC.
- Phenols very low, typically at ≤ 1mg/l.
- Elevated levels of some metals cf. MSW leachates:

Pb	1–1.5 mg/l,
Ni	~1 mg/l,
Cd	200–600 μg/l.

- Other metals similar to MSW leachate:

Zn	0.2–1mg/l
Cu	0.1–0.3mg/l
Cr	0–0.1mg/l
Hg	0–8μg/l
As	0–7μg/l

- Cyanide at trace levels, up to 0.6 mg/l, mean 0.16 mg/l.
- Fluoride low, at ~2mg/l.
- No data for most major ions, nitrogen content, trace organics, Br, Mo, Se, V.

6.2.6 Overview of leachates from mixed hazardous waste landfills

The leachate quality data from mixed hazardous waste sites may be summarized as follows.

- Large short-term fluctuations in strength may occur; especially where the leachate collection system is maintained empty and where flows from uncapped areas respond rapidly to rainfall events.
- TDS concentrations span a wide range and are generally very high, individual sites giving the following ranges:

15 – 30 g/l
30 – 50 g/l
150 – 200 g/l (inputs thought to contain fly ash)
20 – 70 g/l
2 – 12 g/l
40 – 50 g/l

- Anions are dominated by Cl; SO_4 is typically only ~1,000 mg/l (occasionally several thousand); carbonate/bicarbonate is a surprisingly minor component (<1,000 mg/l). Br, when analysed, has been significant, at 1,000 to 6,000 mg/l; fluoride is always low (<100 mg/l) compared with other ions but in one case was significantly higher than in MSW or ash leachates. NO_x is variable (0 to 500 mg/l) depending on redox and on the presence of anaerobic activity.
- Cations are generally Na > K > Ca > Mg but in some instances Ca and K may be more dominant and/or of a similar order to Na.
- pH values of collected leachate are mostly near neutral with occasional high pH spikes, indicating an alkaline matrix affected by carbonation close to the main flow pathways.
- Conditions range from high redox (+200mV) with no evidence of biological activity, to low redox (≤-200 mV) and clear evidence of anaerobic processes, albeit at a low rate. This appears to be linked to the organic content of the wastes and is clearly dependent upon acceptance criteria.
- Organic content varies over a wide range but is generally much lower than at sites taking bioreactive wastes. Ranges at individual sites:

TOC	20 – 50	mg/l	
	2,000 – 250	mg/l	(falling)
	~30	mg/l	
	40 – 90	mg/l	
	30 – 250	mg/l	
	60	mg/l	

Most data sets have no BOD results and no other information on the nature of the organics. Metals results at some sites suggest that the organics may include strong ligands.

- Ammoniacal nitrogen concentrations vary over a wide range and can be similar to those at sites taking bioreactive wastes. Ranges at individual sites:

150 – 390 mg/l	(old area 200 – 600)
200 – 800 mg/l	(old area up to 1,500)
20 – 50 mg/l	
20 – 100 mg/l	

It is likely that the ammonium is present in the incoming treated wastes as there is little potential for it to be generated *in-situ* by degradation.

- Some elevated concentrations of heavy metals occur. These vary from site to site and often the data sets covered only a small list of metals:

10 – 100 mg/l	V
1 – 10 mg/l	Zn, Ni, Mo, Cu, Cr
10 – 1,000 μg/l	Pb, Cd, Se, As.

- Trace organics have only been analysed to a very limited extent (e.g. AOX, oil, PAH, PCBs VOCs) but have generally been low compared with typical MSW and co-disposal leachates:

occasionally phenols at 1 – 10 mg/l
(older sites) 10 – 100 mg/l

occasionally VOCs at 10 – 100 μg/l
(older sites) 100 – 1,000 μg/l

VOCs, when detected, have been simple halogenated aliphatic solvents.

- Cyanide has usually been absent, or present at very low concentrations (≤0.1 mg/l).

6.3 Landfills accepting pretreated hazardous wastes

6.3.1 Solidified or stabilised hazardous wastes

Stabilisation/solidification processes based on Portland cement and other inorganic ingredients have become established in several EU states. It is expected that they may be re-established commercially in the UK when the Landfill Directive is implemented. The literature search revealed no examples of leachate data for this type of landfill. Data were obtained for this study from a current operational facility, Site G, described in Appendix 3.

Site G

Site

Cells used for cement-stabilized wastes, including fly ash and APC residues from municipal and hazardous waste incineration, sludges from liquid waste treatment, and metallurgical wastes, foundry sands etc. Containment cells, waste depth ~3 m, L/S ~0.1.

Collected leachate is a composite of percolation and surface run-off from three similar cells, two of which have geomembrane capping and one of which is exposed. Therefore concentrations are expected to have been diluted to approximately one-third of what they might be from the uncovered cell. Leachate quality data are shown as time series graphs in Figures 6.2 and 6.3.

Leachate quality

- Concentrations of major parameters very spikey.
- TDS high, ~20,000 to 40,000 mg/l
  Cl (6-20,000) > NO_3 (1-3,000) > SO_4 (1-2,000)
  Na (4-8,000) > K (2-6,000) > Ca (1-3,000)
- pH initially ~7.5 but rose by ~2 units to 9.5 – 10, part-way through data set.
- No evidence of biological activity; high NO_3 concentrations suggest non-reducing conditions.
- Significant concentration of organics:

TOC	40-180 mg/l
phenols	2-30 mg/l

- No data on NH_4-N or TKN.
- Some elevated metal concentrations prior to the rise pH value:

Ni	0.5-3 mg/l
Cd	50-150 μg/l
As	100-600 μg/l

- No data for trace organics.

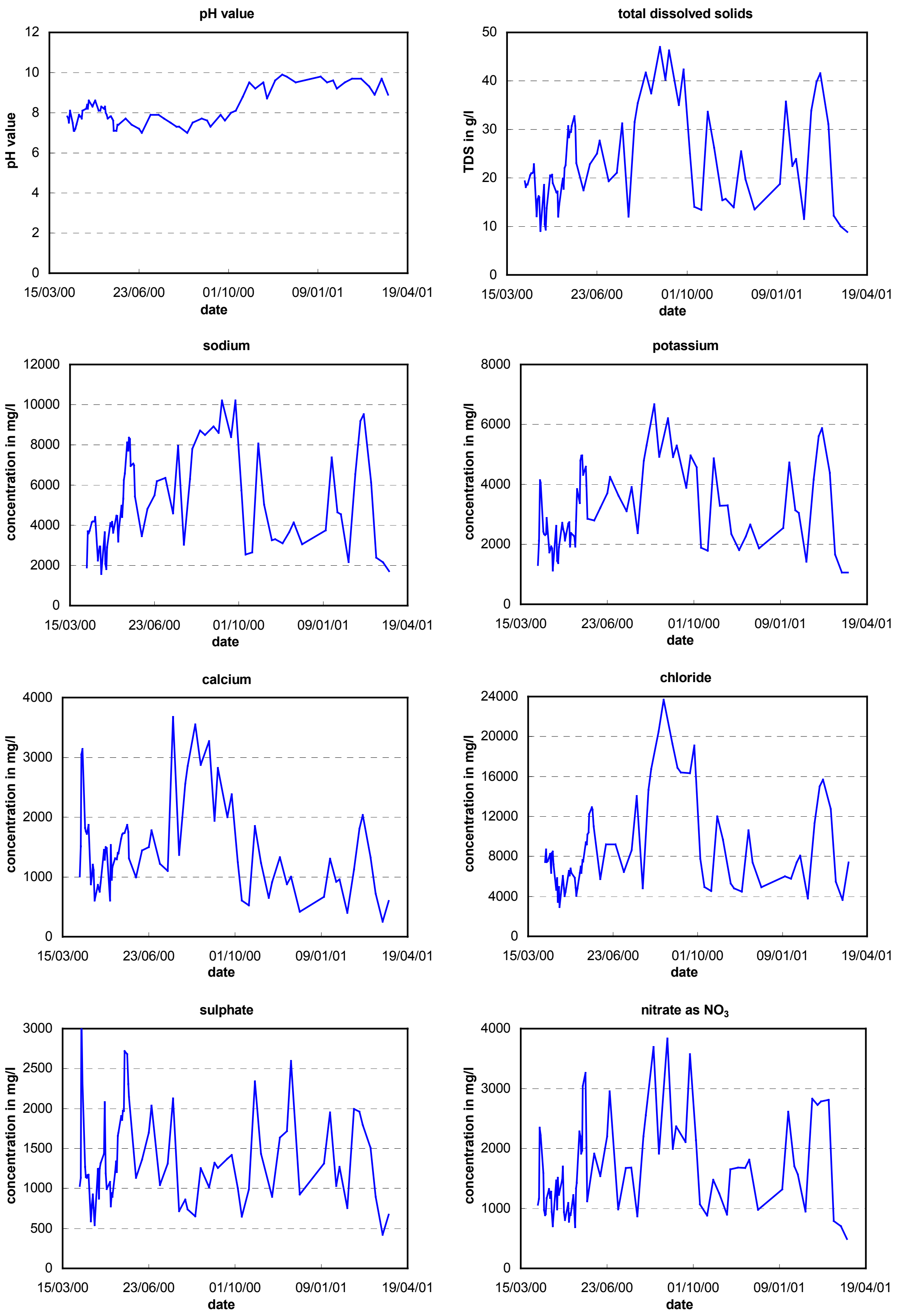


Figure 6.2 Leachate quality at Site G: cells for solidified hazardous wastes

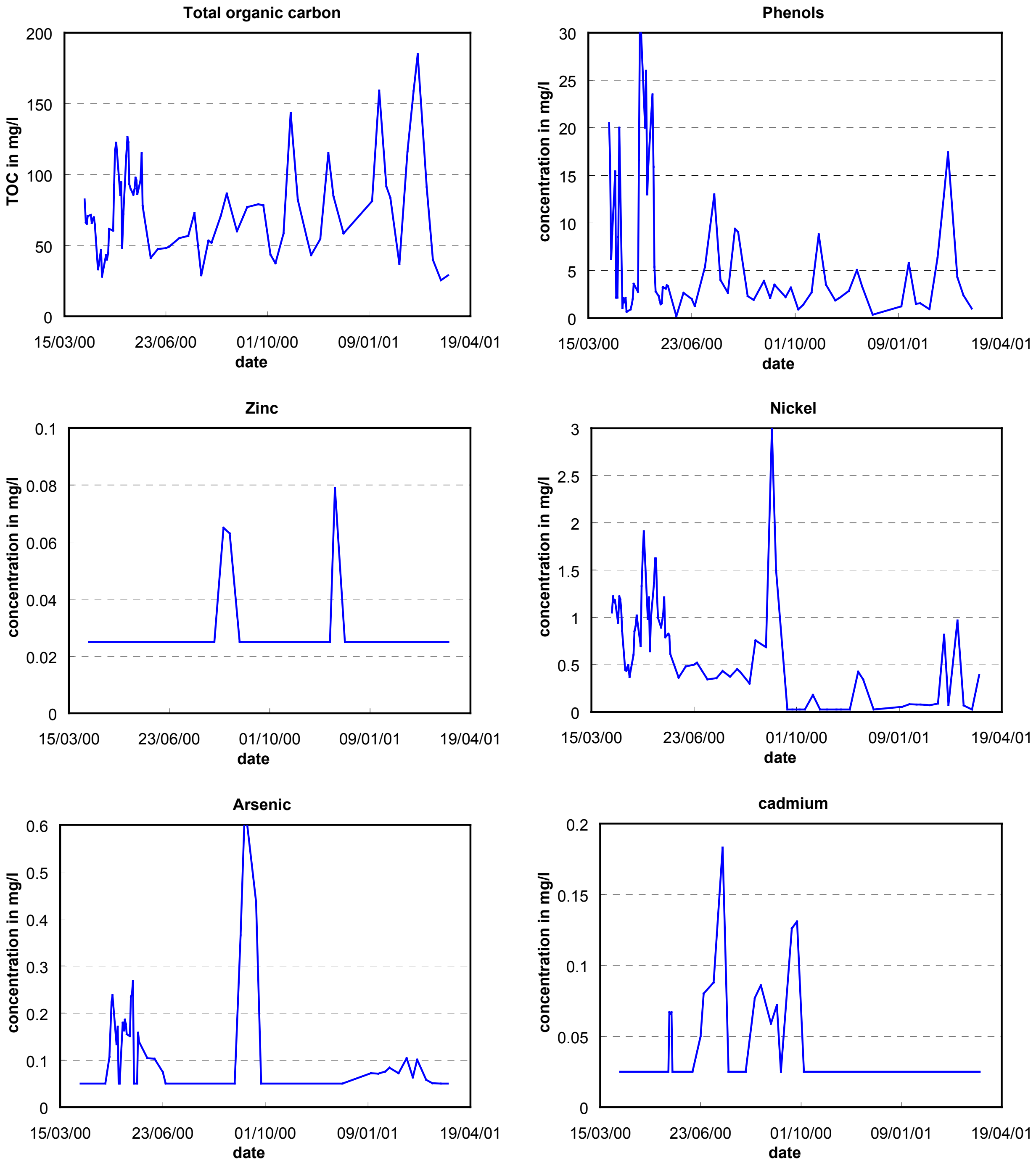


Figure 6.3 **Leachate quality at Site G: cells for solidified hazardous wastes**

6.3.2 Bottom ash from incineration of hazardous wastes

High temperature incineration of hazardous wastes produces a partly vitrified, granular bottom ash or slag. The literature search revealed no leachate quality data for this type of material.

Site F

Site

Data for this study were obtained from Site F (see Appendix 3), where four cells have been used solely for HWI bottom ash. The cells were filled in sequence, from 1975 to the early 1990s. The first cell is thus 15 years older than the most recent. The cells remain un-capped. L/S ratios calculated for these cells range from 0.1 to 0.6 but are subject to a high degree of uncertainty. Leachate quality data from annual sampling from 1995 to 1999 are shown in Figures 6.4 to 6.6 and summarized below:

Leachate quality

- Annual samples show no obvious spikiness
- Significant TDS levels (3-15,000 mg/l):
  SO_4(~2,000) > Cl^- (300-2,000) > CO_3 (500) >> NO_x ~ Br^-
  Na (1-4,000) >> K (200-700) ~ Ca (50-400) > Mg (10-60)
- Some atypical results e.g. high SO_4 in Cell 4, high carbonate in Cell 2
- pH slightly alkaline, decreasing slowly over the period of the data (? carbonation). Cell 2 significantly higher than the others
- Redox potential uniformly high (+400 mV)
- Organic content mostly low:
  TOC ~5 mg/l
  COD 25-50 mg/l
  [Cell 2 has higher organic content (TOC up to 40 mg/l)]
- No significant ammoniacal nitrogen:
  NH_4-N $\leq$ 0.5 mg/l
- Metal concentrations low but Cell 2 has anomalous As, Cr, Pb:
  Cr 0.1-0.3 mg/l
  As 60-150 µg/l
  Pb 180 µg/l
- Fluoride $\leq$ 2 mg/l
- Cyanide $\leq$ 0.1 mg/l
- No trace organic analysis

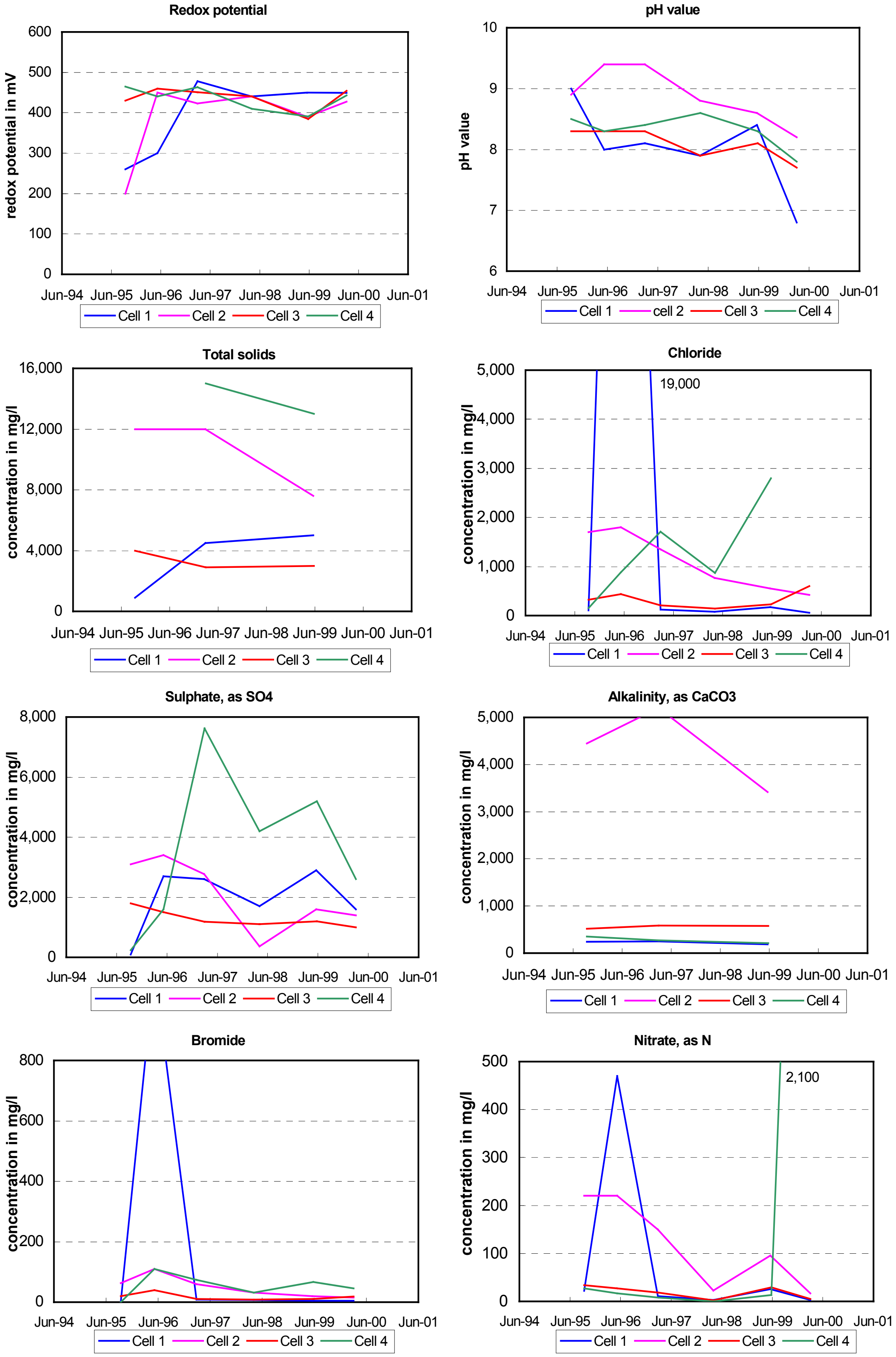


Figure 6.4 **Leachate quality from cells receiving bottom ash/slag from incineration of hazardous wastes**

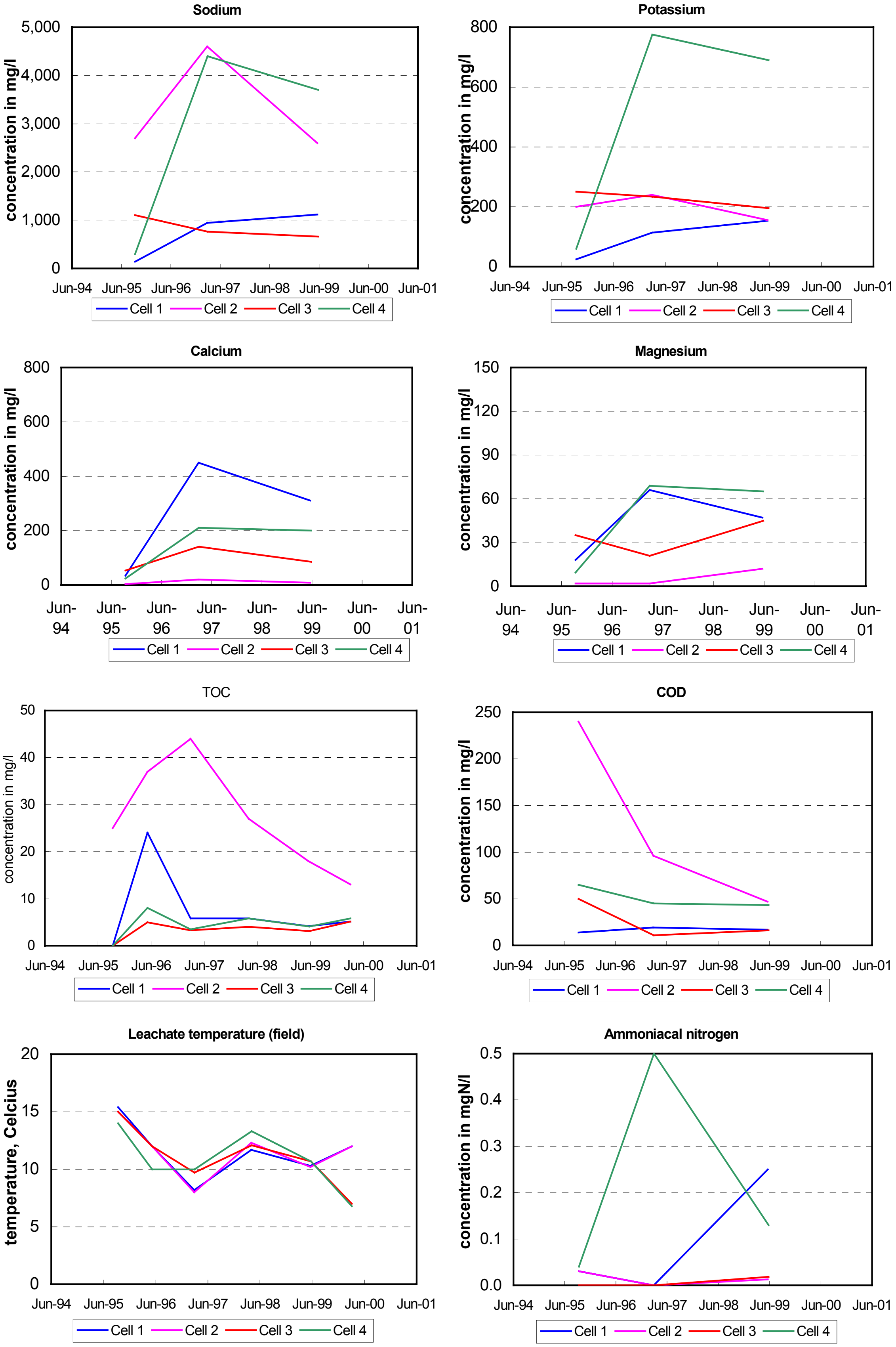


Figure 6.5 **Leachate quality from cells receiving bottom ash/slag from incineration of hazardous wastes**

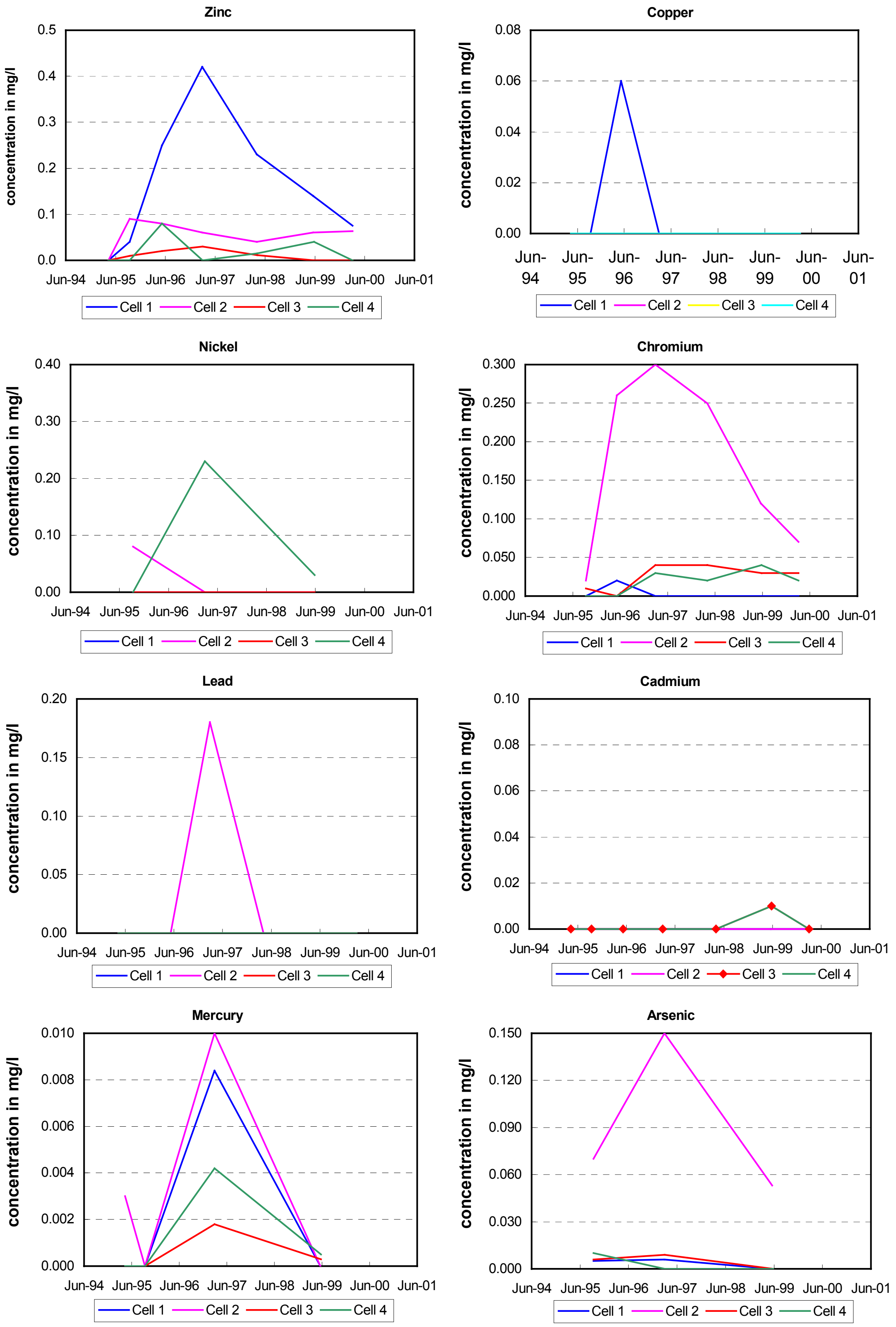


Figure 6.6 **Leachate quality from cells receiving bottom ash/slag from incineration of hazardous wastes (Site F)**

6.3.3 Fly ash from incineration of hazardous wastes

The literature search revealed no leachate quality data for this type of material.

Site F

Site

Data were obtained for this study from Site F (see Appendix 3), where HWI fly ash has been deposited in big bags in a dedicated cell. The bags contain only fly ash: APC sludges are not present in this cell. The cell has been in use since 1994 and was still operational and uncapped at the time of this study. The estimated L/S ratio by the beginning of 2001 is 0.26.

Leachate quality

Data from twice yearly sampling since 1994 are shown in Figures 6.7 to 6.9 and summarized below:

- The results show dissolved solids increasing sharply ~2-3 years after the cell began operation and remaining more or less steady since then.
- Very high TDS (~100-150,000 mg/l)
    Cl (60-80,000 > Br (5-8,000) > SO_4 (1-3,000) >> CO_3 (50) > NO_3 (~10)
    Na (30-40,000) > K (10-15,000) > Ca (3-6,000) >> Mg (20)
- pH near neutral (~7.5 – 8.5)
- Redox potential mostly very high (≥ 300 mV); (two low results may be erroneous).
- Organic content low:
    TOC 6-10 mg/l
- Ammoniacal nitrogen content low:
    NH_4-N 10-15 mg/l
- Metals generally very low:
    Cr peaks to ~1.7 mg/l, mostly <0.5 mg/l
    Cd peaks to 60 μg/l, mostly <10 μg/l
- Cyanide absent
- Fluoride <0.5 mg/l
- No trace organic analysis

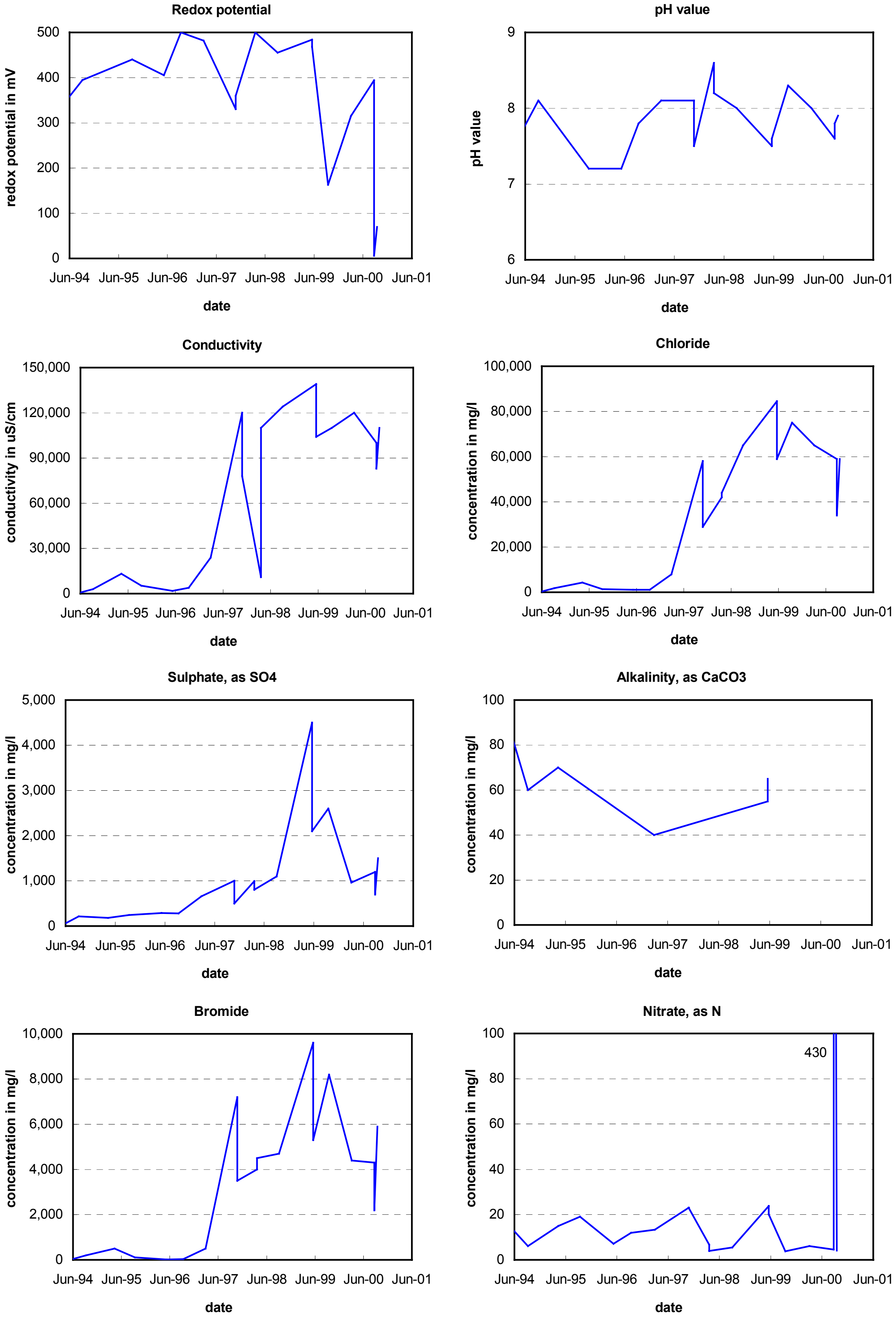


Figure 6.7 **Leachate quality from cells receiving fly ash from incineration of hazardous wastes (Site F)**

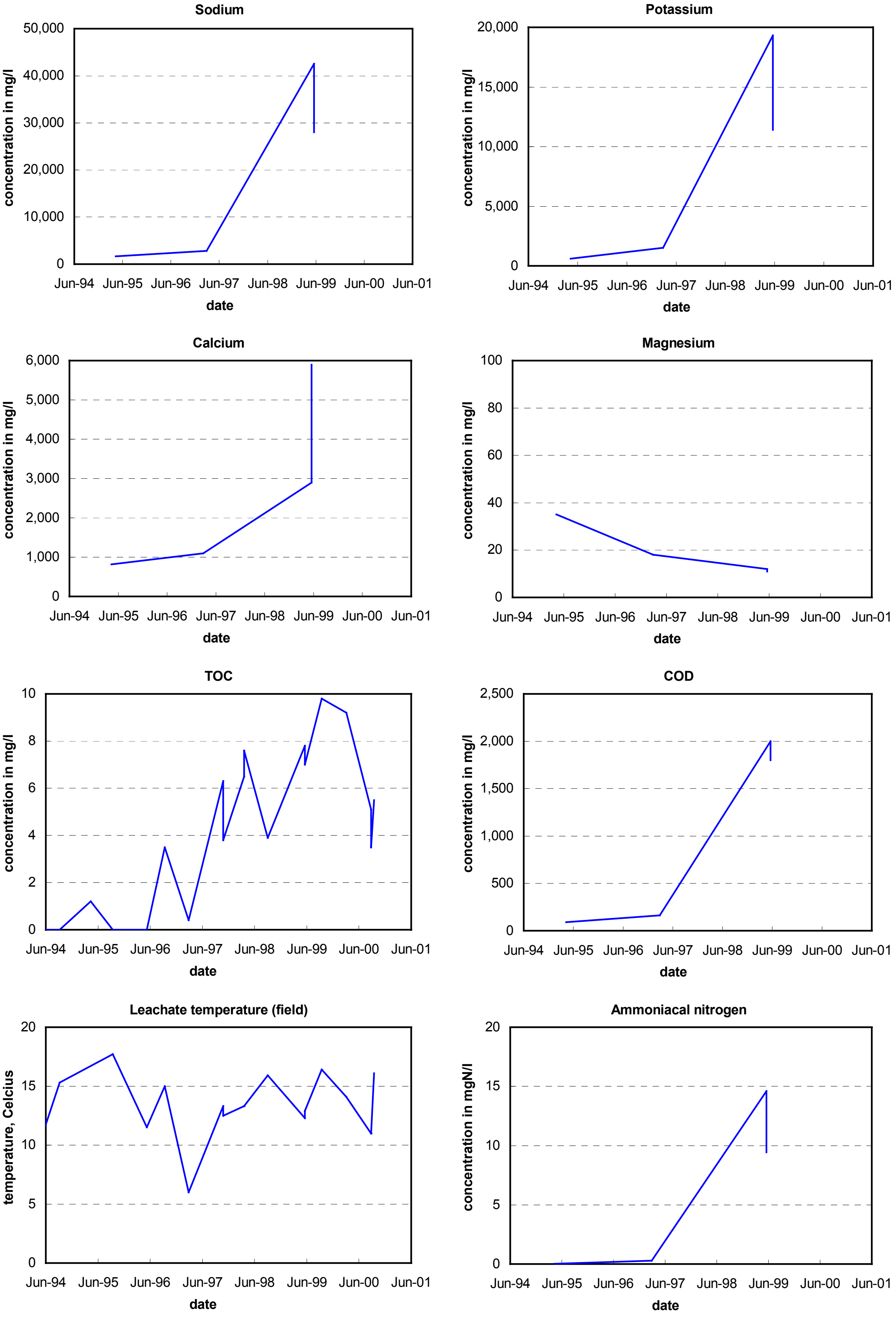


Figure 6.8 **Leachate quality from cells receiving fly ash from incineration of hazardous wastes (Site F)**

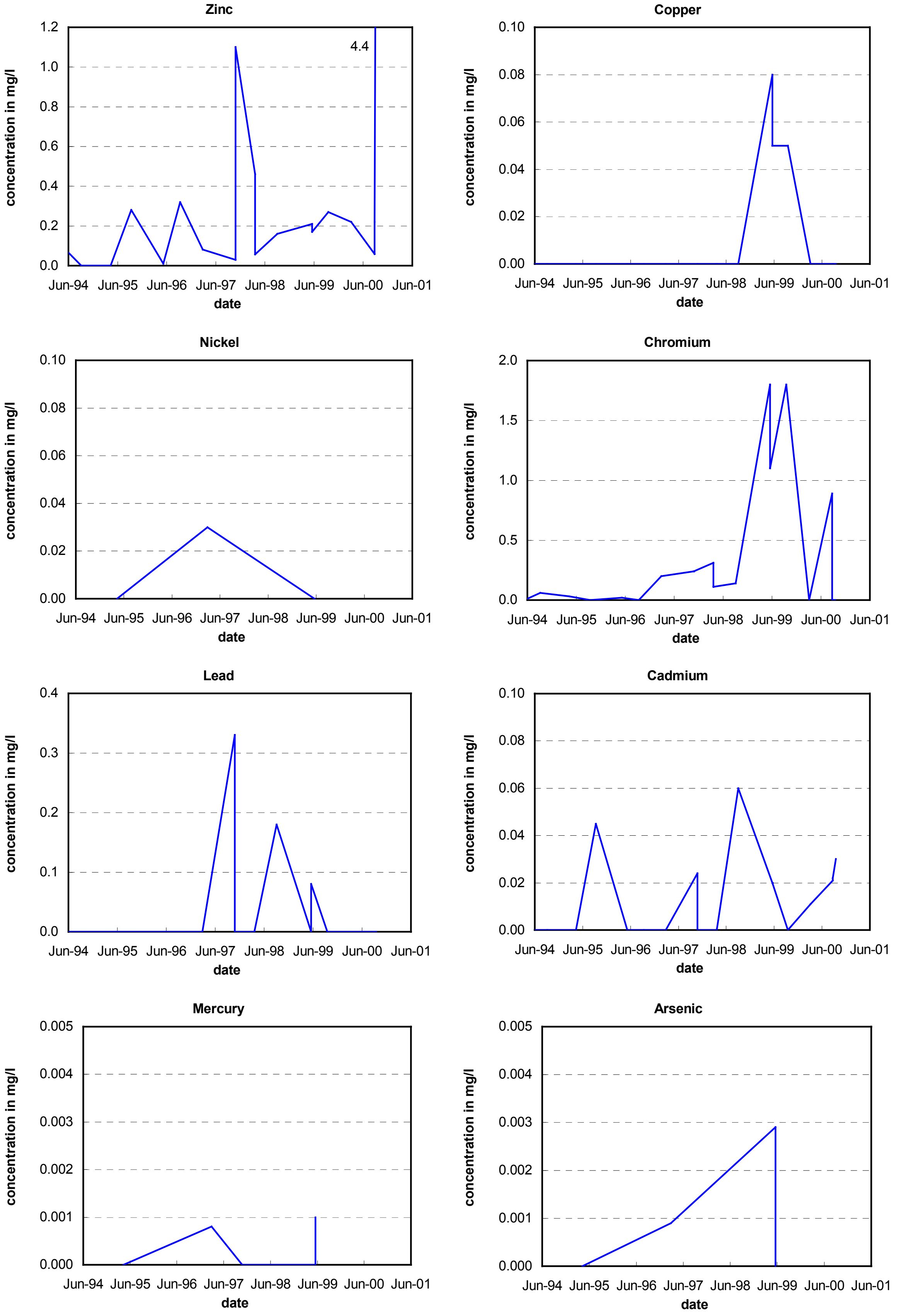


Figure 6.9 **Leachate quality from cells receiving fly ash from incineration of hazardous wastes (Site F)**

6.4 Overview of leachate from mixed hazardous waste landfills

6.4.1 Determinands of leachate quality

The UK is at the beginning of a learning curve in its understanding of hazardous waste landfills. There does not yet exist a body of knowledge on the behaviour of modern hazardous waste landfill sites, comparable to that developed in the UK for co-disposal. Biological processes, dominant in landfills receiving bioreactive wastes, are relatively unimportant in landfills for hazardous wastes. Largely inorganic chemical reactions and physical processes determine leachate quality, including:

- leaching/dissolution from solids;
- hydration, especially of free lime, CaO;
- carbonation of lime by $CO_{2;}$
- complexation by organic ligands;
- neutralisation of acids and alkalis;
- precipitation (as hydroxide, sulphide, carbonate etc.);
- adsorption;
- ion exchange; and
- filtration.

