CRANFIELD UNIVERSITY

R SMITH

TOWARDS SUSTAINABLE LANDFILL MANAGEMENT

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Richard Smith

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Supervisor: Professor Simon Pollard

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ABSTRACT

The UK is reliant on landfill as a waste management option with some 72% w/w of municipal waste landfilled in 2003/04. This thesis advances an argument that landfill, as practised historically and currently, is unsustainable. This thesis demonstrates, specifically, that current legislative aftercare provisions of 30-60 years are inadequate with reference to modelled landfill completion times (the achievement of equilibrium status) of up to 2,000 years. Uniquely, the research quantifies the scale and significance of methane emissions during the early stages (up to 28 months waste age) of landfill operations at 21 UK landfills using a modified flux box. The onset of methanogenesis is quantified for the first time, using a series of *in situ* monitoring probes installed at one UK landfill site.

A significant and novel finding is that the advective flow of landfill gas is preferentially dominated by lateral movement, evidenced here by examination of a predominantly municipal solid waste (MSW) landfill site in Southern England. The direct implications of this finding for the design of landfill gas management systems are discussed.

For future landfills, this research has examined a number of UK scenarios in which the gas and leachate characteristics from waste residues going to landfill are modelled to the point of completion or achievement of equilibrium status. This analysis now allows for a comparative assessment of the future performance of landfills. Under these scenarios, completion times can be reduced in some instances (*e.g.* landfilling of compost and mechanical biological treatment residues) and extended in others (incinerator bottom ash). Problematic contaminants remain; notably arsenic, chromium and lead.

Using the research herein, the work describes the application of a landfill gas management hierarchy. In part response to the requirements of the EU Landfill Directive, this provides a science-based framework for operators to maximise protection of the environment and human health from gaseous emissions. The research provides evidence that can be used by landfill managers seeking to maximise landfill collection efficiency. This contributes to regulators managing public and environmental health and is increasingly significant for climate change.

Keywords: landfill, gas, risk, regulation, management, sustainability, waste treatment

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A significant proportion of this research was carried out whilst my role was that of *Principal Scientist* and *Landfill Topic Leader* for the Waste Research and Development Programme at the Environment Agency. I thank my former colleagues at the Agency and the team of researchers who between them were responsible for developing and delivering a substantial landfill research programme, part of which is contained herein. Particular thanks go to Professor Jan Gronow for her inspiration, unwavering support and professionalism. To the community of landfill scientists out there that I have learned so much from I thank you all. Landfill research may not currently be *en vogue* but landfills are here to stay, so I urge you to stick with it. After 15 years of working with landfills, I still live in hope of one day setting foot on a sustainable landfill in the UK.

I am grateful for the dialogue with and support from landfill operators which has been vital for without it the site-specific data that underpins much of our research would not exist.

To my family, especially wife Rachel and sons Joseph and Benjamin, you can all expect to see more of me. I'm grateful for the endless delight, sustenance and sanity that you've all given me without question, as well as providing a constant reminder for me to return home in the evening.

I remain in gratitude to the wise counsel and inspiration of the late Professor Colin Ferguson, Nottingham Trent University, who encouraged me to embark on the PhD road of discovery.

DEDICATION

This thesis is dedicated to my family and in particular the memory of my father William Roger Smith and his unwavering optimism for life.

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ABBREVIATIONS

AD	Anaerobic digestion
APC	Air pollution control
BMP	Biochemical methane potential
BMW	Biodegradable municipal waste
BOD	Biochemical oxygen demand
CBA	Cost benefit analysis
COD	Chemical oxygen demand
DOC	Dissolved organic carbon
GHG	Greenhouse gas
IBA	Incinerator bottom ash
MBT	Mechanical biological treatment
MRF	Materials recovery facility
MRV	Minimum reporting value
MSOR	Mechanically sorted organic residues
MSW	Municipal solid waste
NFFO	Non-Fossil Fuel Obligation
PPC	Pollution Prevention and Control
RDF	Refuse derived fuel
ROC	Renewable obligation certificate
TOC	Total organic carbon
WAC	Waste acceptance criteria
WMP	Waste management paper
WQS	Water quality standard

1. GENERAL INTRODUCTION

This section of the thesis provides a synthesis and overview of the intellectual contribution made in the work that follows and a contextual discussion of significance. The principal results and discussion for each paper is described in subsequent chapters.

1.1 BACKGROUND

The UK is heavily reliant on landfill as a waste management option with some 72% w/w of municipal waste still being landfilled in 2003/04 (Department of the Environment Food and Rural Affairs 2005) compared with 38% w/w in France and 20% w/w in Germany (National Audit Office 2006). We are starting to kick the landfill habit; the UK landfilled 80% w/w of MSW in 2000 (Department of the Environment Food and Rural Affairs 2000). Reduced landfill reliance in England has so far cost Defra £0.34 billion, contributing to an increased recycling rate from 13% in 2001/02 to 23% in 2004/05 (National Audit Office 2006). Whilst this cost might seem high, it compares with a predicted UK fine of £0.18 billion per year if we fail to meet our biodegradable waste diversion targets under the Landfill Directive.

Of growing importance is the impact of landfill emissions on climate change. The Framework Convention on Climate Change entered into force on 21 March 1994. Having ratified this treaty, the main responsibility of the UK government is to develop a national strategy for addressing greenhouse gas emissions and adapting to expected impacts. The strategic target set was to return UK methane emissions to their 1990 levels by 2000. In Kyoto in 1997 it was agreed to reduce annual emissions of methane to an average of 5% below 1990 levels over the period 2008-2012. The increased control of landfill gas emissions is an essential part in the UK

Climate Impacts Programme to reduce methane emissions. The Environment Agency, as regulator, has encouraged greater collection and utilisation of landfill gas and promoted the use of methane oxidation as an effective gas control option where gas yields are insufficient to sustain flares or utilisation plant.

Landfill operators are facing a more pressing need for change. The EU Landfill Directive (Council of the European Union 1999) is changing the way waste is disposed of in the UK by setting specific requirements for the design and operation of landfills, and for the types of waste that can be accepted in landfills. In many instances, the Directive introduces a new approach to landfill management *e.g.* classification of sites into hazardous, non-hazardous and inert and the implications for future gas regimes are far from being fully understood. The Directive places emphasis on reducing methane emissions to global atmosphere which itself requires operators to install active gas management systems, usually in the form of combustion (flares or engines for utilisation). These combustion systems produce their own emissions which can be at the expense of local air quality objectives.

Research into landfill gas is not new. During the last two decades, since landfill gas schemes were first installed in the UK, a considerable portfolio of landfill gas research has been developed primarily by Government Departments and the Environment Agency for England and Wales to investigate the various aspects of landfill gas management. For example, fundamental research carried out at the Brogborough Test Cells (see **Chapter 2**) has demonstrated the role of landfill management techniques such as leachate recirculation, controlled air injection and sewage sludge addition, in accelerating the degradation timescales and their influence on gas production. The greater body of research has underpinned a series of policy and regulatory guidance documents that have culminated in the two most significant landfill gas outcomes to emerge in recent years; a move towards emissions based regulation (see **Chapter 3**) (Deed *et al.*

2003) and the development of a risk assessment framework for landfill gas management(Environment Agency 2004f).

Since the second edition of Waste Management Paper 27 *Landfill Gas* (WMP27) was published in 1991, the fundamental understanding of landfill gas science has improved significantly. For example, knowledge of trace gas composition is improving and it has been established that there are more than 500 trace components potentially present in landfill gas (Environment Agency 2003b). Whilst the underpinning knowledge of gas composition and emissions has improved, there have been a number of regulatory and legislative pressures that have forced the landfill gas agenda. Site-specific implementation of appropriate landfill gas management is arguably one of the biggest challenges facing site operators. WMP27 was not risk based and this was inconsistent with the Environment Agency's developing risk based framework for waste management regulation (Deed *et al.* 2003). Whilst landfills are now perhaps perceived as old technology, as the agenda turns towards waste reduction and treatment, landfill is nevertheless here to stay and the risks associated with landfill gas in particular must be well characterised and understood with appropriate risk management procedures in place. Broadly, there are two issues of concern; managing our landfill legacy and understanding changing waste streams and their impact on future landfill management.

1.2 POLICY CHALLENGES

For Government and its regulators, who are charged with providing signals and incentives to landfill operators, a number of policy challenges exist. These are explored in detail within the published papers within this thesis, but in summary are:

1. **Theoretical gas production.** The level of confidence in modelling versus reality is gradually improving. Modelled theoretical gas production provides a basis for landfill

operators to make strategic investment decisions about how best to manage landfill gas on a site-specific risk basis. There is a growing awareness that models such as GasSim tend to over-predict gas production. Timescales associated with landfill gas production are discussed in **Chapters 2 and 4**.

2. **Changing waste streams.** The characteristics of landfill gas will change as waste streams evolve, particularly as a result of classification of landfills and diversion or banning of selected waste streams under the EU Landfill Directive. **(Chapters 5, 6 and 7)**. The impact of waste treatment residues in particular, as they become the dominant waste that is sent to landfill, requires characterisation and assessment comparative to conventional landfills. Potentially, there are significant difficulties with landfilling some treatment residues (many residues are largely inorganic) as this material represents a much greater volume and concentration of waste and there may be specific contaminant problems, as demonstrated by metals and chloride from incinerator bottom ash (see **Chapters 6 and 7**). In particular, **Table 7.2** shows the distribution of specific elements in waste fractions, potentially allowing targeted contaminant removal at source. This in itself could have a potential impact on improving the quality of and reducing emissions from solid recovered fuel.
3. **Collection efficiency.** This is linked with the need to minimise emissions. Deriving a reasonable collection efficiency is challenging. Clearly it is not practical to require a 100% collection efficiency. Conversely if setting, for example, an 85% collection efficiency at the regulatory or operational level, it must be accepted that potentially 15% of gas generated will be emitted and these emissions will need to be managed appropriately. Flux box data from 21 UK operational landfills (see **Chapter 4**) has demonstrated the need for the timely installation of active gas management systems in working cells.

4. **Requiring operators to capture gas early.** Research shows that the scale of landfill gas emission in the early stages of landfilling can be significant yet there are operational practicalities of extracting gas from a working face *i.e.* vehicle movements often result in damaged gas collection infrastructure. Operators still only tend to collect gas from working cells to control odour rather than to manage the gas for environmental or health protection reasons (see **Chapter 4** and **Appendix C**).
5. **Nature of emissions from gas combustion.** Whilst utilisation of landfill gas as renewable energy is the preferred option in the landfill gas hierarchy (see **Chapter 3, Figure 3.2**), such systems produce their own emissions. As one example, emissions from engine exhausts may pose a greater problem than those from flare systems due to poorer dispersion in the local environment. The trade-off between reducing global emissions (of methane in particular) and having a negative impact on local air quality is a major consideration. The relative contribution of landfill gas emissions towards Local Air Quality objectives needs to be understood *e.g.* NO_x emissions from gas engines versus transport.
6. **Longevity of gas production.** Whilst it can be in the operator interests, for financial reasons, to prolong the utilising life of a landfill, the exponential decay aspect of gas production is predicted to last many decades. It is only in recent years that landfill gas production has started to peak in parts of some UK landfill sites, establishing timescales for the classical gas production curves (Farquhar and Rovers 1973) for the first time. However, there has been little incentive for operators to consider reducing uncertain timescales beyond peak production. The remaining third of landfill gas production curves remain largely theoretical; they are not underpinned by real data.
7. **Reappraisal of accelerated landfill stabilisation techniques.** There are techniques available to accelerate landfill stabilisation (*e.g.* pre-treating incoming waste, controlled

air injection, leachate recirculation, bioreactor landfilling). Whilst invariably such techniques have cost implications and are not without difficulty, the potential benefit in better environmental protection (potentially less emissions) and reducing liability over shorter timescales seems attractive and requires further consideration (see **Chapter 2** and **Chapter 7**).

8. **Financial aftercare provision.** Landfill operators are currently required to have financial aftercare provision in place for a 30-60 year period. Aside from the fact that landfill stabilisation criteria, on which to judge a landfill permit surrender application, are yet to be defined it is clear, based on the modelling evidence to date, that many UK landfills will require a much longer timescale likely to be measured in centuries rather than decades to achieve completion (see **Chapters 5, 6 and 7**). This brings into question the sustainability of landfills.
9. **Charging basis.** A more speculative challenge is that operators could in future charge customers according to the implications that accepting a particular waste stream has on achieving completion *e.g.* when considering the acceptance of waste, operators may wish to assess the impact of a waste stream on gas production or pollution potential.

1.3 LANDFILL SUSTAINABILITY

The sustainability debate that followed the Earth Summit in 1992 (United Nations 1992) focused attention on the longevity of waste degradation processes (Department of Environment 1995). Ultimately, economies must consider what place landfill has in the waste hierarchy and, accepting there will always be waste residues, understanding how landfill can become more sustainable. The waste strategy policy on landfill set out in the White Paper, *Making Waste Work* (Department of the Environment and the Welsh Office 1995), is to promote landfill practices which will return the products of stabilisation to the environment in a controlled

manner so as to prevent harm. This was reiterated in non-statutory Waste Management Paper 26B (WMP 26B) (Department of Environment 1995) that aimed to ensure that landfill remained a viable and sustainable waste management option by achieving stabilisation of landfilled waste within one generation *i.e.* to ensure "no unacceptable burdens" for future generations. The emphasis in WMP26B however was on groundwater protection. One potential way to achieve an "acceptable burden" is to manage bioreactive wastes such that the waste mass degrades to a stable, non-polluting state. WMP26B considered one generation to be 30-50 years. It is defined in this thesis as a 30 year period.

WMP26B recommends that a landfill should be operated as a highly active, wet bioreactor, together with a sufficient degree of flushing to remove the products of decomposition, although difficulties in this, such as achieving flushing through the entire waste mass, were recognised. Leachate recirculation is one technique that can help achieve this. To date few landfills have installed the necessary infrastructure needed to recirculate leachate within a landfill, although there is a growing interest in retrofitting such systems to older suitable sites in order to enhance the degradation processes. However, there is little practical experience of how well such systems can be retrofitted into older sites and whether the initial benefits of fitting the system are maintained over many years. The Landfill Directive (Council of the European Union 1999) does not consider sustainability specifically but it does, for the first time, provide a statutory basis for landfill gas management; it requires landfill gas to be collected, treated and, where possible, used to produce energy. Emphasis is on reducing methane emissions to global atmosphere and active gas management is required throughout the life of a site.

1.4 REQUIREMENT FOR LANDFILL GAS EMISSIONS REGULATION

Policy makers and their regulators are now embracing emissions-based regulation. The concept offers a principal means of controlling landfill gas with the objective of sustainability in mind. Landfill emissions have been under increasing regulatory scrutiny during the last three decades. The Control of Pollution Act (1974) (COPA) introduced a system of local authority control for waste management through site-specific licensing. COPA was not applied to landfill gas emissions unless they resulted from illegal disposal operations. Regulation at this time was based on best practice (see **Chapter 3**). Factors that influence the rate, quantity and mitigation of methane production from landfill are necessarily site specific and are many. Quantitative evidence obtained through early research on methane emissions (Environment Agency 1999c) showed that emissions from different landfill sites varied according to eight main factors: waste quantity, composition, age and depth, site area, filling regime, type of cap and/or cover material and the landfill gas collection efficiency (assuming an active system is present). Landfill surfaces can be considerable in size (sites can be many hectares and individual cells are typically 10,000 m^2). Until recently, the scale of methane emissions and the requirements for their control from landfill surfaces was poorly understood. Evidence gathered from 21 UK landfill sites (55-100% MSW, age up to 28 months and depth of 5-40 metres) established that methanogenesis was occurring after only 1-2 months before which time surface methane fluxes were measurable (0.06 mg m^{-2} s^{-1}). Average surface methane flux rates of 0.1 mg m^{-2} s^{-1} were derived for waste up to 20 months old. Significantly, emissions at side slopes were generally four times higher (see **Chapter 4**). This has an impact on the need to control and manage gas at peripheral areas, perhaps through the use of shallow sacrificial pin wells, or horizontal gas collection systems. The efficiency and role of different types of landfill gas management infrastructure is a research

gap. An agreed flux box measurement protocol, based on research findings across a range of UK landfill sites, was subsequently developed as regulatory guidance (Environment Agency 2004e). The regulator has developed a series of emission standards for landfill gas (see **Chapter 3**) which serve as minimum requirements for landfill sites to achieve. However, there is flexibility for site-specific standards to be agreed, subject to particular parameters being identified through risk assessment.