Carbonation is of particular importance, as it may often result in the pH within parts of the waste matrix changing from strongly alkaline to near-neutral. Carbonation (also referred to as carbonatisation) occurs when carbon dioxide in the atmosphere reacts with lime to form calcium carbonate, which precipitates as calcite. This causes the pH locally to fall, typically from around 11-12 to around 8, rendering some metals less soluble and others more soluble than at higher pH values. Under these conditions the pH may be fairly strongly buffered by the solid phase calcite. The extent to which carbonation occurs depends on local factors such as the lime content and density/porosity of the waste, relative humidity, and the access of air.

Covering with a low permeability cap greatly restricts access to the upper waste surface, while maintaining a leachate collection system in a flooded condition restricts the access of air to the lower layers. The greater the depth of the landfill the less the opportunity for access by atmospheric carbon dioxide. The extent of carbonation may thus be highly varied, being most advanced along preferential flow channels and the reaction may continue at a slow rate for many decades to several thousands of years (Sabbas *et al.*, 2003) within the landfill.

The quality of collected leachate may also be affected by two further factors:

- the spatial heterogeneity of the wastes, especially the variable extent of carbonation, may result in corresponding variations in leachate quality reaching the base of the landfill; and
- the wastes placed immediately above the leachate collection layers may interact with components leached from higher layers and thus have a controlling effect on leachate quality.

6.4.2 Long term aftercare liabilities from hazardous waste landfills

The hazardous waste landfills encountered in this study were generally at L/S ratios of not more than 0.3 and it is clear that they are nowhere near final storage quality, even though there may be negligible biological activity.

The time needed to flush hazardous waste landfills to reach completion may be controlled by chloride and other salts. Typical concentrations for chloride and ammoniacal nitrogen are shown in Table 6.2

below, together with typical discharge limits and the dilution that would therefore be needed to reduce the leachate concentration to a dischargeable level. [Discharge standards for an inland surface watercourse are taken as an indicator of the quality needed before active management of the leachate could be discontinued.] The comparison shows that dilution by up to two orders of magnitude may be needed for conservative components, which is slightly less than the flushing requirement for NH_4-N at MSW landfills.

Table 6.2 Dilution required for conservative contaminants

Component	Units	Typical leachate concentration	Typical discharge limit (inland watercourse)	Dilution/reduction needed
Cl	mg/l	20,000	250	80
NH_4-N	mg/l	200	10	20

There is some evidence that hazardous and other types of inorganic landfills exhibit dual porosity behaviour, similar to that for domestic and non-hazardous wastes. The complete mixing model that has been used to approximate the behaviour of non-hazardous waste landfills (Knox, 1990) may therefore be equally applicable to hazardous waste landfills. Dilution by two orders of magnitude would therefore need up to 4.6 bed volumes of flushing water, or a L/S ratio of around 2:1. However, this remains speculative and would need to be determined by long-term lysimeters trials and experimental cells containing representative mixtures of wastes.

For some wastes, the flushing time may be dictated by non-conservative components. Experimental studies have shown that high concentrations of lead can continue to leach from APC residues at L/S ratios up to 20:1. Similarly, phenols have been found to continue to leach from some filter cakes at high L/S. For these, the flushing times would be very much longer than for MSW.

6.4.3 Management of leachate collection systems at hazardous waste landfills

The waste matrix within hazardous waste landfills is generally alkaline and leachates may absorb carbon dioxide, leading to the formation of calcium carbonate scale in leachate collection systems. At two of the hazardous waste landfills in this study, where the leachate collection system was allowed to drain freely at all times, the operators had to undertake de-scaling of the main leachate collection pipes up to three times per year, at a cost of ~€30,000/annum. Other operators found that maintaining the leachate collection system in a flooded condition did not lead to scale build-up. Maintaining a saturated zone is therefore an important issue and would have the additional benefit of reducing short term fluctuations in flow and quality by providing a buffer for instantaneous percolation flows

6.4.4 Treatment and disposal of leachates from hazardous waste landfills

The high salinity of hazardous waste leachates may limit the volumes dischargeable into non-saline water bodies without the use of separation techniques such as reverse osmosis or leachate evaporation. This may have a significant influence on the selection of locations for this type of landfill.

The levels of NH_4-N found in this study are likely to require treatment. Biological nitrification, similar to that currently used for MSW/co-disposal leachates, was used as part of the process at Site C, evidently without inhibition. However, this was the most dilute of the six leachates and the effectiveness of nitrification for the stronger leachates has not yet been confirmed.

The levels of TOC would, in themselves, be within consent limits at the majority of existing UK leachate treatment plants. However, the nature of the TOC has not been characterised – not even its BOD. The organic compounds could cause problems of heavy metal mobility in the environment and difficulty meeting heavy metal discharge limits if the TOC includes ligands that are not readily degraded during biological treatment. They could therefore require the development of alternative physical-chemical treatment processes.

The elevated levels of heavy metals would need to be lowered before discharge to surface water and possibly in some cases before discharge to sewer.

The methods of treatment in use at the European sites visited for this study included:

- tankering to off-site evaporation plant;
- tankering to off-site liquid waste incinerator;
- use as process water for hazardous waste solidification; and
- site-specific combinations of various aqueous treatment processes including: coagulation, flocculation, ultrafiltration, hydrogen peroxide addition, biological treatment and reverse osmosis.

6.4.5 Impact of EU waste acceptance criteria

The European Commission issued waste acceptance criteria in December 2002. Their impact on leachate quality will not be known with any certainty until some years after they come into force, scheduled for 2005. However, many of the national criteria that controlled the inputs to the landfills in this study are similar to the EU criteria recently adopted. For many parameters such as TOC, nitrogen (which is not covered at all by the EU criteria) and heavy metals, the leachate quality data in this study are likely to be an adequate guide to what may be expected from LANDFILL DIRECTIVE-compliant landfills. A difference could occur in the levels of chloride and other dissolved solids: in most of the national criteria, these were either not controlled or were controlled at levels higher than set in the EU criteria. Nevertheless, the EU criteria allow leachable TDS of up to 100,000 mg/kg at L/S 10. These limits could easily lead to leachate TDS at similar levels to those reported in the present study. Also the EU criteria allow a relaxation to up to three times the limit, dependent upon a risk assessment, and the individual member states determine how this provision is to be implemented.

6.5 Proposed leachate source term values

The data obtained during the study indicate a wider range for hazardous wastes, than that due purely to hydraulic factors. This undoubtedly reflects a greater variation in the sources and chemical nature of the waste inputs than occurs for MSW and its treatment residues. A factor of 10 was therefore used between maximum and minimum for some parameters. An even greater range was indicated by the data for some of the heavy metals.

For cement-stabilised wastes, there were no data for many parameters and it was deemed unreliable to assume that they would be similar to mixed hazardous wastes. Cement-stabilised wastes were assumed to be less variable than mixed hazardous wastes, so a factor of 5:1 was used for the max:min ratio for most parameters, and 10:1 for the heavy metals, unless available data indicated otherwise.

Sources of variation

Sources of variation from the default values in Table 6.6 relate to the chemistry of the waste streams, and include:

- a preponderance of a single source of waste, effectively creating a mono-disposal situation;
- site-specific variations in the types of metal-bearing waste, which will influence which metals may be at elevated concentration in the leachate; and
- large inputs of unusual soluble anions such as fluoride or bromide; and
- the placement of certain wastes as the basal layer.

The leachate quality may also be variable, as a function of the flow conditions within the landfill - whether dominated by flow through the waste matrix or preferential pathways.

Table 6.3 Proposed contaminant distributions

Parameter		Mixed hazardous waste		Cement-stabilised waste	
	units	max	min	max	min
pH					
Conductivity	µS/cm	80,000	16,000	50,000	10,000
DOC	mg/l	250	25	180	35
COD	mg/l	750	50	550	110
BOD	mg/l	-	-	-	-
NH_4-N	mg/l	400	20	-	-
TKjN	mg/l	400	20	0.5	0.1
Na	mg/l	20,000	2,000	10,000	2,000
K	mg/l	6,000	600	6,000	1,500
Ca	mg/l	10,000	500	3,000	600
Mg	mg/l	300	30	-	-
Cl	mg/l	40,000	5,000	20,000	4,000
SO_4	mg/l	5,000	500	2,000	400
NO_x-N	mg/l	1,000	25	3,000	600
Br	mg/l	5,000	500	-	-
F	mg/l	5	1	-	-
Cyanide (total)	mg/l	-	-	0.5 [*1]	0.1
Alkalinity pH 4.5 as $CaCO_3$	mg/l	3,000	600	-	-
As	ug/l	600	60	600	60
Hg	ug/l	8 [*2]	0.1	-	-
Cd	ug/l	500	20	150	15
Cr	ug/l	4,000	10	-	-
Cu	ug/l	4,000	10	-	-
Pb	ug/l	1,000	50	-	-
Ni	ug/l	8,000	100	1,500	100
Zn	ug/l	15,000	100	25	2.5
Mo	ug/l	2,000	200	-	-
V	ug/l	55,000 [*3]	50	-	-
Se	ug/l	50	5	-	-
phenols	ug/l	10	1	30	6
VOCs	ug/l	50	5		

[*1] based on Site H
[*2] based on max. at Site H
[*3] based on max. at Site A

7 Non-hazardous, low organic wastes

7.1 Mixed wastes

Few data were collected on mixed non-hazardous, low organic waste landfills, and there are insufficient data to provide a meaningful overview of leachate quality or to propose source term data. This chapter discusses leachate quality at one mixed waste landfill and another that accepted vehicle fragmentiser wastes.

At Site D (see Appendix 3) mixed non-hazardous, low organic wastes have been deposited in a series of containment cells since 1985. The site is still operational. Cells that have reached final levels are capped with restoration soils. The waste inputs since operations began are shown in Figure 7.1. The 'miscellaneous inorganic sludges' comprises more than half the inputs and include small quantities (<5%) of some wastes that the site operator regards as borderline hazardous/non-hazardous, e.g. sludges from industrial production processes, residues from soil decontamination and dewatered dredging sludges. Local regulations at Site D require that all contaminated soils are cleaned before landfilling. Figure 7.1 shows that ~30% of the inputs have a significant organic content, despite the exclusion of household wastes. The average organic content of the waste mix is ~10%. Leachate from all of these cells is combined and treated on site before discharge. Results for a restricted range of parameters have been obtained for monthly analyses of the influent to the plant, from 1986 to 2000 and are shown in Figure 7.2 and 7.3 as time-series graphs. General conclusions are summarised below.

- The leachate has developed as a typical dilute/medium strength methanogenic leachate, with chloride, TKN, COD and BOD concentrations similar to those found at many UK landfills.
- Landfill gas is collected and used for combined heat and power generation, confirming that there is a significant rate of anaerobic biological activity within the site.
- The leachate exhibits marked seasonal fluctuations in strength, similar to many UK landfills, with the lowest concentrations occurring during winter.
- There has been a long-term increase in TKN. This may indicate that an increasing proportion of the organic components are degrading at increasing rates. This is similar to the gradual increase that occurs at many UK landfills as the moisture content of the wastes increases towards the optimum for degradation.
- Short-lived extreme values in concentrations of heavy metals occurred at a time in the mid 1990s when COD and BOD were also erratic. This may have been associated with acetogenic conditions at the start of operations in new cells. Apart from arsenic (1,700 μg/l) and cadmium (64 μg/l) these peak metal values were not unusually high compared with those in acetogenic leachates at UK landfills.
- At most other times, levels of most metals are very low. However, arsenic is present consistently in the range 50-100 μg/l: this is higher than would be expected at a typical UK non-hazardous waste site, or co-disposal landfill.
- PAHs are present, typically at few tens of μg/l. Most of the PAHs consist of naphthalene, acenaphthalene and fluorene.
- Extractable organohalogens (EOX) levels, at ~10-50 μg/l, are lower than typically found in MSW leachates.
- Mineral oil is present only at sub-mg/l levels, which is lower than commonly found in existing UK leachates.
- Phenols have been detected only at sub-mg/l levels only, which is no higher than commonly found in existing UK leachates.

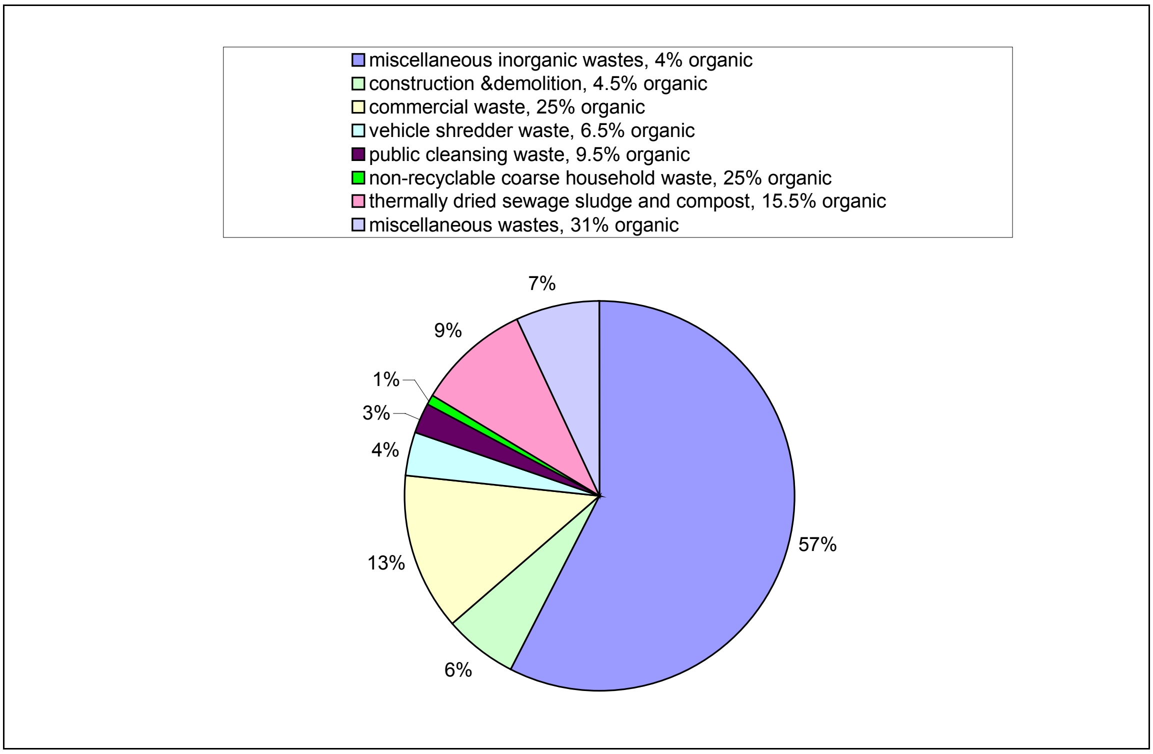


Figure 7.1 **Waste inputs at Site D, 1985-2000**

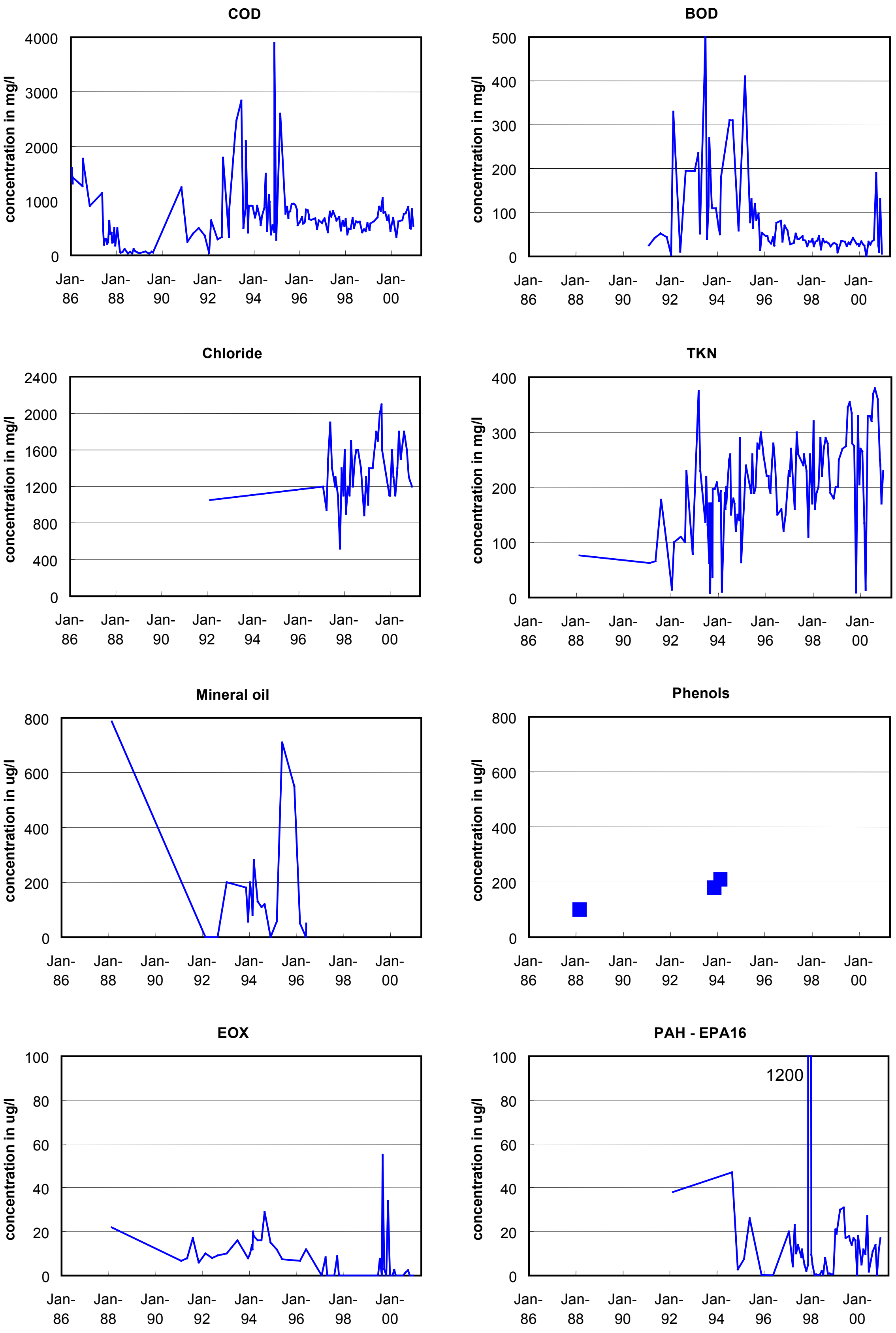


Figure 7.2 **Leachate quality data for low hazard, low organic wastes, Site D (major parameters and organics)**

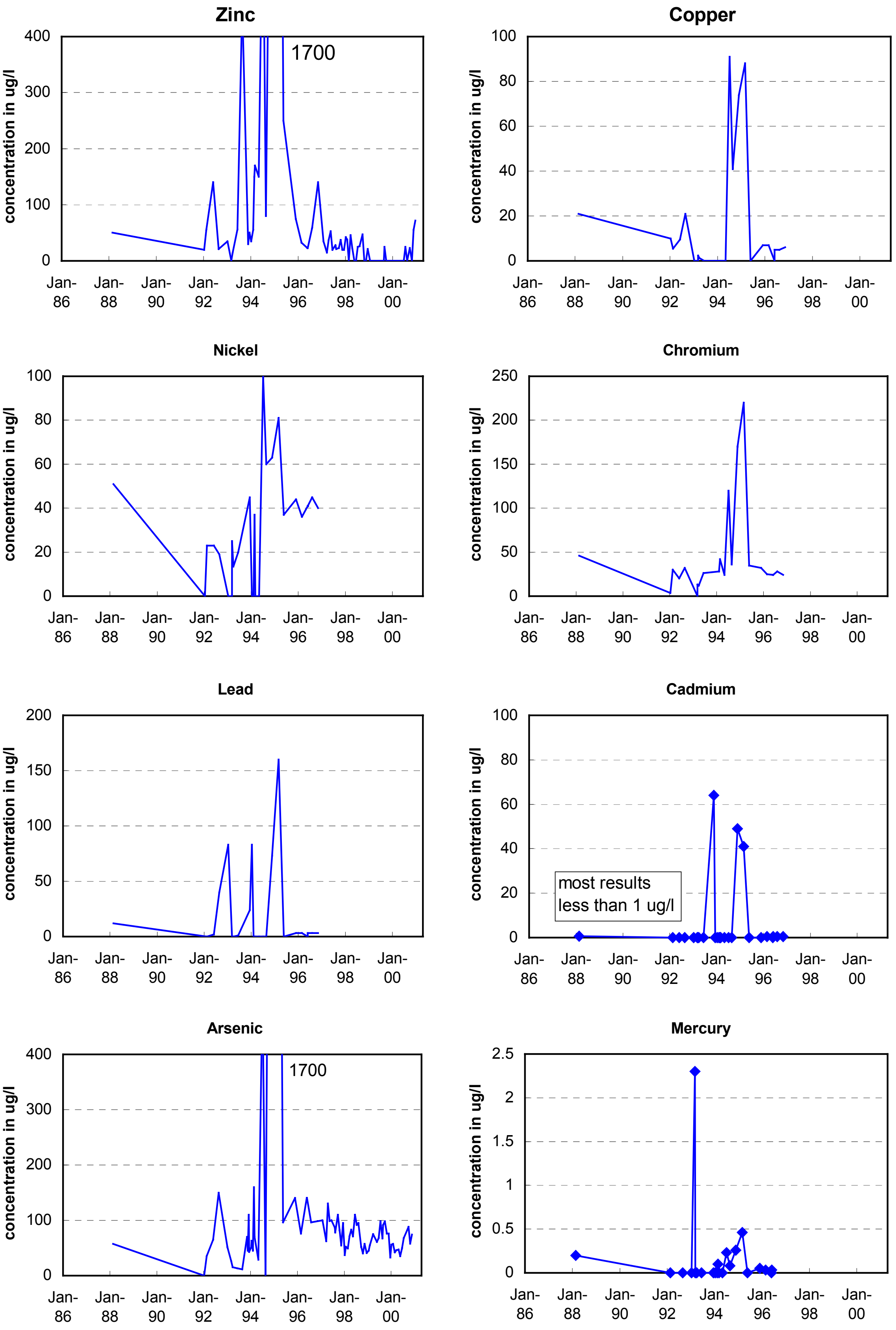


Figure 7.3 **Leachate quality data for low hazard, low organic wastes, Site D (heavy metals)**

7.1.1 Vehicle fragmentiser wastes

At Site E (see Appendix 3), vehicle fragentiser waste has been deposited in a 6 m deep cell since 1994. The cell remains operational and uncovered. The base of the cell is below the level of the local water table and the operator suspects some ingress of groundwater. Dry weather flow data for the whole landfill suggest this could be as much as 50% of the overall flow (see Appendix 3). The estimated cumulative L/S ratio for the cell receiving fragmentiser waste is ~0.15.

Although the cell has received predominantly fragmentiser waste, it has ~1 m of bottom ash from sewage sludge incineration in the base. This was placed with the objectives of protecting the leachate collection system and attenuating any metals that might leach from the fragmentiser waste.

Leachate quality data from 2-monthly sampling have been received for a limited range of parameters. Most are shown as time series graphs from 1994 to 1999 in Figure 7.4. General conclusions include:

- the pH value is near neutral;
- metals concentrations are very low;
- mercury (not graphed) was consistently below the detection limit of 0.05 µg/l;
- oil was consistently below the detection limit of 3 mg/l; and
- no other conventional analyses or trace organic analyses were undertaken.

These results, while of interest, do not include enough parameters to provide a clear impression of leachate quality from vehicle fragmentiser waste. Also, it is possible that the basal layer of ash may have affected the collected leachate quality. In view of concerns over the presence of PCBs and other contaminants in leachates from fragmentiser wastes additional data should be sought.

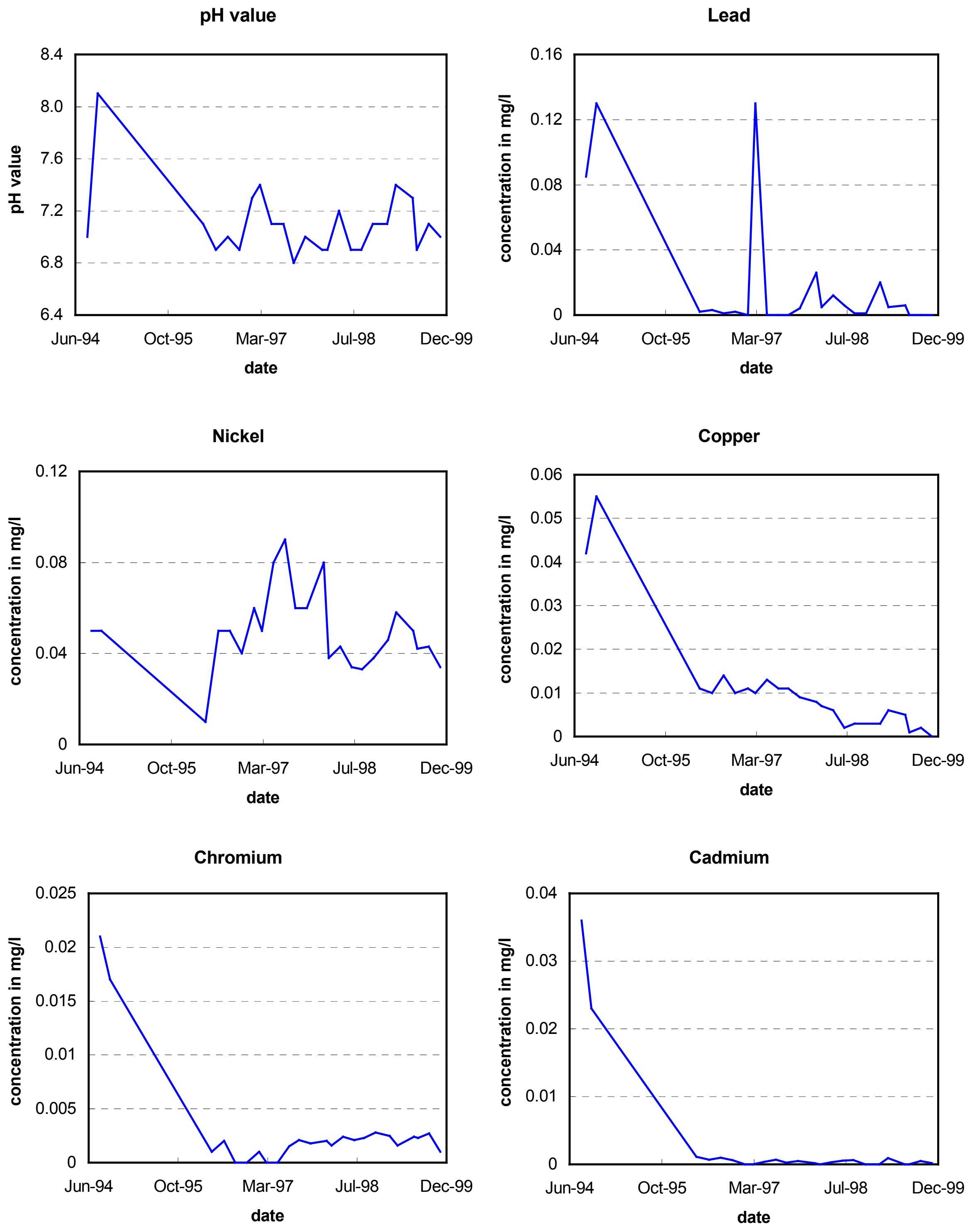


Figure 7.4 **Leachate quality from cell receiving vehicle fragmentiser waste (Site E)**

8 Discussion

8.1 Overview

The preceding chapters provide the most comprehensive review of leachate data from landfill sites in the EU that are likely to be developed in the UK following implementation of the Landfill Directive. Key issues associated with leachate quality and landfill management are discussed in the respective chapters, and research needs summarised in the following sections. Despite the large quantity of data compiled in this review, the picture remains incomplete for many aspects of leachate quality of concern in the UK. This chapter summarises the research needs from each of the preceding chapters and discusses significant knowledge gaps for the attainment of "final storage quality" within decades rather than centuries.

8.2 Residues from incineration of MSW

There is a considerable body of data on MSW incinerator residues in Europe and beyond (e.g. Hjelmar, 1996, IAWG, 1997). However, there is little information available on the long-term leaching performance of residues either in a re-use (bottom ash) or disposal scenario. Key research needs, many referred to in Chapter 2, are summarised below:

- evaluate the chemistry and mineralogy of residues; in particular the origin and composition of DOC and its impact on metals mobilisation;
- evaluate the potential for gas (e.g. Magel *et al.*, 2001) and heat (e.g. Huber, 1998) evolution, their influence on landfill processes and conditions, and liner integrity;
- evaluate the influence of weathering on the leaching performance of residues (e.g. Bodénan *et al.*, 2000, Polettini and Pomi, 2003, Mostbauer *et al.*, 2003);
- evaluate the influence of dissolution and carbonation on the mechanical properties of residues, treated residues and the stability of incinerator residue landfills; and
- measurement and modelling of landfill hydraulics.

8.3 Leachates from MBP of MSW

Chapters 3 and 4 provide an overview of leachate from MBP wastes, supported by data from full-scale commercial landfills and extensive research from Austria and Germany. Chapter 5 provides information on trace organic contaminants in MBP leachates from a small number of samples collected during this study. Key research needs for MBP leachate and landfill include:

- develop and evaluate compliance tests for indicating the long-term biodegradation potential of MBP residues;
- evaluate the biochemical transformation of nitrogen during composting and the conditions needed to optimise nitrogen removal;
- evaluate the nature and fate of "hard" COD in MBP leachates and implications for the mobility of heavy metals;
- evaluate the fate and transformation of selected trace organic contaminants during composting; and
- measurement and modelling of MBP landfill hydraulics (taking into account the mechanical properties of the waste), and assessment of optimum conditions to accelerate stabilisation.

8.4 Hazardous waste and cement-stabilized hazardous wastes

There is surprisingly little scientific literature on leachate quality from hazardous waste landfill sites, and the implications for their management. Chapter 6 provides an overview of leachate chemistry, primarily based on leachate data from six operational hazardous waste sites in the EU. Key research areas are similar to those summarised for MSWI residue landfills, and additional research needs include:

- evaluate the benefit of mixing largely inorganic hazardous waste streams to optimise landfill processes;
- evaluate the role of basal layers to buffer leachate pH before it is collected for treatment;
- evaluate the relative contribution of inorganic and organic nitrogen in mixed hazardous waste leachates;
- evaluate the TOC in mixed hazardous waste leachates and its significance for mobilisation of heavy metals; and
- develop a system to collate and interrogate waste stream characterisation data to enable a better prediction of leachate chemistry and derivation of site-specific source term data.

8.5 Derivation of kappa

LandSim 2.5 uses a probability distribution function (pdf) to define contaminant source term distribution and a kappa value to define the declining source term (see Section 1.4.1). The calculation of kappa requires the use of empirically derived values, m and c for which there are no default values for ammoniacal-N at present, and no kappa determinations for MBP wastes. There is a need to continue a rolling programme of column tests to derive kappa values for a wider range of both contaminants and types of waste.

Phase 2 of this project (Environment Agency, 2004) provides source term data for both untreated and carbonated residues. The kappa values are considerably different for untreated and carbonated residues for some of the heavy metals, as illustrated in Table 8.1 below, but similar for mobile contaminants such as Cl.

Table 8.1 Kappa values for Pb in incinerator residues (from Environment Agency, 2004)

	BA1	BA2	APC1	APC2
Untreated	0.02	-0.17	-0.21	0.08
Carbonated	-	0.17	0.19	-
$L/S1_U$ - $L/S10_C$	0.83	0.46	0.42	0.77

The use of kappa values is a relatively simple means of expressing a declining source term and it is assumed that kappa remains constant with time. Clearly the release mechanisms will change with time, as demonstrated by the difference in kappa between untreated and carbonated ash. It is interesting to note that for Pb, despite the dramatic reduction in the leachable concentrationfrom carbonated ash, the kappa value is increased in comparison to the untreated ash, indicating that the leachable Pb may be released more quickly. The effect of weathering on kappa should be researched further and the implications for achieving stabilisation assessed.

In addition, biogeochemical changes in landfills containing biodegradable waste would tend to indicate a non-constant declining source term. It is currently unclear whether constant kappa values represent an unacceptably overconservative approach. Research is needed to evaluate the influence of both weathering and biochemical transformations on the value of kappa for a range of waste streams.

8.6 Achieving "final storage quality"

A major research challenge will be to assess techniques to enable "final storage quality" of a landfill to be achieved in decades rather than centuries. This will require the development of technologies to accelerate biodegradation, washout mobile salts, and immobilise less mobile contaminants (e.g. heavy metals) such that emissions of contaminants are at such a rate that can be managed by natural attenuation without further intervention. It is therefore appropriate to look beyond the Landfill Directive at research needs to address the issues associated with the achievement of final storage quality in sustainable landfill scenarios.

This is not a new concept and "sustainable" landfill has been the subject of research and guidance (e.g. Institute of Waste Management, 1999, Brands-van den Esschert *et al*., 2003). However, the concept has not developed beyond the pilot scale. Research is needed to determine when "final storage quality" is actually achieved, laboratory tests that can be used to simulate the stabilisation process and modelling approaches to predict when the end-point will be reached. In addition, research needs are discussed below in terms of pre-treatment, landfill processes and in-landfill treatment technologies, and linked to the need for effective communication of research findings and access to data.

Where a single contaminant or small number of contaminants are driving the time to achieve final storage quality there may be merit in identifying the key waste streams responsible for most of the contaminant loading and consider options for diverting such wastes from landfill. The requirement for waste characterisation should help this process and, if the data produced is used strategically, a case may be made for change at the point of production, either by changing the production process or use of alternative materials.

8.6.1 Pre-treatment

In the EU a number of pre-treatment technologies are being developed to achieve Article 6 of the Landfill Directive requirement for treatment of waste before landfill and/or to meet waste acceptance criteria. Despite receiving such treatment (e.g. MBP of MSW, cement stabilisation of hazardous waste), the data from this project indicates that final storage quality is unlikely to be achieved in decades.

There is currently little data available to quantify the effect of pre-treatment on the stabilisation of wastes. Research is needed to identify how pre-treatment technologies can be optimised to shorten the period to reach final storage quality, both for technologies studied in this report and others for which no full-scale leachate data was found (e.g. pyrolysis, anaerobic digestion). Specific research priorities related to wastes discussed in this report include:

- understanding the fate of ammoniacal-N, residual COD and trace organic contaminants in MBP wastes subjected to varying degrees of composting;
- developing efficient methods to recover soluble salts from inorganic wastes at low L/S ratios;
- developing efficient methods to recover metals from inorganic wastes, in particular looking at opportunities for commercial recovery, e.g. platinum group and precious metals;
- developing efficient methods to immobilise and reduce the rate of release of heavy metals; and
- publication of pilot studies and demonstrations of treatment technologies (e.g. like the CL:AIRE Technology Demonstration Project Reports for remediation of contaminated soil and groundwater).

8.6.2 Chemical and biochemical landfill processes

Data collected during this project suggest that MBP processes can considerably reduce the organic strength of leachate, avoiding the acetogenic phase, and more rapidly produce leachates similar to those from MSW landfills in methanogenic phases of decomposition. However, landfills receiving MBP wastes will continue to pose risks to groundwater, and may require aftercare periods similar to conventional MSW landfills that have become methanogenic. The research priority should be to evaluate ways to optimise the MBP process, particularly with regard to reduction in residual COD and ammoniacal-N.

However, a number of researchers have studied the impact of combined disposal, e.g. of MBP waste and MSW incinerator bottom ash on the rate of biodegradation in a landfill. Boni *et al*. (2001) demonstrated a significant release of contaminants (e.g. COD, Cl, and TKN) into leachate from co-disposed MBP MSW and bottom ash from incinerated hospital waste. Van der Sloot *et al*. (2003) showed a significant increase in DOC (and as a result Cu and trace organics) when organic rich wastes are mixed with inorganic waste. Cossu *et al*. (2003) demonstrated the effect of redox conditions on leachate quality from mixed MBP and bottom ash in laboratory tests. There therefore appears to be merit in researching the optimising of landfill processes to achieve final storage quality by managed combined disposal of MBP and inorganic wastes.

Van der Sloot and others (e.g. van der Sloot *et al*., 2003, van Zomeren *et al*., 2003) have studied the mechanisms controlling leaching from mixed and treated inorganic wastes at laboratory, lysimeter and field scales. It has been demonstrated that mixed waste behaves more consistently than the heterogeneity of the material would otherwise suggest, and that leaching behaviour is controlled by the solubility of the contaminant rather than waste characteristics. There is generally good consistency at the different scales for a wide range of contaminants, suggesting that laboratory tests, such as the upflow column test, can be used to predict long-term leachate quality and hence timescales to achieve stabilisation.

However, this work does not provide sufficient data on the influence of longer-term weathering processes on leaching behaviour. A number of researchers have evaluated the influence of accelerated weathering (primarily carbonation of alkaline MSW incinerator residues) on leaching behaviour at laboratory (e.g. Polettini and Pomi, 2003) and lysimeter scale (e.g. Mostbauer *et al*., 2003). The results have shown that weathering can have a significant impact on leaching performance and reduces pH without loss of acid neutralisation capacity. Mostbauer *et al*. (2003) found good, and in some cases excellent correlation between laboratory-aged samples in leaching tests and naturally aged samples in lysimeters at the same L/S ratio.

Further research is needed to:

- extend the monitoring data available from full-scale landfill cells;
- evaluate the leaching performance of "fresh" and weathered waste and waste mixes;
- assess the importance of other key variables, in particular redox, in the prediction of leaching performance and management of landfill processes;
- further develop the concept of beneficial combined disposal of inorganic and organic wastes, both to improve geotechnical properties and to accelerate biochemical processes; and
- validate the column percolation test (with or without accelerated weathering) as an appropriate tool to estimate the time to reach final storage quality.

8.6.3 Landfill management

The new classes of landfill and requirement for pre-treatment of waste will change the management of landfill sites in the UK. It is therefore important to understand the landfill processes and design new cells accordingly. The design should include, where appropriate, measures to be taken to accelerate stabilisation of the waste.

Landfill hydraulics

A number of researchers have identified preferential pathways in landfill as a mechanism of short-circuiting waste mass in a number of landfill scenarios (e.g. Johnson *et al*., 1998, Ludwig *et al*., 2000, Maloszewski *et al*., 1995, van der Sloot *et al*., 2003). Such short-circuiting will have an adverse effect on stabilisation - prolonging the time to reach final storage quality. Mechanical properties of waste should, as far as possible, be engineered to minimise the development of preferential pathways and promote either uniform flow through the waste mass or diffusion-controlled contaminant release along engineered pathways. This may become more challenging in the future as non-hazardous waste landfills will contain higher proportions of pre-treated MSW.

For example, MBP wastes are more homogeneous and finer grained than untreated MSW, have lower biological activity, will undergo less settlement and have low gas and water permeabilities. Such properties are not ideal to actively manage and accelerate longer-term stabilisation. There may therefore be merit in researching the influence on landfill hydrology of combined disposal of MBP wastes with coarse, granular inorganic wastes.

Stegmann and Heyer (2004) have proposed other landfill design concepts for effective management of MBP wastes. One proposed option is to compact MBP waste to a high density with chemically stable coarse layers placed for drainage, aeration and gas collection. This appears to be a viable option for disposal of such wastes as settlement is likely to be minimal due to the low biological activity and high density of the waste. The other option discussed is to landfill baled MBP waste, with the space between bales utilised for passive aeration.

Leachate collection system

This study has identified variable performance of leachate collection systems (LCS) at hazardous waste landfills. Carbonation moderates the pH of leachate, accelerating the formation of calcium carbonate scale in leachate collection systems. At two of the hazardous waste landfills in this study where the LCS was allowed to drain freely at all times, the operators had to undertake de-scaling of the main leachate collection pipes up to three times per year, at considerable expense. Other operators found that maintaining the LCS in a flooded condition did not lead to scale build-up. Maintaining a saturated zone is therefore an important issue and would have the additional benefit of reducing short term fluctuations in flow and quality by providing a buffer for instantaneous percolation flows. There is a need to understand the mechanisms of scale formation, mineralogy and key geochemical processes (e.g. White *et al*., 2002), timeframe of scale formation and influence on permeability of the LCS, and effective techniques to de-scale.

One approach may be to design a buffer layer above the LCS to neutralise alkaline leachate and potentially remove heavy metals from the leachate. Van Zomeren et al. (2003) has found this at a test landfill cell for hazardous waste, cement stabilised for disposal in a non-hazardous waste cell. The life time of the 1 m thick slightly contaminated soil buffer layer has been modelled based on its density, (alkali) neutralisation capacity and variable rate of infiltration to remain effective for 50 to 450 years. Although the effect of precipitation on the permeability of the layer was not modelled, it is clear that the inclusion of a buffer layer shows great potential in moderating the leachate quality in terms of pH and heavy metal concentration.

In-situ flushing and aeration

In the UK there has been traditional scepticism toward accelerated stabilisation of waste using flushing techniques due to preferential pathways and the volume of water required. In the past flushing, or leachate recirculation trials have been retrofitted to conventional MSW or co-disposal landfills and yielded ambiguous results; most likely due to poor flow distribution and differential settlement.

Brands-van den Esschert *et al*. (2003) describe a pilot sustainable landfill trial for MSW and industrial wastes with a recirculation system comprising two perpendicular distribution systems at different levels in the waste. The wastes were characterised, pre-treated (e.g. shredded) where necessary and properly mixed to achieve a relatively homogeneous, open structure to reduce the development of preferential flow paths. Preliminary results indicate that the pilot is performing to expectations, but further monitoring is required to evaluate whether the predicted degradation rate can be maintained to achieve final storage quality within 30 years. The development of effective flushing and leachate recirculation techniques, if to be used in the UK, will need to be supported by research and good practice guidance on the design and management of flushing landfills.

In-situ aeration is a technique that is well-developed for the remediation of hydrocarbon-contaminated soil (bioventing) and is being increasingly applied to stabilise old landfill sites, addressing issues such as odour, settlement, landfill gas generation and migration, and leachate quality. A significant number of papers were presented at Sardinia 2003 - a testament to the development of the technology, particularly in The Netherlands and Germany. However, it is a comparatively young technology for in-situ waste stabilisation with little current use in the UK. The design of any aeration systems for new landfill cells, as opposed to retrofitting to closed landfills, will need to take into account the geotechnical properties of the waste - a further reason to asses the benefits of combined disposal.

In-situ leachate treatment

A number of researchers are examining novel methods for the in-situ treatment of leachate to reduce the cost of ex-situ treatment and/or reduce the environmental risks in the event of liner failure. The preliminary findings of van Zomeren *et al.* (2003) on the performance of a buffer layer of soil are discussed above. Another concept subject to recent research is that of an low permeability attenuating lining system constructed from mixed industrial wastes (see www.miro.co.uk for further information). The composite barriers are designed to ultimately fail, but in a predictable way and with benign consequences. This project is in its second phase to evaluate the performance of a large-scale trial.

Other leachate treatment technologies could also be applied in a landfill scenario. For example, leachate collection sumps could be designed to contain reactive materials to pre-treat leachate prior to recirculation, ex-situ treatment or discharge. Combined with hydraulic control, such systems could be designed as easily maintained, almost closed systems with little above ground infrastructure.

Landfill management - summary of research needs

Further research is needed to:

- evaluate the influence of preferential pathways on leachate quality and the stabilisation of waste for different landfill scenarios and to provide recommendations for waste placement and leachate/liquid recirculation to promote more uniform flow conditions;
- evaluate the hydraulic and leaching performance of combined disposal of MBP and inorganic wastes;
- evaluate the performance and remediation of leachate collection systems for different landfill scenarios;
- evaluate the performance of a buffer layer between the LCS and waste;
- develop the concept of novel, reactive, low permeability lining systems; and
- evaluate technologies for in-situ treatment and develop research proposals to understand the treatment processes and key factors relevant to their optimisation.

8.6.4 Data management

At a time of rapidly changing practice it is important for new technologies and information to be accessible to the waste management industry, consultants and regulators alike. In view of the need to collect waste characterisation data and the historical collection of leachate data, a substantial body of data is potentially available. An open exchange of such information within the European Union may assist member states in meeting Landfill Directive targets. Indeed, Hjelmar and van der Sloot (2003) state that "*The advantages of concerted and co-ordinated efforts are obvious, and it is particularly recommended, that European-wide databases on waste characteristics and landfill leachate characteristics are established. In view of the difficulties experienced in getting EU funding for such activities, it is recommended that such databases are set up in co-operation between interested parties in the individual European countries, including the national environmental authorities*." Specific recommendations include (from Hjelmar and van der Sloot, 2003):

- to determine which information is essential in terms of testing and monitoring (minimum requirements), and gather such data on a continuous basis;
- to encourage the production of such data;
- to ensure they are published under conditions acceptable to industry and operators (anonymise single data sets);
- to exchange data and data interpretation information;
- to make the data publicly available and useful, e.g. in databases coupled to expert systems; and
- to educate the public, authorities and politicians.

A similar case could be made for co-ordinating data on treatment technologies across the EU through the compilation and publication of technology research and demonstration projects.

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Glossary of terms and acronyms

AOX	Adsorbable organo halogens. This is a blanket parameter, used in many European countries in discharge consents. See also EOX.
APC residues	Solid wastes arising from air pollution control processes used to remove pollutants (e.g. HCl, SO_2) from incinerator flue gases.
Big bags	Woven plastic bags, ~$1.3m^3$ volume, commonly used for the transport of fine grained materials.
DOC	Dissolved organic carbon. Similar to TOC and NVOC.
EOX	Extractable organo halogens. Based on a solvent extraction procedure, this parameter is used in some countries in preference to AOX, which is based on adsorption by activated carbon.
Fly ash	Fine particulate matter carried over from the combustion chamber during waste incineration; particle sizes typically in the range 1 to 500μm.
GPR_{90}	Gas production rate – total in 90 days
L/S ratio	Liquid:solid ratio. In a landfill, the ratio of the cumulative volume of leachate that has passed through the site (or cell) to the mass of waste contained in the site (or cell).
LoI	Loss on ignition
MBP	Mechanical-biological pre-treatment
MCPP	Mecoprop
MRF	Materials recovery facility
MSOR	Mechanically separated organic residue
MSW	Municipal solid waste
NVOC	Non-volatile organic carbon. Roughly equivalent to TOC and DOC.
RDF	Refuse derived fuel
Residual waste	Something of an imprecise term, generally referred to as what remains after having removed all recyclable waste fractions. It should be used only with further explanation and definition in specific situations, but this is not always possible.
TDS	Total dissolved solids
TOC	Total organic carbon - in most waste waters this will be similar to NVOC and DOC.
TS	Total solids

Appendix 1: Austria - background information

The Austrian political and legislative background
Austria has a population of about 8 million people, and has practised waste composting on a significant scale for about 30 years. The city of Vienna has been at the heart of advances, with separate biowaste collections from households for half of this period. This collection scheme probably represents the largest and most progressive such scheme in Europe at present (Konrad, 1997). Pilot tests for a separate biowaste collection system began in 1986, and have since been expanded, with regular intensive experiments in householder behaviour (e.g. Goldshmid and Hauer, 1997).

Quantities of biowaste being collected in Vienna rose from 66,000 tonnes in 1993, to 96,000 tonnes in 1994, and have continued to rise. By 1995, two large plants were already in operation, one occupying 5 hectares and treating 100,000 tonnes of biowaste per year, the other with a capacity of 30,000 tonnes per year. Each used a natural ventilation system with periodic turning.

A major reason why composting of separately-collected biowaste increased so rapidly, was that the Austrian Waste Management Law was amended in 1992 (effective from 1 January 1995) to make the collection and composting of organic wastes obligatory. On a regional level, procedures for compliance had already been established in several provinces. By the mid-late 1990s one million tonnes of household biowaste was being collected each year, in addition to another million tonnes of garden waste, and green wastes from churchyards, parks and urban areas. Most of this was being composted in various ways.

By 1994, about 300 biowaste composting plants with a capacity of more than 50 tonnes per year, were already working in Austria. In addition, there were 200 composting plants dealing with purely green wastes. In total, over 415,000 tonnes of wastes were being composted, with 220,000 tonnes of low emission compost produced annually (Binner and Raninger, 1995).

Latest Austrian waste statistics, published in July 2001 (Environment Daily, 10 July 2001), report data for 1999, when of 3.1 million tonnes of household wastes, about half was being separately collected. Only 43% of household waste was landfilled (cf. 55% in 1993), but in recognising an inevitable trend towards household waste incineration, the Austrian Environment Agency identified a 400,000 tonne per year deficit in capacity (Perz, 2001).

A total of 526 biological treatment plants currently process 1.1 million tonnes of separately-collected biowaste each year. A further 12 plants process 0.39 million tonnes of residual wastes annually. A major feature of the Austrian system is that many of these MBP composting plants are relatively small. Biowastes are often composted locally, in very small plants, with typical throughputs in the range 1000 – 3000 tonnes per annum. In many rural areas, a far greater total quantity of biowaste is composted in MBP plants handling <2000 tpa, than in larger facilities (Szlezak, 1997). This is likely to change, as rules for air emissions from MBP plants are increasing rapidly, following the German example (see Appendix 2). It appears certain that biowastes will either require to be transported greater distances (to 50km) for treatment in larger plants, or that waste incineration will become a more important pre-treatment option, to comply with EU legislation.

Characteristics of composted wastes
The Austrian Landfill Directive (1997) set standards for design and construction of landfill sites, but equally importantly, specified characteristics of wastes that are allowed to be landfilled. The limits that wastes have to meet include maximum organic content (TOC >5% or Volatile Solids <8%), which cannot be met by MBP. The intention of these limits is to prevent emissions of “greenhouse gases” as waste decompose. However, the TOC also contains carbon such as plastics etc, which are not degradable by micro-organisms.

The Austrian Directive allows the disposal of MBP wastes in a so-called “Massenabfalldeponie”, if their calorific value is less than 6,000 KJ/kg dry substance (Mostbauer and Schneider, 1997). This parameter may be more easily measurable than TOC, but nevertheless is still unable to describe the reactivity of the wastes, and their emission potential, when landfilled.

Extensive practical trials have therefore been carried out on behalf of the Austrian Environment Ministry (Binner *et. al,* 1997a) to investigate reactivity of MBP residual wastes, derived from 4 full-scale plants. As in most countries, the output of individual MBP plants in Austria is extremely variable (e.g. maximum particle size from 20 mm to 100 mm; duration of treatment from 3 weeks to more than 5 months; natural aeration in static pile systems or forced aeration in-vessel systems). A simple technique for representative sampling therefore had to be developed first.

Alternative parameters for characterising the biological reactivity of wastes, before and after pre-treatment, needed to be determined. The focus of such tests is not on leachate contaminant potential (although this is obviously related closely to biological reactivity of the materials), but is more commonly measured as their gas generation potential, in a standard incubation test (after Binner, 1996a).

These type of tests provide a very clear indication of the extent of improvement effected by the various MBP processes, and by their duration in specific instances. Because leaching data for MBP wastes are relatively rare, a standard incubation test is often a starting point for description of wastes (given that it is widely recognised that often-applied, short-term “shake tests” on degradable waste materials are of only very limited value, since they exclude potential for products of biological decomposition processes).

Figure A1.1 below shows the cumulative gas production of MSW, determined in a typical incubation test, after different forms of MBP. Such trials have shown promising results for describing the effects of MBP of MSW in a helpful way, albeit in terms of net gas production. Non-pretreated residual waste produced about 64 litres of gas per kg dry solids (l/kg DS). In contrast, pretreated MSW produced only 5 l/kg DS when treated for 6 months by windrow composting (similar to wastes exhumed from a 10 year old MSW landfill), or 2 l/kg DS when treated for 10 weeks in forced aeration windrows, followed by 20 weeks of less-intensive maturation.

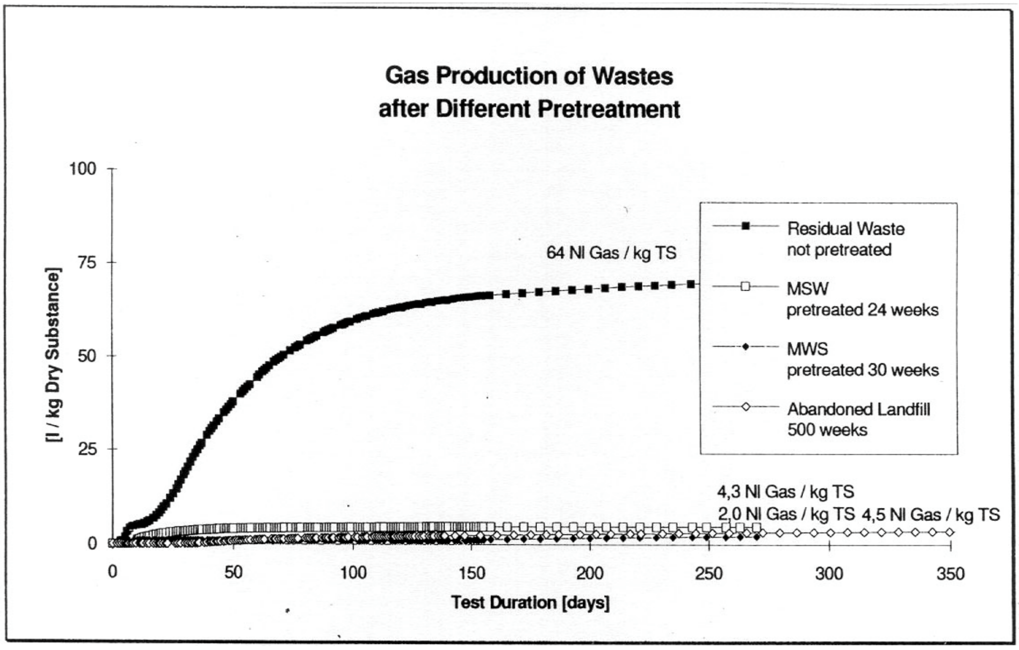


Figure A1.1 **Gas production potential of different pretreated wastes, determined in an incubation test (Binner *et al.* 1997a)**

The authors also looked at gas production of residual wastes, after various MBP treatments, at four full-scale plants in Austria (see Figure A1.2 below).

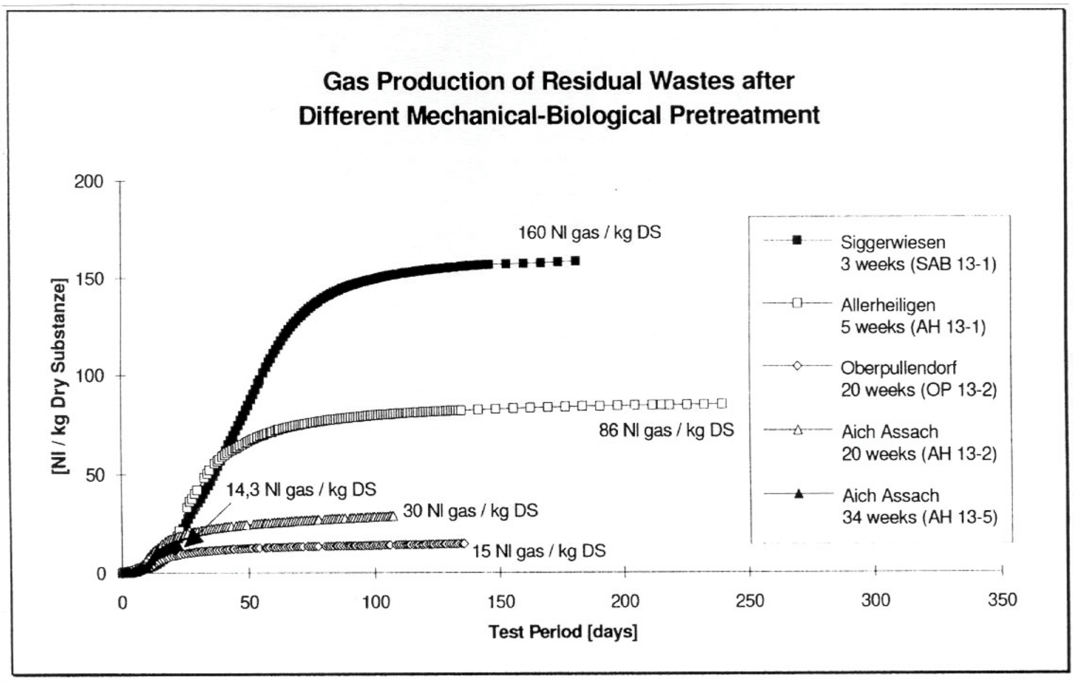


Figure A1.2 **Gas production potential of MBP wastes from 4 Austrian plants, determined in incubation tests (Binner *et al.*, 1997a)**

All samples were sieved to a standard particle size of <20mm, to ensure comparability. A parallel series of tests was also undertaken, over a shorter timescale (see Figure A1.3), which instead examined respiration activity (i.e. oxygen consumption rate) during the first 10 days of the composting process. Although the authors concluded that materials should be tested in triplicate in such trials, to overcome natural variability, nevertheless results showed great promise that within a 10-day trial, the extent of reduction of an MBP wastes, pollution potential can be assessed with some confidence. Respiration activity is normally tested by a 4-day Sapromat-test, but results demonstrated that this may be too short (Widerin, 1996) producing much higher standard deviations than for the 7 or 10-day tests. To get best results, it was recommended that respiration activity should be calculated during the period from 4-10 days, after an initial lag period, when maximum activity has started.

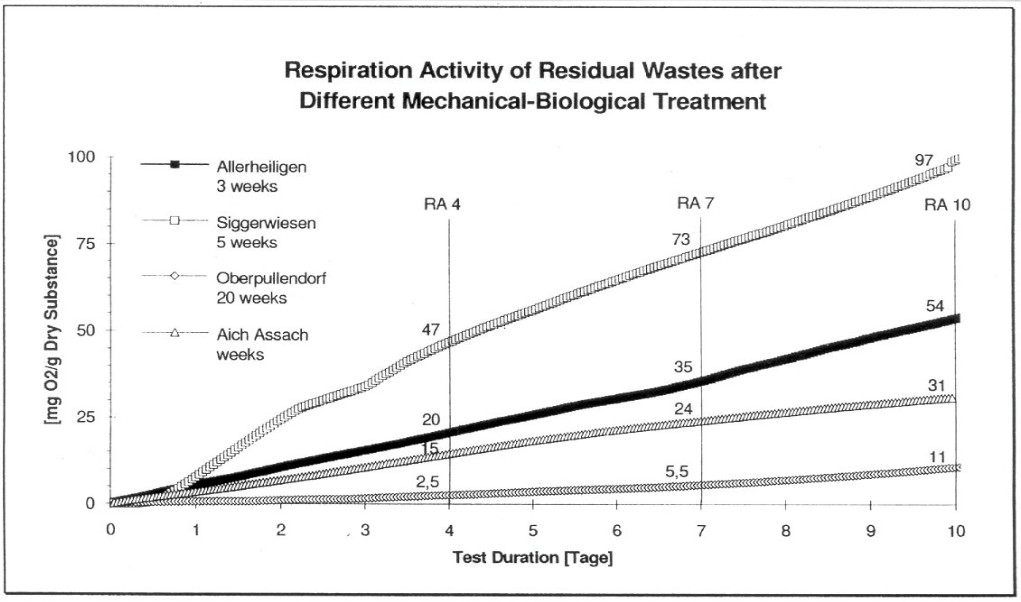


Figure A1.3 **Respiration activity of residual wastes after different MBP treatments, at different full-scale plants in Austria (Binner *et al.*, 1997)**

Nevertheless, a note of caution is appropriate. Obviously there are reasons why landfilled wastes take many decades, or even centuries, to decompose fully, whereas incubation tests may appear to reach a steady-state, with minimal further gas emissions within a matter of weeks. Non-optimal conditions within full-scale landfills are certainly one reason – another may well be toxic effects that can be enhanced within the low liquid:solid ratios typical within full-scale landfills. Work by Zach *et al.* (2000) has demonstrated how toxic effects within an incubation test can result in significant lag effects (see Figure A1.4).

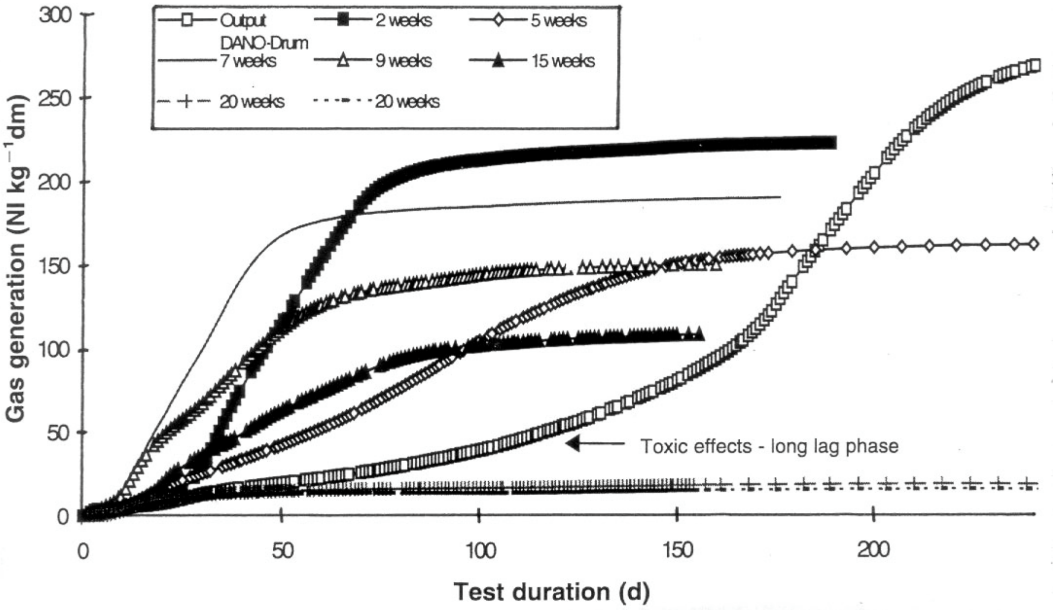


Figure A1.4 Gas generation in incubation tests for MSW subjected to different durations of MBP – influence of toxic effects (Zach *et al.*, 2000)

Such effects, in individual cases, can only be determined and observed by comparison of 4 day and 7 day respiration tests (e.g. see Figure A1.5).

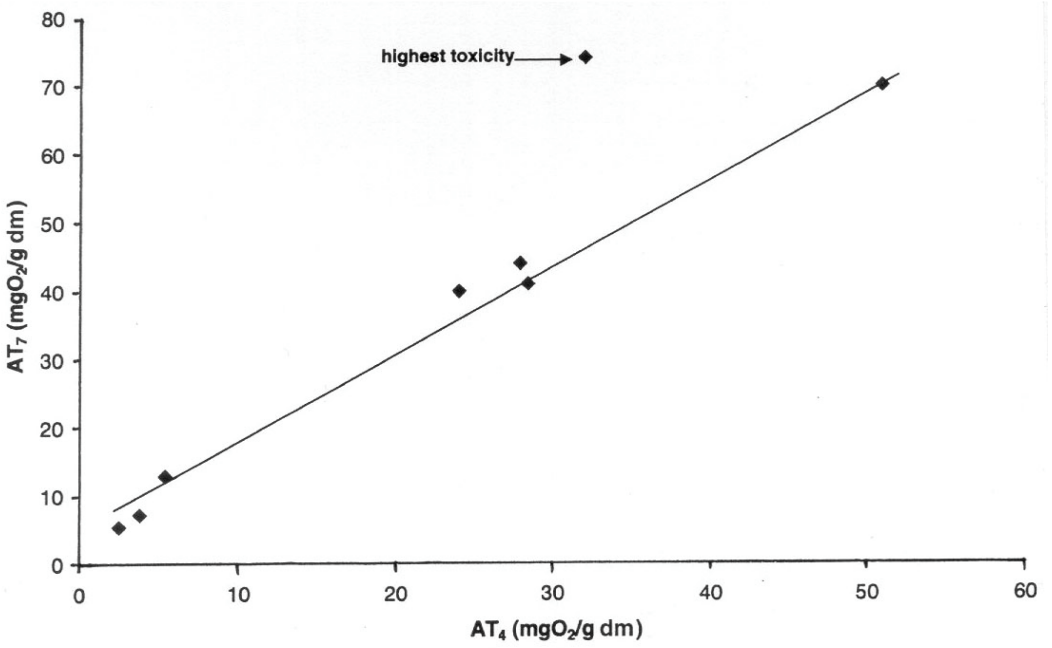


Figure A1.5 Relationship between results from 4 day and 7 day respiration tests (AT_4 vs AT_7) for samples of MSW subjected to MBP (Zach *et al.*, 2000)

Overall, the research project has demonstrated the promise of both gas production-based tests (although these typically take 100 – 200 days to complete) and respiration activity tests, for assessing the extent of pollution potential reduction by MBP of residual wastes from MSW pre-treatment. Although not directly related to release of specific contaminants should these MBP materials subsequently be landfilled, nevertheless such trials give a useful indication, and if relationships can be drawn between the two characteristics, may prove to be extremely valuable.