1.4.1 Flare emissions

A regulatory cost benefit analysis for requiring the phasing out of open flares with enclosed flares was developed in 1998 to gauge the potential impact on industry (for discussion see **Appendix A**). The CBA (see **Appendix D**) took account of the number of flares in operation, the future number of flares required, typical design life of a flare and availability of manufacturers. As a result of policy implementation in 1999, a Ministerial decision was made to set a compromised four year transition period to require enclosed flares by 31st December 2003, except for emergency short-term use. The benefit of this was that for the first time, flares could be reliably monitored and emissions quantified to assess the level of environmental impact.

An emission standard was developed for flares (Environment Agency 2002a) based on achievability if operators install and maintain systems appropriately. Monitoring parameters are selected as indicators of good combustion *e.g.* elevated carbon monoxide can indicate the system has imbalanced air and mixing. Minimum monitoring requirements are specified for CO, NO_x and unburned hydrocarbons to allow assessment of combustion performance.

1.4.2 Engine emissions

The Energy Act of 1983 encouraged private electricity generation and COPA provisions were made for this (Part 1, Section 21) specifically from waste. However there were no restrictions on engine exhaust gases from engines fuelled by gas (Department of Environment 1991). At this time, it was perceived that the benefit of utilising gas far outweighed any potential impact on the environment or human health and emissions monitoring was not generally required. It was only through research and development providing monitoring methods to quantify emissions and increasing knowledge of gas composition that this impact could start to be addressed.

The location and design of gas utilisation plants has significant implications on the dispersion of emissions from engine exhausts. Once emissions data started to emerge, it was recognised that emissions from landfill gas engines were potentially a greater problem than from flares due to poorer dispersion, particularly in instances where engine exhausts were horizontal (Environment Agency 1999a). Emissions monitoring for combustion systems is still in its infancy and further research is required.

1.4.3 Landfill gas yield and collection efficiency

The greatest uncertainty for landfill gas management is predicting the available landfill gas resource, whether at the full-site scale or according to time and space dimensions. Landfill gas production models typically predict ± 20%. When this uncertainty is combined with changing waste streams (post Landfill Directive implementation), in particular the reduction in biodegradable waste sent to landfill, it highlights a need to capture and control landfill gas in a timely way. There is a reasonable body of evidence to characterise gas production at conventional (large-scale modern, engineered) landfills to the point of peak production. There is

much greater uncertainty regarding the exponential decay phase of gas production. Further uncertainty surrounds the landfill gas regime at inorganic waste and hazardous waste landfills. There is also a tendency to consider emissions in isolation of each other in the absence of knowledge concerning the cumulative and/or additive effects of emissions. The cumulative impact of multiple emission sources (*e.g.* gas engine exhausts) also requires additional understanding.

The yield composition of bulk gases (methane, carbon dioxide and oxygen) within conventional landfill gas varies between sites but by and large it is reasonably predictable using a range of simple or complex models available. These models usually use different methods to calculate the available gas yield, all of which require validation using real monitoring data. The long-term stability of gas production models has not been assessed and requires careful assessment along with consideration of the most appropriate timescales.

Landfill gas collection efficiencies can be calculated to ascertain the degree of success that a gas management scheme is operating at and the Environment Agency (Environment Agency 2004f) has an aspirational target of 85%. Achievable collection efficiencies can be predicted using the GasSim model and the optimal efficiency determined to assess the maximum utilisation over the longest timescale.

A number of rules of thumb for the viability of landfill gas (Environment Agency 2002b) have been used in the landfill sector *e.g.* 600-750m^3 $hour^{-1}$ at 50% CH_4 is required to generate 1 megawatt (MW).

As biodegradable waste is progressively diverted from landfill sites, in accordance with Article 5 of the Landfill Directive (Council of the European Union 1999), the potential for future gas recovery will diminish in the longer-term. Even though the 2020 target seems a long way off, now is the time to install the necessary infrastructure where it is currently lacking (and where justified) to maximise gas collection efficiencies.

1.4.4 Landfill gas management options

A hierarchy of landfill gas management options was co- authored as part of a programme of work developing regulatory guidance (Environment Agency 2002a). The hierarchy (see **Chapter 3, Figure 3.2)** shows the general relationship for landfill gas management options throughout which gas management systems must be kept flexible so as to control gas through the hierarchy to final completion. A landfill operator must decide which landfill gas management option is the most appropriate for a particular site at any moment in time, whilst striving to be as high up the hierarchy as possible (subject to sufficient gas quality and quantity). In many instances, landfill gas emissions management and reduction does not require the application of new technology.

Investment in gas management systems can be considerable (see **Appendix D)** and installations should be designed flexibly in order to accommodate change over a period of time likely to be measured in decades. Gas control systems are not off the shelf products to be installed, left running and forgotten about. Systems need ongoing maintenance and continued nurturing in order to optimise the high level of control that they can provide. Control systems also have anticipated downtime of typically 5% each year. The implication of this is that operators require standby equipment (*e.g.* enclosed flare systems) for contingency reasons. The operator must

understand that if the landfill gas management system fails, uncontrolled gaseous emissions can result.

Deficiencies in WMP27 (a non-statutory guidance document) were widely recognised by operators and regulators. The overarching requirement for a successive guidance document (Environment Agency 2004f) was to focus on emissions and effectiveness of control. WMP27 was not risk-based and monitoring was centred on detecting lateral landfill gas migration from non-engineered landfills. The regulatory guidance provides a clear approach to landfill gas risk assessment underpinned by monitoring objectives.

1.4.5 Passive venting

Prior to the mid-1990s and since the Loscoe landfill gas incident in 1986 (Derbyshire County Council 1988), landfills were largely dependent on the installation of a series of isolated vertical passive vents through the waste that provided a pressure release mechanism for landfill gas in order to reduce the risk of lateral migration through sub-surface geology or service ducts for example. In 1997, the regulatory position on passive venting was that a landfill operator had to justify its use on the grounds of sustainability *i.e.* sustaining a flare or utilising without the use of support fuels (the Non-Fossil Fuel Obligation (NFFO) requires <10% support fuel, set on the basis that dual-fuel landfill gas engines typically operate on 5% diesel and 95% landfill gas. The Landfill Directive was a decade in the making and so there was early warning that global methane emissions to atmosphere would be a priority. In accordance with Directive objectives, it was decided at the regulatory policy level that large-scale passive venting was no longer an option (Environment Agency 1999b). This policy was co-authored by Ian Cowie, Chair of the Environment Agency's Landfill Gas Task & Finish Group as well as the author of this thesis, a

Member of the same group. Waste Management Paper 26B had already discouraged passive venting in preference to flaring or energy recovery (Sections 9.165, 9.166 and 9.170 (Department of Environment 1995) and the position was reiterated in later regulatory guidance on landfill gas flaring (Environment Agency 2002a).

1.5 ACCELERATED STABILISATION

Recirculation of leachate within the landfill body is a means of achieving and maintaining the required distribution of moisture. However, it is far from clear how well leachate recirculation will enhance the waste degradation processes. Recirculated leachate may encounter areas containing preferred pathways or areas of impermeable waste which will inhibit the uniform distribution of leachate through the waste mass and may lead to uneven waste degradation in the fill. Leachate may need to be treated during recirculation to prevent the build up of inhibitory compounds such as ammonia or may need to be heated before being reintroduced to prevent excessive cooling of the waste mass, though this is likely to be subject to seasonality. Another problem likely to be encountered concerns the requirement to maintain a low leachate head on the liner (typically 1 metre as specified in many landfill site permit conditions).

Techniques are available whereby landfill gas production can be influenced and potentially enhanced. The role of the following techniques and their relationships require further exploration to assess the future longevity of landfill gas production curves:

- Waste composition
- Pre-treatment
- Moisture content optimisation (dry entombment, merits of leachate recirculation)
- Addition of inoculums (*e.g.* sewage sludge)

- Buffering capacity
- Nutrient addition
- Temperature optimisation
- Implications from maximising biodegradation in organic waste treatment processes prior to residues disposal.

Data from the Brogborough Landfill Test Cells provides what is thought to be the longest validated dataset on landfill gas production worldwide that for the first time has now peaked and placed a real timescale against the classical gas production curves (Farquhar and Rovers 1973) shown in Figure 1.1.

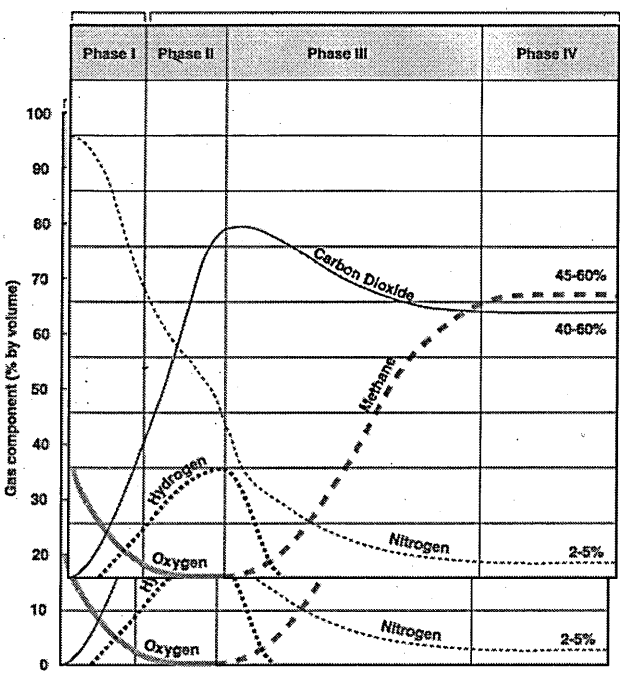

Figure 1.1 Landfill gas production

Though designed for experimental purposes, the test cells have also demonstrated that observed total yield data are well in excess of 100 m^3 $tonne^{-1}$ of waste are recoverable for utilisation. This

is a factor of 2 or 3 above lower estimates of useful recoverable yield used by the industry. Gas yields are shown in **Chapter 2, Figure 2.3**. The Brogborough data (see **Chapter 2**) demonstrate that in reality, gas production does not necessarily behave as models predict. Peak gas production at Brogborough was reached after around 10 years, but some 25-50% of remaining gas production was yet to occur, for which the timescales are uncertain.

Other research, including an applied research project on minimising methane emissions from landfill (Barry and Smith 2002a; Barry and Smith 2002b; Barry *et al.* 2004a; Barry *et al.* 2004b) has since confirmed the timescales recorded at Brogborough and additionally provides evidence for a much shorter period for the onset of methanogenesis than was previously thought. This suggests that the timely installation of utilisation infrastructure is warranted (see **Chapter 4** and **Appendix C**).

Biodegradability test methods such as the Biochemical Methane Potential (BMP) test (see **Appendix B**, also known as the BM_{100} method in recognition of it being a 100 day test), provide an opportunity to predict residual methane potential as a peak rate derived in the laboratory environment. Whilst such methods (also including the dynamic respiration method DR_4 as a four day test) are now commonly applied to waste treatment processes such as MBT, they are not yet widely applied to landfill samples. The role of these methods, which could be used to further validate modelled predictions, requires further assessment. One of the main considerations is likely to be confidence in representivity of samples recovered.

1.6 ENVIRONMENTAL EQUILIBRIUM STATUS AND LANDFILL COMPLETION

The Environmental Protection Act 1990 (implemented in 1994 by the Waste Management Licensing Regulations) introduced the requirement for landfill sites to obtain a certificate of completion. In order to do this, operators had to comply with licence conditions and make financial provision for an aftercare period. Previously, operators could hand back their licence to the regulator, which usually happened when a site stopped accepting waste, and walk away from any future liability. Landfill completion was defined as being achieved when the land was unlikely to cause pollution of the environment or harm to human health (Department of Environment 1993).

Regulatory guidance on landfill completion published by the Environment Agency (consultation draft, March 2003) does not require landfill sites to meet set leachate or gas completion criteria. Instead, the proposed approach requires completion criteria to be set on the basis of an assessment of the condition of the landfill that would not pose a pollution risk on a site-specific basis depending on the sensitivity of the environment. The guidance also proposes an understanding of what "returned to a satisfactory state" means for Pollution Prevention and Control (PPC) regulated landfills; any pollution of the surrounding environment from the landfill should be removed or mitigated as part of the process of returning the environment to a satisfactory state, as might be defined under environmental equilibrium status.

As a landfill evolves, it becomes less contained or artificially isolated and it becomes more integrated with the environment as leakage occurs. By definition, the concept of environmental equilibrium status (also referred to in the literature as Final Storage Quality) accepts that landfills will not achieve 100% degradation and that the controlled release of substances into the surrounding environment can be acceptable if equilibrium status can be achieved. Samples

extracted from the Brogborough test cells were used to quantify the level of stabilisation achieved during the 14 years since filling began (see **Chapter 2**) by comparing current waste characteristics with published data for the characteristics of MSW that had been pre-treated by composting and anaerobic digestion. The work considers what might be achieved, through a range of site management techniques, if a further 16 years had been available, giving a total 30-year life *i.e.* the notion of achieving completion in one generation (Hall *et al.* 2006a; Hall *et al.* 2006b; Hall *et al.* 2007) (see **Chapters 5, 6 and 7**).

Landfill completion has implications for financial aftercare provision. The potential benefits to landfill operators of accelerated waste stabilisation include reduced post-closure site management costs, an enhanced gas production making energy recovery schemes more viable, earlier re-use of the land and a shortened settlement period. Further work is required to develop completion criteria at the UK and EU level, methods to meet the criteria with confidence and how to achieve representative sampling *e.g.* the BMP method (Harries *et al.* 2001a; Harries *et al.* 2001b) (see **Appendix B**) only requires < 1g sample for analysis. In the USA, regulations also dictate a 30 year aftercare period and it is assumed that beyond this period, monitoring will be discontinued as sites will no longer pose a threat to the environment. Similarly, technical criteria to define stability are lacking in the USA (Barlaz *et al.* 2002).

Waste streams that are being sent to landfill are changing significantly with an increasingly proportion of waste treatment residues being landfilled. The implications for landfill completion or environmental equilibrium timescales have been modelled and benchmarked against conventional landfills (**Chapters 5, 6 and 7**). Many Local Authorities are selecting MBT to achieve short-term landfill diversion targets in accordance with the Local Authority Trading Scheme (LATS) to implement Article 5(2) of the EU Landfill Directive. MBT systems are either

designed to achieve a certain biodegradability reduction before sending the material to landfill, or, they are designed to produce solid recovered fuel (SRF) by removing moisture content by biodrying. MBT residues sent to landfill will, in most cases, have residual gas potential. MBT with composting performs well when equilibrium timescales are modelled (see **Chapter 6**) yet MBT processes are energy intensive, raising the question of sustainability. MBT residues could be sent to thermal treatment facilities such as incineration, but it has been demonstrated that untreated incinerator bottom ash cannot be landfilled and meet sustainable landfill criteria *i.e.* meeting environmental equilibrium status within 30 years. The achievement of environmental equilibrium status must be balanced against the likelihood of landfill engineering failure which could potentially result in a catastrophic release of contaminants to the environment. The Danish standard for landfill liners, for example, has a projected lifetime of 60-70 years.

1.7 REGULATORY POLICY AND GUIDANCE

1.7.1 Policy development

Appendices E and F contain two landfill gas policies that were developed by the Environment Agency. This was a significant step change in the way that landfill gas was regulated (see **Chapter 3**). An overarching (general) landfill gas policy was required in the first instance that stated:

> *"Gaseous emissions from licensed landfill sites will be regulated by the Environment Agency according to site specific risk to minimise the impact on health, the local environment and global atmosphere".*

A second policy was developed specifically covering landfill gas flaring. This policy served two purposes. Firstly, it required the phasing out of open flares to be replaced with enclosed flares[1]; and secondly operators were required to demonstrate operational performance and meet a prescribed emission standard.