Biological parameters, such as incubation tests and respiration rate tests, which measure the biological reactivity of wastes subjected to MBP, widely reduce to very low levels after 20 weeks of efficient composting. Optimised MBP can reduce overall emissions of gas, from subsequent landfilling of such wastes, by more than 90 per cent, compared to untreated wastes.

The organic matter content of residual solid wastes remains very high, but the issue is to what extent this comprises reactive materials, and extensive research continues in this field in Austria. Many workers (e.g. Völker, 1991; Lepom and Henschel, 1993) recognise that chemical parameters such as ignition loss, TOC, or even calorific value, are not appropriate to predict the emission behaviour (either gaseous or in leachate) of landfilled MBP wastes. There is a strong need for additional biological test methods such as the incubation test and the respiration activity test, as well as for long-term monitoring results at full-scale MBP waste landfill sites, against which to assess these.

Heavy metals in biowastes and composts
Table A1.1 below presents the strict limits for heavy metal contents of composts, that are presently applied in Austria (Önorm, 1993). Two quality standards are specified, based on maximum levels of individual heavy metals. For compost achieving Quality I, use is not restricted, but at Compost Quality II, heavy metal concerns restrict allowable rates of application. Specified heavy metal concentrations are based on parts per million dry solids.

Table A1.1 Limits of heavy metal concentrations for application of Austrian composts (expressed as ppm dry solids)

	Quality I	Quality II
cadmium	0.7	1
chromium	70	70
copper	70	100
mercury	0.7	1
nickel	42	60
lead	70	150
zinc	210	400

In general, these limits provide relatively few constraints for composts derived from the small, rural composting schemes, but are of more concern for composts derived from the larger, urban pre-treatment plants.

For example, the large Vienna MBP plants produce composts that are almost entirely used within the city horticultural departments. In spite of the fact that biowastes being composted have been collected separately, nevertheless, some heavy metals (lead, zinc, chromium and copper) have been shown to often be present at levels which exceed limits for Class I compost, which may therefore compromise their continued use, (Goldschmit and Hauer, 1997).

The Austrian Institute of Ecology (Konrad, 1997) was commissioned to carry out a study to locate the major waste sources for these various heavy metals, and to evaluate their impact. There are four different categories of pollutant source in the Vienna biowaste system:

- green wastes from gardens and road embankments, and kitchen wastes;
- “biowaste-accompanying materials”, which many users do not know whether or not to put into the biowaste container – such as newsprint and other papers, and ashes;

- other polluting materials, such as plastic bags and packaging; and
- by-products from biowaste processing, including abrasion products as well as materials released during the composting process.

Empirical data for these categories of materials were obtained from literature, and were apportioned to the fractions found in Vienna. The following figures show the results of this calculation. Because of seasonal changes in the distribution of fractions, (e.g. no ashes in summer) separate analyses were made for each of the summer and autumn seasons.

Results (see Figures A1.6 and A1.7) show that, partly because of their large volumes, and especially during summer months, green wastes from road verges and gardens have a high source potential with regard to all examined heavy metals. For example, 60 percent of the total zinc potential is contained here. The same is true of kitchen wastes, although to a lesser extent. Kitchen waste contains a significant proportion of vegetable and fruit peels, which also can contain relatively high heavy metal concentrations – demonstrating that even “pure” biowaste material may be contaminated with heavy metals. Plastic bags can contribute lead and chromium (although the extent to which these might leach is questionable), and during the autumn season ashes become a major source potential for heavy metals – indicating that where possible they should be excluded from the biowaste stream.

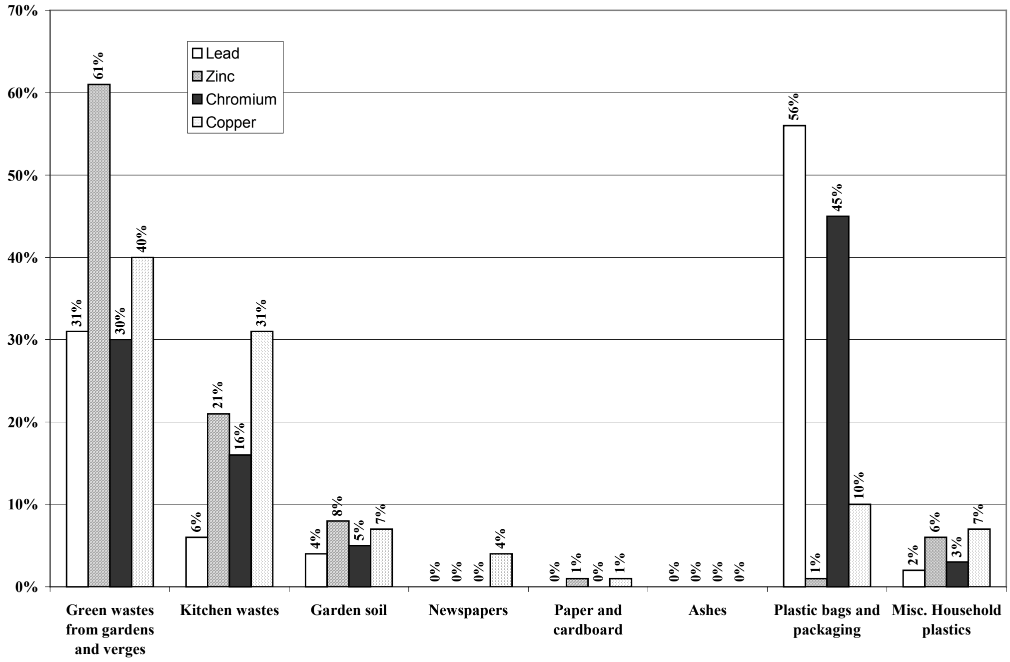


Figure A1.6 **Heavy metal concentrations in various categories of source-separated biowastes, collected in Vienna during summer months**

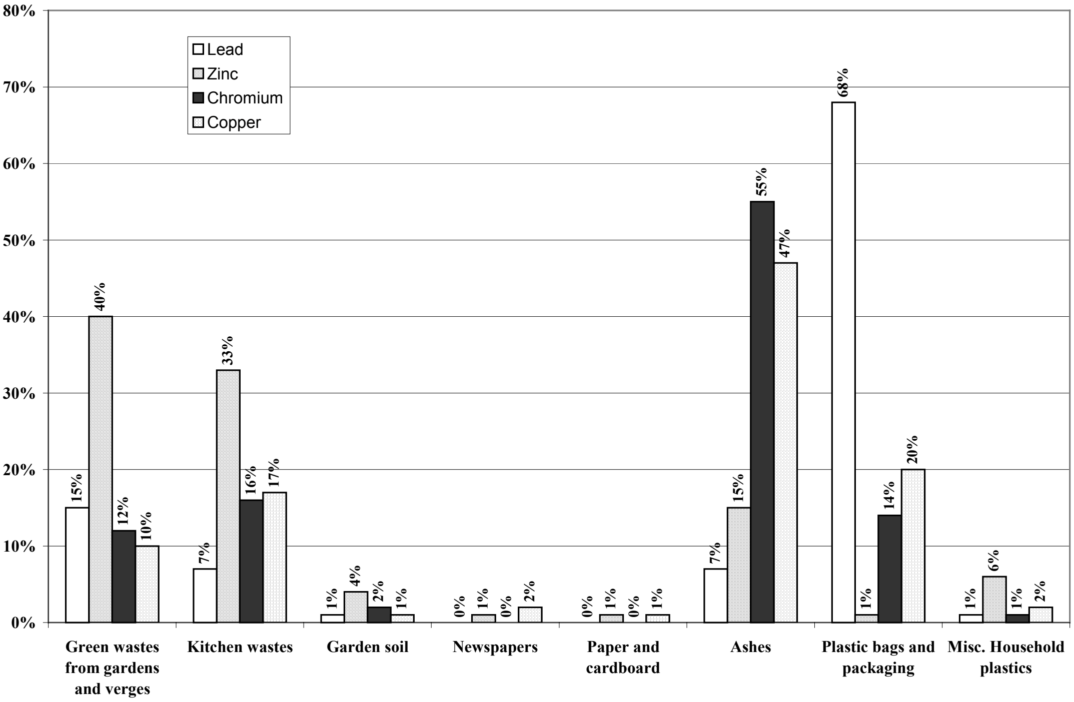


Figure A1.7 Heavy metal concentrations in various categories of source-separated biowastes, collected in Vienna during autumn months

Appendix 2: Germany - background information

The German political and legislative background
In Germany, the management of wastes is governed by the “Kreislaufwirtschaftsgesetz”, (or Closed Cycle Economy and Waste Act), which gives priority to waste avoidance. Wastes that cannot be avoided must be recycled or reused where possible, so long as this is environmentally beneficial (Soyez *et al.*, 1999). Residual wastes must be pretreated before landfill disposal, in order to reduce negative effects of emissions from their decomposition, and of landfill settlement.

This has been defined within the TASi (“Technische Anleitung Siedlungsabfall” – Technical Instructions on Waste from Human Settlements, Anonymous, 1993), which defines landfill standards and minimum standards for pretreated wastes, in a prescriptive manner. Only wastes with a volatile solid content of less than 5 percent can be landfilled, with the practical result that only incinerated wastes can meet this standard. Consequently, mechanical-biological waste pre-treatment plants and processes are only accepted as “experimental plants”. Nevertheless, in Germany over one million tonnes of residual waste are presently pretreated by MBP, in about 20 plants of various sizes, including large scale facilities, and new plants are currently being developed.

This situation is clearly not sensible, and many in Germany acknowledge that MBP processes can have a valuable role, technically, economically and environmentally, within integrated waste management systems. To answer the many questions arising, including those related to behaviour of pretreated wastes in landfills, in 1995 the German Federal Ministry of Education and Research (BMBF) launched a major research programme, which has involved 18 research projects in this field (Soyez *et al.*, 1997). The research programme has also co-ordinated and publicised work being carried out by various groups of researchers across the country.

One area of research has focussed on the physical behaviour of MBP wastes in landfill sites. It has been widely found that, although these materials can achieve landfilled densities as high as 1.5 tonnes per cubic metre (fresh weight), this results in extremely low hydraulic conductivity values, in the order of 1×10^{-8} m/sec (Scheelhaase *et al.*, 2000; Dach and Tiebel-Pahlke, 2000; Scheelhaase and Bidlingmaier, 1999; Von Felde and Doedens, 1997), or in the range 5×10^{-9} to 1×10^{-10} m/sec (Horing *et al.*, 1999), in spite of beneficial properties in terms of much reduced settlement of the pretreated wastes (Bidlingmaier and Scheelhaase, 1997). Other work has concentrated on attempts to correlate emission potential of landfilled MBP wastes with the results of short-term leaching and fermentation tests, with some success (Bidlingmaier and Scheelhaase, 1999). A large body of research has considered emissions from MBP wastes, including contaminants in leachates.

However, German legislation has continued to progress. The 1993 regulations divided waste into 3 classes:

Class 1 = “inert” wastes;
Class 2 = municipal solid wastes; and
Class 3 = hazardous wastes.

During February 2001, new national landfill regulations divided MSW into two classes, based essentially on weight loss on ignition (“glühverlust”), on a dry weight basis, and specified in Appendices 1 and 2 to the report (Anonymous, 2001). Primary legislation also allows the TOC of solid wastes to be used as a test – wastes being milled, burned and CO_2 emissions measured – but this is widely seen as inappropriate, since it does not necessarily relate to emission potential of landfilled MBP waste. It is apparent that such a test includes CO_2 generated by burning non-biodegradable plastics, and that such a test would not allow HDPE liner materials into landfills. A further alternative law is more favourable towards MBP wastes – under S6, measurement of “biological oxygen consumption”, over a 4-day period in a specified test vessel can also be used, and gas production during the test is also specified. Between 12-16 weeks of good composting is required for MBP residual wastes to meet the specified standards.

The 2001 national laws will ban landfill of untreated MSW or residual wastes after 2005, to allow some time for the German regions to achieve compliance. It is acknowledged that the laws may require subsequent modification, in order to comply with EU regulations.

A major factor affecting the MBP of residual MSW, which is rapidly becoming more important, is the control of air emissions from compost plants. Much work is being done at the University of Hannover on this subject (e.g. a PhD thesis is presently being completed by C. Cuhls). Measured data have been obtained from real, full-scale composting plants, for a wide range of components – many of which are actually generated during the composting process. As a result of this work, legislation is now being implemented specifically to control air emissions from compost and other MBP plants, and this is diverting much research into this area (see Figure A2.1 below).

Limits within the legislation are as follows:

Organic Carbon:
- <55g per tonne of input wastes
- <20 $\mu g/m^3$ air (average reading per day)
- <40 $\mu g/m^3$ air (half hour average)

N_2O:
- <100g per tonne of input wastes

Odour:
- <500 odour units (sometimes <300)

Biofilters are unable to achieve these standards, so thermal treatment of air emissions is necessary – this, of course, makes it advantageous to minimise use of air/oxygen during the composting process, thereby reducing volumes of off-gas needing treatment.

It also means that milling of wastes to 10mm, with pre-composting, will not be adequate to achieve the new air quality regulations. There will be a focus on “in-container” composting, and wastes will need 4-6 weeks of this, before secondary outdoor composting in windrows can be contemplated, to achieve the specified limit of 20mg oxygen consumption per gramme dry solids in the 4-day test.

Legislation in Germany is therefore driving current MBP research in the following direction:

- work to minimise air use in composting (to minimise air emissions requiring treatment);
- work to look at efficiency of air cleaning/treatment systems for MBP emissions, to comply with existing and coming regulations.

The pilot scale facility at the University of Hannover (Figure A2.1) comprises 8 vessels, 24 litres each, of stainless steel, each individually temperature-controllable, and sealable. The research objective is to reduce the fresh air requirements of the composting process, by having continuous monitoring of CO_2, CH_4 and O_2 in the atmosphere of each reactor, which triggers inputs of fresh air, as oxygen levels fall to a pre-set concentration. The intention is to reduce composting costs, by reducing off-gas volumes and treatment costs, and recirculating air wherever possible. Moisture content within the reactors can also be closely controlled.

Figure A2.1 **P**ilot-scale experimental composting facility, nearing completion at the University of Hannover.

In Germany, not only political pressure, but also financial pressure, is being applied to the concept of MBP of residual wastes. Prices of MSW incineration were as high as €350 per tonne, but are reducing rapidly. A recent state-of-the-art MSW incinerator was constructed in Hamburg in 1998/99, and achieves a price of DM240 per tonne – with most ash being used in road construction (APC residues going to deep mine disposal).

Such costs compare with typical recent prices in the order of say €60 to 70 per tonne for an MBP plant (size about 80,000 tpa) to include plastic separation etc. Given that preparation of a new landfill to meet current legislation may cost an additional €60 to 75 per tonne, total costs are similar to incineration. In practice, MBP is seen as the best option at present at existing landfills, where investment has already been made in engineering and preparing the site – however, where development of a new landfill would be involved, incineration is increasingly being preferred on cost grounds.

Figure A2.2 Detailed laboratory studies into biological pre-treatment of MSW, at the Technical University of Hamburg-Harburg, Spring 2001

Nevertheless, extensive sound research work has been completed on behaviour of MBP systems for residual wastes, using both detailed and sophisticated laboratory-scale systems (see Figure A2.2).

Appendix 3: Description of sites visited during study

SITE A

Description of site

Site A serves a region of ~2.5 million people. The region includes a large city whose industries include motor vehicle manufacture and metallurgical processing. The landfill has been the predominant disposal facility for the majority of landfillable hazardous wastes produced in the region. The landfill has been in operation since 1985 and has been constructed and filled in two phases:

Phase I	Landfilled 1985-89; now restored.
Phase II	1989-present; still operational, partly restored and containing wastes that meet stricter acceptance criteria than the earlier phase.

The landfill has a low permeability engineered base, a leachate drainage layer, and collection pipes. It is sub-divided into cells, each with its own leachate collection pipe, laid to a fall of 3%. The leachate collection pipes from each cell flow into a common main, which then flows by gravity through the side wall of the landfill to a leachate management facility. The leachate pipes are not kept in a flooded condition but are allowed to drain freely, part full. Previously treated by reverse osmosis, the leachate is currently tankered off-site for treatment in an evaporation plant at another landfill.

Separate leachate mains serve the leachate pipes from the older Phase I and the more recent Phase II, so it is possible to compare the two to see any effect of the stricter waste acceptance criteria in Phase II. In addition, two of the cells in Phase II have been the subject of intensive research studies carried out since they were first infilled in 1992/93. Leachate from these cells, labelled A1 and A2 for this report, has been analysed separately and data from 1992 to 1999 have been obtained for this study. Data on the dimensions of Site A and its sub-areas are shown below.

		Phase I	Phase II	A1	A2
filling dates		1985-89	1989-present	3/92-6/93	6/92-6/93
Area	m^2	11,000	37,000	1,690	3,500
volume of waste	m^3	228,500	302,890	11,000	15,400
average depth	m	20.8	8.2	6.5	4.4
weight of waste	t	342,750	454,335	16,500	23,100

Waste inputs

The wastes deposited in phase II were characterised in intensive detail during the filling of cells A1 and A2. In total, 38 different waste types were accepted, of which only 12 constituted more than 1% of the inputs. In practice ~90% of the inputs consisted of seven generic types of waste:

- filter cake from a commercial liquid waste treatment plant	21%
- bottom ash (slag) from hazardous waste incineration	18%
- electroplating sludges	15%
- miscellaneous sludges from treatment plants	13%
- contaminated soil or contaminated construction/demolition wastes	8%
- foundry/furnace demolition waste.	8%
- metal hydroxide sludges.	5%

No incinerator fly ashes are deposited at the landfill. These are excluded because of their high chloride content.

The wastes have a low organic content. This is a result of restrictions both on loss on ignition and a leaching test limit. In practice, the average loss on ignition was ~4% during the detailed characterisation study.

The average moisture content of the wastes when deposited is ~40%w/w. No compaction is used during placement and no significant settlement is observed. (Total settlement so far was quoted as “a few centimetres.”) The Operator stated that the in situ density is consistently 1.5t/m^3, based upon comparison of airspace use with recorded tonnages.

Data obtained
The following data have been obtained from information and reports supplied by the landfill operators.

Leachate volumes

- half-yearly volumes for A1,A2; 1992-1999
- monthly volumes for A1, A2; 1996-1999
- annual volumes for Phase I, Phase II; 1999.

Leachate quality

- detailed annual analyses for Phase I, Phase II; 1997-2000
- annual means of weekly analyses for A1, A2; 1992-1999
- weekly analyses for A1, A2; 1992-1999 for a small number of parameters only.

Presentation and interpretation of data
Half-yearly leachate flows for cells A1 and A2 are shown in Figure A3.A.1a, which also shows cumulative L/S ratio calculated for each cell. To illustrate the shorter term variation, monthly flows for cell A1 are shown in Figure A3.A.1b.

Approximate L/S ratios for Phases I and II have been estimated as follows.

Phase II

Leachate flow in 1999	12,668 m^3
mass of waste in place	454,335 t
∴ annual L/S ratio in 1999	0.0279 /a
∴ cumulative L/S ratio at same rate 1989-99	**0.307**

Phase I

Leachate flow in 1999	2,825 m^3
mass of waste in place	342,750 t
∴ annual L/S ratio in 1999	0.00824 /a
∴ L/S ratio 1990-1999, at same rate	0.0824
L/S ratio 1985-89, at present phase II flow (5x12,668) ÷ 342,750	0.185
∴ cumulative L/S ratio 1985-1999: (0.185 + 0.0824)	**0.267**

These estimates are based on two assumptions:

(i) That 1999 rainfall and leachate generation were typical; and

(ii) That most of the higher flow generated in phase II is generated through the operational areas, which are assumed to have been similar in phase I when it was operational.

The estimates for L/S ratio in phases I and II are very crude but are of a similar magnitude to the more accurate estimates for cells A1 and A2 shown in Figure A3.A.1. The expected extent of dilution due to flushing can be estimated from the L/S ratios and the moisture content of the wastes, as follows.

	A1	A2	Phase I	Phase II
L/S at end of 1999	0.25	0.35	0.267	0.307
average H_2O content (%)	40	40	40	40
∴ no. of BV passed	0.625	0.875	0.668	0.768
∴ Ct/Co (=exp[-BV])	0.54	0.42	0.51	0.46

Leachate quality

Leachate quality data are shown in Table A3.A.1 for phase I, phase II, cell A1 and cell A2. The data for A1 and A2 are shown as time series graphs in Figures A3.A.2 to A3.A.5. All four data sets show some common characteristics as well as differences that may indicate spatial variations and perhaps changes in waste inputs from phase I to phase II. Common characteristics are as follows.

- All have higher concentrations of dissolved solids than are typical in UK landfills. They are dominated by inorganic ions Na, K, Ca, Cl, SO_4. Consequently, electrical conductivity is also elevated (up to ~40,000 μS/cm) compared with current UK landfills.
- All have significant concentrations of oxidised nitrogen, typically several hundred mg/l.
- Redox potentials are ~ +200 mV, which is much higher than in biologically active landfills.
- Concentrations of TOC and COD are very low (TOC less than 100 mg/l).
- Carbonate/bicarbonate concentrations are low compared with typical UK landfills, where bicarbonate, derived from biological processes, is often the dominant anion.
- Elevated concentrations of some heavy metals.

The redox, nitrate and sulphate data indicate non-reducing conditions, consistent with an absence of significant biological activity. There was no detectable odour during a site visit and the operator reported that there is no measurable gas generation.

The data sets show that there is considerable spatial variation in some leachate components.

- Ammoniacal nitrogen is present only at very low concentrations in the leachate from the two cells studied in detail (typically ≤10 mg/l). However, the combined leachate from Phase II as a whole, and that from phase I, contains concentration of several hundred mg/l. This is an important observation. It implies that the source is a waste that was deposited in the earlier phase I, but was not accepted during the period 3/92 to 6/93 when cells A1 and A2 were filled, but was accepted at other times in phase II, possibly pre-1992. It is also assumed that the source contains ammoniacal nitrogen, whereas in current UK landfills the predominant source of leachate NH_4-N is the biological mineralisation of proteins, present in non-hazardous wastes.
- The inorganic ions Na, SO_4, NO_3, are present at higher concentrations in the combined phase II leachate than in the leachate from cells A1 and A2.
- All four data sets show elevated concentrations of some minor components, including As, Cu, Ni, Zn, Cd, V, F^- and cyanide. As with some of the major components, these show evidence of large spatial variations within the landfill:
    - arsenic and cyanide are present at much higher concentrations in cell A2 than in either A1 or the combined Phase II leachate;
    - copper, nickel, zinc and cadmium are present at much higher concentrations in A1 than in A2.
    - the combined leachate from phase II shows elevated concentrations of Cu, Ni, Zn, Cr and V, but not As, Cd or cyanide.

There is some evidence of differences between the present phase and the older phase. Phase II has lower pH (~8.2 cf. ~9.2) and lower concentrations of some minor components, both organic and inorganic, namely F, V, As, VOCs and oil. It is assumed that these reflect changes in waste acceptance criteria. The magnitude of the elevated metal concentrations in the phase II leachate, despite its pH of

~8.2, suggest that the organic content of the leachate, albeit fairly low, may contain strong organic ligands.

The time series graphs may be compared with expected dilution for the complete mixing model, calculated earlier. Dilution to approximately 50% of the initial concentration would be expected for a conservative component, from 1992 to 1999. The most useful parameters for initial evaluation are conductivity, chloride, and Na. These show a clear difference in behaviour between cells A1 and A2. In A1, they rise to an early peak, then decline steadily. The extent of decline is more or less consistent with the expectation. Several other parameters, while not usually conservative in biologically dominated landfills, show similar behaviour (SO_4, Ca, Mg, NO_3). In A2, there is little evidence of any decline in most of these parameters: Na, K, SO_4, Cl and Mg, for example, appear very consistent over the whole of the monitoring period. This behaviour may indicate continuing release from the solid phase into solution. Anomalous behaviour is also apparent in the results for minor components: CN, F, As, Ni are maintained at relatively constant levels in A2 leachate. In A1, Cu, F and Ni show substantial decline, while Cd shows a steady long-term increase.

The operator has carried out research into the solid phase chemistry of cells A1 and A2. This work has shown that there is a continuous development of new minerals, primarily oxic minerals such as hydrochalconites (basic carbonates). Many of the minerals identified would remove heavy metals from solution and this may help to explain their non-conservative behaviour.

Insufficient time series data were obtained to allow meaningful examination of dilution/flushing in the combined phase I or phase II leachates. While the data from phases I and II appear to show a general dilution with time for several major components, the data record is too limited to draw this conclusion with confidence.

The annual data, or annual averages, mask a high degree of short-term variability that occurs both with major ions and trace components. This is illustrated in Figures A3.A.6a and A3.A.6b for conductivity, potassium, pH and arsenic. Short term variations of more than 100% of the mean concentration are common in cells A1 and A2. It is not known to what extent this short term variability is reduced in the combined leachate from the whole of phase II. However, the four annual samples shown in Table A3.A1 show considerable variability in all parameters.

Despite the low concentrations of carbonate/bicarbonate, the operator reported that scale builds up in the leachate collection pipes and has to be removed approximately three times per year, at a cost of ~€30,000/year.

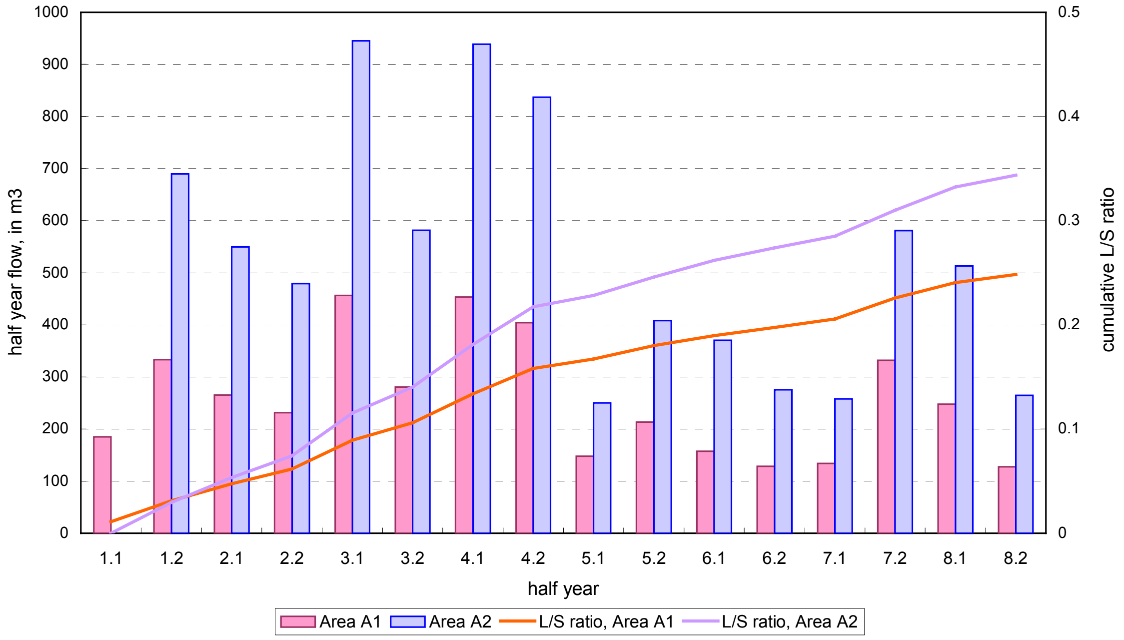


Figure A3.A.1a Half yearly leachate flows at site A, in sub-areas A1 and A2, and cumulative L/S ratios

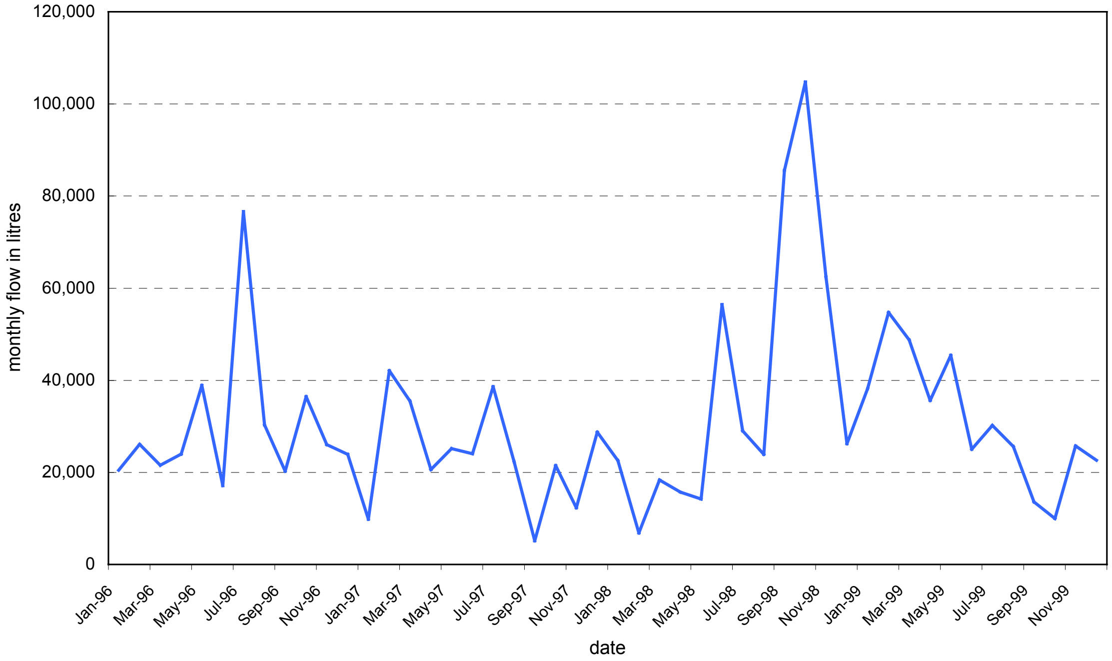


Figure A3.A.1b Variation in monthly leachate flows at site A, sub-area A1

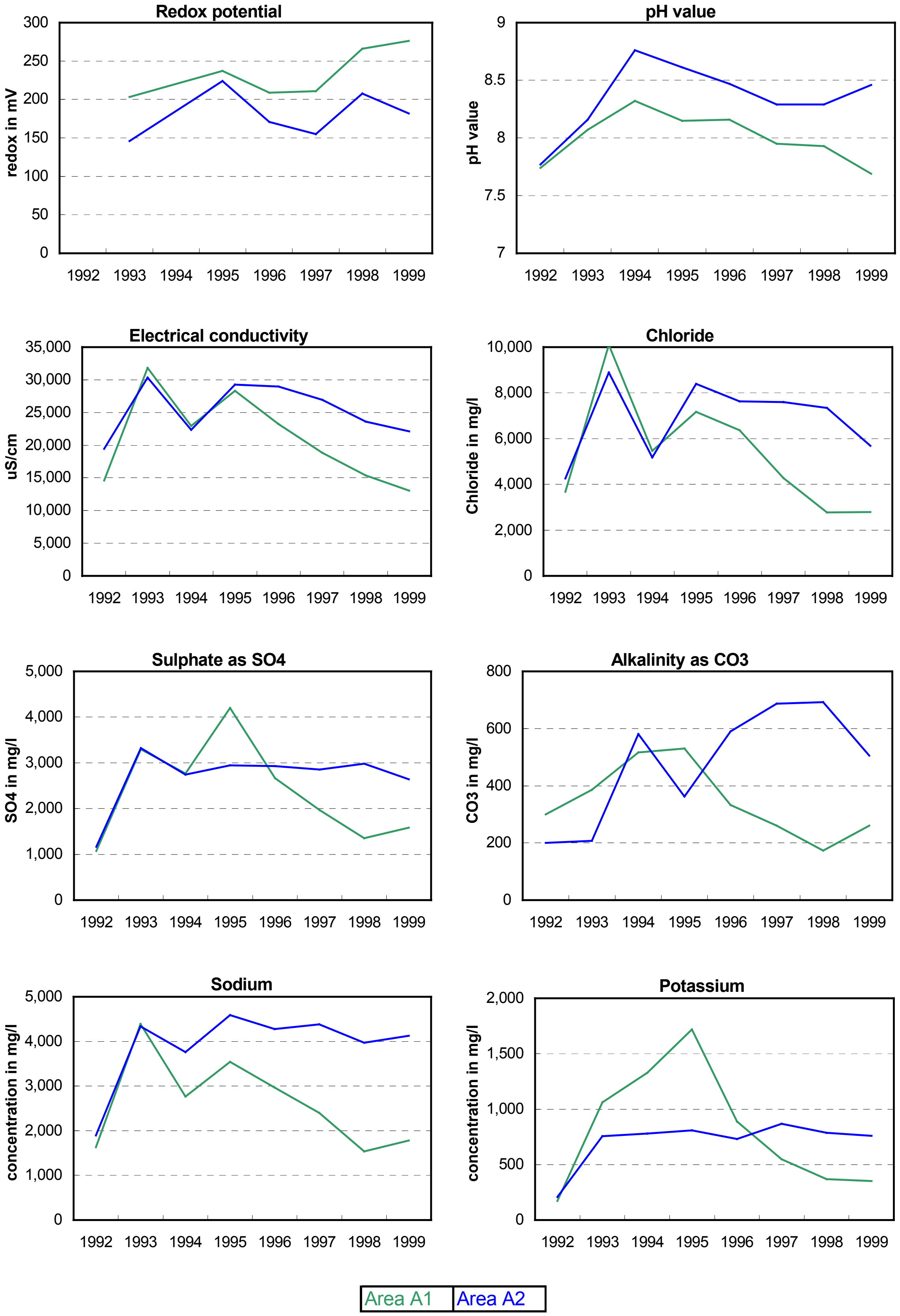


Figure A3.A.2 **Annual averages of weekly leachate quality analyses at site A, sub-areas A1 and A2. I. Major inorganic parameters**

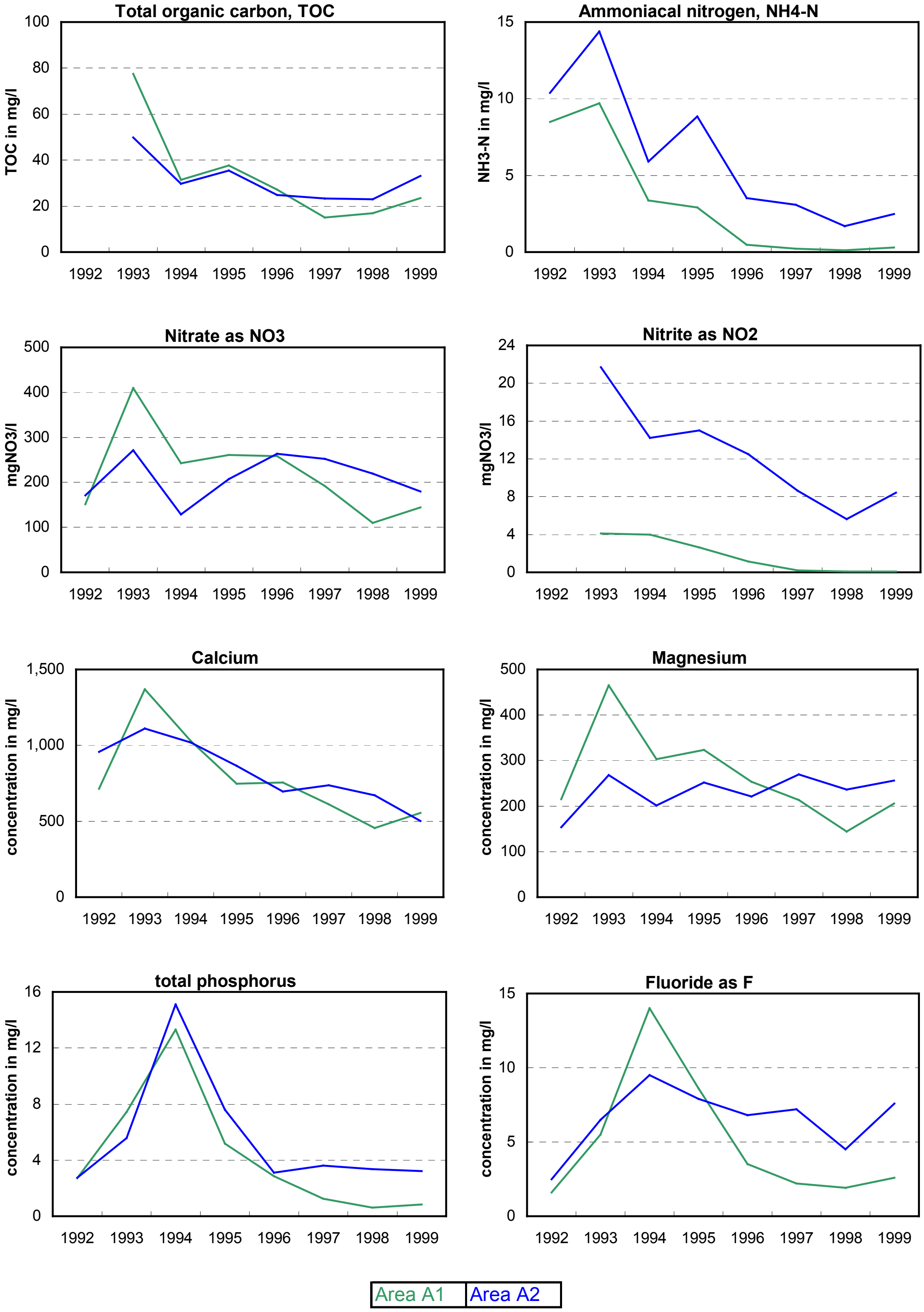


Figure A3.A.3 Annual averages of weekly leachate quality analyses at site A, sub-areas A1 and A2. II. Sanitary parameters plus Ca, Mg, F.

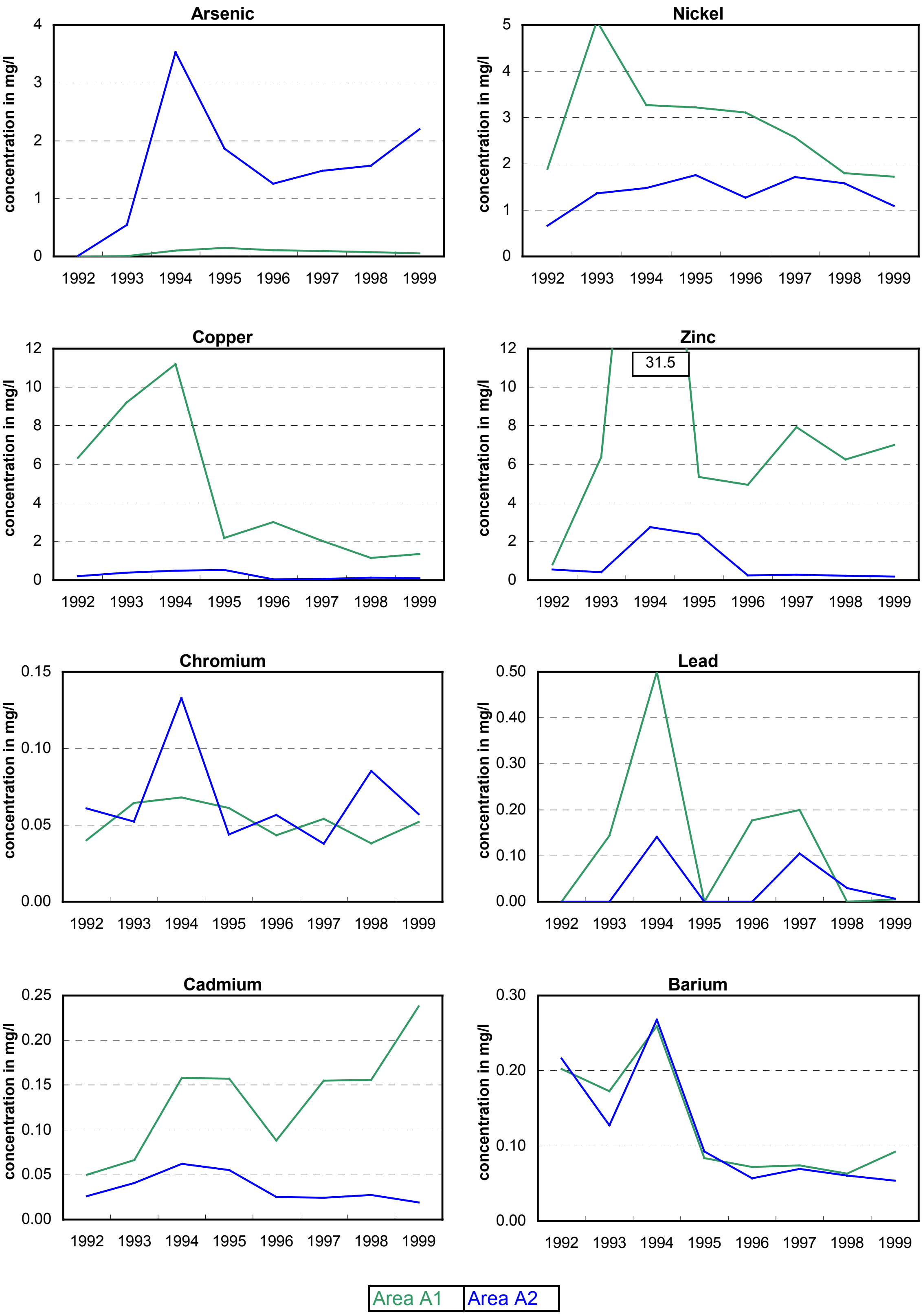


Figure A3.A.4 **Annual averages of weekly leachate quality analyses at site A, sub-areas A1 and A2. III. Toxic metals**

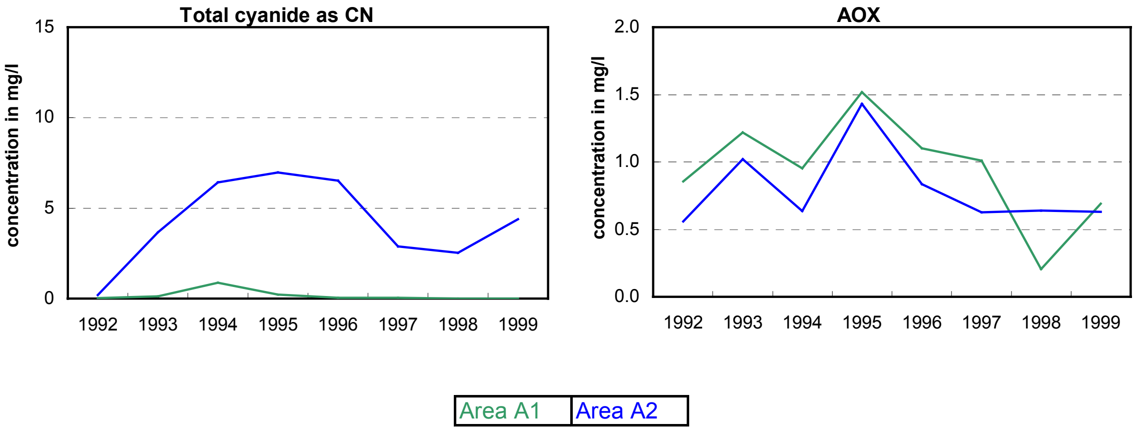


Figure A3.A.5 Annual averages of weekly leachate quality analyses at site A, sub-areas A1 and A2. IV. Minor components

Table A3.A.1 Hazardous waste leachate quality data from composite leachates at Site A

Area A1	redox	pH	Cond	TDS	Chloride	COD	TOC	NH4+	CO3	Na	K	Ca	Mg	SO4	NO3	NO2
Year	mV		µS/cm	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
1992		7.74	14,596		3,680			8.48	300	1630	173	715	215	1070	151	
1993	203	8.07	31,859		10,100		77.5	9.7	386	4390	1060	1370	465	3300	409	4.1
1994		8.32	22,976		5,460		31.4	3.38	516	2770	1330	1030	303	2770	243	4
1995	237	8.15	28,342		7,180		37.6	2.92	530	3540	1720	748	323	4200	261	2.64
1996	209	8.16	23,263		6,370		27.2	0.469	332	2970	890	756	253	2670	258	1.15
1997	211	7.95	18,841		4,279		15.1	0.24	260	2397	547	611	213	1967	192	0.2
1998	266	7.93	15,402		2,770		16.9	0.13	173	1540	369	455	144	1353	110	0.1
1999	276	7.69	13,041		2,793		23.5	0.31	260	1778	350	556	206	1585	144	0.09
Area A2																
1992		7.77	19,430		4,260			10.4	200	1890	208	958	154	1170	171	
1993	146	8.16	30,379		8,890		49.9	14.4	207	4340	754	1110	268	3320	271	21.7
1994		8.76	22,353		5,180		29.7	5.91	580	3760	780	1020	201	2740	129	14.2
1995	224	8.61	29,277		8,400		35.4	8.84	362	4590	808	866	252	2950	207	15
1996	171	8.47	28,954		7,630		24.9	3.53	590	4280	730	697	221	2930	263	12.5
1997	155	8.29	26,940		7,600		23.3	3.08	687	4381	869	736	270	2856	252	8.63
1998	208	8.29	23,638		7,334		23	1.69	692	3966	787	671	236	2984	219	5.63
1999	182	8.46	22,128		5,697		33.1	2.49	505	4128	760	501	256	2638	180	8.44
Phase I																
17/03/97		9.3	42,700	34,600	7,900	320		620		8,800	950	152	177	1,150	730	26
09/03/98		9.1	44,100	31,600	3,500	270		560		8,640	992	95	97	10,500	700	40
08/03/99		9.3	16,700	11,400	2,000	33		200		3,090	293	105	72	3,700	280	17
14/03/00		9.3	40,900	31,700	4,500	105		650		10,100	945	117	123	9,100	1,030	37
Phase II																
17/03/97		8.4	43,400	31,900	11,400	450		390		7,060	820	1,590	480	6,750	810	50
09/03/98		8.2	40,700	23,300	7,700	180		230		5,970	721	972	255	6,150	660	15
08/03/99		8.2	22,400	15,200	5,200	57		160		3,520	459	1,110	1,110	1,900	270	23
14/03/00		8.3	24,000	12,700	5,200	80		150		2,830	295	832	147	3,400	350	8

Table A3.A.1 (cont.) Hazardous waste leachate quality data from composite leachates at Site A

Area A1	As	Cd	Cr	Cu	Ni	Pb	Zn	P	Ba	CN tot	F	Fe	Mn	Mo	V	Se	B	AOX	PAH	oil	PCB	VOC	phenols
year	Mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	µ~~u~~g/l	mg/l	µ~~u~~g/l	µ~~u~~g/l	mg/l
1992	< 0.0001	0.05	0.0401	6.33	1.89	<0,1	0.8	2.73	0.202	0.034	1.6							0.855					
1993	0.0039	0.0665	0.0645	9.19	5.08	0.144	6.37	7.44	0.173	0.131	5.5							1.22					
1994	0.0994	0.158	0.0681	11.2	3.27	0.5	31.5	13.3	0.259	0.881	14.0							0.953					
1995	0.148	0.157	0.0611	2.17	3.22	<0,1	5.35	5.19	0.0839	0.217	8.6							1.52					
1996	0.11	0.088	0.0433	3	3.11	0.177	4.94	2.87	0.072	0.042	3.5							1.1					
1997	0.096	0.155	0.054	2.03	2.57	0.2	7.93	1.25	0.074	0.059	2.2							1.01					
1998	0.074	0.156	0.038	1.15	1.8	< 0,1	6.26	0.6	0.063	< 0,02	1.9							0.207					
1999	0.053	0.238	0.052	1.35	1.72	0.0047	7	0.82	0.092	0.012	2.6							0.69					
Area A2																							
1992	0.0126	0.0259	0.061	0.207	0.665	<0,1	0.545	2.74	0.216	0.204	2.5							0.559					
1993	0.547	0.0407	0.0523	0.381	1.36	<0,1	0.407	5.56	0.127	3.66	6.5							1.02					
1994	3.53	0.0619	0.133	0.474	1.48	0.141	2.74	15.1	0.268	6.41	9.5							0.638					
1995	1.86	0.0553	0.0438	0.531	1.76	<0,1	2.36	7.59	0.0924	6.97	7.9							1.43					
1996	1.26	0.0253	0.0567	0.0407	1.27	<0,1	0.241	3.1	0.0568	6.52	6.8							0.836					
1997	1.4790	0.0245	0.0379	0.07	1.711	0.105	0.285	3.61	0.069	2.89	7.2							0.626					
1998	1.5688	0.0275	0.0853	0.128	1.577	0.030	0.229	3.36	0.061	2.54	4.5							0.639					
1999	2.2	0.019	0.057	0.093	1.09	0.007	0.191	3.22	0.054	4.39	7.6							0.63					
Phase I																							
17/03/97	0.78	<0.1	5.30	0.17	2.40	<0.2	0.40		0.03	0.03	46	1.03	1.1	2.2	161	0.81	41	0.20		3.1	0.32	12.8	0.037
09/03/98	1.07	0.011	5.20	0.70	2.46	<0.1	0.44		0.02	0.02	52	1.02	1.6	2.2	127	0.15	25	0.28		3.1	0.09	23.4	0.037
08/03/99	0.44	0.002	0.40	0.22	0.69	<0.1	0.09		0.03	<0.005	31	0.18	0.4	1.0	24	0.04	12	0.20	0.017	3	0.033		0.037
14/03/00	0.71	<0.01	0.21	0.60	2.76	<0.1	0.31		0.04	0.02	63	0.39	1.8	2.3	68	0.16	23	0.40	0.046	2.5	<0.005	62	0.03
Phase II																							
17/03/97	0.20	<0.1	4.30	4.20	4.55	<0.2	3.53		0.16	0.04	6.4	0.15	5.6	1.6	55	0.26	32	0.30	0.87	0.2	0.27	0.3	0.034
09/03/98	0.11	0.079	2.40	1.04	3.82	<0.1	5.94		0.09	<0.005	4.8	0.06	4.3	0.8	19	0.03	25	0.16	0.87	0.2	0.02		0.034
08/03/99	0.24	0.048	0.60	0.50	2.27	<0.1	2.04		0.08	<0.005	3.9	0.04	2.5	1.9	19	0.022	16	0.30	<0.01	0.6	<0.002		0.034
14/03/00	0.16	0.041	0.73	0.51	1.97	<0.1	2.70		0.096	<0.005	2.4	0.03	2.2	1.0	14	0.023	11	0.40	0.044	0.1	<0.005	0.7	0.072

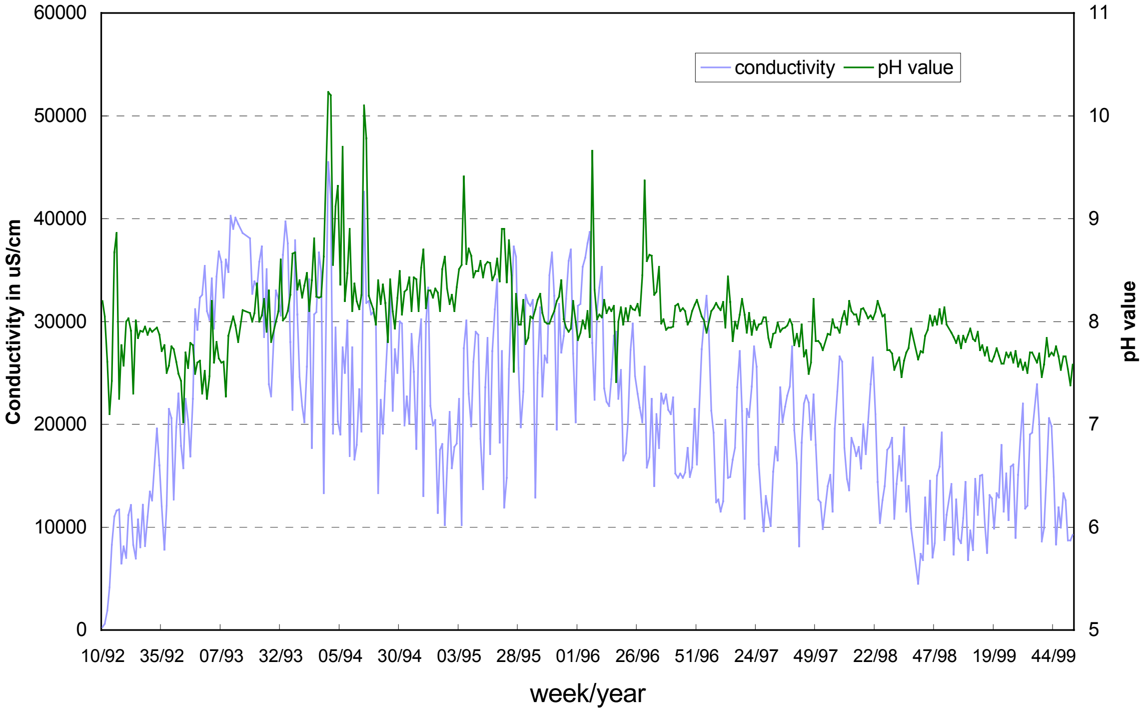


Figure A3.A.6a Site A, mixed hazardous wastes, sub-area A1: weekly leachate analysis for pH value and conductivity

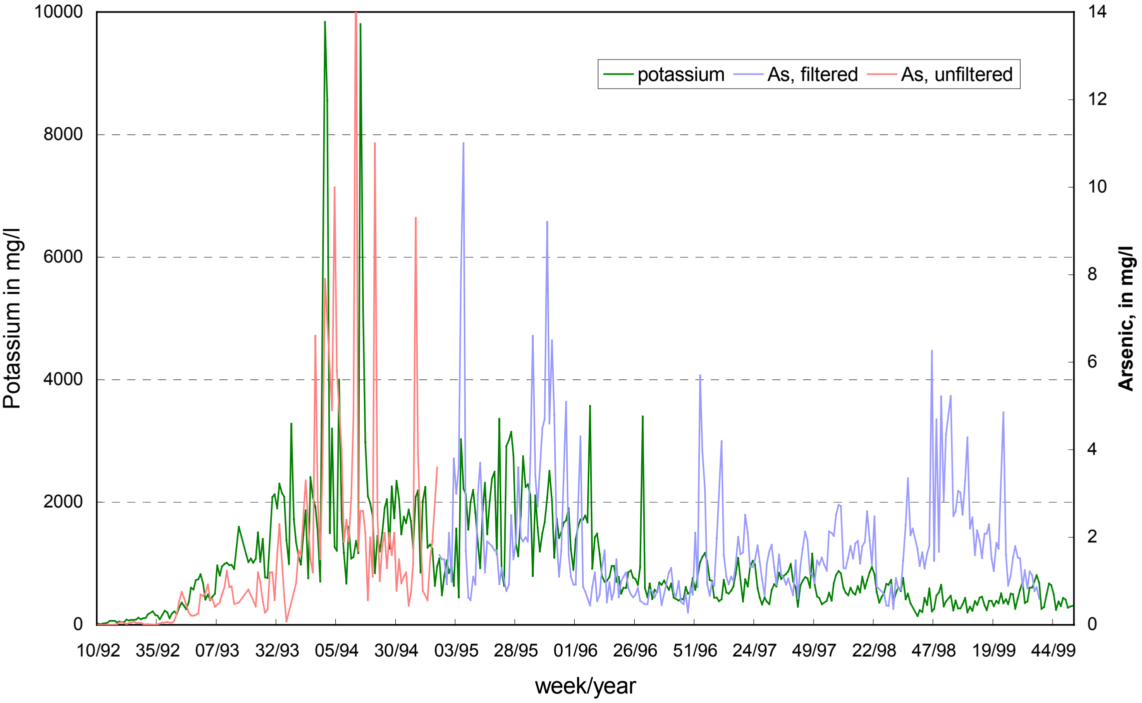


Figure A3.A.6b Site A, mixed hazardous wastes, sub-area A1: weekly leachate analysis for potassium and arsenic

SITE B

Description of site
Site B serves a densely populated region containing several industrial towns and cities. Industries in the region include motor vehicle manufacture, metal processing and petrochemicals. The landfill has been the predominant disposal facility for the majority of landfillable hazardous wastes produced in the region.

The landfill has been in operation since 1976 and has been constructed and filled in two phases:

Phases I/II	Landfilled 1976-90; now clay-capped and restored.
Phases III/IV	1990-present; still operational, some interim geomembrane capping, and containing wastes that meet stricter acceptance criteria than the earlier phases.

All phases of the landfill have a low permeability engineered base, a leachate drainage layer, and collection pipes. These pipes flow to leachate mains, one for phases I/II and one for phases III/IV. The mains pass through the side wall of the landfill and flow by gravity to holding tanks, from where the leachate is tankered off-site for treatment. Treatment includes air-stripping, flocculation and activated carbon. Some hydrogen peroxide dosing is carried out just up-stream of the holding tanks, to remove sulphide odours.

Separate leachate mains serve the leachate pipes from the older phase I/II and the more recent phase III/IV, so it is possible to compare the two to see any effect of the stricter waste acceptance criteria in phase III/IV.

The landfill contained a total volume of 1.7 Mm^3 of waste by March 2001, at an average density reported to be ~1.6 t/m^3. Data on the dimensions of Site B and its sub-areas are shown below.

		Phase I	Phase II	Phase III	Phase IV
filling dates		1976-82	1982-90	1990-95	1995-present
area	m^2	45,000	30,000	22,100	23,500
volume of waste	m^3	450,000	450,000	800,000[1]	400,000[1]
average depth	m	10	15	36	17
weight of waste	t	720,000	720,000	Note 1	Note 1

Note 1: volumes for phases III/IV refer to total capacity; 388,000m^3 remains unfilled, as at March 2001.

The total weight of wastes in phase I/II is estimated by the operator to be ~1.44 M tonnes and the total so far in phases III/IV is estimated to be ~1.28 M tonnes.

Waste inputs
Input rates have typically been on the order of 60,000 to 110,000 tonnes per year and are currently ~80,000 t/a.

The principal waste inputs to Phases I/II were:

- residues from incineration; these are thought to have included fly ash and air pollution control residues;
- filter cake from commercial liquid waste treatment plants;
- miscellaneous sludges from treatment plants;
- sludges from metal finishing processes;
- contaminated soils and demolition wastes, including some with oil contamination.

Phases III/IV have received a similar mix of wastes except for three major changes:

- no incinerator fly ashes are deposited in Phase III/IV, only bottom ash (slag) from hazardous waste incineration. Fly ash is excluded because of its high chloride content;
- restrictions have been placed on the organic content, both total and leachable;
- wastes must pass leaching test criteria on a wide range of parameters.

The wastes in phase I/II have an average organic content of 10 to 15% dry weight, and an average moisture content in situ of ~30%, based on a drilling investigation in the mid 1990s. The wastes in phase III/IV are thought to have a much lower organic content, due to the restrictions applied.

Compaction is sometimes used during placement, mainly because of the presence of remnants of steel containers in the bottom ash from hazardous waste incineration and also because of occasional inputs of contaminated packaging and demolition materials. No significant settlement is observed. The operator stated that the in situ density is consistently 1.6 t/m^3, based upon comparison of airspace use with recorded tonnages.

Data obtained
The following data have been obtained from information and reports supplied by the landfill operators.

Leachate volumes
- annual volumes for phase I/II from 1976 to 1999
- annual total volumes for the whole site from 1976 to 1999.

Volumes for phase III/IV have therefore been calculated by difference.

Leachate quality
- quarterly analysis from both phases, for a wide range of parameters, from 1977 to 2000.

Estimation of L/S ratio
Annual leachate volumes for both phases are shown in Figure A3.B.1, which also shows cumulative L/S ratio calculated for each phase. In both areas, the L/S ratio has remained relatively constant, as the waste quantity in place has increased. In phase I/II, the annual leachate volumes declined dramatically since capping (around 1989-1991), so that there has been only a small increase in L/S ratio since waste inputs stopped. By late 2000, the L/S ratio in phase I/II was ~0.2 and that in phase III/IV was ~0.14. The difference is partly due to the younger age of phase III/IV but equally importantly to its greater average depth of waste. In both parts of the site, it is clear that the L/S ratio is low.

The expected extent of dilution due to flushing in a completely mixed reactor can be estimated from the L/S ratios and the moisture content of the wastes, as follows.

	Phase I/II	Phase III/IV
L/S at end of 2000	0.2	0.14
average H_2O content (%)	30	30
$\therefore$ no. of BV passed	0.67	0.47
$\therefore$ Ct/Co (=exp[-BV])	0.51	0.63

note: BV = bed volume

Thus, significant dilution might be expected for conservative components already in solution, if the landfill behaved as a completely mixed reactor.

Leachate quality
Leachate quality data are shown as time series graphs in Figures A3.B.2 to A3.B.5 for phase I/II (old area) and in Figures A3.B.6 to A3.B.9 for phase III/IV (current area). The following observations may be made.

Old area, Phase I/II

- The leachate has very high concentrations of dissolved solids (~110,000 mg/l);
- The dissolved solids consist mostly of simple inorganic ions, predominantly sodium (~30,000 mg/l), potassium (~12,000 mg/l) and chloride (~60,000 mg/l) ions;
- Calcium and sulphate concentrations are much lower, ~400 and ~1,000 mg/l respectively, consistent with the solubility of calcium sulphate;
- Concentrations of carbonate/bicarbonate are also relatively low, at ~3,500 mg/l as $CaCO_3$.
- There is no evidence yet of washout or declining concentrations of conservative components. On the contrary, the concentrations of chloride, sodium and potassium have increase steadily since capping.
- The leachate pH is near neutral, slightly alkaline, similar to UK non-hazardous and co-disposal landfills.
- There are significant concentrations of organics (TOC ~2,000 mg/l), which appear to include a significant percentage of degradable compounds (BOD ~2,000 mg/l).
- The concentrations of degradable organics have declined with time, in contrast to the conservative salts. This is consistent with some biological activity within the landfill. Low but measurable rates of gas generation have been reported by the operator, with a few percent of methane and carbon dioxide.
- The leachate contains no detectable oxidised nitrogen (not graphed) but does contain sulphide, often at concentrations of several tens of mg/l. These are consistent with anaerobic activity.
- Ammoniacal nitrogen is present at concentrations similar to those found in UK landfills. The concentration has been rising since the early 1990s, possibly as a result of capping, and is currently at ~1600 mg/l. It is most likely to have been present in the waste inputs as NH_4^+-N rather than being produced in situ by biological mineralisation of organic nitrogen.
- The concentrations of many parameters show large short term fluctuations, consistent with an influence of fluctuations in rates of infiltration.
- Several minor components are present at significant concentrations:
  - phenol at tens of mg/l;
  - hydrocarbons at tens of mg/l;
  - AOX at a few mg/l (though interference by high chloride concentrations should be considered);
  - halogenated aliphatic hydrocarbons at tens or hundreds of μg/l (mainly trichloroethylene and tetrachloroethylene).
- Heavy metal concentrations are generally low (sub-mg/l) but some had initial higher concentrations during the first few years: Ni (~5 mg/l), Pb (~1 mg/l), Cr (~0.5-1.0 mg/l) and Cd (~0.2 mg/l). Most of the heavy metals have shown an overall decline with time (perhaps as redox has fallen and sulphide concentrations have increased). However, arsenic shows a rising trend with time, and is currently at 100-400 μg/l.

  No significant concentrations of mercury have been detected (usually < 1 μg/l, max 29 μg/l).

New area, Phase III/IV

- The leachate has a high concentration of dissolved solids, at ~40,000 mg/l, though this is clearly much lower than in phase I/II.
- The dissolved solids are, like phase I/II, dominated by sodium, potassium and chloride but there is a greater contribution from calcium (~1200 mg/l) and sulphate (~4,000 mg/l) than in the older area.
- Carbonate/bicarbonate remains a minor component (1,000 to 3,000 mg/l as $CaCO_3$)
- Dissolved salt concentrations and major ions exhibit a very slight long-term decline, but much less than calculated above for a completely mixed reactor.
- Sulphate concentrations exhibit far more extreme fluctuations than the other major inorganic ions.

- The leachate pH value has remained neutral to slightly alkaline.
- There are significant concentrations of organics, but they are lower than in the old area. Like the old area, they decline with time: TOC fell from an initial ~2,000 mg/l to ~250 mg/l after ~8 years. BOD fell from ~1,000 mg/l to ~200 mg/l in the same period. The BOD:TOC ratio shows that there is a significant degradable content in the TOC. The fact that the decline is greater than that for inorganic salts, is consistent with the decline in organics being due to biological activity within the landfill.
- Oxidised nitrogen concentrations have varied widely: for the first ~4 years, oxidised nitrogen was absent. Subsequently it has been present intermittently at tens or hundreds of mg/l, interspersed with periods when it has been absent. The peak concentrations have occurred primarily in winter. Summer concentrations are often close to zero.
- Sulphide was initially present at up to 35 mg/l in the early years but has been absent since ~1994.
- The results for sulphide and oxidised nitrogen suggest that conditions were fairly strongly anaerobic in the first few years but have become increasingly aerobic subsequently.
- Ammoniacal nitrogen has been present at significant concentrations, up to 800 mg/l, which is similar to those in MSW landfills. However, concentrations have declined steadily since 1995, to recent values of ~200 mg/l. Like the organics, this decline is greater than that of the dissolved inorganic salts and is greater than anticipated from dilution calculations. A possible explanation is therefore biological degradation.
- Minor components are present at significantly lower concentrations than in the old area:
  - phenol was detectable at ~10 mg/l or less for the first few years but has been undetectable since ~1996;
  - hydrocarbons are present but generally at less than 10 mg/l;
  - AOX is at a similar concentration to the old area but is subject to the same concern regarding interference by high chloride concentrations;
  - halogenated VOCs are virtually absent;
  - cyanide is virtually absent.
- Heavy metal behaviour has been different from the old area. Concentrations of some metals have been much higher and several have increased in concentrations, whereas in the old area, most declined over time:
  - nickel was at 1 - 2 mg/l in years 1-6 then increased to 6-8 mg/l;
  - zinc was at 2 – 4 mg/l in years 1-6 then increased to ~8 mg/l;
  - copper was at 0 – 1 mg/l in years 1-6 then increased to ~2 mg/l.

  Lead did not show the initial high concentrations that occurred in the old area, but has fluctuated at relatively low levels, around 0-0.4 mg/l. Cadmium has fluctuated and reached 1 mg/l in 1998, though most values are less than 0.1 mg/l. Chromium and arsenic have been at similar sub mg/l levels to those in the old area. Chromium (VI) has generally been absent.

  Mercury has not been detected in any sample.

Research studies conducted at the site by the operator have indicated redox potentials in the range -150 mV to -300 mV. These are consistent with the anaerobic biological activity suggested by the data for organics, sulphide and oxidised nitrogen. The operators research studies have suggested that the biological activity is concentrated in the leachate at the base of the site far more than throughout the main waste body.

It is assumed that the extremely high TDS concentrations in phase I/II are due to its having received fly ash from incineration in its early years, though this has not been specifically confirmed by the operator.