1.7.2 Development of regulatory guidance

The thesis author was a key member of the Environment Agency's landfill gas guidance team with responsibility for drafting, consulting on and publishing a suite of six related non-statutory guidance documents on landfill gas underpinned by associated research. The six documents between them form the basis of an emissions based strategy (see **Chapter 3**) and the guidance is outlined in the table below. The relationship between the guidance documents is highlighted in **Appendix E**. A regulatory impact assessment carried out for the portfolio of landfill gas guidance (Environment Agency 2004d; Environment Agency 2004a; Environment Agency 2004c; Environment Agency 2004f; Environment Agency 2004e; Environment Agency 2004b) demonstrated that the environmental benefits outweigh the costs, even when conservative assumptions were made on potential environmental benefits (Environment Agency 2003a).

1.7.3 Environment Agency Waste R&D Programme - related research underpinning landfill gas regulation

The related projects that fit in with the overall theme of the proposed thesis are listed in Appendix G with a brief description of each. These projects are particularly important in taking

[1] For a discussion on flare types, refer to Appendix A, current technology section.

a forward look at the likely impact of changing waste streams and the effect on landfill gas management requirements.

The timing of key decisions made during the development of the landfill gas regulatory guidance is summarised in Table 1-1. Gaps in the chronology are attributable to ongoing research.

Table 1-1. Key aspects in the development of landfill gas emissions based regulation during the last decade

Date	Milestone
February 1997	Landfill operators must justify the use of passive venting.
March 1997	Environment Agency Landfill Gas Task & Finish Group established.
August 1997	Health effects from landfill gas research started (Helga model) (Environment Agency 2000)
September 1997	Recognition that guidance was needed for when to install landfill gas flares.
November 1998	Cost benefit analysis for landfill flares developed with support from National Centre for Risk Analysis and Options Appraisal (see Appendix D).
December 1998	First landfill gas policy drafted and submitted to National Environmental Protection Group (see Appendix E). Deficiencies in WMP27 became increasingly recognised.
January 1999	Landfill gas flaring policy developed (see Appendix F).
March 1999	National Landfill Gas Conference held (launch of Best Practice Flaring of Landfill Gas guidance).
April 1999	Landfill Directive text agreed.

Date	Milestone
August 1999	National library of licence conditions and working plan specifications (Volume 1, Edition 2), including those for landfill gas, published as a reference framework to represent the default position on a consistent basis.
August 1999	Landfill gas management hierarchy developed by John Keenlyside and Richard Smith.
June 2002	*Gassim - Landfill gas risk assessment tool (Model and user manual)* Published.
July 2002	*Measurement of gas potential: development and application of a biochemical methane potential (BMP) test* published.
September 2002	Environment Agency Landfill Gas Task & Finish Group disbanded.
November 2002	*Guidance on landfill gas flaring* published.
June 2003	*Guidance on landfill completion* published for consultation.
December 2003	Impact assessment of Environment Agency landfill gas management guidance.
July 2004	*Guidance on the assessment of risks from landfill sites* (consultation) published.
August 2004	*Guidance on gas treatment technologies for landfill gas engines* published (consultation March 2003).
September 2004	*Guidance on the management of landfill gas* published (consultation February 2003).
September 2004	*Guidance for monitoring enclosed landfill gas flares* published (consultation March 2003).
September 2004	*Guidance for monitoring landfill gas engine emissions* published

Date	Milestone
	(consultation February 2003).
September 2004	*Guidance for monitoring trace components in landfill gas* published (consultation February 2003).
September 2004	*Guidance on monitoring landfill gas surface emissions* published (consultation July 2003).

1.8 AIM AND OBJECTIVES

The overall aim of this thesis is to gain an improved understanding of landfill sustainability with respect to conventional landfill in the UK (pre-Landfill Directive) and post-Landfill Directive, particularly with regard to gaseous emissions and changing waste streams that are landfilled and the implications for landfill completion, as determined by achieving environmental equilibrium status.

This aim is underpinned by five objectives:

- To critically assess the timescales for the onset of methanogenesis *i.e.* the scale and significance of gas production during the first few months following waste deposition.
- To provide modelled evidence for the timescales and drivers for achieving landfill environmental equilibrium status (stabilisation) using a range of scenarios.
- To develop waste biodegradability methods and understand their role for predicting residual gas potential for landfill completion.
- To understand the role of specific techniques for accelerating the stabilisation of landfilled waste.

- To recommend changes in management practice and identify further research needs to improve landfill sustainability.

1.9 FUTURE RESEARCH PRIORITIES

Gaseous emissions monitoring tends to be carried out on a periodic snapshot basis that captures "moment in time" data. An understanding of how representative this data is in relation to temporal and spatial dimensions is currently lacking. Continuous emissions monitoring or process type approval for landfill gas combustion systems may have a role.

Now that monitoring protocols are developed and emissions monitoring is required for different aspects of landfill gas management (landfill surfaces and combustion systems) on a consistent basis, there is an opportunity to capture and interpret data nationally. This could provide a much greater understanding of landfill gas emissions and potentially help identify specific components of concern for further research. For example, vinyl chloride (chloroethene) is being found at an increasing number of sites in the landfill source term but spatial and temporal variability is not known on an inter-site basis. The origins of vinyl chloride (which could be formed by biogenic processes or during degradation) and the significance of emissions is uncertain yet there are important health and safety considerations for vinyl chloride in terms of exposure.

Particular aspects of gas management require further work. For example, guidance on pipework (types of material, pressure testing etc.) is one aspect of a gas management system that is often poorly specified. Research is needed on the role and efficiency of different landfill gas management infrastructure *e.g.* use of pin wells towards site/cell edges and on side slopes, horizontal versus vertical collection systems and optimal spacing of gas wells. The gas

production regimes at post-Landfill Directive hazardous and inorganic waste landfills are still emerging, resulting in an uncertain need for gas management systems.

As part of the landfill gas management hierarchy, methane oxidation is promoted for managing residual gas production beyond the 'useful' combustion period. Most modern engineered landfills have not been in existence for long enough for this period to yet be of concern. However, operators should be encouraged to consider the role of methane oxidation and plan for it in a timely manner *e.g.* to accommodate appropriate capacity for methane oxidation during landfill restoration. The long-term performance and role of passive versus actively pumped methane oxidation systems requires research.

Finally, understanding of the profile of landfill gas trace component production and depletion over time is limited. It is assumed that destruction efficiencies for trace components in landfill gas combustion systems are good, but there is little evidence available. Emissions of landfill gas from landfill surfaces tend to be limited to methane and fugitive emissions of trace components could be significant. The fate of trace components under methane oxidising conditions is uncertain. This fundamental knowledge is required to inform risk and exposure assessments.

1.10 FORMAT OF PRESENTATION

This thesis is presented in the form of an introductory chapter followed by a series of two international peer reviewed conference papers (Chapters 2 and 3) and four journal papers (Chapters 4 to 7). Chapter 8 provides a summary record of peer review, integrating conclusions and recommendations. The chapters are supported by a series of three international conference papers presented as Appendices A to C. References are provided for each chapter.

It is the policy of the Centre for Resource Management and Efficiency for authors to be listed in order of contribution with the exception of the corresponding author for journal papers, who is listed last. The author of this thesis was corresponding author for Chapters 4, 5, 6 and 7 and therefore appears last in each case.

Figures 5.1 and 6.1 are necessarily repeated in two companion journal papers (Chapters 5 and 6).

1.11 PERSONAL CONTRIBUTION

This section (Table 1-2) provides an estimate of the author's personal contribution to each journal and conference paper that has co-authors.

Table 1-2. Personal contribution to chapters and appendices that have co-authors

Chapter	Intellectual (%)	Data collection (%)	Analysis and interpretation (%)	Paper writing (%)
2	30	10	40	25
3	20	N/A	20	10
4	80	10	65	75
5	60	30	60	90
6	20	10	60	90
7	70	0	75	90
Appendix A	45	N/A	N/A	30
Appendix B	20	0	15	20
Appendix C	25	10	20	25

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2. THE BROGBOROUGH TEST CELLS: CONCLUSIONS FROM A 14-YEAR FIELD-SCALE LANDFILL GAS EXPERIMENT

BARRY CROFT[1], RICHARD SMITH[2], MALCOLM CAINE[3], KEITH KNOX[4], JIM WHITE[5], IRENE WATSON-CRAIK[6], CHRIS YOUNG[7], AND JULIAN ELLIS[7]

[1] *Mouchel Consulting Ltd, West Hall, Parvis Road, West Byfleet, Surrey KT14 6EZ, UK*

[2] *Environment Agency for England and Wales*

[3] *AEA Technology Environment*

[4] *Knox Associates Ltd*

[5] *University of Southampton*

[6] *University of Strathclyde*

[7] *Water Research Centre plc*

ABSTRACT

Six full-scale municipal solid waste (MSW) landfill test cells were constructed during 1986-88, at the Shanks and McEwan Brogborough landfill in Bedfordshire, England. The cells were constructed primarily to demonstrate and compare the effects of selected pre- and post-placement management techniques on methane production, for waste-to-energy and for environmental control purposes. The cells have been regularly monitored and this paper summarises the findings with particular emphasis on gas production. Cell characteristics,

methane yields, CH_4:CO_2 ratios and gas flow rates are discussed for all six cells. Many other chemical and physical parameters were monitored over time in the gas, leachate and solid phases, and the cells were comprehensively measured and sampled as part of a final characterisation study, providing data for analysis by statistical and numerical modelling methods. The field experiment is now over and the cells have been buried.

2.1 INTRODUCTION

Six full-scale municipal solid waste (MSW) landfill test cells were constructed during 1986-88, at the Shanks and McEwan Brogborough landfill in Bedfordshire, England. Each cell was constructed using contemporary full-scale landfill methods and equipment, with a base of puddled callow clay, bunds of blue clay and compacted layers of MSW from London in 2m lifts (4m in cell 2), with a 2-3m thick clay cap. Each cell was constructed to be approx. 40m long, 25m wide and 20m deep, so contained approximately 20000 m^3 (~15000 tonnes) of MSW. Figure 2-1 shows a plan of the cells as they were laid out for most of the experiment and Figure 2-2 shows a schematic cross-section. The cells were constructed primarily to demonstrate and compare the effects of various pre- and post-placement techniques on landfill gas production (monitoring gas quality and production rate over time), to determine the gas profile (a) for gas-to-energy plants (the period of adequate rate and quality), and (b) for environmental impacts and design of controls (Croft and Fawcett, 1992). This paper focuses on the gas results.

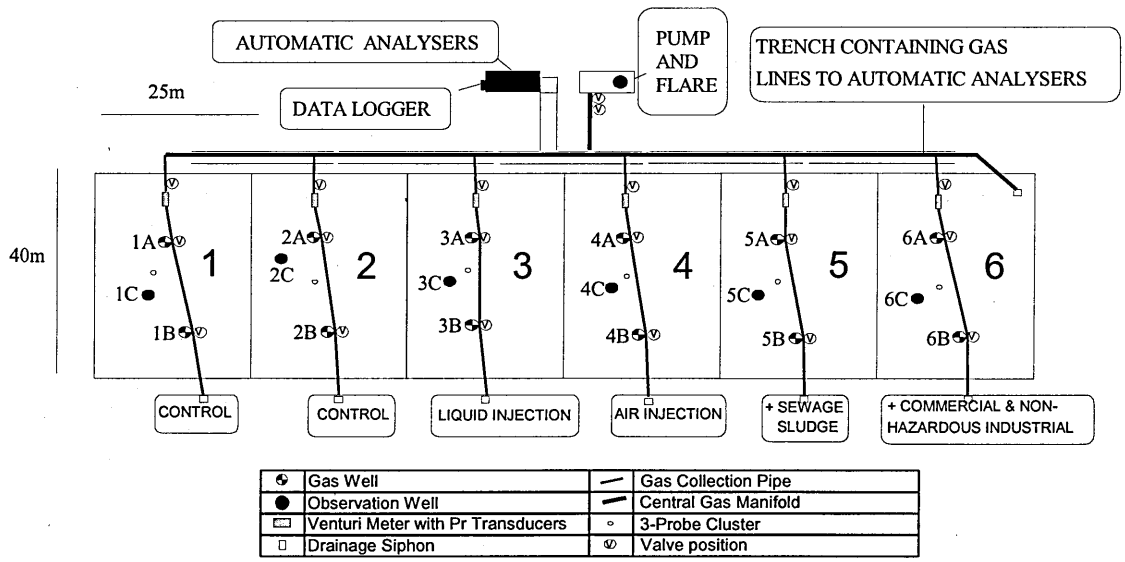
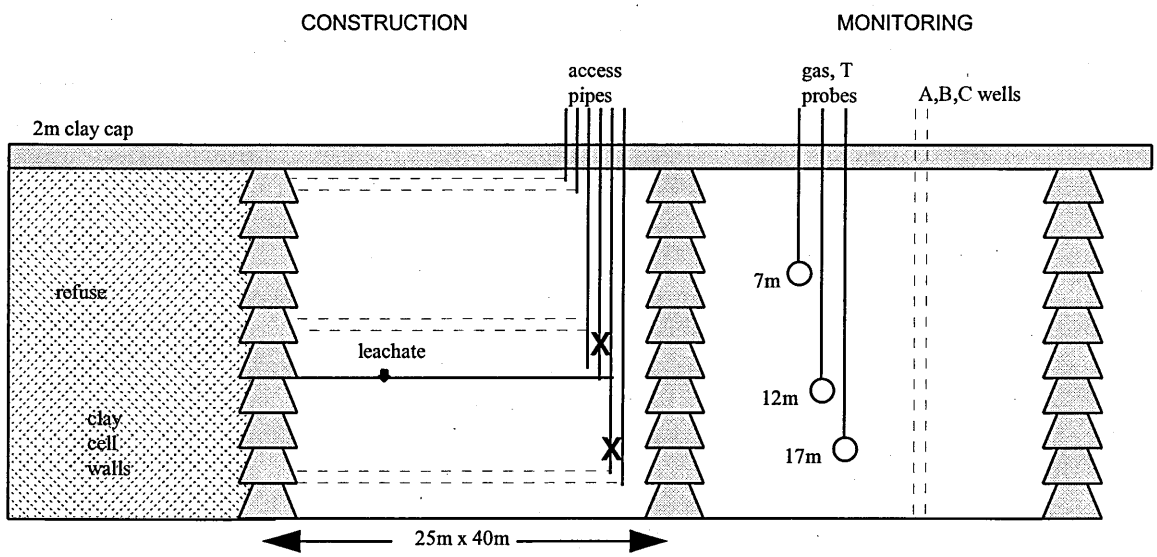


Figure 2-1. Schematic plan of the Brogborough Test Cells



--- = Herringbone trenches
X = Connection to trenches lost during second stage of waste emplacement

Figure 2-2. Schematic section and construction details of Brogborough Test Cells

Cell 1 was the control cell, constructed using Shanks' full-scale methods and using typical MSW (puddled base, 2m wedge bunds and 2m thick cap constructed of local clays, MSW sourced

from London/Bedfordshire and placed in thin lifts, compacted by steel wheels). Cell 2 was placed in thicker lifts and intended to be lower density, but the in-situ density achieved was very similar to Cell 1. Cell 2 therefore became a useful replicate control. Cell 3 was constructed as Cell 1 so was another control until it received three liquid-injection treatments: 100 m^3 in July 1992, 21 m^3 in April 1993, and 231 m^3 in February 1994. Cell 4 was constructed as Cell 1 then received three air-injection treatments: 500 m^3 in April 1992, 250 m^3 in February 1993, and 11000 m^3 in August 1993. Cell 5 was constructed as Cell 1 but placing intermediate layers of primary dewatered sewage sludge with the MSW, to achieve a final sewage concentration of 9% by weight. Cell 6 was constructed as Cell 1 but placing alternate layers of MSW and commercial and industrial (non-hazardous, cellulose-rich) waste, to a final concentration of 45% by weight. Construction took more than two years, during which the cell design was adapted from 10m deep to 20m deep, and a horizontal gas collection system was discarded in favour of two vertical abstraction wells per cell, connected to a dedicated gas pump and flare. A third well created during waste sampling was used for leachate sampling and temperature profiles. A cluster of probes sampling from approximately 5, 10 and 15m below cap was placed mid-cell. Gas composition monitoring was manual, using portable equipment, and flows were measured using hot wire or positive-displacement meters. By 1994 measurement of gas flow was by venturi, and gas composition using an automated analyser with remote data-logging. Methane was optimised to 1995 by extracting as hard as possible consistent with maintaining O_2 concentration below 1%, from 1995 on by ignoring oxygen but maintaining methane above 50% (Caine *et al* 1996). Some of these changes may have affected absolute estimates of methane generation, but the 6 cells were always subjected together to any change, so relative differences between cells are considered reliable. Settlement, waste temperatures, leachate level and composition were measured. Waste was drilled, sampled and analysed in 1993, and again prior to burial. A final comprehensive sampling and monitoring study took place in September/October of 2000 and

then the experiment was completed and buried to allow resumption of normal landfill operations. Prior to this final phase of work, a review of the Brogborough test cell data was reported (Knox 1997).

2.2 GAS MONITORING

Gas monitoring began in earnest after final capping in 1989, at which time production was slow but increasing rapidly. The CH_4:CO_2 ratio was initially below 1:1 but within 3 years settled at between 1.4:1 and 1.5:1 in all cells. However, extreme variability in flow rate from day to day and even week to week was noted and this feature continued in all cells throughout the experiment. Table 2-1 shows details of gas flow and quality over time.

Table 2-1. Total yield (m^3/t), mean specific methane (M) and total gas (LFG) yields (m^3/t/y)

	Cell 1		*Cell 2*		*Cell 3*		*Cell 4*		*Cell 5*		*Cell 6*	
Period	M	LFG	M	LFG	M	LFG	M	LFG	M	LFG	M	LFG
Oct-89 to Jul-90	3.0	6.1	2.4	4.7	2.0	4.8	2.3	4.4	3.3	6.2	3.9	6.4
Oct-90 to Jul-91	3.1	7.2	2.0	4.5	2.0	5.4	3.3	7.7	6.8	13.3	3.8	8.6
Aug-91 to Jul-92	3.5	7.2	2.8	5.7	2.7	5.4	4.4	8.2	5.4	9.9	3.9	8.2
Aug-92 to Jul-93	4.8	8.6	4.0	8.0	5.5	9.8	8.4	14.9	7.3	12.6	6.1	11.5
Aug-93 to Jan-94	5.7	10.1	4.6	8.5	7.0	11.7	10.5	17.7	7.9	13.2	5.8	10.6
Feb-94 to Jul-94	6.4	11.1	5.4	10.3	10.1	16.8	12.2	21.5	7.9	13.7	5.8	9.3
Aug-94 to Dec-94	4.6	8.2	4.0	6.8	9.4	16.1	9.1	16.0	7.0	12.4	5.8	8.4
Jan -95 to Oct-95	7.1	12.2	6.4	11.0	9.2	15.7	9.9	17.1	7.0	12.1	8.1	14.0
Nov-95 to Jul-96	8.4	14.2	7.8	13.2	13.0	22.2	11.2	19.1	6.2	10.6	6.5	11.3
Aug-96 to Feb-97	7.8	12.7	5.5	9.1	9.8	16.0	9.6	15.6	3.9	6.3	8.5	13.9

	Cell 1		*Cell 2*		*Cell 3*		*Cell 4*		*Cell 5*		*Cell 6*	
Total yield to Feb-97	37.9	69.1	32.1	59.1	47.7	84.7	54.1	95.4	43.2	76.2	40.3	72.2
Specific yield Feb-97	5.6	10.1	4.7	8.7	7.0	12.4	7.9	14.0	6.3	11.2	5.9	10.6
Mar-97 to Dec-97	6.9	12.1			8.7	15.2	7.6	13.3	4.7	8.1	6.1	10.4
Jan-98 to Jun-98	7.0	11.6			7.4	12.5	8.3	13.9	4.3	7.2	6.4	10.5
Jul-98 to Dec-98	8.3	14.1			6.9	12.3	9.3	15.7	2.0	3.3	6.9	11.7
Jan-99 to Apr-00	5.2	8.5			2.3	3.8	10.5	17.1	0.8	1.3	7.4	12.0
Total yield to Apr-00	58.0	103.0			65.3	114.7	82.3	142.5	51.6	90.2	61.3	107.1
Specific yield Apr-00	5.5	9.8			6.2	11.0	7.9	13.6	4.9	8.6	5.9	10.2

Cell 2 monitoring was discontinued after Feb 97. Note that averaging periods are of irregular length.

2.2.1 Cell 1 (Control)

Figure 2-3 shows landfill gas flow rate over time. Day to day and week to week variations were high, and even the 9-week moving average shown exhibited high variability. The first and last year of monitoring showed many discontinuities due to variations in monitoring and control, so have not been included. However, displaying only the 9 years of relatively uninterrupted data[2] and using 6-monthly or higher averages produced the much smoother and clearer gas generation profile shown.

Methane production averaged 5.3 m^3/t/y over 11 years, and peaked at 8.4 m^3CH_4/t/y in 1995-1996 (7-8 years after waste placement). The trend in flow at completion was slowly falling. Cell 1 cumulative methane yield to 2000 was 58.0 m^3/t (over 11 years).

[2] The swing between 1994 and 1995 was associated with a discontinuation then recommencement of monitoring with new control objectives, and affected all cells.

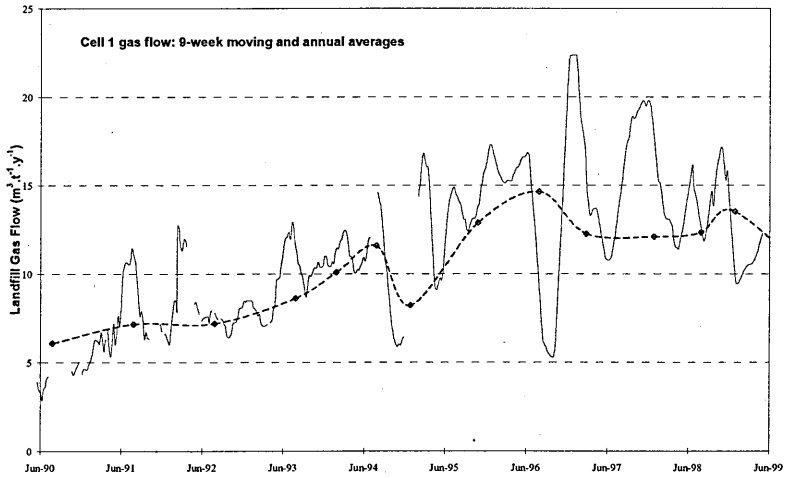


Figure 2-3. Cell 1 (control) 9-week moving average and smoothed annual average

2.2.2 Cell 2 ('Low Density')

Figure 2-4 shows Cell 2 gas production. Placed in 4m deep lifts, in practice this cell achieved almost the same in-place density as Cell 1 (0.82 $t.m^{-3}$ compared to 0.83 $t.m^{-3}$ in Cell 1), so effectively became a replicate control. Detailed flow behaviour (daily and weekly peaks and troughs) differed from Cell 1. Overall yield and annually-smoothed gas production behaviour up to discontinuation were parallel to but approximately 15% lower than Cell 1. Gas monitoring was discontinued so that it could be used for hydraulic studies from 1997 (see section 3.4).

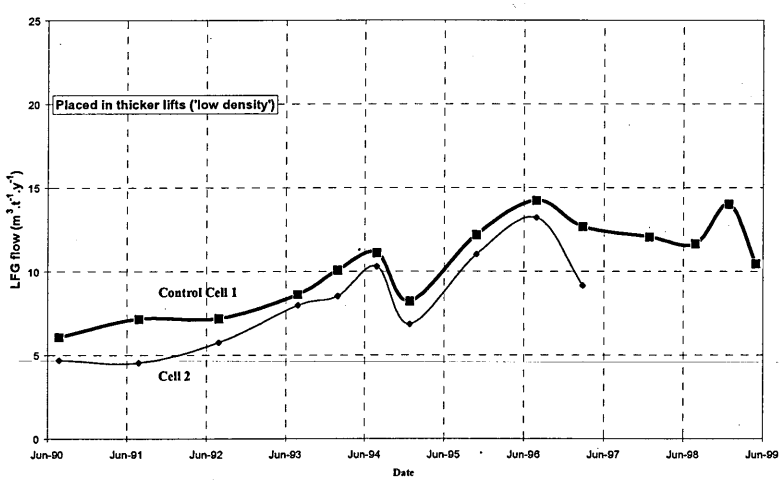


Figure 2-4. Annualised landfill gas flow: Cell 1 (control) and Cell 2 (low density)

2.2.3 Cell 3 (Liquid Injection)

Similar in behaviour to cells 1 and 2 prior to injection, this cell was subjected to three different water-injection or leachate-recirculation trials during 1992-4. Gas flow was stimulated to a peak of approximately 1.7 times that of Cell 1 two years after the third injection, with enhancement lasting a further 1-2 years. However, gas flow at completion was below Cell 1, and although the cumulative methane yield was still higher than Cell 1 at 65.3 m^3/t, they were converging.

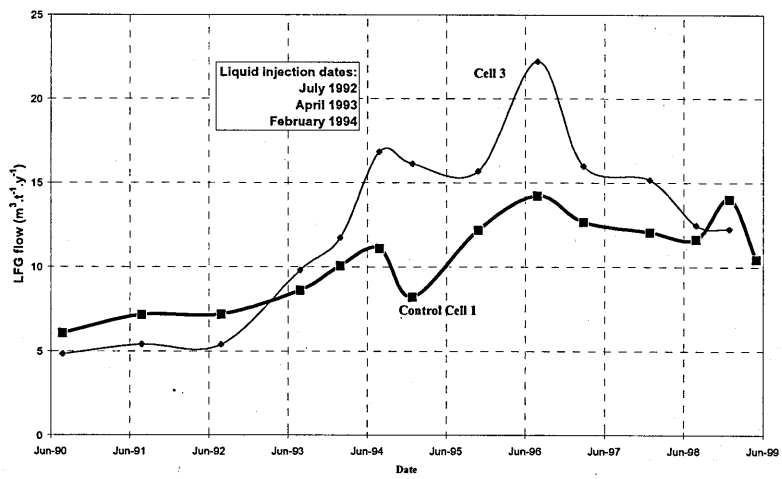


Figure 2-5. Annualised landfill gas flow: Cell 1 (control) and Cell 3 (liquid injection)

2.2.4 Cell 4 (Air Injection)

Figure A2-6 shows Cell 4 gas production over time. Constructed as per cells 1, 2 and 3, and similar in behaviour until treated, this cell was subjected to three air injection trials during 1992-3. Gas flow was stimulated to a peak of approximately 1.8 times Cell 1, peaking in 1994 and lasting until 1996 before gradually converging on Cell 1. At completion Cell 4 was still

producing more gas than Cell 1, and the total methane yield to 2000 was the highest of all the cells (82.3 m^3/t over 11 years).

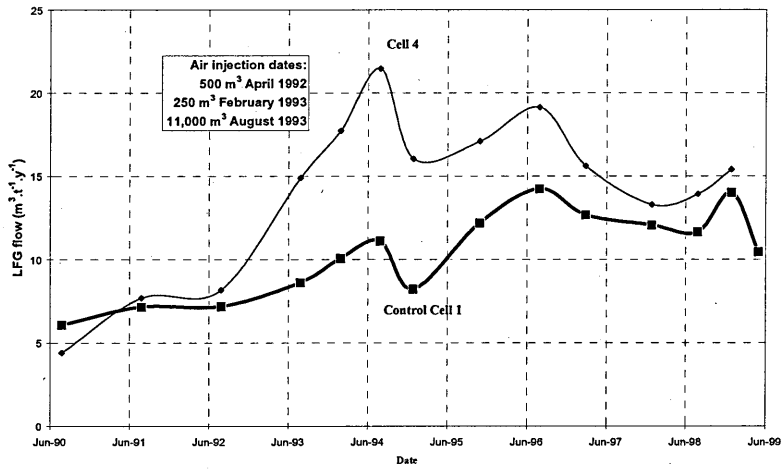


Figure 2-6. Annualised landfill gas flow: Cell 1 (control) and Cell 4 (air injection)

2.2.5 Cell 5 (9% Sewage Sludge)

Figure 2-7 shows Cell 5 gas production. This cell was the first to achieve a 1.5:1 CH_4:CO_2 ratio, and showed early and marked enhancement of gas flow. The greatest relative difference from control cells was in 1990-1, when Cell 5 was producing 6.8 $m^3CH_4/t/y$ and cells 1-4 ranged from 2.0 to 3.8 $m^3CH_4/t/y$. Cell 5 production peaked at 7.9 $m^3CH_4/t/y$ in 1993-4, by which time cells 1 and 2 were also peaking, but then showed a sharp reduction in landfill gas flow relative to controls. Flow rate fell below Cell 1 after 1995, to rates so low that they were difficult to measure accurately by 2000. As a result, cumulative methane yield to 2000 was only 51.6 m^3/t, *i.e.* less than Cell 1. However, this fall in flow did not necessarily indicate a fall in production, since Cell 5 also had higher leachate levels (possibly impeding gas recovery).

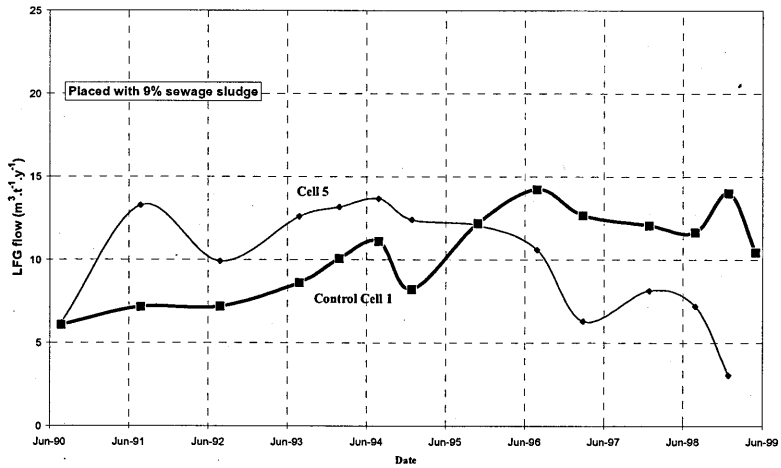


Figure 2-7. Annualised landfill gas flow: Cell 1 (control) and Cell 5 (9% sewage sludge)

2.2.6 Cell 6 (45% Commercial Waste)

Figure 2-8 shows gas production over time for Cell 6. This cellulose-enhanced cell may have shown modest stimulation in the early years (differing most from Cell 1 most in 1992-3) but from 1994 onwards it was indistinguishable from Cell 1. Cumulative methane yield over 11 years was very similar to Cell 1 at 61.3 m^3/t.

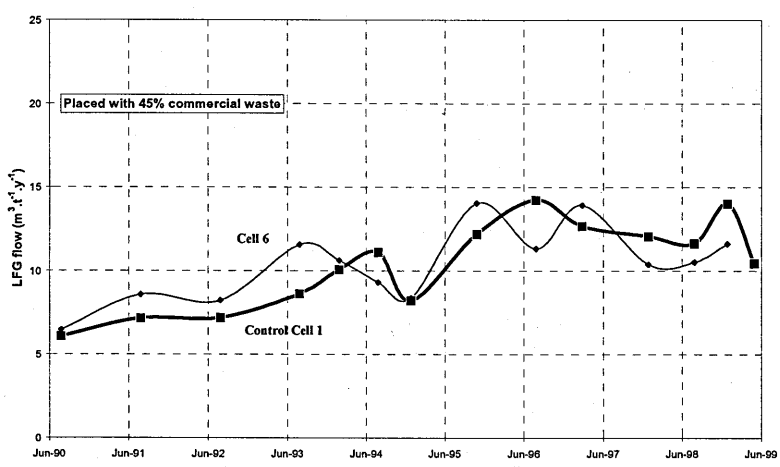


Figure 2-8. Annualised landfill gas flow: Cell 1 (control) and Cell 6 (45% commercial)

2.3 LANDFILL GAS SUMMARY

Landfill gas quality was measured as the methane to carbon dioxide ratio. Initially this was low in all cells (1:1 or less) and variable, both from cell to cell and over time. However, by 1996 (1993 in cell 5) the ratio in all cells had risen and converged on a ratio of ~ 1.5:1, where it remained until completion.