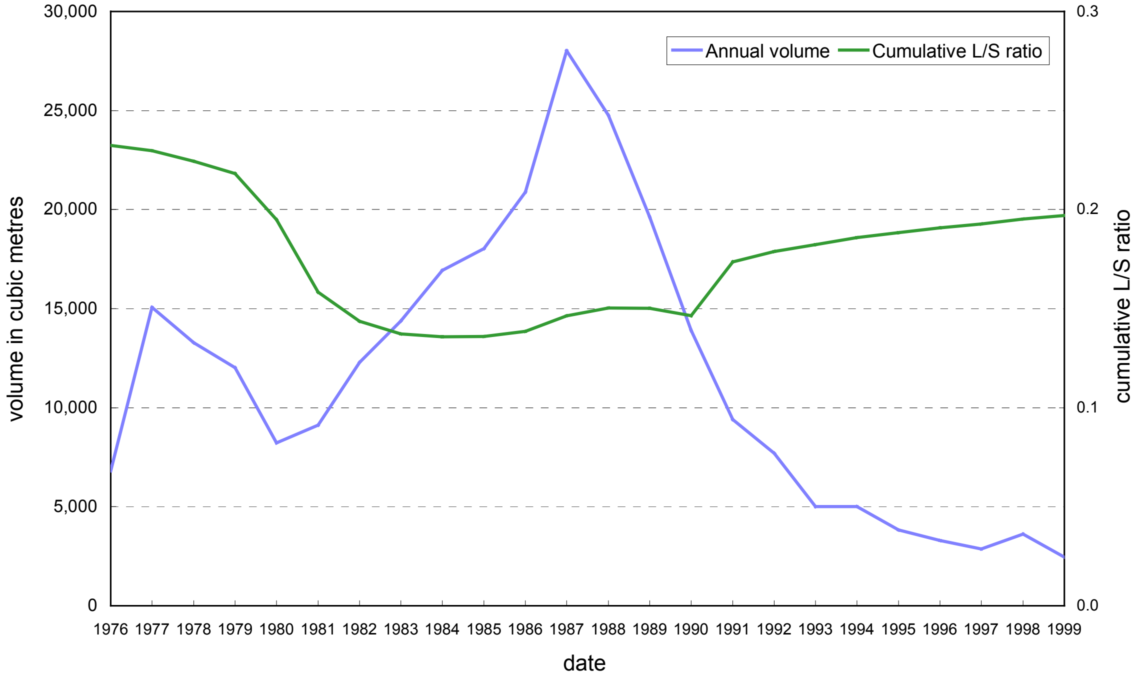


Figure A3.B.1a **Annual leachate volumes and estimated cumulative L/S ratio at site B (old area)**

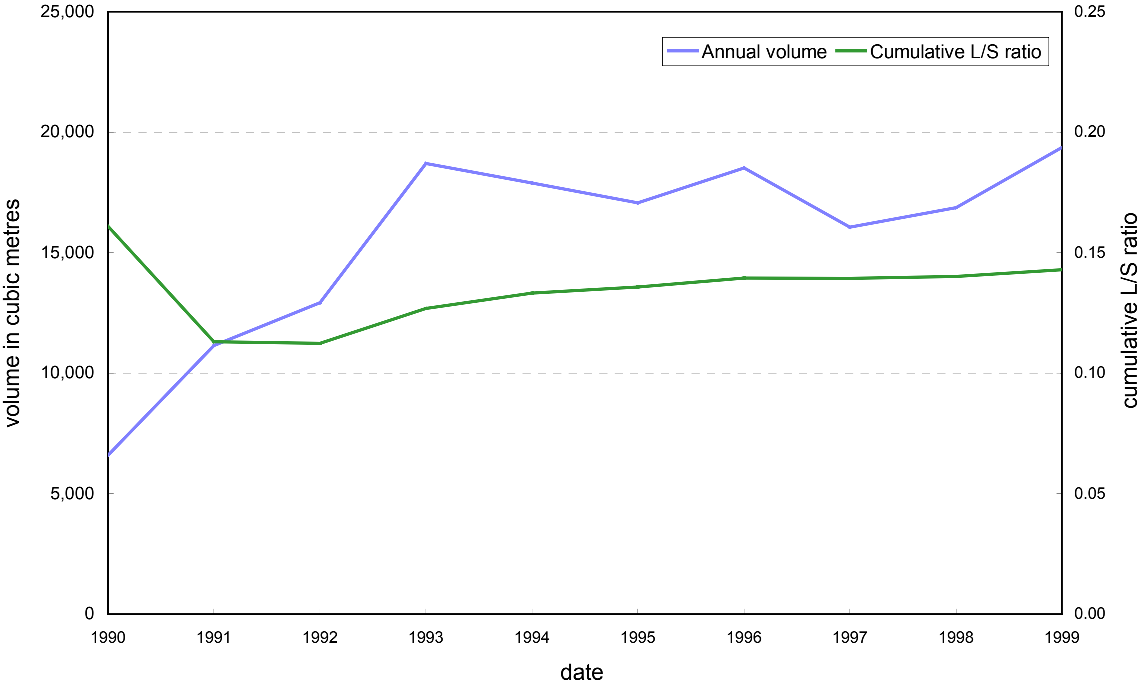
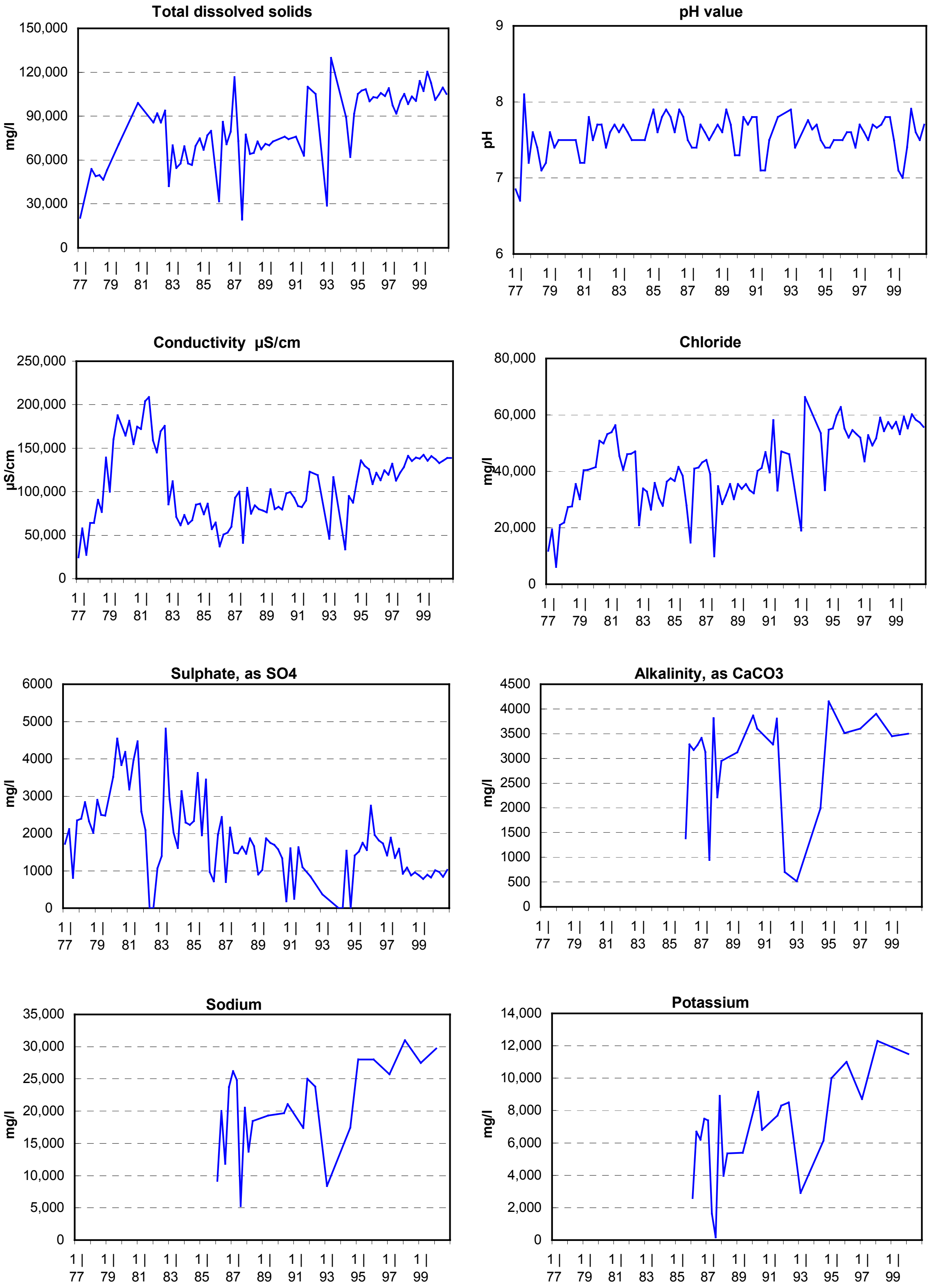


Figure A3.B.1b **Annual leachate volumes and estimated cumulative L/S ratio at site B (new area)**


Total dissolved solids

pH value

Conductivity µS/cm

Chloride

Sulphate, as SO4

Alkalinity, as CaCO3

Sodium

Potassium

Figure A3.B.2 **Leachate quality analyses at site B (old area). 1 Major inorganic parameters [date shows quarter and year]**

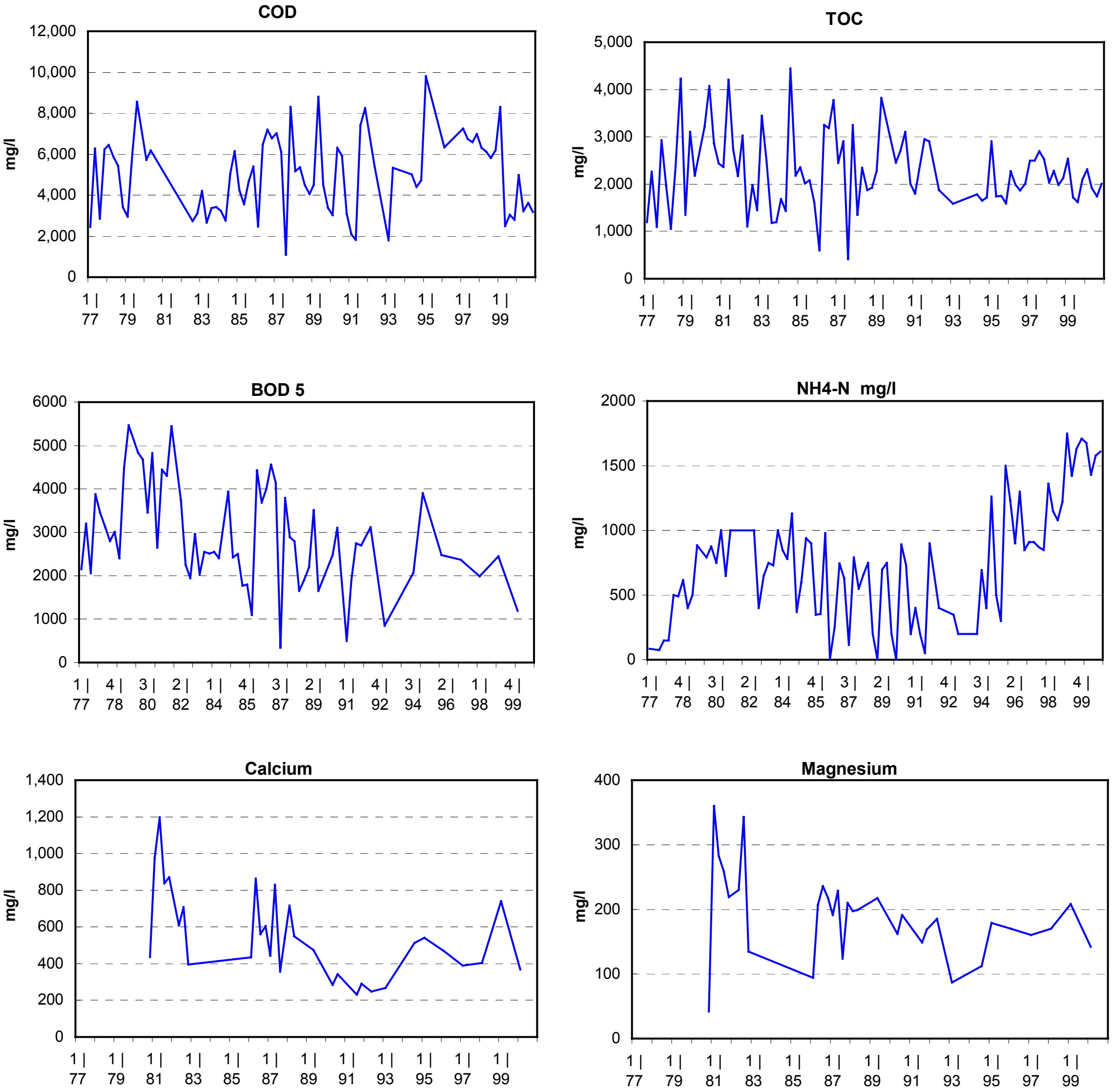


Figure A3.B.3 Leachate quality analyses at site B (old area). 2. Sanitary parameters, Ca and Mg [date shows quarter and year]

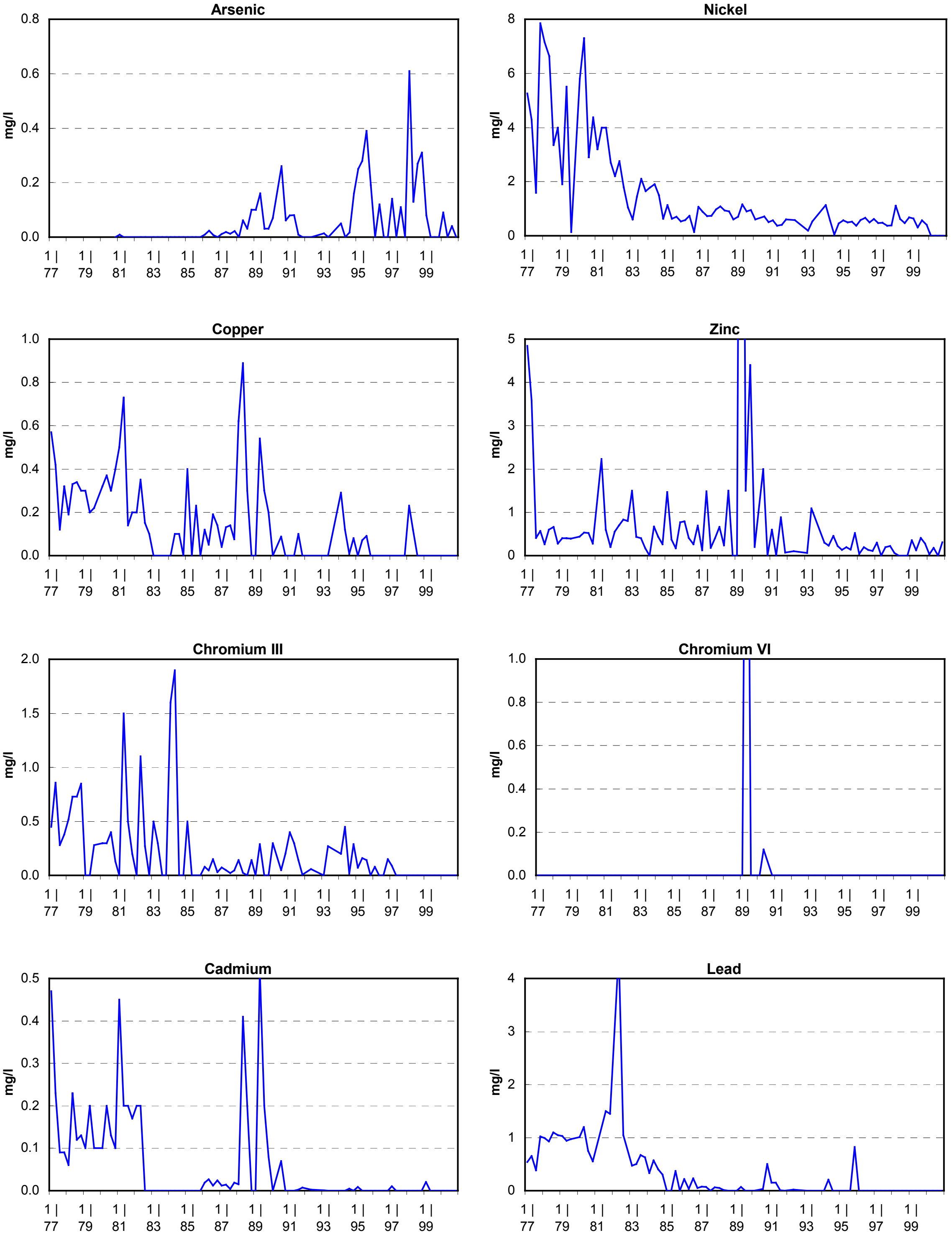


Figure A3.B.4 Leachate quality analyses at site B (old area). 3. Heavy metals (date shows quarter and year)

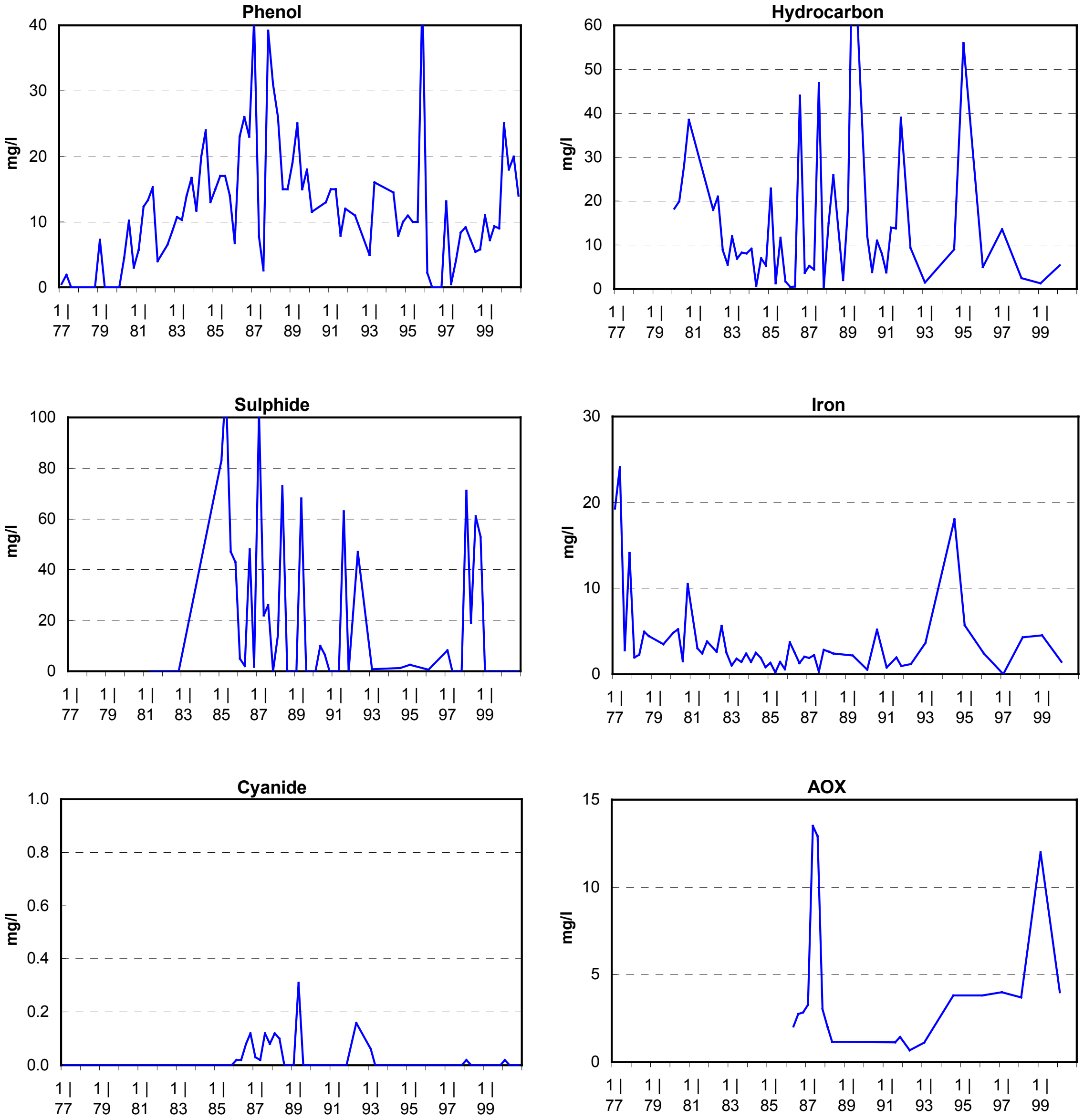


Figure A3.B.5 Leachate quality analyses at site B (old area). 3. Minor components (date shows quarter and year)

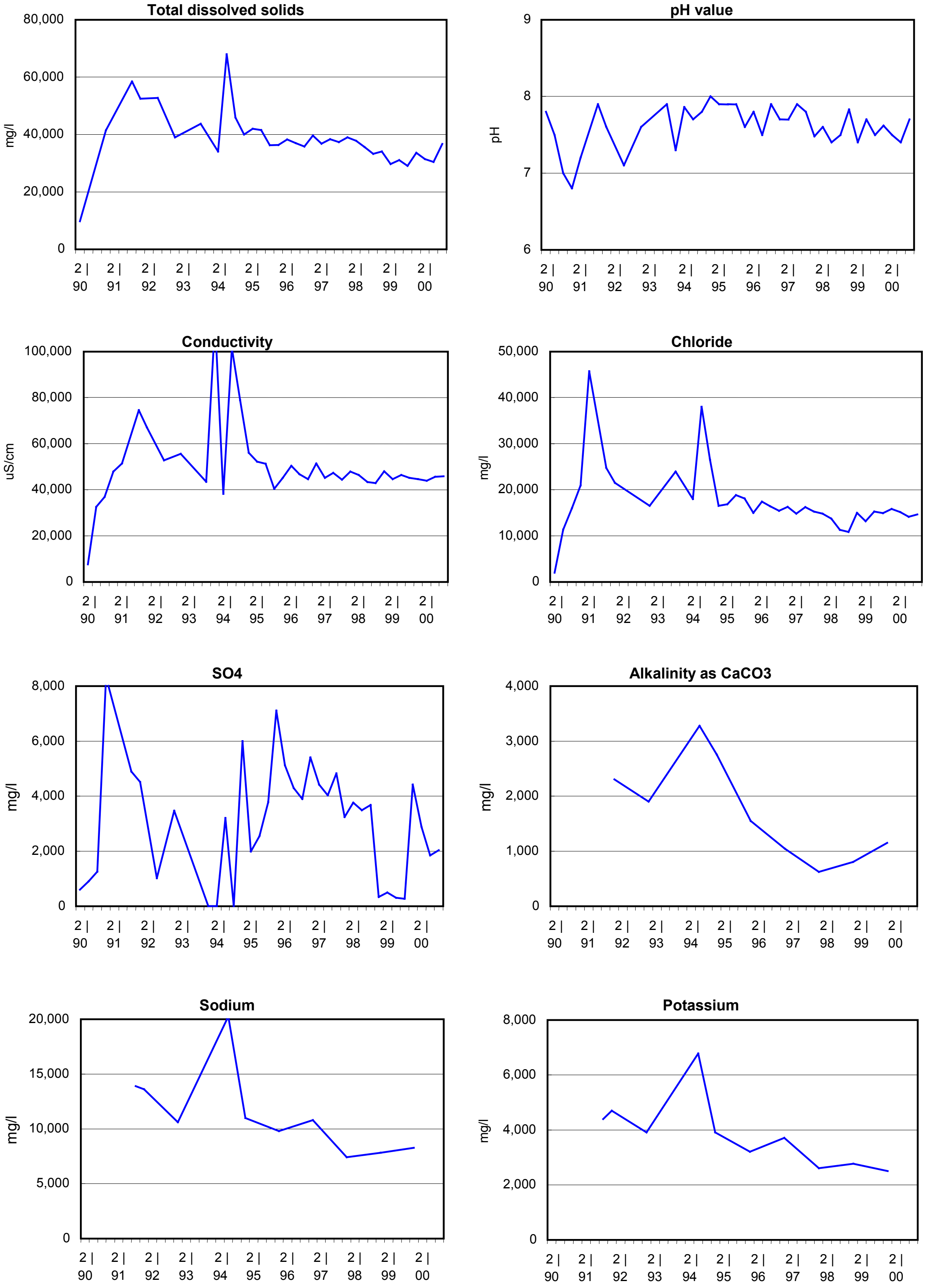


Figure A3.B.6 **Leachate quality analyses at site B (new area). 1. Major inorganic parameters (date shows quarter and year])**

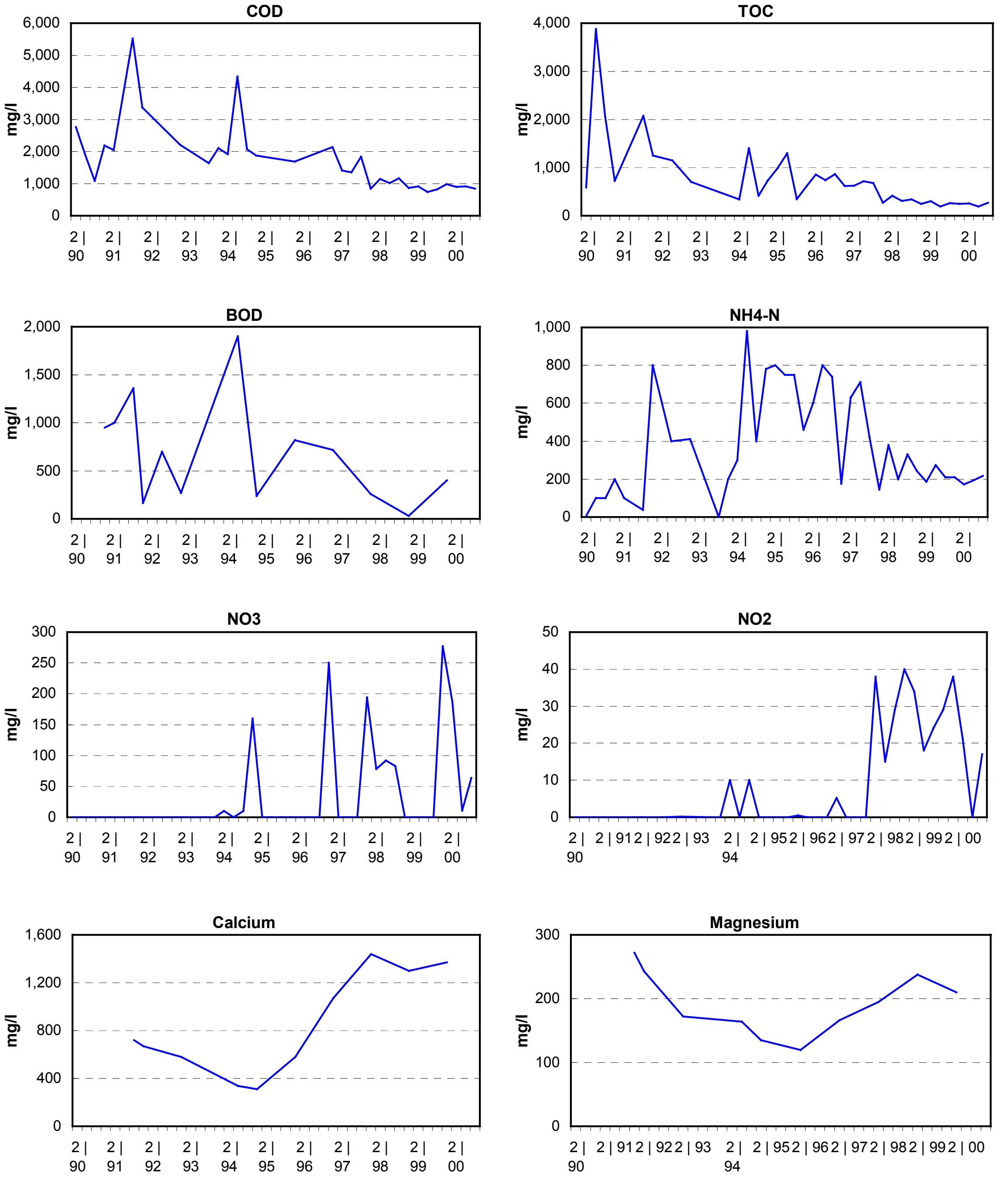


Figure A3.B.7 Leachate quality analyses at site B (new area). 2. Sanitary parameters, Ca and Mg (date shows quarter and year)

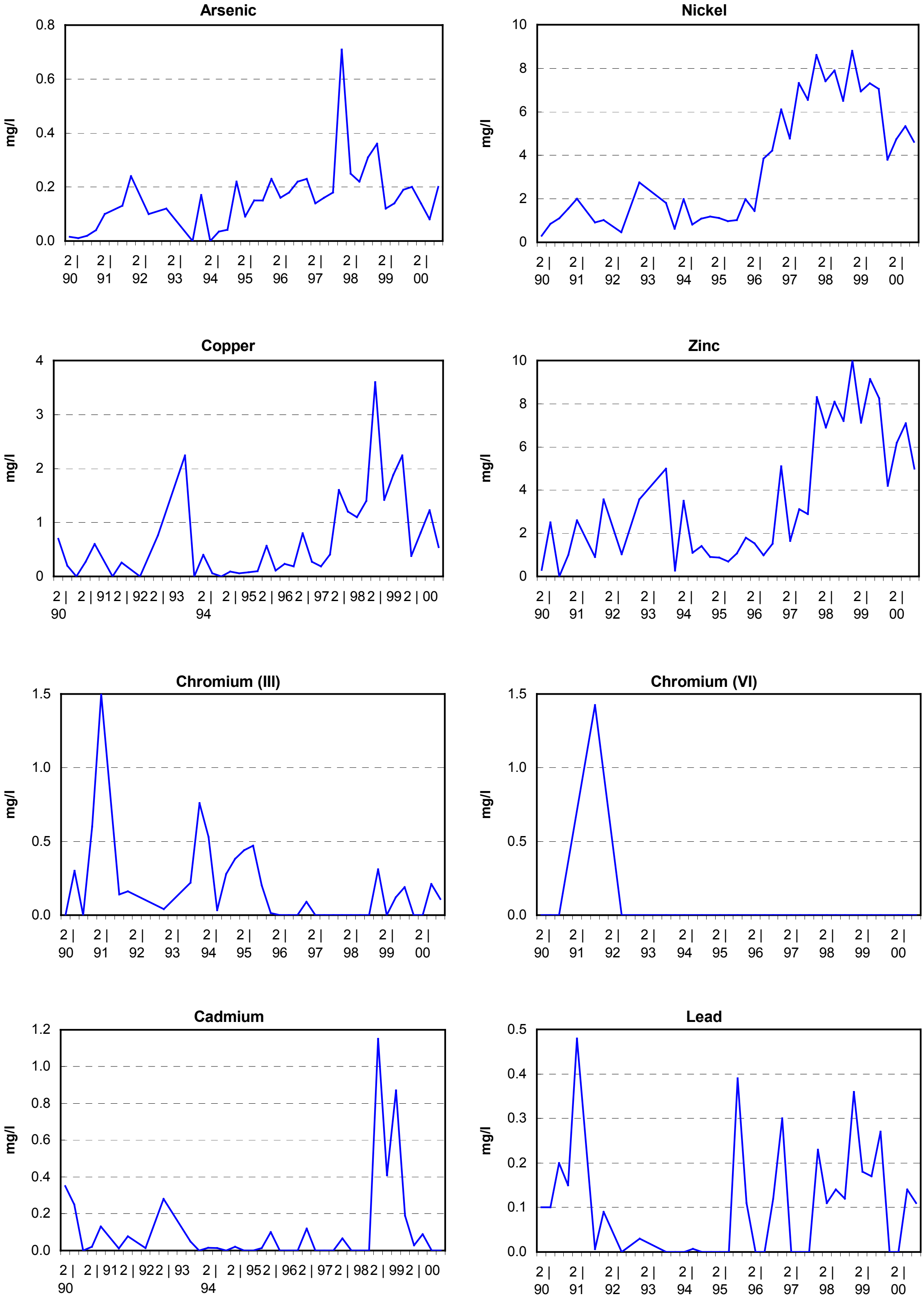
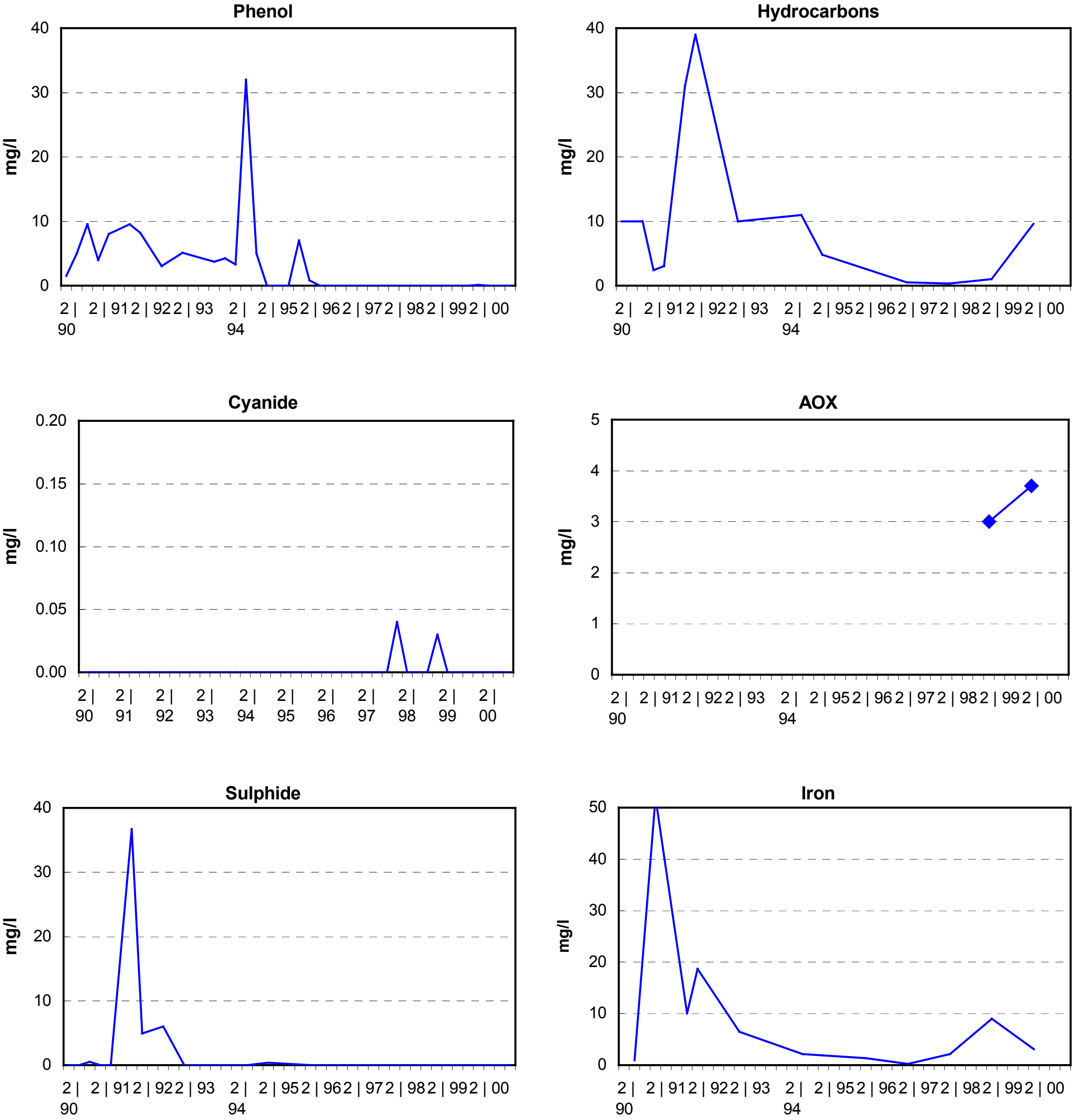


Figure A3.B.8 **Leachate quality analyses at site B (new area). 3. Heavy metals (date shows quarter and year)**


Phenol

Hydrocarbons

Cyanide

AOX

Sulphide

Iron

Figure A3.B.9 **Leachate quality analyses at site B (new area). 4. Minor components (date shows quarter and year)**

SITE C

Description of the site
Site C is located on the outskirts of a large industrial coastal city and is the major outlet for hazardous wastes in this area.

Landfilling started in 1988. Since 1993 it has focussed only on hazardous wastes. Currently the operational areas cover 14 ha, out of a total of 17 eventually. There is 1.3 Mm^3 of waste in place, which will eventually increase to 3.4 Mm^3. The average waste depth is currently 9.3 m. The lifetime of the site is expected to be another 30 years.

The base of the landfill is 2.5 m above the saline groundwater table. A double composite liner is in place with leak detection in between the liners. Sand/bentonite is the mineral layer. Each cell is capped with a single composite liner with geomembrane and sand/bentonite mineral layer, restored to grass.

Wastes are placed in discrete cells. Samples of leachate are taken from vertical abstraction pipes in each cell. There are currently 19 separate flows leading to the leachate treatment plant. As well as the sumps from individual landfill cells, these include domestic wastewater from the site offices and surface run-off from some operational areas.

A leachate collection system layer comprises of sand, with gravel only used around the pipes. Leachate collection system pipes are at ~5 m spacing, slope 2%. All the leachate collection system pipes slope outwards to the outer edge of the landfill, where the leachate is abstracted from vertical sumps. Leachate depth is thought to be variable. About 47,000 m^3 was abstracted in 2000. This is equivalent to an average percolation of 336 mm/a on the 14 ha current site area.

Leachate discharge is to a tidal river/creek. The plant includes physical/chemical treatment, as well as biological, to reduce TKN/ammoniacal nitrogen concentrations. The physico-chemical processes include $FeCl_3$ addition, coagulation and settlement. The consent includes 13 quality parameters and maximum flow of 15 m^3/hour, with the following limits:

parameter	Spot sample (mg/l)	24 hour composite
COD	300	250
$BOD_{20/5}$	20	15
TKN	35	25
SS	30	-
As	0.010	0.005
Cd	0.010	0.010
Hg	0.005	0.003
sum of Cr, Cu, Pb, Ni, Zn.	1	0.500
CN (tot)	0.100	0.050
PAH (Borneff)	0.010	0.005
mineral oil	10	-
EOCl [actually they do EOX]	0.100	-

No problems have been reported on clogging of the leachate collection system pipes. They are inspected every year and the operator is obliged to clean them every two years. They get only small amounts of scale, including iron and manganese compounds, although no analyses were seen during the visit.