At the scale chosen for the experiment it was not possible to replicate any of the experiments, so although an updated statistical appraisal of the data is planned it will not be possible to assign confidence limits to the results. However, there are clear qualitative differences in cumulative gas production from treated cells, relative to the control(s):

- Designated control Cell 1 had higher production than identical Cells 2, 3 or 4 before they received treatment, but all four lay within ±10% of their mean value.
- Cell 2 was lower than but parallel to Cell 1 until monitoring was discontinued.
- Cell 3 (liquid injection) was below Control prior to treatment and then gas production increased to nearly twice its own pre-treatment yield.
- Cell 4 (air injection) appeared to respond more quickly than Cell 3, and the effects appeared to last longer, with the greatest final yield. However, gas production may already have been increasing prior to injection for unknown reasons, making interpretation difficult.
- Cell 5 (sewage) clearly enhanced production early on, but at the expense either of poor production or poor recovery later, with an overall negative effect by 11 years.
- Cell 6 (commercial) probably enhanced production early on, but by the 11th year its cumulative yield was indistinguishable from the Control.

All cells at completion had peaked and were in declining production, although the shape and length of the decline phase had not become clear. However, even in the highest yielding cell to

date (Cell 4) we have only seen approximately half of the theoretical yield to date, suggesting that gas production will continue for many years at an ever-decreasing rate.

A suite of over 100 trace gases/volatiles was also measured on all 6 cells at completion, which we plan to examine in detail in the final phase of data analysis and interpretation, to try and draw conclusions about the effects of treatments (if any) on odour, corrosion and toxicity.

2.4 OTHER MONITORING RESULTS

It is only possible to give brief summaries of other results here. For further information contact the authors.

2.4.1 Leachate Level and Composition

The initial in-place water content of all cells was in the range 31.4-31.7 % (wet weight), except cell 5 which was 34.5 % due to the sludge addition (which contained 76% water). Leachate levels were monitored soon after capping then throughout the experiment, and as expected generally increased slowly as rainfall infiltrated and the waste degraded and settled. Cell 5 started significantly higher (8.5m above base, compared to a 3.3m mean for the other cells) and levels rose faster than in the other cells, which may have inhibited effective gas collection in this cell in the later stages of the experiment.

Leachate samples were taken at capping then at intervals over the life of the cells, including just prior to burial. Initially all were high strength, particularly cell 5, but by 1992 all 6 cells had settled into methanogenic mode, and the leachate compositions were very similar (even for cell 5). It is clear that ammonia has been accumulating steadily in all cells and correlates broadly

with cumulative methane yield. A careful examination of the data for any other trends and correlations is planned.

2.4.2 Settlement

Settlement rates were monitored closely at first, and showed between 500 and 700 mm of settlement over the first 2 years after capping, with the rate falling over time. Settlement correlated quite well with gas production for cells 1-4 and 6, accounting for at least half of the settlement observed (with cell 5 a notable outlier, settling faster than predicted by the other cells). Physical settlement (collapse of void space) accounted for a large proportion of early settlement. Further measurements have been taken at intervals, including a thorough survey of cap and well-heads just prior to burial, and it is planned to analyse this data and present the full picture in the near future.

2.4.3 Summary of Other Results

The initial in-place density of cells 1-4 and 6 all lay within 0.80±0.03 t/m^3. Cell 5 was 0.89 t/m^3. Temperatures at selected depths, with less frequent depth profiles, have been taken at intervals in all cells over the years. Peak temperatures were seen in 1991, up to 46°C in cell 6 (mean 38°C), usually at or close to the leachate surface, and this timing was probably associated with the development of a more balanced anaerobic microbial population. Thereafter peak temperatures fell but even the coolest parts of the cells (generally just below the cap) warmed, to give mean temperatures approaching 37°C in all cells. Prior to capping winter cooling was significant, but after capping no seasonal effects were evident. Profiles were measured again just

prior to burial and appear to show a cooling since the last measurement, which is consistent with the observed recent falls in gas production.

Solid waste sampling has been performed, although only twice – once in 1993 (with a limited suite of microbiological, physical and chemical analyses), and again just prior to burial (samples preserved but not yet analysed). A full chemical and physical analysis of these samples is planned, including Biochemical Methane Potential and other key measures of final quality, and we have proposed sufficient replication to make a statistically reliable appraisal both of the waste composition and of the methods used to assess it.

Meteorological data has been collected in various ways since the start of the experiment, including a period of several years during which a weather station was operational on the cells, with data logged frequently and automatically. This data will be incorporated in the planned final appraisal, particularly to compare atmospheric pressure and rainfall with gas flow and quality.

The cells have also provided valuable experience in practical management of landfill, including drilling of waste and placement of gas/leachate monitoring points, selection and operation of discrete and continuous gas flow equipment and quality measurement, data-logging, waste and leachate sampling and analytical techniques, and optimum measurement of settlement. The planned final report will discuss these issues.

It has been possible in the past to draw certain statistical conclusions, but a thorough statistical analysis has not been attempted for some years. Statisticians were consulted again prior to final sampling, and a thorough statistical re-appraisal is planned in the final phase of work.

Data from the test cells is perhaps uniquely complete, and has been used in the past to help develop and validate 2- and 3-D mathematical models of waste degradation, gas generation and migration, and of leachate flow and quality. Modellers at Southampton University were consulted prior to final sampling and will use the data to validate an integrated landfill model.

2.4.4 Cell 2 Leachate Recirculation

In 1997, replicate control Cell 2 gas monitoring was sacrificed to allow a 3-year study of the hydrodynamics of leachate recirculation, using the whole cell to examine achievable recirculation rates, movement and fate of added tracers, and effects of this on leachate chemistry. Flow and pressure findings were modelled mathematically. This work was significant and the key findings are reported separately in this Symposium (Beaven *et al* 2001).

2.4.5 Decommissioning and Final Studies

The test cell experiment has drawn to a close. Gas production had fallen, maintenance of the ageing cells was more difficult, and Shanks and McEwan needed the void space associated with the cells. The test cells have therefore been thoroughly characterised, and gas, leachate and waste sampled, prior to being buried. Results from these 'end of life' exercises will be reported in the future. A limited number of preserved waste and leachate samples might be released to researchers for further studies, given adequate justification.

2.5 DISCUSSION

Gas flow rates were very erratic throughout the experiment, from the scale of changes over a few minutes (primarily due to atmospheric pressure and wind) to a few hours/days (due primarily to rainfall and cap condition) to seasonal effects associated with the above. Overlying these real variations were anthropogenic variability due to various changes made, including week to week rebalancing of the well-head, cell and master valves, changes in flow monitoring method, and particularly the change in abstraction strategy from "O_2 minimisation" to "CH_4

maximisation" in 1995. Statisticians and modellers will be examining the data in detail in the final phase of work.

Apart from the valve adjustments required to meet the strategy, all these changes affected all cells more or less equally, so although absolute values may be questioned the relative performance of the cells is thought to be valid. The specific yields measured on control cells were perhaps double those recorded by others on operational landfills (Willumsen *et al* 1992), suggesting that the test cells were abstracted more efficiently than real life systems, or that methane flows were over-estimated. In contrast, the reliability of relative cell performance is perhaps best demonstrated by comparison of flow in Cells 1 and 2 (Figure 2-4): expression of results relative to Cell 1 eliminates most short- and medium-term variation. Absolute values should therefore be treated with caution, but relative differences are probably more significant.

To discuss the treatments in turn:

- Liquid injection (cell 3) appears to have stimulated methanogenesis significantly, an effect lasting for several years. This may have been due to addition of substrate to 'starved' areas, but water alone was as effective as leachate, suggesting that simply wetting dry areas was the most important factor. Mixing was probably also important – during the recirculation experiments on cell 2 (not reported here) some areas of the waste even below leachate were initially found to still be acetogenic in nature some 10 years after placement, but quickly became methanogenic when recirculation commenced. Landfill evidently may be a mosaic of gassing and inactive areas for many years, unless mixed in some way.
- Air injection (cell 4) appears to have had an even greater effect than liquid injection. The response was more rapid, and persisted for longer. Possible mechanisms so far considered include stimulation of the whole ecosystem via the aerobic heterotrophic organisms, the mass transfer warming of cool areas of waste, direct or indirect introduction of moisture and

associated substrate into dry areas of waste, the flushing of inhibitory materials from passive areas, and possibly the development of new physical gas pathways, aiding recovery.

- Sludge addition (cell 5) appeared to have positive effects on gas production in the early years, creating the conditions for rapid onset of methanogenesis and high rates of production of good quality gas. Probable factors include: water added, degradable carbon, trace nutrients and vitamins, and 'seeding' with active methanogens. In terms of gas-to-energy schemes, sludge addition would make the economics more attractive, with earlier and higher returns on investment. In the longer-term, gas production fell below control, and by 11 years the cumulative yield had fallen below control. This may have been due to toxicity, or to landfill conditions not suiting sewage methanogen species. In terms of environmental impact, sludge appears to prolong the time required for waste to stabilise, and gives rise to more leachate.
- Commercial waste addition (cell 6) may have promoted gas production in the early years, but this cannot be defended statistically, and by the conclusion of the experiment this cell had almost exactly the same cumulative yield as Cell 1.

A final programme of work is planned to include a final workshop. Please contact the authors if interested in attending or receiving proceedings from such an event.

2.6 CONCLUSIONS

- Waste from typical municipal sources and placed by normal late-1980's methods achieved a ratio of CH_4:CO_2 close to 1.5:1 within two years, and produced on average 5.3 m^3/t/y methane over 11 years, peaking at 8.4 m^3CH_4/t/y between 7 and 9 years after placement.
- Liquid- and air-injection (post-placement treatments) have shown significant enhancement in landfill gas production rates relative to the Control cell. Air-injection in particular was inexpensive and simple to carry out yet gave rapid and prolonged gas enhancement, with a net

increase in yield (mean 7.5 $m^3CH_4/t/y$ over 11 years) and low associated environmental impacts.

- Co-disposal with sewage sludge produced early onset of high volumes of methane-rich landfill gas relative to untreated waste, so might be of interest in sites designed for gas utilisation. However, these benefits were short-lived, with possibly a lower long-term yield. It was also associated with worse environmental impacts, particularly a prolonged time to achieve waste stabilisation, and enhanced volumes of leachate requiring collection and treatment.
- Co-disposal with commercial (paper) waste may have enhanced gas production modestly in the early years, and with no added environmental impacts.

2.7 ACKNOWLEDGEMENTS

The Authors wish to thank the many who have provided advice, assistance and support over the years of this experiment, particularly Shanks and McEwan Ltd. The Environment Agency provided funding through the Waste Regulation and Management Research Programme. Views expressed in this paper are those of the authors and not necessarily those of the Environment Agency.

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3. A STRATEGY FOR EMISSIONS BASED REGULATION OF LANDFILL GAS

C. DEED, J. GRONOW, A. ROSEVEAR, R. SMITH AND P. BRAITHWAITE

Environment Agency, Rio House, Waterside Drive, Aztec West, Almondsbury, Bristol, BS32 4UD United Kingdom.

ABSTRACT

The Environment Agency (the Agency) is responsible for the regulation of landfill sites in England and Wales. Since its formation, the Agency's regulatory strategy for landfill gas has been to require operators to demonstrate best practice. However, this approach does not allow environmental outcomes from site-specific landfill gas management to be easily illustrated or quantified. Greater clarity is now given to these environmental outcomes by augmenting best practice regulation of landfill gas with emissions-based regulation. This will require a "step change" in the management of landfill gas. However, it will enable the operator and regulator alike to respond to public concerns regarding landfill gas, including increasingly complicated health-related issues.

3.1 INTRODUCTION

The Environment Agency is the leading public body for protecting and improving the environment in England and Wales, a responsibility that includes the regulation of landfill sites. The major legislative requirements relevant to landfill gas at permitted landfill sites are the

Waste Framework Directive (Council of the European Communities, 1975), Landfill Directive (Council of the European Union, 1999) and the IPPC Directive (Council of the European Union, 1996).

From the perspective of a regulator, landfill gas is a "waste" that results from the landfilling of waste. Mature landfill gas from biodegradable waste is a mixture predominantly made up of methane and carbon dioxide and small amounts of hydrogen. It also contains varying amounts of nitrogen and oxygen derived from air that has been drawn into the landfill. Landfill gas will also contain a wide variety of trace components. Over 550 trace components have been identified in landfill gas (Environment Agency, 2002a), and together they normally comprise no more than 1% of the gas by volume.

In 2000, it was estimated that approximately 660 thousand tonnes of methane from UK landfills were released, accounting for 27% of the total UK emission of methane (NETCEN, 2002). There is an increasing level of concern about the uncontrolled release of landfill gas in the UK. This concern is reflected by the significant number of complaints relating to landfill gas that the Agency received from the general public. In 2002, the Agency received over ten thousand complaints related to odour, the majority of which related to landfill sites. Indeed the most significant of these cases result in a number of "landfill gas-related" prosecutions by the Agency every year. This concern has been exacerbated by the increased awareness of the potential health impacts of landfill gas.

Several epidemiological studies conducted around landfill sites have indicated an association between adverse health outcomes and the proximity to landfill (*e.g.* Dolk *et al.*, 1998), although no causal link has been established. In another recent example, a draft report by the United States Department of Health and Human Services on the Nant-y-Gwyddon Landfill site in Wales (ATSDR, 2002) concluded that it was likely that off-site exposure to landfill gas had caused an

increase in reporting of short-term health conditions such as respiratory ailments, severity of asthma attacks, headaches and skin rashes.

Historically, the Agency's strategy for the regulation of landfill gas has been based upon a "best practice" approach. However, this approach is limited because regulating the engineering doesn't easily allow site-specific outcomes associated with landfill gas management to be clearly demonstrated, particularly to local residents. There is also a need to increase the quantity of landfill gas that is currently collected and/or treated from many landfill sites in the UK, partly as a result of implementing the Landfill Directive and as a result of the UK Renewables Obligation Order (2002).

3.2 EMISSIONS BASED REGULATION

The Agency's approach to the future regulation of landfill gas is based upon a strategy of "environmental outcomes" and follows the principle of "emissions-based regulation". It takes the view that if you cannot monitor "it" you cannot manage "it". Historically, landfill gas has been a difficult emission to manage and regulate because of the variability in composition and rate of production. In addition, landfill gas is generated as an area source and limited toxicological information is available for many of the trace components present within the gas.

The strategy augments existing best practice methods with the concept of emissions-based regulation. The strategy is designed to achieve a balance in reducing global emissions without incurring the expense of increased local impact. Emission standards are introduced as minimum requirements to be met at all landfill sites, whilst requiring site-specific risk assessment to identify additional parameters or more stringent standards. The strategy provides for a step change reduction in the quantity of landfill gas that is uncontrolled and an improvement in the ability of the operator and regulator alike to directly respond to the concerns regarding landfill gas. It is recognised that many modern engineered landfills already have gas control systems that

are capable of meeting the emission standards. Such systems are not off the shelf packages and they must be adequately maintained.

Over the last three years the Agency has completed a significant research programme into landfill gas emission measurement. This research culminated in the production of a number of guidance documents and tools to enable the regulator and waste industry to deliver this strategy. Figure 3-1 lists the guidance, tools and underlying research projects that inform the delivery of this strategy.

At a fundamental level, the Agency's strategy for emissions-based regulation requires an understanding of landfill gas through the development of a gas management plan, developed as the result of an assessment of the risks posed by the site to human health and the environment. The core of the gas management plan is the monitoring and assessment plan, which includes the monitoring of:

- landfill gas composition at source, including quantification of trace components;
- engine emissions monitoring and compliance assessment;
- enclosed flare emissions monitoring and compliance assessment;
- surface emissions monitoring and compliance assessment; and
- air quality and meteorology.

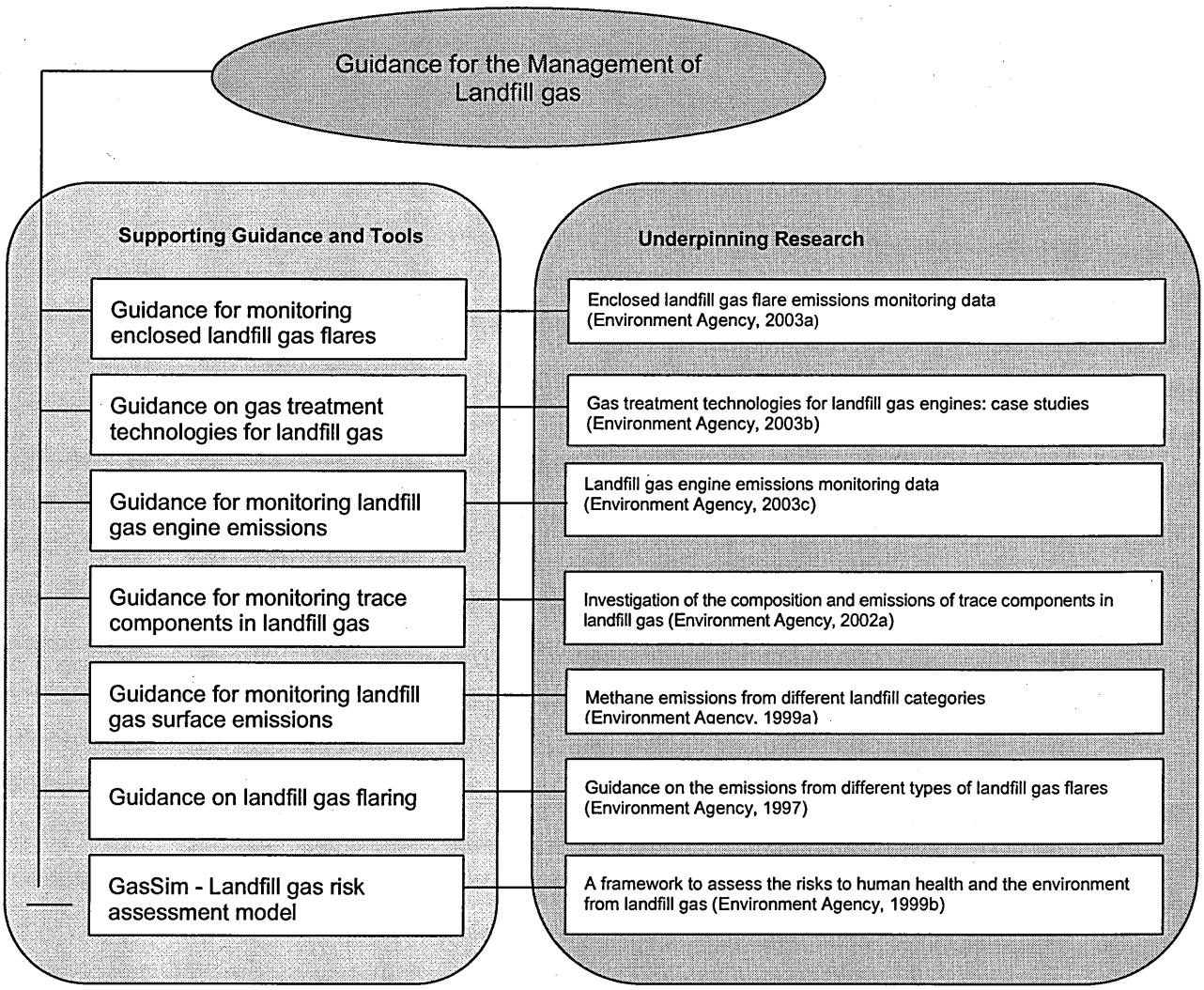

Figure 3-1. Relationship between the Agency's landfill gas guidance and supporting documents

3.3 THE MANAGEMENT OF LANDFILL GAS

The Agency guidance on landfill gas management (Environment Agency, 2002b) is an overarching document that sets out a structured approach to the management of landfill gas. This involves the assessment of the impacts, the implementation of control methods and the

monitoring required demonstrating proper performance of those controls. The gas management plan provides a framework within which the methods, procedures and actions for the control of landfill gas are provided.

The guidance also sets out a number of landfill gas management principles that the Agency expects landfills to adopt.

- A structured approach to the assessment of the risks posed by the landfill to health, environment and amenity. A tiered approach to the risk assessment should be adopted where the level of effort is proportionate to its magnitude and complexity. Tier 1 is risk screening using a basic semi-quantitative assessment. The other two tiers, identified as simple and complex risk assessments, consist of quantitative calculations using deterministic and probabilistic techniques respectively.
- All elements of the landfill gas control system are to be subject to Construction Quality Assurance, including the completion of validation reports following completion of all works.
- The establishment of a landfill gas management hierarchy (Figure 3-2) which encourages the utilisation of landfill gas and excludes passive venting of landfill gas.
- Maximisation of landfill gas collection, with an annual collection efficiency of 85% identified as a target.
- The provision of assessment criteria to determine when utilisation of landfill gas on site is feasible (Environment Agency, 2000). The criteria specified are (i) size of landfill (ii) geometry of landfill (iii) gas flow rate (iv) waste composition and (v) site location.

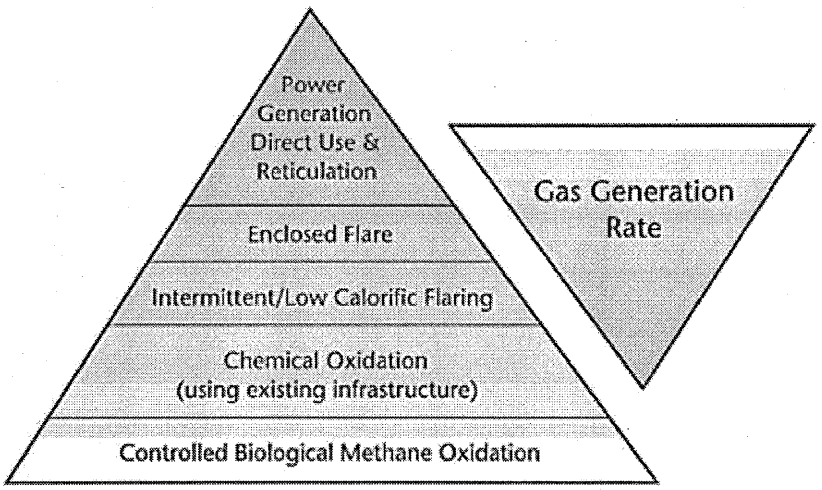

Figure 3-2. The Agency's landfill gas management hierarchy

3.4 MONITORING TRACE COMPONENTS IN LANDFILL GAS

Knowledge of the source-term trace gas composition at a landfill site provides vital information for the development of the Gas Management Plan. The Agency has developed a method for ranking the significance of individual substances based on the potential health or odour impacts of landfill gas (Environment Agency, 2002a). A database of measured concentrations was created and from this, typical UK landfill gas concentrations were derived. Separate potential odour and toxicological importance ranking scores were produced. The ranking value took into account toxicity data, odour threshold concentrations and physical properties. These values were combined with a range of measured landfill gas component concentrations, obtained from the database to derive the potential significance score. This score has been used to prioritise the typical landfill gas components relative to one another and thus identified important trace components for possible emission reduction (Table 3-1).

In particular this information will:

- provide an integral part of the demonstration of compliance with the objectives of the Landfill Directive;
- assist in defining the source-term gas composition for use in a site-specific risk assessment of gaseous emissions and the initial gas management plan;
- provide compositional data to the annual review and refinement of an existing gas management plan; and
- contribute to the source-term for gas generation models that estimate the emissions of landfill gas and that are used to produce the Pollution Inventory of specified substances released from a permitted site (*e.g.* the Agency's landfill gas risk assessment model GasSim; Environment Agency, 2002h). This has been designed to meet the requirements of Article 15(3) of the IPPC Directive to produce an inventory of principal emissions.

This Agency guidance (Environment Agency, 2002c) also recommends a suite of sampling and analytical methods that could be used to monitor these significant trace components in typical landfill gas. The guidance indicates that the priority trace components (Table 3-1) should be monitored annually as a minimum frequency, subject to site-specific circumstances *e.g.* significant changes to the gas management system or waste composition.

Table 3-1. Priority trace components to be monitored in landfill gas

Significant Trace Component	Sampling Method	Analytical Method
Chloroethane	Dual solid sorbent	ATD-GC-MS [1]
Chloroethene (vinyl chloride)	Dual solid sorbent	ATD-GC-MS
Benzene	Dual solid sorbent	ATD-GC-MS
2-butoxy ethanol	Dual solid sorbent	ATD-GC-MS
Arsenic (as As)	Solid sorbent	ICP-MS/AAS [2]
1,1-dichloroethane	Dual solid sorbent	ATD-GC-MS
Trichloroethene	Dual solid sorbent	ATD-GC-MS
Tetrachloromethane	Dual solid sorbent	ATD-GC-MS
Methanal (formaldehyde)	Reactive sorbent	HPLC [3]
Hydrogen sulphide	Direct on site measurement of raw gas	Hand-held instrument
1,1-dichloroethene	Dual solid sorbent	ATD-GC-MS
1,2-dichloroethene	Dual solid sorbent	ATD-GC-MS
Carbon disulphide	Dual solid sorbent	ATD-GC-MS
Methanethiol	Dual solid sorbent	ATD-GC-MS
Butyric acid	Solid sorbent	GC-FID [4]
Ethanal (acetaldehyde)	Reactive sorbent	HPLC
Ethyl butyrate	Dual solid sorbent	ATD-GC-MS
1-propanethiol	Dual solid sorbent	ATD-GC-MS
Dimethyl disulphide	Dual solid sorbent	ATD-GC-MS
Ethanethiol	Dual solid sorbent	ATD-GC-MS
1-pentene	Dual solid sorbent	ATD-GC-MS
1-butanethiol	Dual solid sorbent	ATD-GC-MS
Dimethyl sulphide	Dual solid sorbent	ATD-GC-MS
1,3-butadiene	Dual solid sorbent	ATD-GC-MS
Furan	Dual solid sorbent	ATD-GC-MS

[1] Automatic thermal desorption - gas chromatography –mass spectrometry

[2] Inductively coupled plasma mass spectrometry/atomic absorption spectrometry

[3] High pressure liquid chromatography

[4] Gas chromatography with flame ionisation detection

3.5 MONITORING LANDFILL GAS ENGINE EMISSIONS

To minimise the risks associated with landfill gas, it should be collected and combusted either in a landfill gas engine or enclosed flare. Currently in the UK, there are approximately 200 landfill sites generating electricity for the national grid. Landfill gas as a resource is estimated to be equivalent to around 6.75 TWh per year, around 2 % of the current UK electricity demand. However, as concerns over the potential global impacts of raw landfill gas have been addressed by combustion of the gas, this itself has led to concerns about the potential local impact from these emissions.

In response to these concerns the Agency has produced guidance (Environment Agency, 2002d), which specifies emissions standards for landfill gas spark-ignition engines. This provides a tiered approach where the generic emission standards are based on best practice, but are combined with stricter, site-specific, risk-based standards where appropriate. The emission standards are set in the light of research undertaken by both the Agency and the waste management industry. It has demonstrated that the operational emissions are achievable with a well-constructed, maintained and operated landfill gas spark-ignition engine of a particular age (Environment Agency, 2003c; Biogas Association, 2002). The proposed emission standards are given in Table 3-2.

The waste management industry in the UK is keen to develop a type approval system. Under this system, specific landfill gas flare and engine plant could be shown to be capable of meeting the emissions standards set by the Agency and could be demonstrated to do so reliably in the field, when operated at optimum conditions. The Agency would be supportive of a move

towards this approach, as it may lead to less expensive but more frequent monitoring of landfill gas combustion equipment, in the knowledge that its emissions standards were being met.

Table 3-2. Proposed emission standards for landfill gas engines and flares

Emission Parameter	Reference Method	Enclosed Landfill Gas Flares Emission Standards (Existing flare)* (mg. m^{-3})	Enclosed Landfill Gas Flares Emission Standard (Flare commissioned after Nov. 2002)* (mg. m^{-3})	Landfill Gas Engines commissioned between January 1998 & November 2004 Emission standard mg. m^{-3}**	Landfill Gas Engines commissioned after 1 November 2004 Emission standard mg. m^{-3}**
Nitrogen Oxides (NO_x)	ISO 10849: 1996	150	150	650	500
Carbon Monoxide (CO)	ISO 12039: 2001	100	50	1500	1400
Total Volatile Organic Compounds (VOCs)	BS EN 12619:1999*** BS EN 13526:2002****	10	10	1750	1000
Non-Methane Volatile Organic Compounds (NMVOCs)	BS EN 13649: 2002	5	5	150	75

Notes:

* These limits are based on normal operating conditions and load. (Temperature - 0°C (273 K), Pressure - 101.3 kPa and Oxygen - 3 % (dry gas)

** These standards are based on normal operating conditions and load. (Temperature - 0°C (273 K), Pressure - 101.3 kPa and Oxygen - 5 % (dry gas))

*** At sites with low total VOC concentrations

**** At sites with low to moderate total VOC concentrations.

Site-specific considerations may necessitate a stricter emission standard based on risk, either in terms of the generic emissions standard stated (*e.g.* a reduction of NO_x from 500 to 400mg/m^3) or in terms of additional parameters. These specific issues may include consideration of atypical

raw landfill gas (*e.g.* elevated levels of chlorinated compounds), and assessment of dispersion modelling (*e.g.* a potential breach of a local air quality objective).

In addition to the numerical emission standards given in Table 3-2, the Agency's guidance also recommends that:

- discharges should be vertically upwards and unimpeded by cowls or any other fixture on top of the stack;
- sampling sockets should be fitted;
- crankcase emissions must be managed to minimise their release to the environment; and
- methane and the rate of flow of the inlet gas must be continually assessed.

For landfill sites where the engines are unlikely to meet the Agency's emissions standards, the Agency has produced separate guidance on the potential for pre and post combustion clean up based on a cost benefit appraisal (Environment Agency, 2002e).

3.6 MONITORING ENCLOSED LANDFILL GAS FLARES

Agency guidance on landfill gas flaring (Environment Agency 2002f) details combustion principles and provides indicative operating conditions required to meet emission standards. Recognition of the need to manage and regulate emissions from landfill gas flares has mirrored that of landfill gas engines.

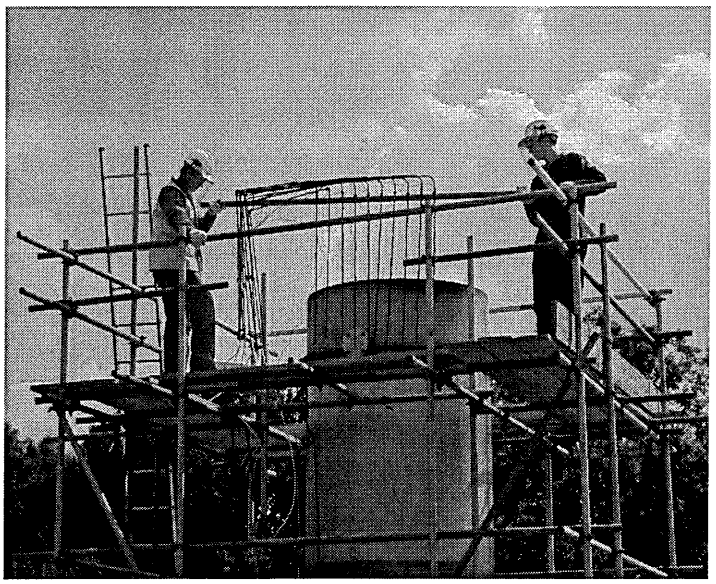

Plate 3-1. Multiple-probe system for end of pipe sampling

Further Agency guidance (Environment Agency, 2002g) provides emission standards and an associated monitoring protocol. The primary emission standards are a tiered requirement based on best practice and age of equipment, combined with a stricter, site-specific, risk-based standard where appropriate. The emission standards (Table 3-2) are based on data collected from ten operational landfill sites. This has been further informed by other data collected by the Agency and the waste industry. In addition to the emission standards and monitoring protocols, the guidance sets out a number of additional principles/requirements:

- the phased removal of open flares from landfill sites;
- enclosed flares to include sufficient shroud to fully enclose the flame;
- installation of sample ports/*in situ* probes for enclosed flares;
- sampling to be undertaken downstream of the flame; and
- sampling to consist of multi-point sampling (see Plate 3-1).