Waste inputs:
The landfill receives some hazardous wastes that are landfilled directly, in admixture with each other, and others that have been treated on site in a stabilisation process (solidification with cement and fly ash) so that they meet leaching criteria for acceptance in the landfill. The solidified wastes

are landfilled in separate cells from the other wastes. Some fly ash and APC residues are deposited direct from the producers in 'big bags' in separate mono-disposal cells.

Total inputs to the site are in the range 100,000 to 150,000 t/a.

type of waste	annual tonnage
contaminated soil and residues from soil washing	30,000
wastes from paint manufacture and recycling	40,000
stabilised/solidified wastes	60,000
metal processing sludges	10,000
grit blasting wastes	?
miscellaneous small arisings e.g. TiO_2	30,000

Wastes landfilled directly must have less than 10% LoI (and also a limit on digestible organic content). However, wastes for solidification/stabilisation must have less than 5% LoI. The main acceptance criteria are for eluates from a standard leaching test.

Data obtained

Leachate from the individual cell abstraction sumps is sampled and analysed four times per year. Data for the annual averages of these samples from the mixed waste cells have been obtained for fourteen determinands, six of which are heavy metals. Results for twelve determinands are shown as time series graphs in Figure A3.C.1. The cells are identified in the figure by the year in which samples were first taken. Results for oil and AOX are summarised below.

	Oil, mg/l			EOX, μg/l		
Cell	**mean**	**min**	**max**	**mean**	**min**	**max**
1988 cell	5.1	0	44	16.5	0	134
1990 cell	5.5	0	60	14.6	6	39
1991 cell	0	0	0	6.8	0	21
1994 cell	0	0	1	3.6	0	13
1997 cell	0	0	0	0	0	2

Unfortunately, no leachate quality data have been obtained for the cells containing solidified wastes.

Estimation of L/S ratio

An approximate L/S ratio for the whole site was calculated as follows.

Infiltration 1988-2000 (13 years) at annual rate of 336 mm/a	4368	mm
	4.368	m^3/m^2
Waste depth x estimated density (9m x 1.5 t/m^3)	13.5	t/m^2
L/S ratio (4.368/13.5)	0.32	

This L/S ratio is subject to much uncertainty because it is based on a single annual leachate volume and on the average current waste depth, rather than on actual annual leachate volumes and actual quantities of waste in place. The density of 1.5 t/m^3 is an assumed value, based on values reported at other sites receiving similar waste inputs.

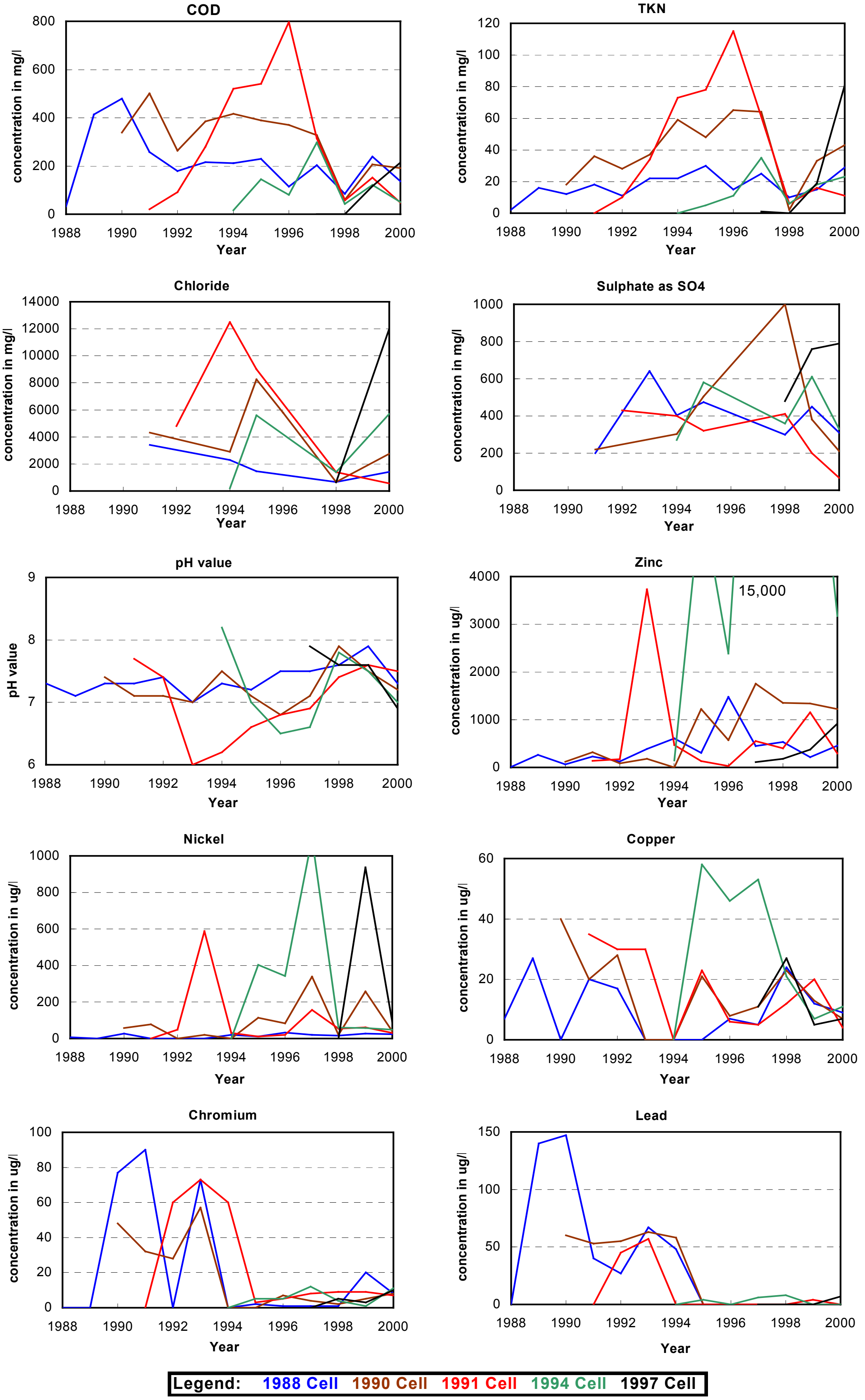


Figure A3.C.1 **Leachate quality at Site C, mixed hazardous waste cells**

SITE D

Description of Site

Site D is located close to two industrial centres and is the major regional outlet for non-hazardous, low organic wastes. It also receives some MSW incinerator fly ash, deposited separately.

Construction began in 1983 and waste disposal operations began in 1985. The operational area of the site is ~70 ha and the quantity of waste in place by the end of 2000 was ~7.7 Mt. Waste inputs are typically ~500,000 t/a with a range of 300-700,000 t/a. The site has been filled in discrete cells. Each cell receives mixed wastes, reflecting the composition given below. However, fly ashes are deposited in separate cells, not mixed with other wastes.

The landfill is methanogenic and has a gas extraction and utilisation system installed, generating electric power and heat, which is used to heat the leachate treatment plant. Gas generation currently is estimated by the operator to be ~1000 m^3/hour and it typically contains ~55% CH_4.

The landfill has a geomembrane basal liner and the base is slightly below the external groundwater level. A composite top liner of geomembrane plus sand/bentonite mixture, overlain by 1 m of clean top soil is proposed but had not yet been installed at the time of this study.
The groundwater in this area is brackish to saline. The annual average rainfall for 1989-2000 was 869 mm (range 610-1113 mm).

The leachate collection system consists of a gravel drainage layer, with pipes leading to a single abstraction sump at the outer edge of each cell. The gravel surrounding the collection pipes is coarser than that used for the main drainage layer. The drainage layer is overlain by sand, rather than geotextile, to separate it from the wastes. The leachate collection pipes and drainage layer are maintained in a saturated condition, but the leachate level in the site is not known. The operator reports that cleaning has only been carried out every 3 years and that very little material is recovered during cleaning.

Leachate is combined and treated on site in a biological plant that achieves nitrification and denitrification of ammoniacal nitrogen. The plant is heated using waste heat from a landfill gas power plant. Discharge is to a brackish water body. Leachate samples are taken from the abstraction sumps in each cell. The sampling method was not established.

Waste Inputs

The waste inputs since operations began are shown in Figure A1.D.1. This does not include MSW incinerator fly ash, which is deposited in separate cells in ‘big bags’.

‘Miscellaneous inorganic sludges’ comprise more than half the inputs to the site and include small quantities (<5%) of some wastes that the site operator regards as borderline hazardous/non-hazardous, e.g. sludges from industrial production processes, residues from soil decontamination and dewatered dredging sludges. Some ~30% of the inputs have a significant organic content, despite the exclusion of household wastes. The average organic content of the mixed wastes is ~10%.

Waste inputs have to be within limit values for a wide range of hazardous elements or compounds and must pass leaching test eluate criteria for a more restricted range of substances.

Data obtained

Data from one of the cells receiving fly ash in big bags is discussed in the main report. Data have been obtained for the combined leachate from the non-hazardous cells as it enters the leachate treatment plant. These data are presented and discussed in the main report.

No leachate is recirculated. Removal rates have increased as the size of the site has increased and are shown in Table A3.D.1, which includes an estimate of L/S ratio.

Table A3.D.1 Leachate flow rates at Site D

	precipitation	leachate	cumulative	waste input	waste input	cumulative
year	mm/year	m^3/a	leachate,m3	tonnes/a	tonnes/a	L/S ratio
1985		12,027	12,027	262,723	262,723	0.046
1986		31,189	43,216	394,085	394,085	0.066
1987		48,767	91,983	478,800	478,800	0.081
1988		51,028	143,011	595,200	595,200	0.083
1989	763	47,006	190,017	424,018	424,018	0.088
1990	758	45,191	235,208	391,024	391,024	0.092
1991	713	30,965	266,173	462,824	462,824	0.088
1992	1,099	79,699	345,872	584,818	584,818	0.096
1993	963	122,817	468,689	701,084	701,084	0.109
1994	1,060	206,708	675,397	602,971	602,971	0.138
1995	720	125,860	801,257	579,092	579,092	0.146
1996	610	96,371	897,628	341,462	341,462	0.154
1997	682	173,601	1,071,229	231,923	231,923	0.177
1998	1,113	267,694	1,338,923	419,341	419,341	0.207
1999	963	296,045	1,634,968	566,187	566,187	0.232
2000	983	287,363	1,922,331	642,906	642,906	0.250

The cumulative L/S ratio by the end of 2000 is estimated to be 0.25.

SITE E

Description of site

Site E serves a large coastal city and its surrounding municipalities. The landfill has been in operation since 1989 and is the disposal facility for the majority of municipal waste residues generated in the area. These include fly ashes from two different MSW incinerators, one of which uses a wet-scrub system, while the other uses a semi-dry scrubbing process.

The landfill is sub-divided into cells, which are used for the mono-disposal of wastes from different sources. The landfill is clay-lined and each cell has a leachate collection system consisting of a gravel drainage layer with slotted pipes at 15 m spacing, leading to a single collection well. Samples are taken for analysis at 2-monthly intervals from these wells. Leachate flows by gravity from the collection wells to a sump and is abstracted from the sumps and pumped into a common main, and thence to a treatment plant. The control over leachate levels is that there should be no standing leachate on the site base in an observation well located at the opposite corner of each cell to the that is collection well.

The base of the site is approximately 2m below the external groundwater level and leachate levels are normally maintained below this with the collection pipes part full, so there is thought to be some seepage of groundwater into the landfill. All cells are filled with waste to a depth of 6m and finished to a flat contour. When full, cells containing APC residues are capped with a geomembrane, over-topped by restoration soils. Cells containing vehicle fragmentiser waste are covered with 200mm of root-blocking gravel, 800 mm soil and 200 mm of top soil.

The total volume of waste in place by June 2000 was ~1.5 Mm^3. The total tonnage in place by the end of 1999 was 1.74 Mt, suggesting an in situ density on the order of 1.15 t/m^3.

Waste inputs

Waste inputs during 1999 were 103,000 t. The waste inputs since the site opened have comprised:

- combustible waste (stored temporarily, awaiting incineration);
- mixed inorganic wastes (insulation, porcelain, etc.);
- MSW incinerator fly ash and APC residues from wet scrub process;
- MSW incinerator fly ash from semi-dry process; APC residues from semi-dry process;
- street sweepings
- contaminated soil;
- asbestos;
- vehicle fragmentiser wastes.

Wastes are segregated into different cells.

The combined fly ash and APC residues from wet scrubbing are placed in dedicated cells, together with fly ash from the semi-dry scrubbed incinerator. The APC residues from dry scrubbing are mono-filled in separate cells. The vehicle shredder wastes are mono-filled in separate cells.

Water balance and L/S information

Limited data on daily leachate volumes from the whole site since early 1994 are shown in Figure A1.E.1. Monthly volumes during 1999 are shown in Figure A1.E.2.

Total site capacity	1.939 Mm^3
Proportion of site base now covered with waste	100 %
Total capacity divided by 6m waste depth	323,000 m^2
Approximate site dimensions ~360 m x ~900 m	324,000 m^2

∴Site catchment area taken as	32.3 ha
Leachate volume in 1999	74,570 m^3

∴Average percolation over 32.3 ha	231 mm/a

Monthly leachate volumes in 1999: min (July)	3,142 m^3
max (Nov)	10,687 m^3

Assume July volume is entirely derived from groundwater seepage, at uniform rate.

∴ Annual groundwater contribution (12 x 3,142)	37,704 m^3
∴ Groundwater contribution as % of total flow	50.6 %
∴ Rainfall-derived leachate, averaged over whole site is	~115 mm/a

This infiltration figure has been used to derive estimates of L/S ratio for the cells of interest to this study. The calculations are shown in Table A3.E.1 below. The estimates are subject to uncertainty because several of the cells have been membrane capped or part-capped. The site average percolation estimate of 115 mm/a is probably an under-estimate of percolation for un-capped areas and an over-estimate for percolation on capped areas. Unfortunately, the operator does not have separate flow data for individual cells, so it is not possible to estimate L/S ratio any more accurately.

Table A3.E.1 Estimation of L/S ratio in cells at Site E

Area (m^2)	Age, to end 2000 (years)	Annual leachate (m^3/a)	Total leachate (m^3)	Mass of waste (t)	L/S ratio
6,090	7	700	4,900	28,279	0.17
2,678	4	308	1,232	16,605	0.07
7,893	3	908	2,723	8,918	0.3
16,673	11	1,917	21,087	92,375	0.23
3,154	4.5	363	1,634	18,924	0.09
9,833	4	1,131	4,524	39,707	0.11
7,290	7	838	5,868	39,858	0.15

Notes to table:
1. Cell areas have been derived from cell capacity data, divided by 6 m waste depth.
2. Annual leachate volume is based on site average percolation of 115 mm.
3. Mass of waste is based on an assumed density of 1 t/m^3 and the operator's data for the volume of airspace used so far.
4. The age of the cells is derived from the start of the leachate equality data.

The calculated L/S ratios ignore the contribution from groundwater seepage. This seepage is assumed to be largely removed in the leachate collection system and to have very little contact with the wastes.

Leachate quality data

Time series data have been obtained for the seven cells listed in Table A3.E.1, and are presented and discussed in the relevant sections of the main report.

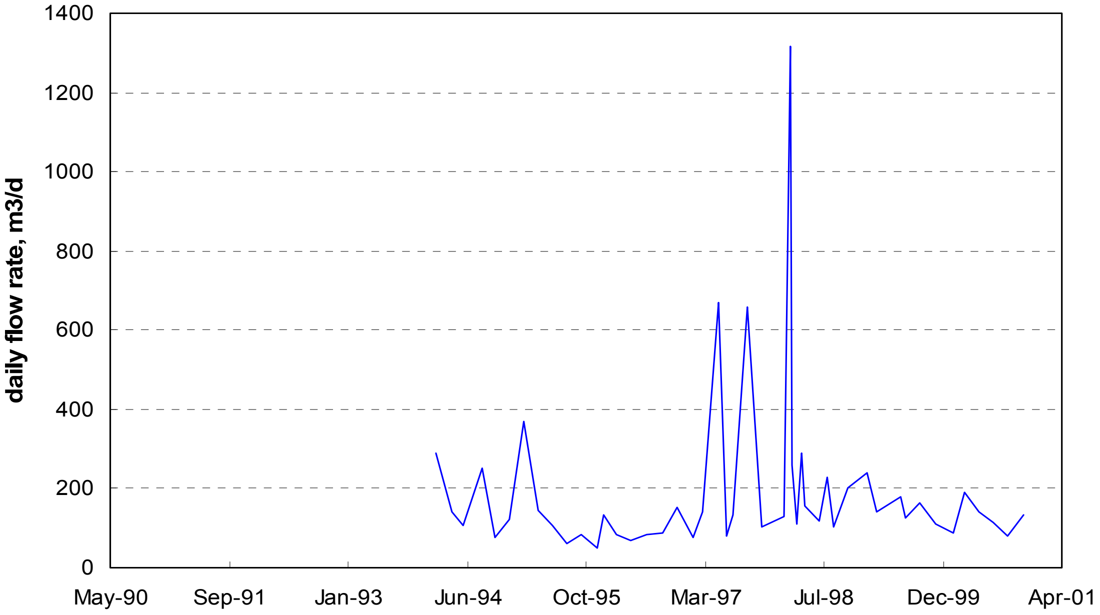


Figure A3.E.1 **Recorded values of daily flow rate for leachate from Site E**

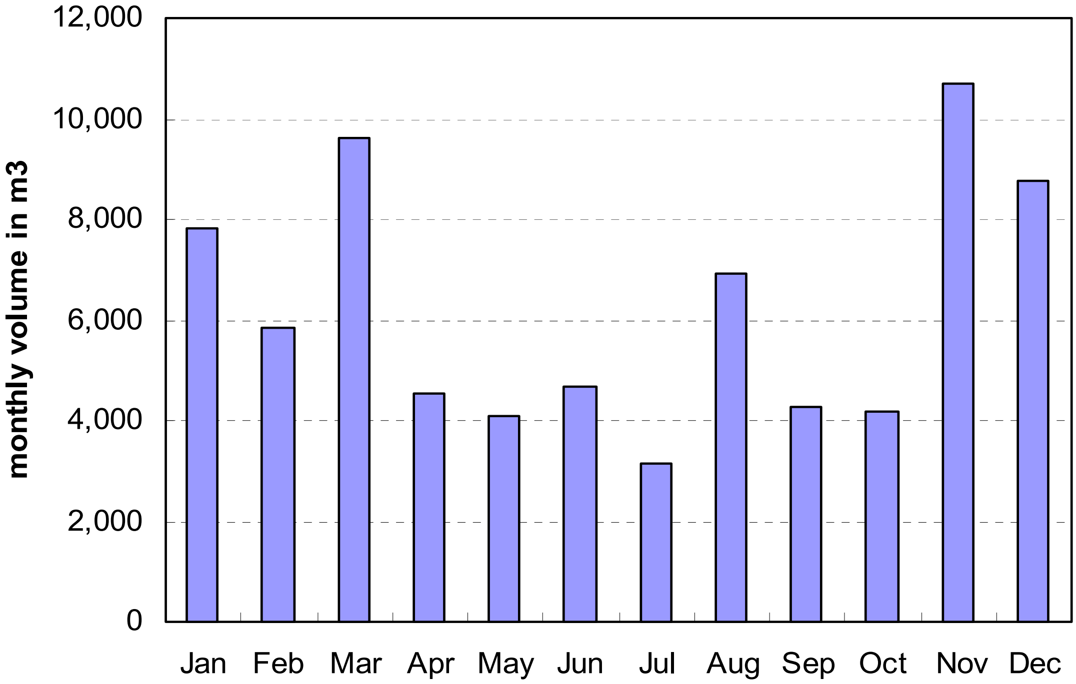


Figure A3.E.2 **Monthly leachate volumes from Site E during 1999**

SITE F

Description of site

The site receives predominantly residues from a single treatment facility for hazardous wastes plus small quantities of treated hazardous wastes from external sources.

It is located on limestone strata in a rural setting, adjacent to the coast. The landfill has been in operation since 1975. As a result, the extent of cell engineering varies, reflecting the evolution that has occurred during the last 25 years: up to the mid 1980s, the cells were unlined and leachate from these cells soaks away into groundwater and thence to the sea. Small lined collection lysimeters (typically 100 to 1,000 m^2) were constructed in the base of these early cells to allow leachate to be sampled and to allow the total quantity for the cell to be estimated (by proportion). Since the mid 1980s, cells have been constructed with engineered liners (either geomembrane or compacted clay, and a 250 mm gravel leachate collection layer. Leachate from these later cells is removed for disposal, with the disposal route being dictated by the analysis of the leachate.

The footprint of the area currently filled with wastes is ~9.1 ha and the volume of waste in the landfill is ~700,000 m^3. There are currently 37 different points at which leachate is sampled and this is undertaken every 6 months.

Apart from cells used for slags, all cells at final level are capped with a geomembrane, drainage layer and restoration soil. Some cells have had a geomembrane cap placed at an intermediate level and have then been over-tipped with slag.

Little or no compaction is used in the placement of the wastes. An area ~8 m deep was said to have settled less than 100 mm since its inception in 1993.

Waste inputs

Annual waste inputs to the landfill are currently ~30,000 t/a. The main types of waste received are:

- slag (bottom ash) from hazardous waste incinerators;
- filter cakes from inorganic liquid waste treatment;
- filter cakes from treatment of incinerator APC washwaters;
- gypsum from removal of SO_2 from incinerator flue gas;
- fly ash from hazardous waste incinerators;
- inorganic low leaching solid wastes in drums (small quantities).

The incinerators burn much larger quantities of hazardous waste per capita than is currently incinerated in the UK. The average inorganic content of the incinerated wastes is therefore likely to be considerably higher than the average in the UK and this may result in the quantity and characteristics of the residues differing from those currently produced in the UK.

The wastes are segregated before landfilling. Separate cells are currently used for the following wastes or combination of waste:

- HWI fly ash in big bags;
- filter cakes and gypsum;
- bottom ash (slag);
- drums.

Historically, fly ash was landfilled loose, in the same cells as the filter cakes. Since 1994 it has been landfilled separately and contained in 'big bags'.

The average organic content of the landfilled wastes is low and is controlled by a leaching test limit.

Leachate régime and L/S ratio

The leachate collection system is normally maintained virtually empty, with no significant zone of saturation. The operator reports some evidence of calcium carbonate deposition in the pipes and the drainage materials.

Leachate volumes have been either measured or estimated by the operator, for several areas of interest to this study. These data have been used to estimate L/S ratio for these areas, as follows.

Current operational stage (1994 to present)

This stage has separate cells for fly ash in big bags, mixed filter cakes and drummed wastes. Leachate volume data are for the combined area of all 3 cells:

waste inputs	~30,000 t/a
∴estimated total waste quantity 1994 to 2000	~200,000 t
combined leachate volume 1994 to 1999	43,390 m^3
leachate volume estimated for 2000	8,500 m^3
∴total leachate volume, 1994 to 2000	~52,000 m^3
∴L/S ratio	~0.26

Previous operational stage filled ~1990 to 1993

This stage has two separate cells, one for filter cakes, and one for a mixture of concrete plus drummed wastes. Leachate volume data are for the combined area of both cells:

area of stage ~95m x ~120m	11,400 m^2
estimated depth of wastes	8 m
∴estimated volume of wastes	91,200 m^3
assume in situ bulk density	1.5 t/m^3
∴mass of wastes in stage	136,800 t
leachate volume 1990 to 1999	19,397 m^3
∴L/S ratio by end of 1999:	0.14

Previous cells for HWI bottom ash (filled 1975 to early 1990s)

Four unlined cells were mono-filled with bottom ash, from 1975 up to the early 1990s. Information on leachate volumes was only available for this study from 1987 onwards. In all cases these are estimated, based on the volumes collected in lined lysimeters in the base of each cell, the assumption being that each lysimeter collected leachate in proportion to its area. The waste quantity in each cell has been estimated from its dimensions. The calculations of L/S ratio are thus subject to a great deal of uncertainty, particularly regarding leachate produced before 1987. The calculations are summarised below.

	Cell 1	**Cell 2**	**Cell 3**	**Cell 4**
filling dates	1975-79	1979-1981	1982-1985	1986-2001
area (m^2)	4,000	3,700	8,750	2,,650
depth (m)	~8	~8	~8	~8
volume (m^3)	32,000	29,600	70,000	21,200
∴estimated mass at 1.5t/m^3 (t)	48,000	44,400	105,000	31,800
∴estimated leachate volume 1987-99 (m^3)	3,282	1,144	25,713	16,650
∴L/S ratio, 1987-99	0.07	0.03	0.24	0.52
estimated total L/S ratio by 2000	0.15	0.1	0.3	0.6

Leachate quality data

The operator has supplied hard copies of laboratory analysis sheets for all leachate sampling points, from 1994/5 to 2000. From these, it has been possible to derive time series graphs of 24 parameters for four categories of waste relevant to this study:

- hazardous waste incinerator fly ash, landfilled 1994 to present;

- hazardous waste incinerator bottom ash, landfilled 1975 to early 1990s;
- mixed filter cakes and gypsum, from liquid waste treatment, plus HWI fly ash, landfilled late 1980s to 1994;
- mixed filter cakes and gypsum, landfilled 1994 to present.

Data for HWI fly ash are shown in the main body of the report as Figures 6.4.1 to 6.4.3. Data for HWI bottom ash (slag) are shown in the main body of the report as figures 6.3.1 to 6.3.3.

The two data sets for mixed filter cakes are shown in this Appendix, as Figures A3.F.1 to A3.F.6. For the current cell (Figures A3.F.1 to A3.F.3) there are sometimes two sets of values for the same date: this reflects results from different collection points within the cell and indicates the spatial variability of leachate quality. Flow data from the different points are not available, so it is not possible to indicate a reliable average for the cell from these analyses.

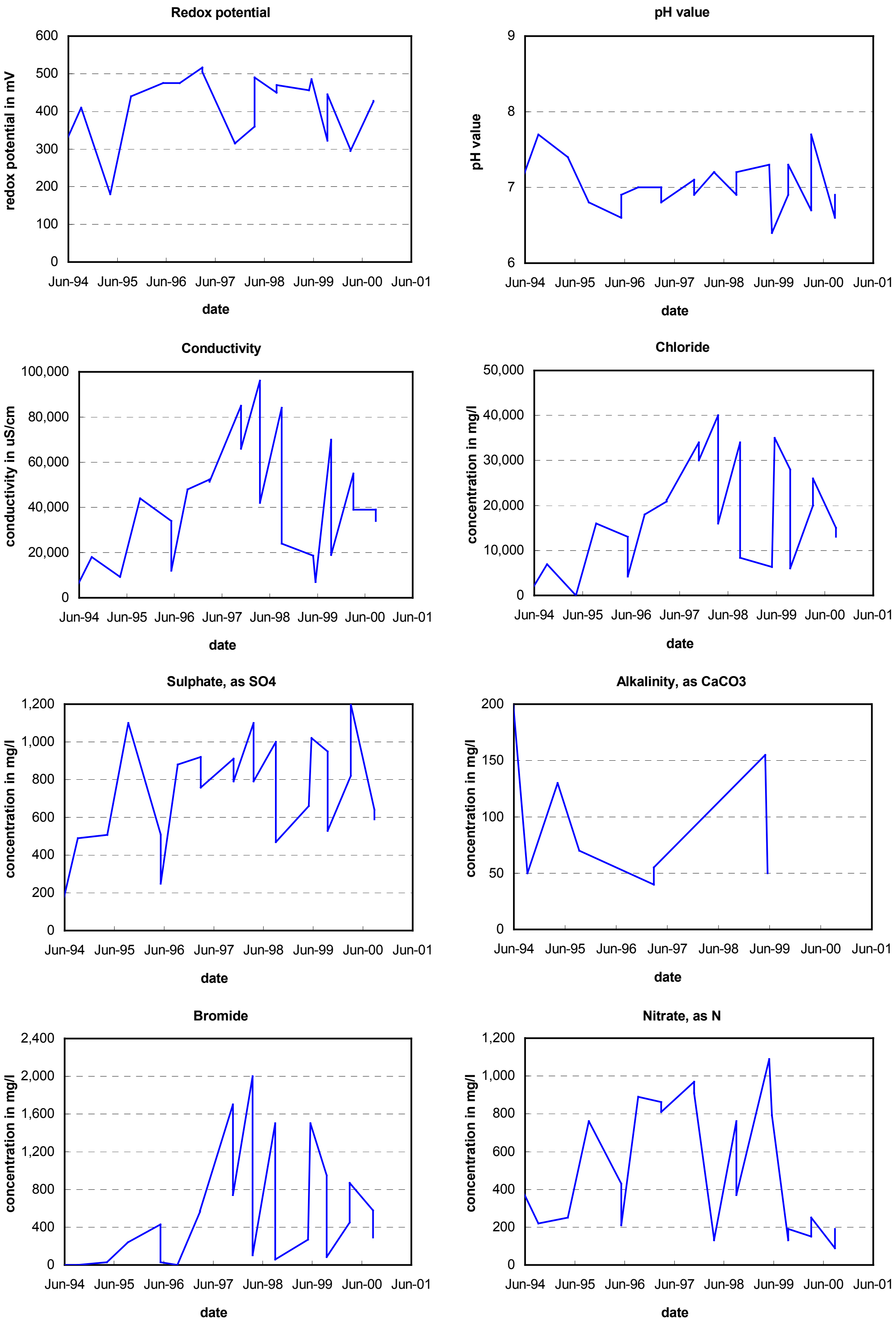


Figure A3.F.1 **Leachate quality from Site F, post-1994, mixed filter cakes including APC wash water sludges**

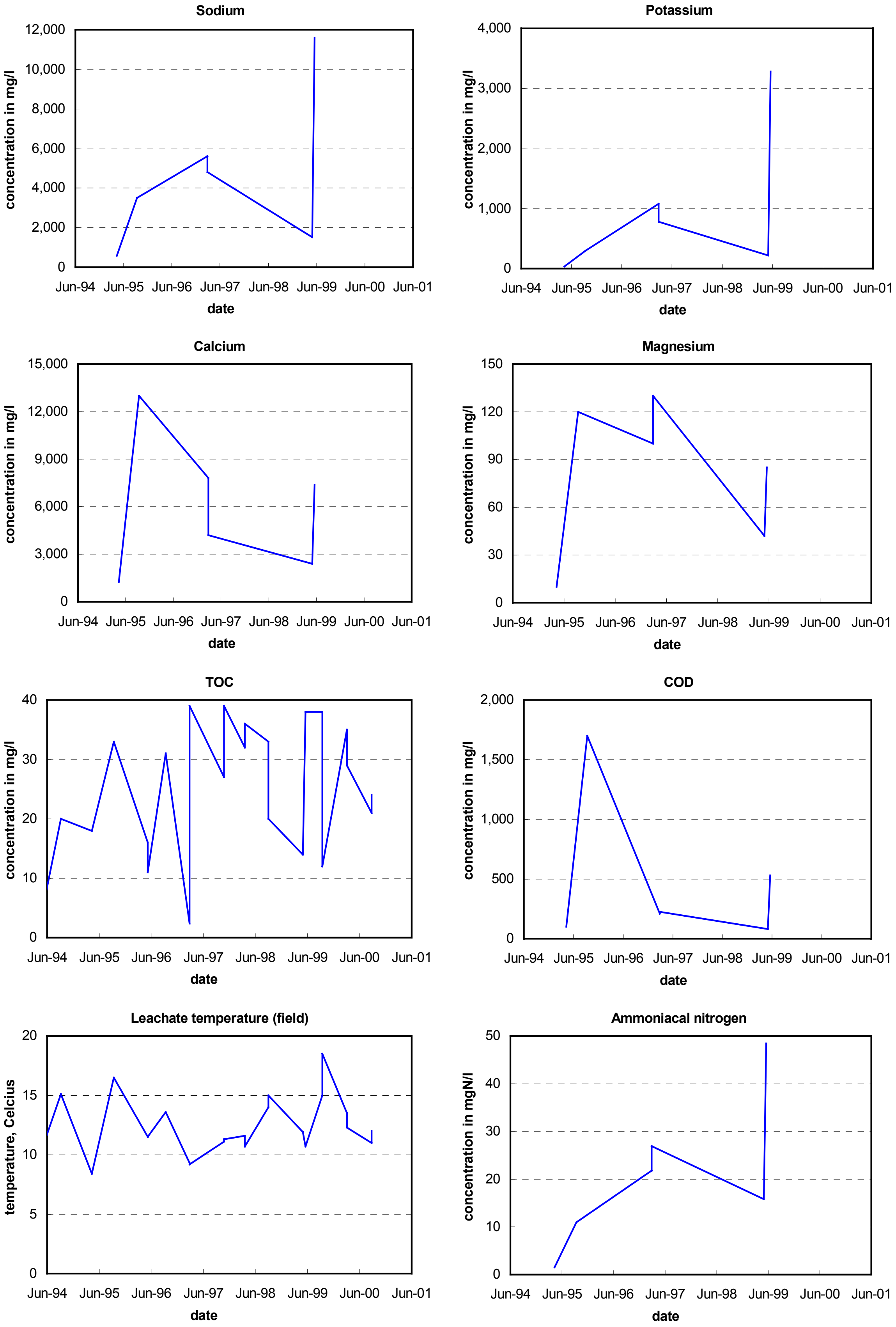


Figure A3.F.2 **Leachate quality from Site F, post-1994, mixed filter cakes including APC wash water sludges**