Paramount to the design of any monitoring for landfill gas systems is the need to consider health and safety aspects. Flares that do not meet the operational standards (*ibid.*) or have not been maintained may not be monitored in a representative or safe manner.

3.7 MONITORING LANDFILL GAS SURFACE EMISSIONS

Methane emissions through the cap of the landfill need to be monitored to identify faults and then to prioritise the remediation required. Additionally, the surface emissions must be quantified to estimate the emission of this important greenhouse gas. Agency guidance on monitoring surface emissions of methane applies to those phases of a landfill with permanent or temporary caps (Environment Agency, 2003d).

The monitoring of emissions through a landfill cap has two stages (Figure 3-3). During the preliminary stage the concentration of methane close to the surface is measured. This aids the identification of inadequacies in the gas containment and collection system. Only when these deficiencies have been remedied and the concentration of the gas above the surface is low, is it appropriate to begin a quantitative survey of surface flux. During the regular survey stage, the flux of methane emitted through the intact cap is measured using flux boxes. These quantify the total release of methane from the capped zones and identify any zones where the gas flux exceeds the Agency's emission standards (Table 3-3).

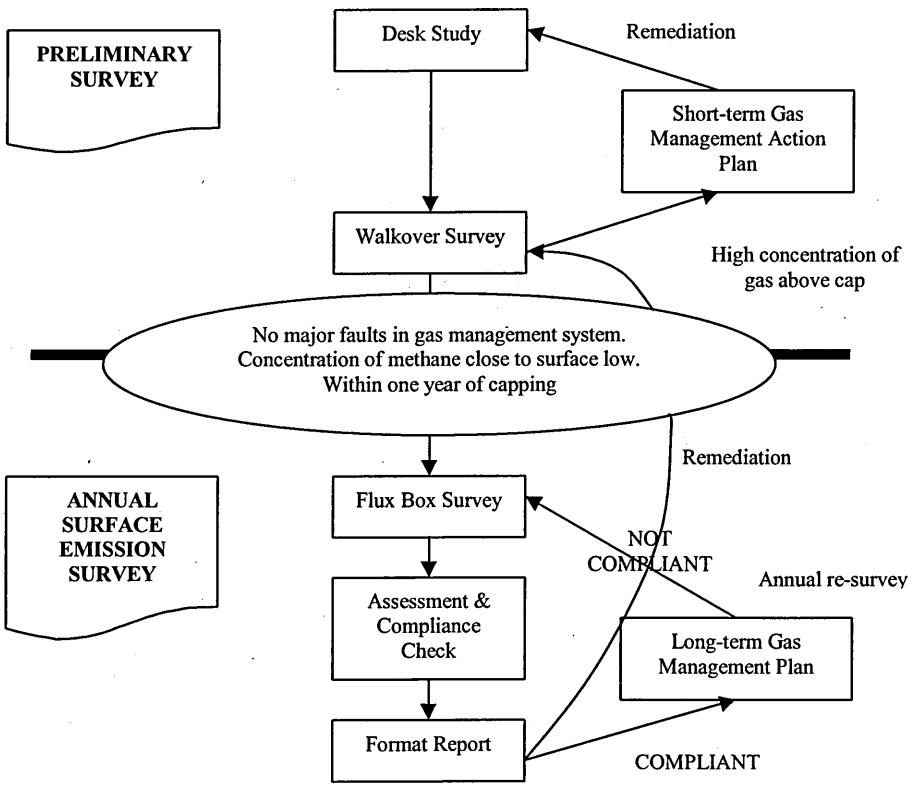

Figure 3-3. Phased approach to surface emissions monitoring

The preliminary stage involves a desk study and walkover survey using a Flame Ionisation Detector to scan the surface of the cap for significant concentrations of methane. This survey should be systematic and give semi-quantitative ranking of the emissions from various features. These data will be used within the gas management plan to remedy inadequacies in active control of the landfill gas. After any remedial work the cap should be resurveyed to identify further features that may need to be rectified.

The survey will not normally proceed to the regular monitoring stage until the concentration of methane in the air is:

- less than 100 ppmv immediately above the surface on the main zones of the cap; and
- less than 1000 ppmv close to any discrete feature.

The regular survey of methane emissions through the surface of an intact cap should use an array of flux boxes. The capped area is categorised into zones (an extensive area of landfill cap that is generally uniform and homogeneous) in which there may be individual features (a discrete area or installation from which emissions are higher than in the surrounding zone).

The flux boxes are sealed on the surface at a number of sampling locations within each zone and feature. It is important to ensure that the individual sampling locations must be representative of the area under investigation. The emission rate for a zone or feature is estimated by aggregating the rates measured by flux boxes at these representative monitoring points.

Field research in the UK (Environment Agency, 1999a; Environment Agency, 2001), showed that a low surface flux of methane can be achieved by following current best practice for site capping and gas abstraction systems. The Agency's proposed standards for methane gas emissions from a landfill surface are detailed in Table 3-3.

Table 3-3. Proposed emission standards for landfill gas surface emissions

Permanently capped zone	$1x10^{-3}$ $mg.m^{-2}s^{-1}$
Temporarily capped zone	$1x10^{-1}$ $mg.m^{-2}s^{-1}$

3.8 CONCLUSION

The management and control of landfill gas requires a co-ordinated and holistic approach to monitoring all the main emissions of the raw gas and its combustion products. It is anticipated that the development of the Agency's strategy for emissions-based regulation of landfill gas will provide for a step change in the regulation and management of landfill gas in England and Wales. Significant improvements in landfill gas collection and emissions reduction are anticipated, which in turn should deliver reassurance to the public that the Agency and the waste industry are responding to the increased concerns associated with landfill gas.

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4. ONSET OF METHANOGENESIS AND SURFACE METHANE FLUXES FROM EARLY OPERATIONAL PHASES OF UK MUNICIPAL SOLID WASTE LANDFILLS

DAVID L. BARRY [†], RICHARD SMITH [‡*], ROBERT G. GREGORY [§], CHRIS HARRIES [II] AND JAN R. GRONOW [‡]

[†] Atkins Environment, Woodcote Grove, Ashley Road, Epsom, Surrey KT18 5BW, UK

[‡] Integrated Waste Management Centre, School of Applied Sciences, Cranfield University, Cranfield, Bedfordshire MK43 0AL, UK

[§] Golder Associates (UK) Limited, Attenborough House, Browns Lane Business Park, Stanton on the Wolds, Nottinghamshire NG12 5BL, UK

[II] Waste Technical Services Ltd, 10 Burlington Terrace, Cardiff CF5 1GG, UK

* Corresponding author phone: +44 (0)1234 754963; fax (0)1234 751671; e-mail: r.smith1@cranfield.ac.uk.

ABSTRACT

Methane flux rates from landfill surfaces were quantified at 21 UK sites and the onset of methanogenesis was studied at one of those sites. For all sites, waste composition was predominantly municipal solid waste (55-100%) with varying waste depth (5-40m) and age (up to 28 months). Methanogenesis was evident within the wastes after 1-2 months and surface methane fluxes were measurable (0.06 mg m^{-2} s^{-1}) before methanogenesis was fully established. Methanogenesis data were obtained from three successive layers of *in situ* monitoring probes and pipes. Surface methane fluxes were obtained using a modified flux box. Overall, 650 flux datasets were analysed. Fluxes were found to be preferentially horizontal. The maximum average surface methane flux rate for all surfaces was found 20-24 months after initial waste placement. The average surface methane flux rate for younger wastes (20 months after placement), was 0.1 mg m^{-2} s^{-1}. Emissions from side slopes were generally four times higher (up to 0.4 mg m^{-2} s^{-1}) than corresponding top surfaces, suggesting that primary landfill gas emission controls can be better targeted towards such zones.

Keywords: landfill gas, methane flux, gas emissions

4.1 INTRODUCTION

Landfills are a significant source of methane which is the second most important greenhouse gas (GHG) after carbon dioxide. Installation of landfill gas management systems can significantly reduce gaseous emissions, depending on the timing of installation. National estimates of methane generation and emission rates (1) rely on a number of assumptions that take no account of the timing of the onset of methanogenesis. Annual methane fluxes from individual landfills through surface emissions in the absence of active gas extraction have been estimated to be >90% (v/v) of generated methane (2). Between 1990 and 2004, overall UK methane emissions have fallen by about 48% with landfill methane emissions decreasing by around 63% (3) in the same period. Methane contributed an estimated 10% of UK GHG in 2000 (5) and 7% in 2004 (3), a decrease due to increasing methane recovery, utilization and flaring at engineered landfills in accordance with European, UK government and Environment Agency legislation, policy and guidance (6-13). More than 70% of UK landfill gas generated is estimated to be flared or utilized (4). As UK landfills are permitted or re-permitted under the Pollution Prevention and Control regulatory regime (14), operators are required to monitor gas emissions from different sources (9,10,15-17), including landfill surfaces.

Previous emphasis has been on controlling emissions from completed cells, phases or entire landfills and in particular encouraging methane oxidation in cover soils (18-20), with little attention being given to emissions control during the earlier filling periods, other than to control odour. UK surface emission rates have been reported extensively (21-23) and relative emission rates are known for different surface and engineering features (10). Prior to this study, no assessments have been made of the scale of surface methane emissions during the pre-control period.

With an improved understanding of surface emissions during the first two years of landfilling, the time after which gas control systems are usually installed following the completion of a cell, better decisions can be made on the cost-effective timing of practical controls.

The idealised landfill gas production associated with degradation processes has been described (24). Methanogenesis is one part of this and its onset is defined as being when concentrations of methane and carbon dioxide are each approximately 40% v/v and when the methane flux is measurable. The first reported estimated timescale for the onset of methanogenesis, produced at a time when engineered landfills were still comparatively new and less well understood, was 7-12 months (25, 26). Data to identify when the generated methane is measurable as a surface flux is also lacking. This study quantified the onset of methanogenesis at one UK landfill site and measured the surface methane flux at the same site and 20 other representative UK landfill sites.

4.2 EXPERIMENTAL METHODOLOGY

4.2.1 Methanogenesis

Monitoring installations were designed to obtain gas composition data at three different layers within an area (approximately 50m x 20m) of a newly commenced landfill. Two sets of vertical multi-line probes and two horizontal perforated pipes were installed in each layer to assess how gas conditions varied. Monitoring pipe connections to the probes and perforated pipes were ducted to the landfill edge, an arrangement that continued for each successive layer of probes and pipe connections as waste levels increased.

The first layer of probes and pipes was installed in mid-September 2001 when the monitoring area had already received two waste lifts (about 6m total depth during 4 weeks) that had been placed over the basal drainage layer. Gas monitoring started mid-October 2001 and continued until December 2002, by which time waste depth was 25m.

Gas monitoring was carried out at approximately 3-4 weekly intervals using portable analysers to measure oxygen, carbon dioxide, and methane. Gas pressure levels were also recorded.

4.2.2 Monitoring installations

Probes were 300 mm long and consisted of 32 mm diameter MDPE tubing with 7mm holes. Probes were connected to a single line in a 5-core, colour-coded group of PVC-sheathed PE tubing, 1mm thick and 4mm outer diameter (OD). Surface connections were terminated with valves for sampling.

Monitoring probes had small-bore sampling lines which were vulnerable on-site and so a parallel supplementary monitoring system was installed. This involved more robust perforated 25 mm OD MDPE sampling pipes which were installed alongside the probes. Two colour-coded pipes, perforated with pairs of 7 mm holes at 500mm intervals along each 30m length of pipe, were set at the same waste levels as the probes and laid open-ended on top of the waste surface at that time. An un-perforated monitoring pipe was taken from each of the perforated pipes to the waste surface alongside the probe sampling lines. These pipes were also terminated with sampling valves.

The perforated pipes yielded a composite gas sample from the surrounding waste layer along the length of the perforated section of pipe but the gas sample taken was likely to have been dominated by the gas regime closest to the sampling point.

The monitoring frequency of 3-4 weeks was not adjusted to try to capture any gas pattern changes in shorter time scales. Measurements of methane flux from the waste surface were also made at regular intervals (on six occasions in total) in the monitoring area, using flux boxes at between 9 and 22 locations.

4.2.3 Surface flux box design

The conventional flux box design developed primarily for measuring surface flux from temporarily or permanently capped sites (9) was not suitable for the higher flux levels expected from uncapped operational sites or the uneven waste surfaces to be monitored. A literature review and site monitoring programme was therefore undertaken to establish the most appropriate technique. Monitoring sites were receiving waste so operational and health and safety issues were paramount (*e.g.* slopes were often >45°).

Monitoring methods included for pilot testing were: (i) static flux chambers; (ii) a modified static flux chamber; and (iii) a dynamic flux chamber, where a pump controlled the gas pressure within the chamber.

Static flux chambers were shown to be generally suitable for this study but the higher flux rates from more gas productive waste areas were difficult to measure due to the speed at which the methane concentrations increased within the flux box. Tests using a static flux chamber with an attached Tedlar Bag had a better reproducibility and a wider range of detection. Dynamic flux chambers were not adopted as they were affected by pump speed and monitoring time.

The pilot tests of surface emissions monitoring showed that: (i) surface fluxes were generally lower than originally expected; (ii) there were significant spatial and temporal variations in near-surface concentrations; and (iii) waste slopes were a major source of emissions. This led to further evaluation of techniques, resulting in the design of a smaller round flux box (Figure 4-1) that would be more appropriate to the range of physical conditions on the top and side slopes of operational landfill surfaces.

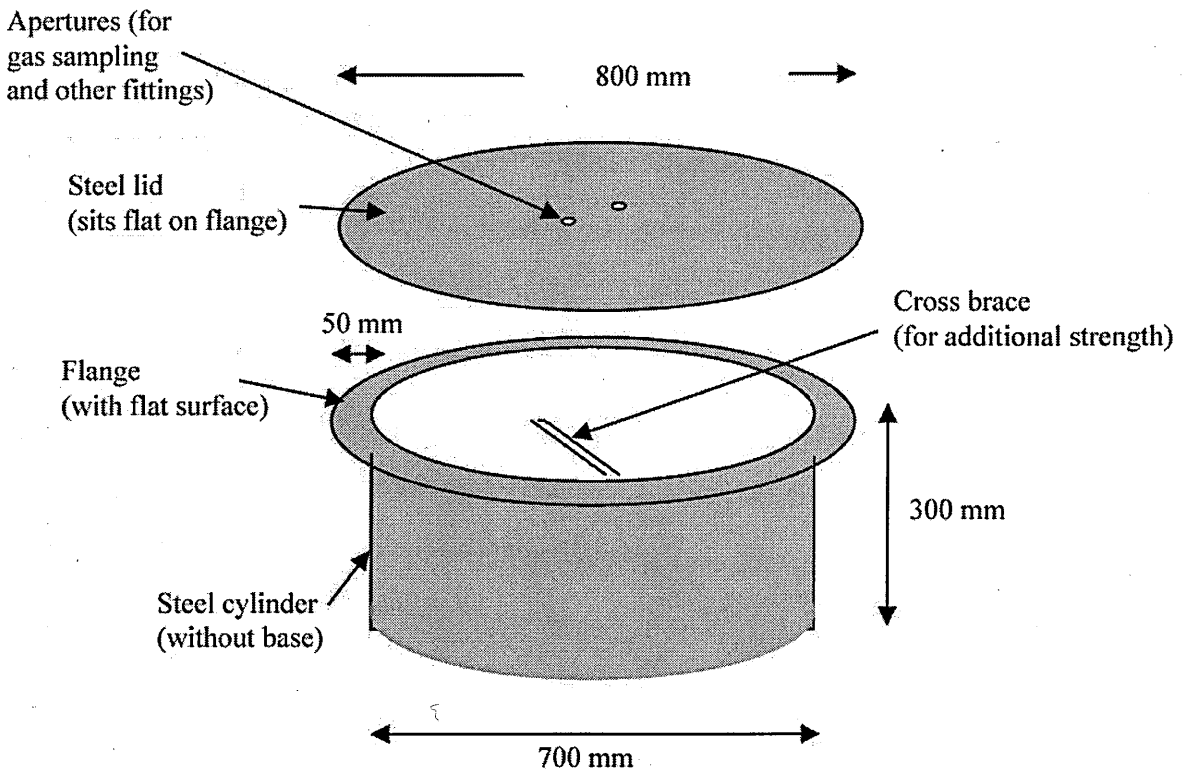

Figure 4-1. Flux box design

4.2.4 Flux monitoring strategy

An assessment of spatial variations in the results of the funnel technique, which used a flame ionization detector for measuring initial variations in near-surface concentrations, informed the size and number of flux boxes required. The strategy needed to reflect the likely range of existing surface flux emission conditions and allow representative flux data. The final flux box design was advantageous because box placement took place before the lid plate was applied so the flux box initially reflected normal near-surface conditions before measurements started.