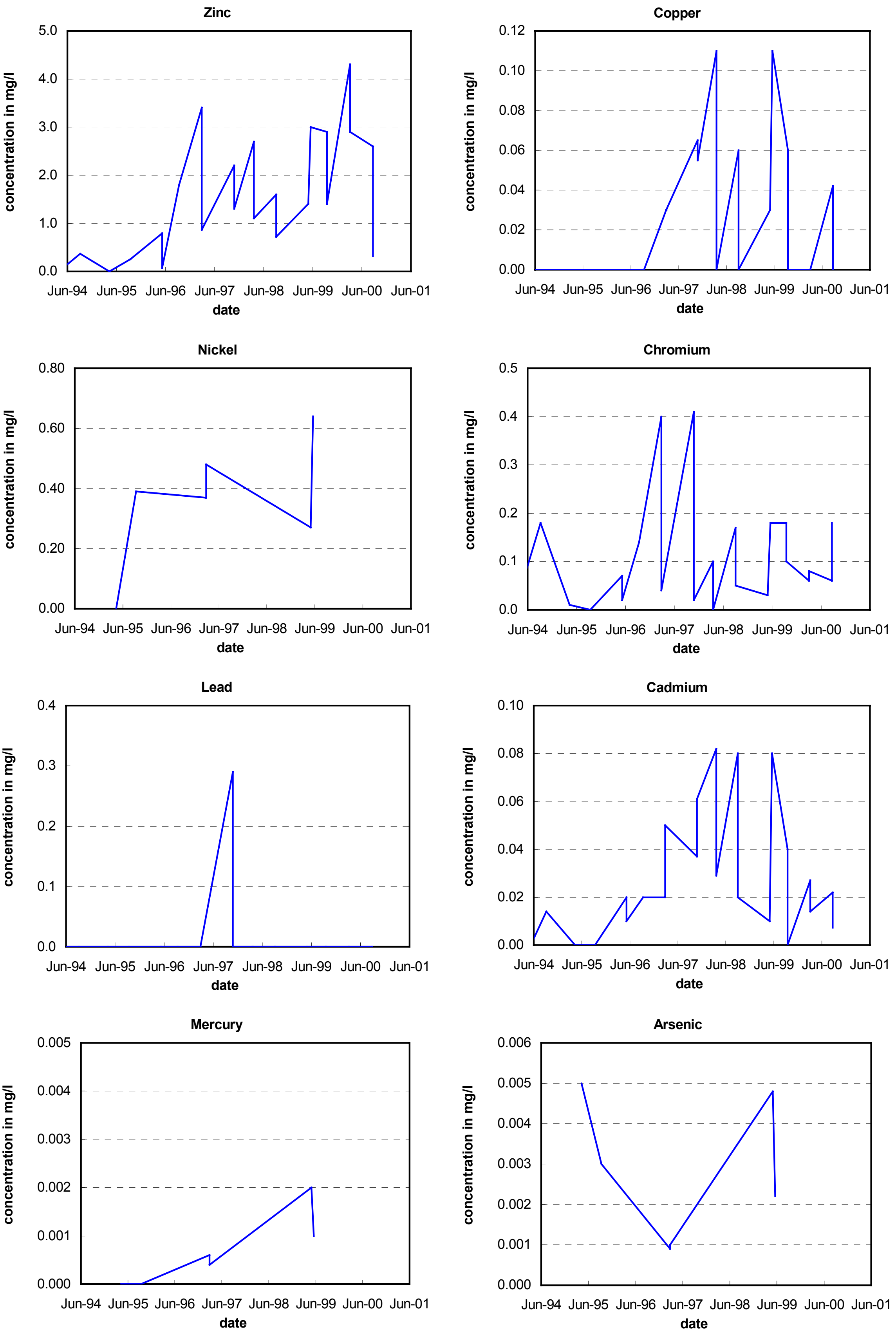


Figure A3.F.3 **Leachate quality from Site F, post-1994, mixed filter cakes including APC wash water sludges**

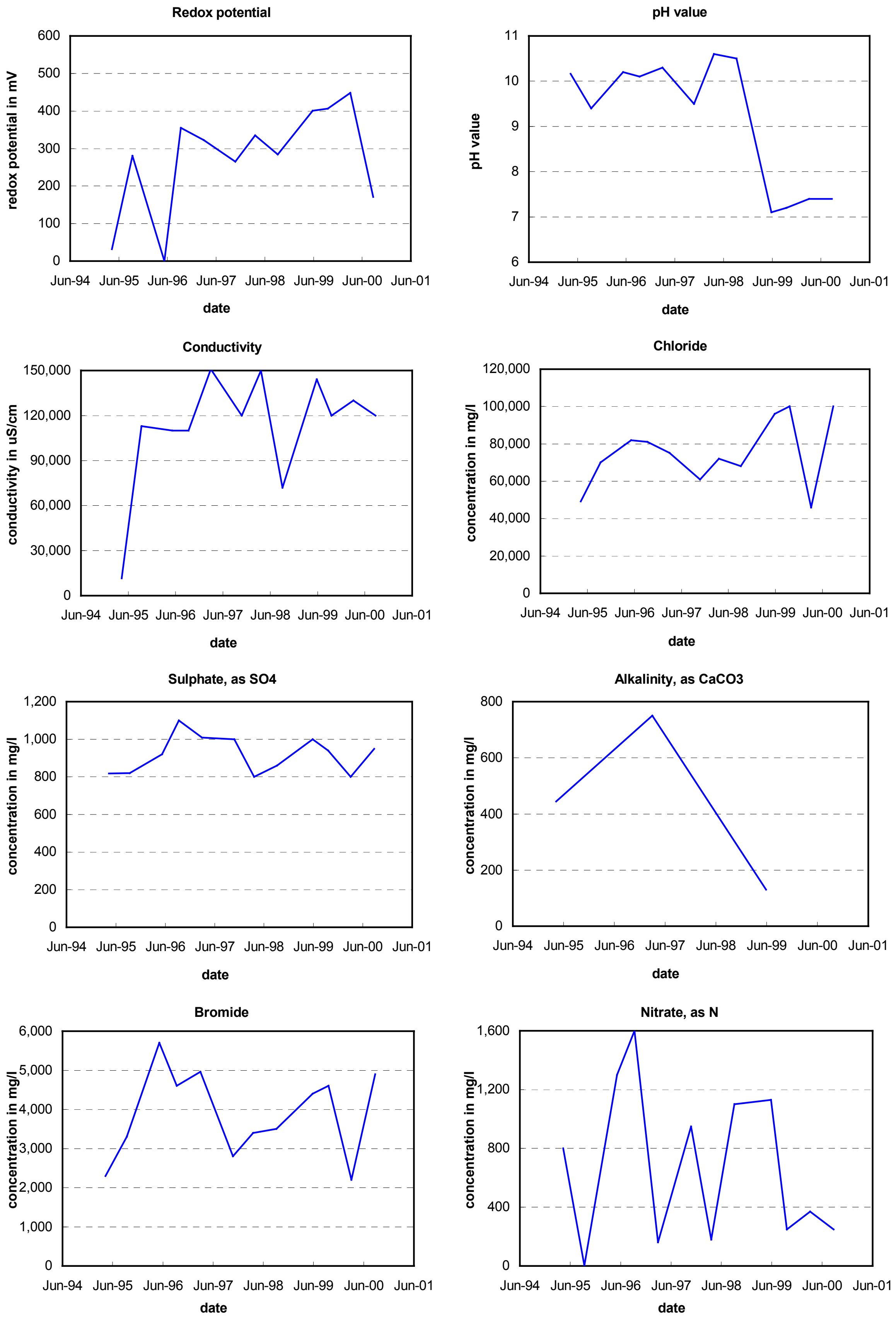


Figure A3.F.4 **Leachate quality from Site F, pre-1994 area, mixed filter cakes including APC sludges and fly ash**

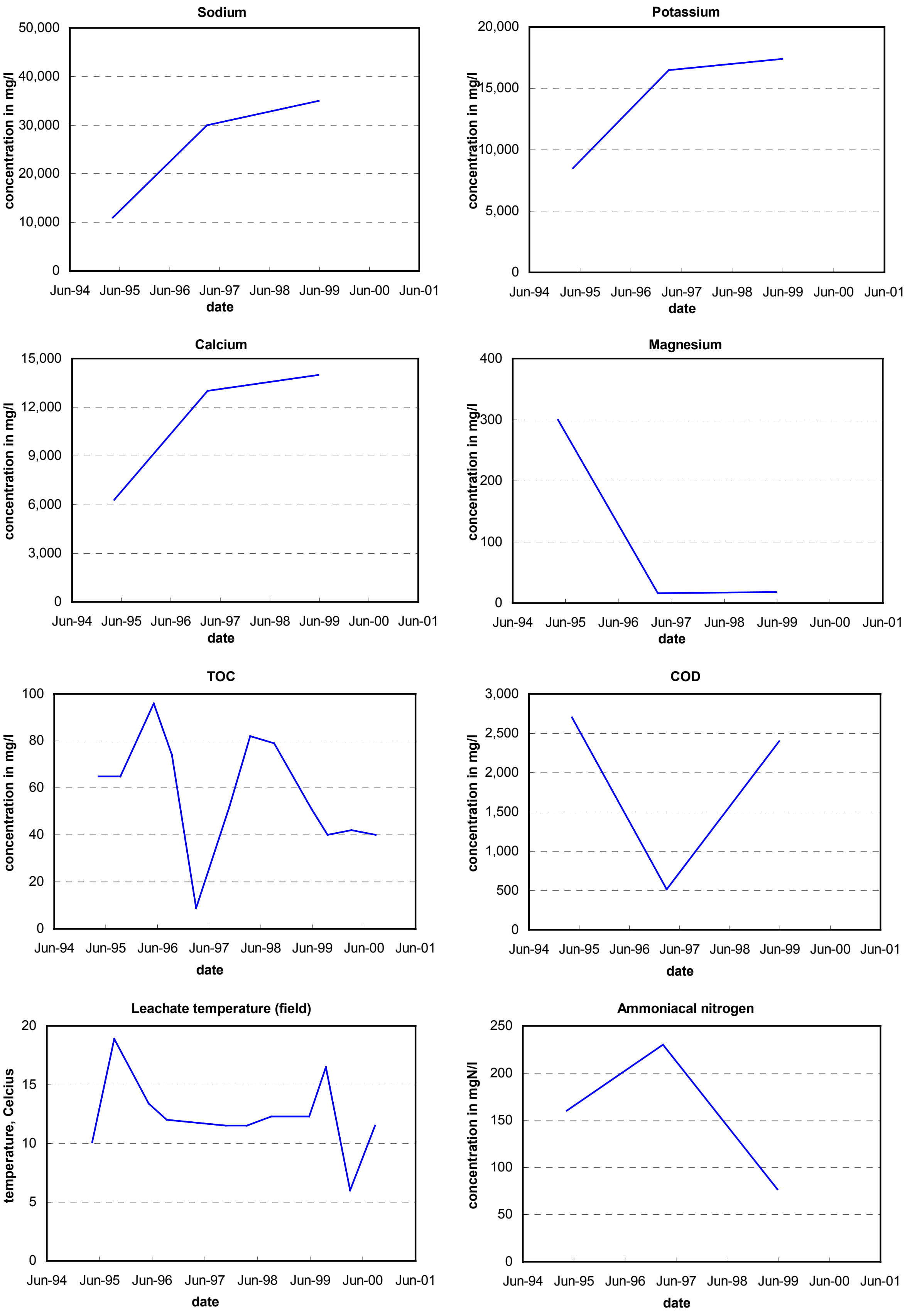


Figure A3.F.5 **Leachate quality from Site F, pre-1994 area, mixed filter cakes including APC sludges and fly ash**

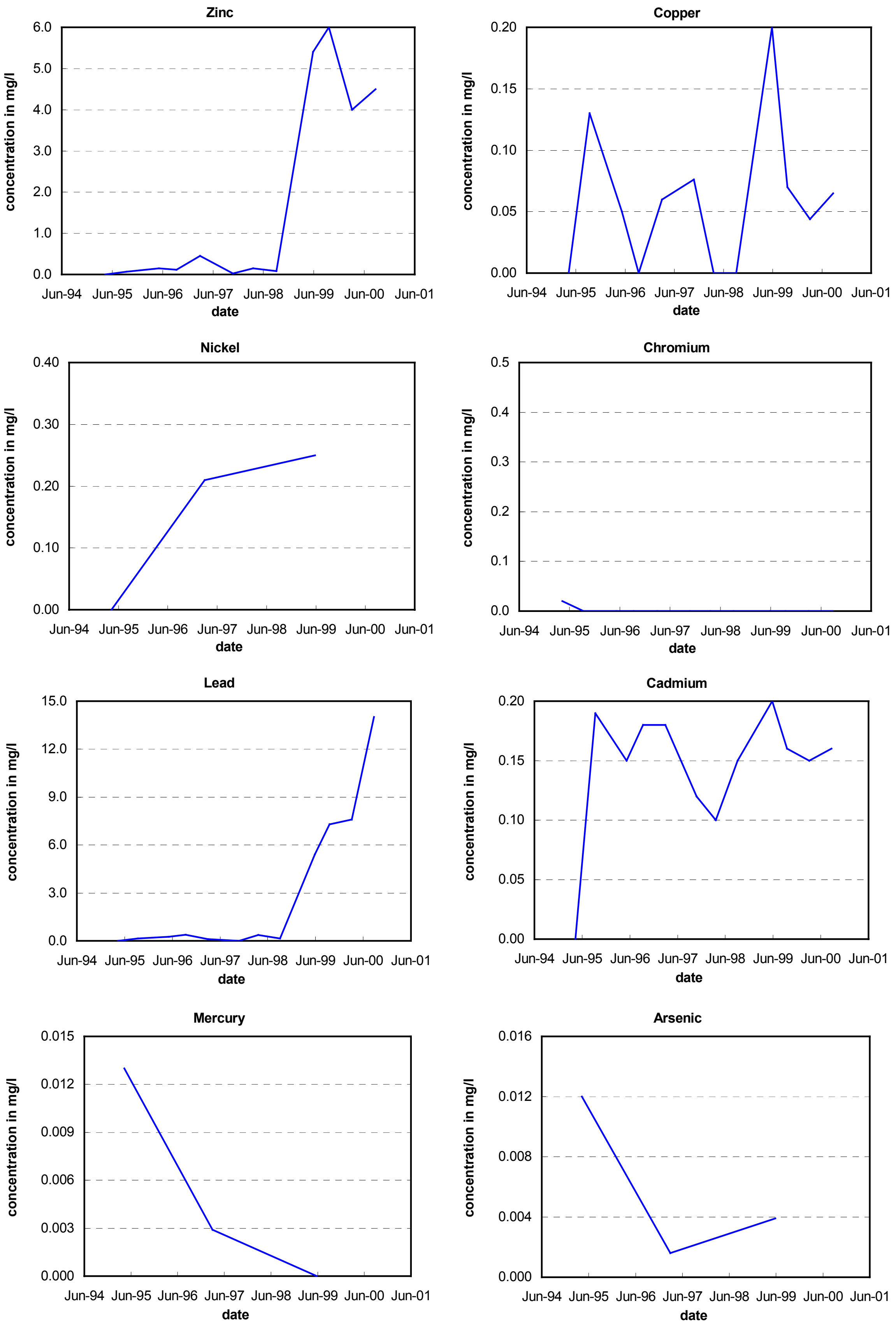


Figure A3.F.6 **Leachate quality from Site F, pre-1994 area, mixed filter cakes including APC sludges and fly ash**

SITE G

Description of site
The landfill is a small, lined concrete-walled structure of ~1 ha. It receives ~4,000 t/a of solidified hazardous wastes, mostly placed in bulk, but including some that have been cast into ~1m cubes. Data for this study come from one part of the site where three rectangular concrete bays were filled during 1999 to a depth of ~3 m. The wastes were finished to a flat profile and then covered with a geomembrane cap. Subsequently, in April 2000, the geomembrane was removed from one of the three bays, allowing rainfall to infiltrate. A single collection system serves all three bays and it also received surface run-off from the geomembranes on the other two bays. The collected leachate that is analysed is thus diluted approximately threefold compared with the actual leachate from the uncovered cell on its own.

Waste inputs
The stabilized wastes come from a wide range of sources, but major inputs include: fly ashes and APC residues from incineration of municipal and hazardous wastes, sludges from metal finishing and other industrial processes, metal working wasted, foundry sands, slags, catalysts etc. Typically, one tonne of solid hazardous waste will be mixed with 300-500 kg of reagents (of which cement is a major portion) and 400 to 500 litres of process water to facilitate mixing, reaction and hardening.

Leachate volumes and L/S ratio
Leachate volume and quality data were obtained for a period of ~1 year, from April 2000 to March 2001. Estimation of L/S ratio during that time is as follows.

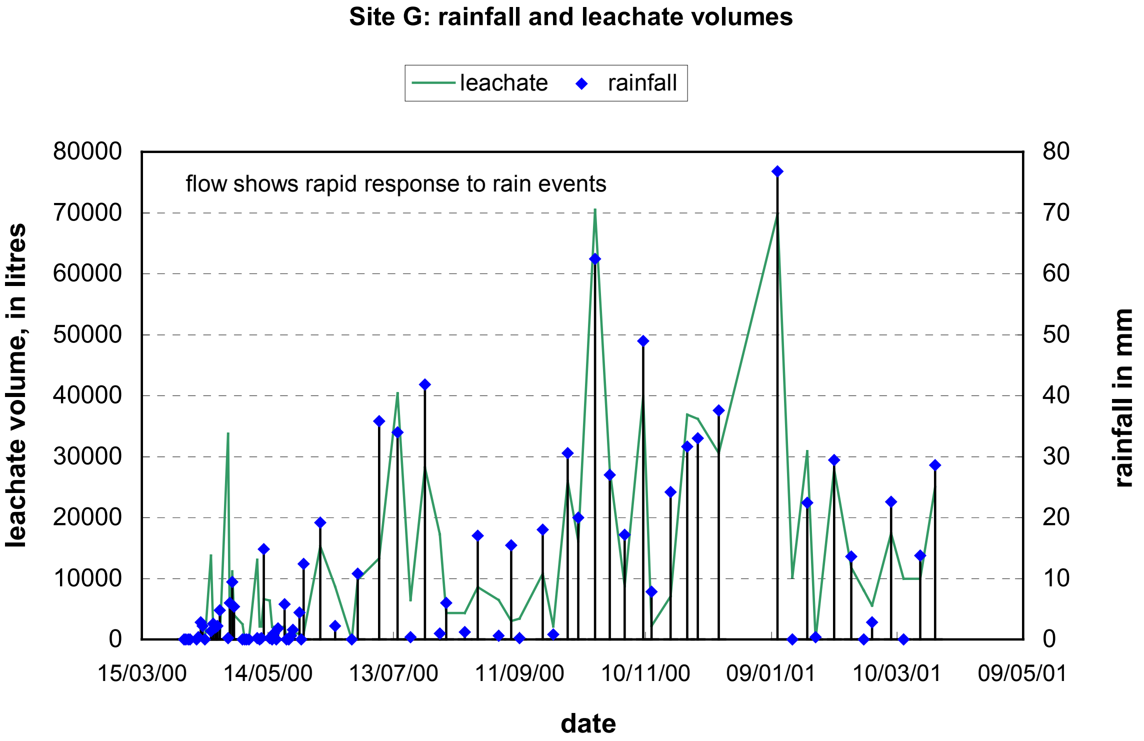
quantity of waste and reagent in uncovered bay	2,313 t
volume of waste and reagent	1,560 m^3
area of uncovered bay	520 m^2
∴depth of waste	3.0 m
area of all three bays	1,850 m^2
∴area of uncovered bay as % of whole	28 %

total leachate collected in period	819 m^3
total leachate as rain depth on 1,850 m^3	443 mm
leachate attributable to uncovered bay	229 m^3
∴L/S ratio in uncovered bay	0.1
actual rainfall in period	838 mm
∴assumed loss by evaporation/absorption	47 %

Leachate volume and rainfall data during the period are shown in the graphs below as Figure A3.G.1.

The graphs show a rapid response of volumes to rainfall events. This is not simply due to the shallow depth of wastes but also because ~70% of the catchment has a geomembrane top cover.

Leachate quality data
Analyses have been obtained for 14 parameters on eighty samples taken over the course of the ~1 year period. These have been used to prepare time series graphs which are presented and discussed in the main body of the report (Section 6.2, Figures 6.2.1 and 6.2.2).



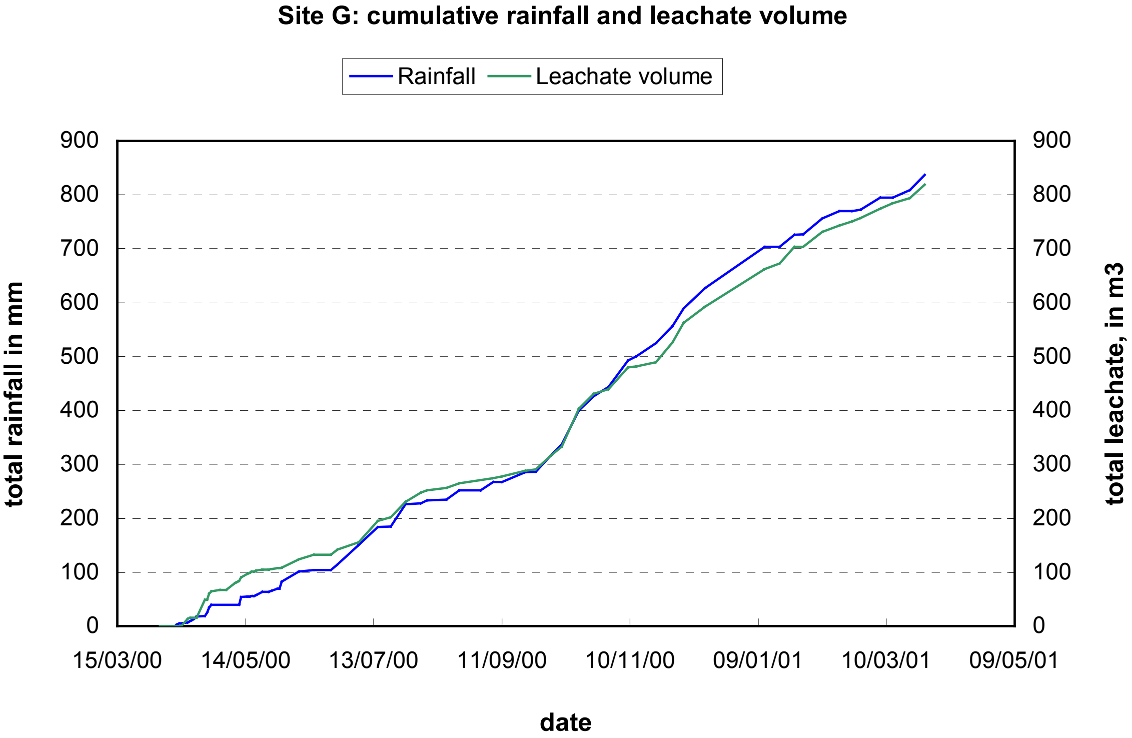


Figure A3.G.1 **Leachate volumes at Site G: two capped and one uncapped cells for solidified wastes**

SITE H

Site H was not visited during the study but data and descriptive material were obtained from the operator.

Description of site
The phase for which leachate data have been supplied began operation in ~1996. Cement-stabilised hazardous wastes are landfilled in one cell, while APC residues from MSW incineration are landfilled in an adjacent cell. The leachate collection systems for the two cells lead to a common collection point so that the leachate is a mixture from the two types of waste. The cement-stabilisation plant is part of the facility and receives a wide range of hazardous wastes, of which the principal categories are: contaminated soils; fly ash; inorganic wastewater treatment plant sludges and filter cakes; paint wastes; metallurgical residues. The stabilised waste is landfilled in bulk within mobile shuttering, at a consistency similar to a dry mix concrete. The APC residues are landfilled in ‘big bags’ but the specification of the bags is not known. All of the wastes have a low organic content, achieved by a limit on leachable COD.

The cells are engineered for containment and have leachate collection systems, but the exact specifications are not known.

No information was obtained on leachate volumes but an upper estimate of L/S ratio can be derived from the reported fact that all of the leachate generated at the site is used as part of the process water in the cement-stabilisation plant. An approximate value of L/S is calculated as follows:

Typical mix in stabilisation plant:	solid hazwaste	1 t
	reagents	300-500 kg
	process water	400-500 kg
	density of product	1.6–1.8 t/m^3
Waste inputs to whole facility		30,000 t/a
Estimate of quantity to stabilisation plant		20,000 t/a
∴ maximum requirement for process water		10,000 m^3/a
∴ inferred maximum rate of leachate generation		10,000 m^3/a
Total mass of material landfilled, including reagents and process water		50,000 t/a
∴ maximum L/S ratio		0.2

It is therefore inferred that the L/S ratio is very low and that leachate strength is close to its maximum.

Leachate quality data
Monthly data were obtained for 19 parameters, for the years 1998 to 2000 inclusive. The data are shown as time series graphs in Figures A3.H.1 to A3.H.3. Two parameters are not graphed, namely:

resistivity	(this is simply the inverse of conductivity); and
Cr(VI)	(total chromium is graphed; Cr(VI) values were all ≤ 0.02mg/l and most were <0.01mg/l).

The data are discussed in the main body of the report, in section 6. There is no way of assessing the relative dominance of the two categories of waste input in determining leachate quality.

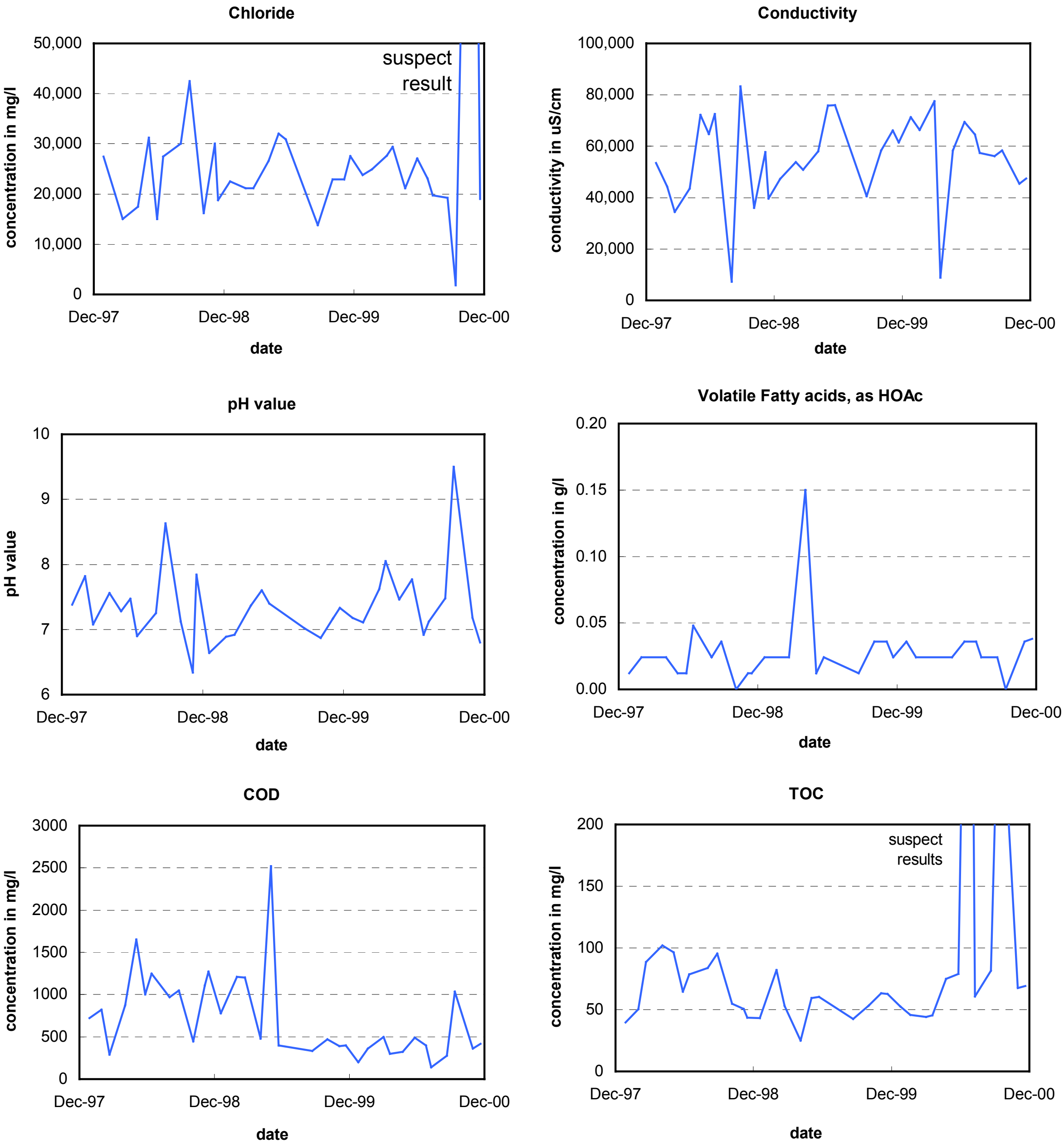


Figure A3.H.1 Leachate quality from Site H

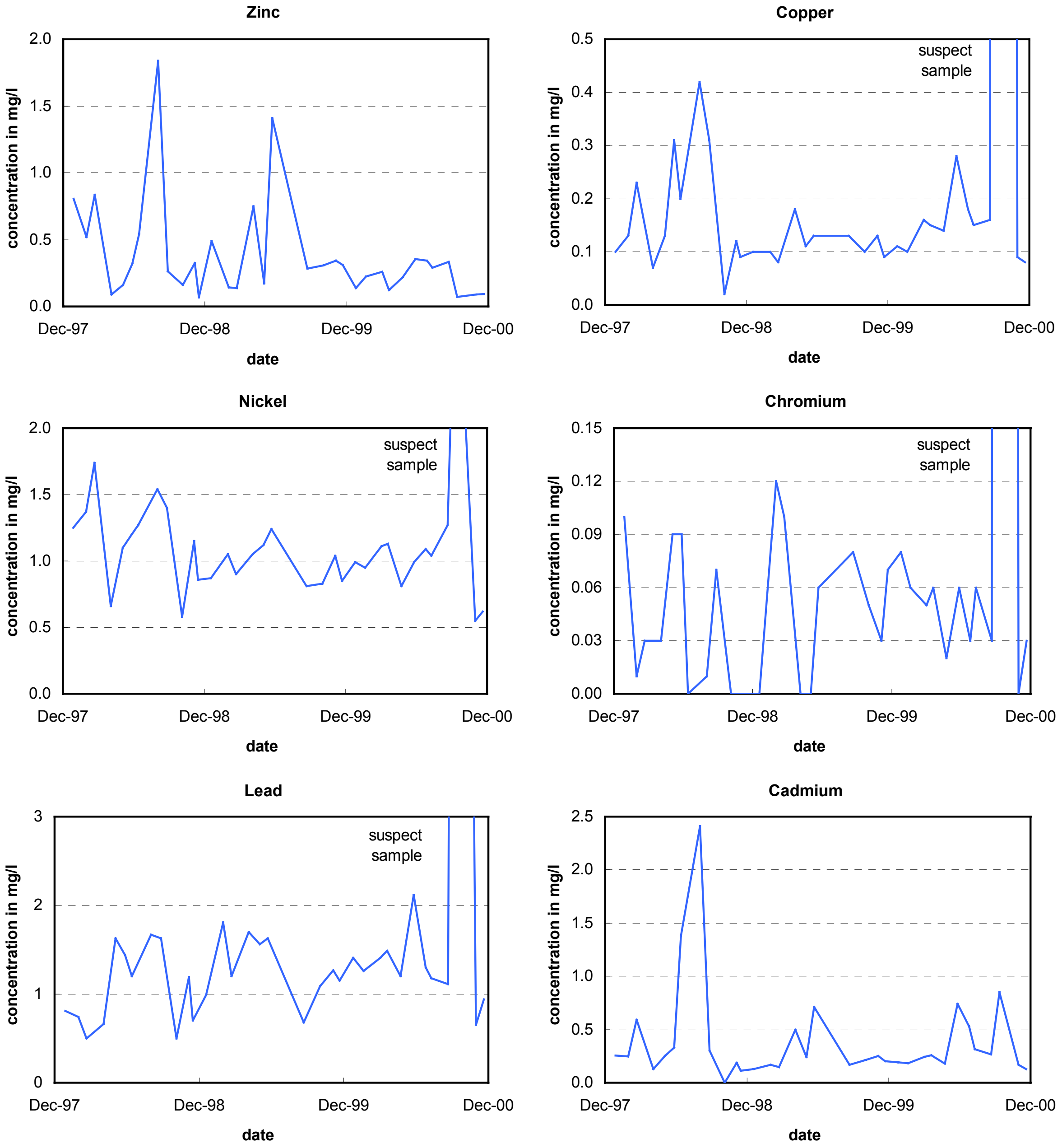


Figure A3.H.2 Leachate quality from Site H

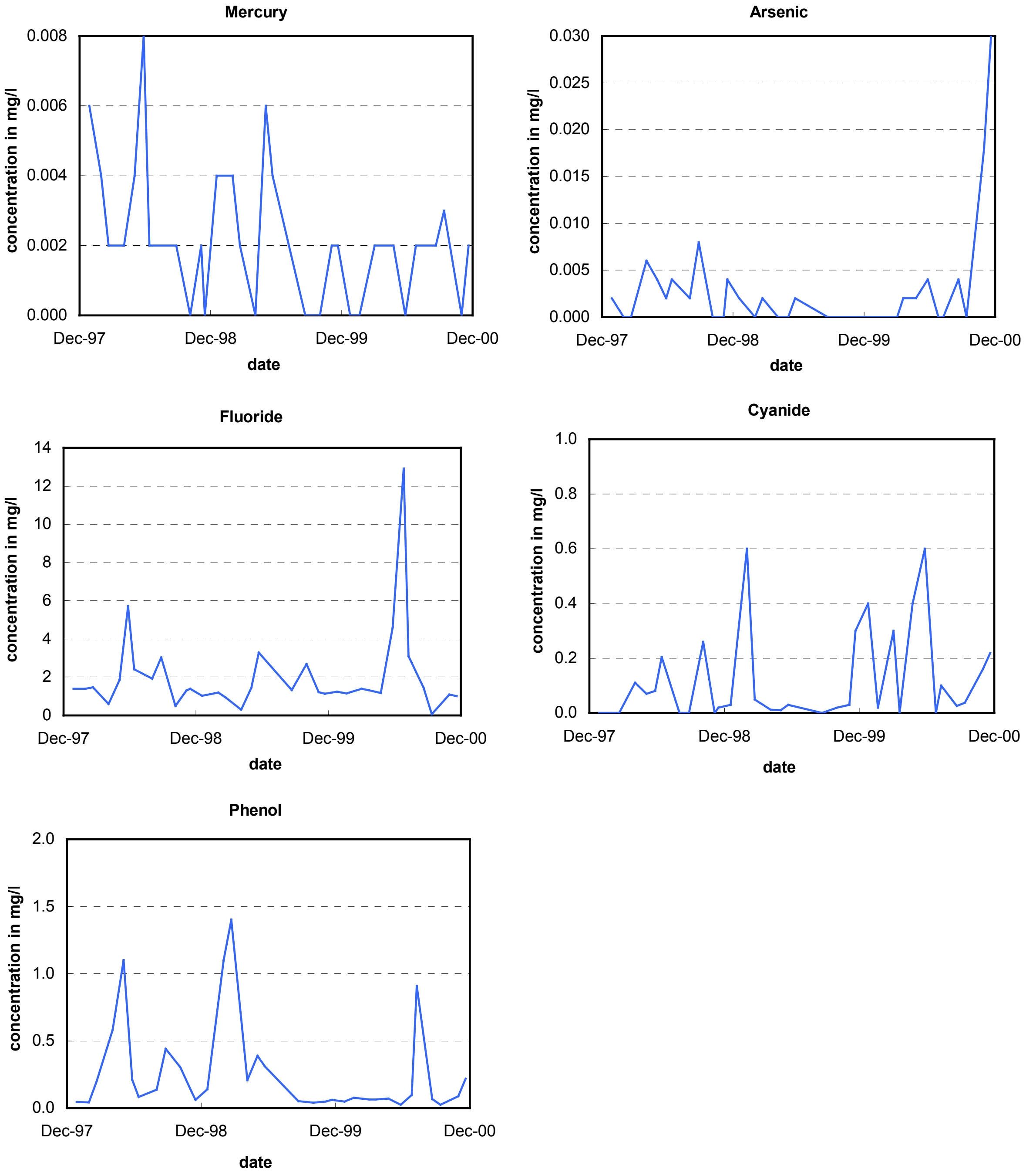


Figure A3.H.3 Leachate quality from Site H

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