4.2.5 Site selection and monitoring programme

The 21 landfill sites chosen for flux monitoring were selected from a range of waste operators on the basis of: (i) geographical coverage across the UK; (ii) variability of waste composition (55 to 100% MSW); (iii) variability of waste age (up to 28 months old) and depth (up to 40m); and (iv) accessibility for monitoring. US experience indicated that for statistical robustness at least 25 sites/visits were needed (27).

Flux data from operational cells were generally collected in a regular grid pattern on top surfaces (360 positions), but were more in linear patterns on side slopes (240 positions). 50 positions were monitored at or near the edge of the landfill. The considerable emphasis placed on measuring side slope emissions was based on our early finding when testing the experimental design that the emission rates from slopes appeared much higher than from the corresponding top surfaces.

Information on the waste age and depth for each cell monitored was provided by the site operator. For waste age, the actual age of different waste layers at any time varied across the entire cell, depending on the filling regime. In the absence of actual cell filling rates, an average value was ascribed at each monitoring visit, assuming that all wastes had been placed in a continuous manner since filling started.

The site monitoring programme started in August 2001 and was completed in October 2002, with 32 site visits providing almost 800 datasets. Some of these (140) were ultimately excluded from the analyses because they suggested a negative flux *i.e.* the dC/dt slope gradient was negative due a decline in methane concentrations after an initial rise. This decline was most likely due to an inward leak of air through the basal seal of the flux box, as might be amplified by a gusting wind for example.

4.2.6 Data presentation

Gas concentration data were processed to calculate flux emission rates for individual locations. The first step was to convert the measured methane concentrations from parts per million (ppm) to mg m^{-3} units using the calculation below.

$$C\ [mg\ m^{-3}] = C\ [ppm] \times \text{molecular weight } CH_4 / \text{molecular volume } CH_4 \quad (1)$$

$$= C\ [ppm] \times 0.714$$

The flux rate was then calculated as being:

$$Q = V/A\ (dC/dt) \quad (2)$$

where **Q** is the flux density of the gas (mg m^2 s^{-1}); **V** is the flux box volume (m^3); **A** is the flux box base area (m^2); and dC/dt is the rate of change of gas concentration in the chamber with time (mg m^3 s^{-1}). Changes in concentration (dC/d*t*) were regarded as significant when $r^2 > 0.85$ in linear regression analysis.

Collected flux data were examined statistically with respect to many site factors. Data were stored in an Access database with defined queries to aggregate data into tables for detailed analysis in Excel and SPSS. Overall about 650 data sets were used in the assessment process. The evaluation of the data involved several different approaches due to the complexity of the many variables associated with the landfills, as well as the different meteorological conditions prevailing during the data collection process. Initially some attempts were made to prepare cluster graphs for different parameters such as age, depth, UK region, and MSW content.

However, it became clear that the only parameters that could be meaningfully evaluated were waste age and, to a lesser extent, waste depth. Even with sites of apparently similar ages, there were some significant differences in the measured emission rates (Figures 4-3 to 4-5).

Two-month average waste age groupings represented the most effective way of presenting generic site data. Data is shown graphically for the 2-month bands for both the individual sites (identified by letter with each number indicating a visit) (Figure 4-3) and the overall site age-groupings (Figures 4-4 to 4-5).

4.3 RESULTS

4.3.1 Onset of methanogenesis

Gas concentration data showed some erratic patterns, particularly with respect to oxygen levels. This suggests that there was a degree of dilution by air through joint weaknesses in some probe and pipe systems, probably near the edges of the site. This was most likely due to mechanical damage from operational plant. Some of the probe sampling lines became blocked, as evidenced by the vacuum created when attempting to take a gas sample.

The variable oxygen concentrations could also have related to air ingress to the wastes particularly during periods of rising barometric pressure. However, due to the shallow depth of wastes when these data were collected, inconsistencies were most likely due to a system weakness, rather than meteorological or other factors. Therefore, data sets considered to have been influenced by operational or mechanical factors were excluded from analysis.

Dilution of gas samples by air ingress was not considered to have had an effect on the methane to carbon dioxide ratios. These ratios showed a clear trend towards a ratio of 1:1 or greater and the time taken to reach this ratio was about the same for each layer (6-7 months actual time).

Figure 4-2 shows the interpolated trends in gas concentrations for all probes and pipes over time. The profiles, which were calculated using a polynomial function, show a pattern of progression through Phases I to III of the idealised evolution of landfill gas production (24). They showed that Phase IV (stable landfill gas production), is reached about 1 year after initial placement of the waste. Phase IV is typified by high rates of gas production and ratios of methane to carbon dioxide in the range of 1 to 1.5.

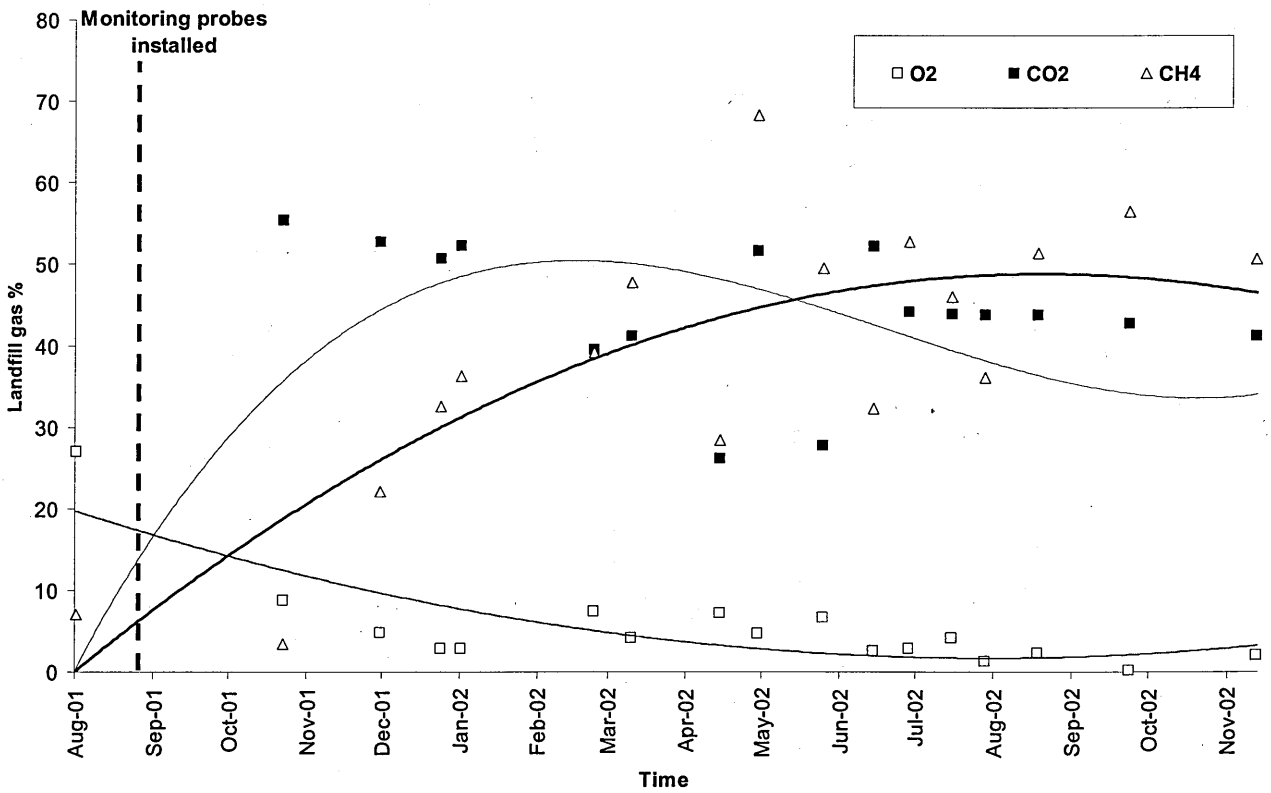

Figure 4-2. Monitored bulk gas concentration profiles defining the onset of methanogenesis

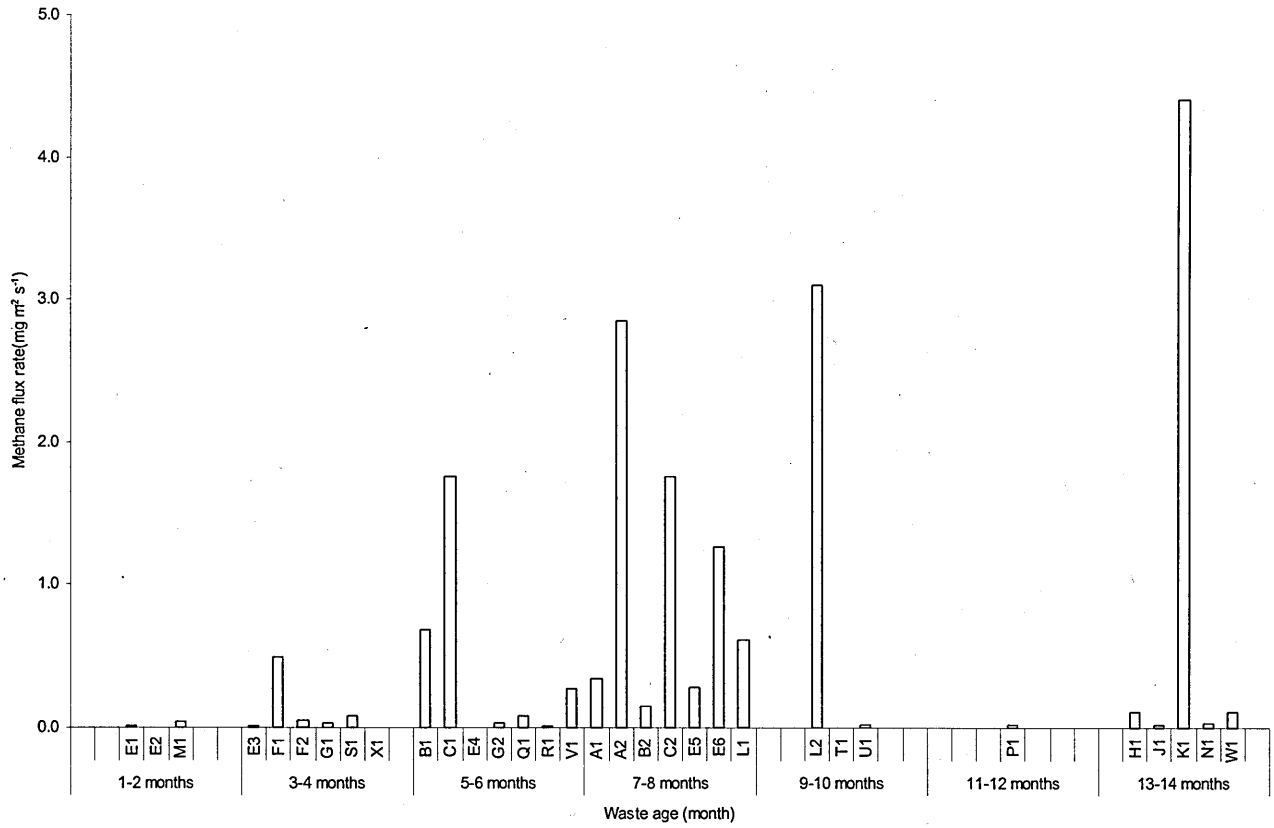

Figure 4-3. Flux rates for sites by age group, all data (top, edge and slope surfaces) up to 14 months

4.3.2 Flux on individual sites

Figure 4-3 shows that the range of overall flux rates for individual site visits varied by a factor of about 25. Some of these variations were due to very high emissions from some areas of some surfaces, such as near a landfill edge (site K1). Some slope flux data were found to be lower than the corresponding top fluxes, although the reverse was expected. No firm conclusion could be deduced solely from the average emission rates for individual site visits although data were of the same order of magnitude as other studies (28).

4.3.3 Flux from sites grouped by age

The most significant initial observation (Figure 4-4) is that the flux rate for the 11-12 months age group appears to be inconsistent with the other age-band patterns. However, there was only one site (P1) in that age band and only eight sample points used in the analysis. Site records showed that a better than normal daily cover (200mm) was being used, enabling consistently low measured flux rates.

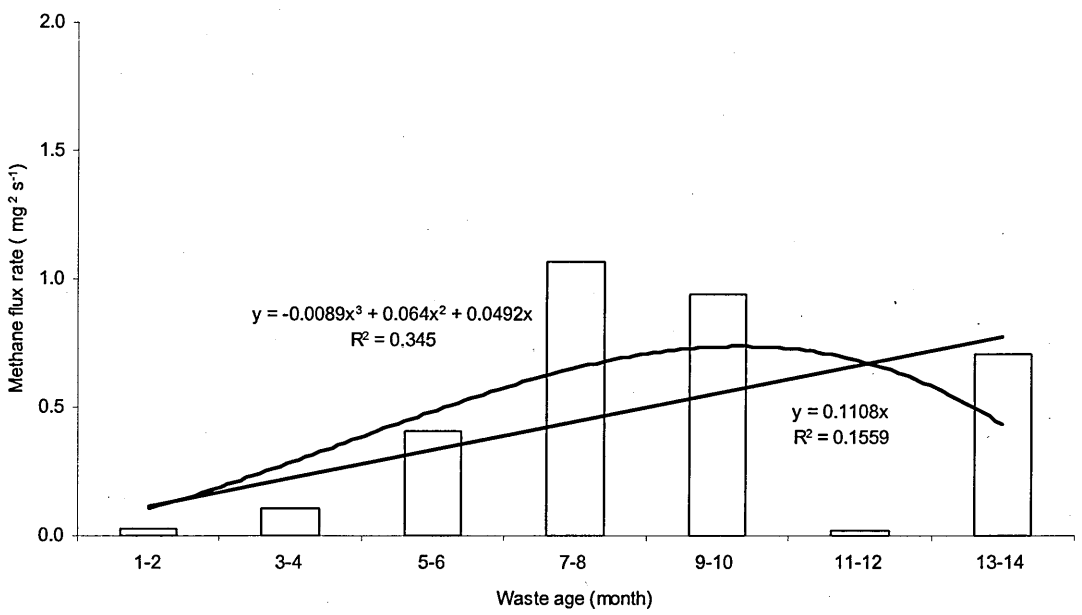

Figure 4-4. Overall individual site flux rates incorporating top and edge surfaces

Despite there being a 122 sample data set used in the 13-14 months age group, the average flux rates did not follow an expected pattern of increase that was evident for the periods up to 10 months (Figure 4-5a). When the higher flux rates from edges are removed from the evaluation

(leaving 188 data sets), the pattern of emissions is even more unexpected for the period >10 months. With regard to the four sites that comprised the 118 data sets, there were no particular site or meteorological characteristics that would help explain why the overall flux rate was much lower than what might reasonably have been expected.

However, because the main project emphasis was on site flux rates during the earlier rather than later phases of waste placement, and because the flux rates at 10 months were already high, it was appropriate to focus the overall data evaluation on the period up to 9-10 months (average). These data could then inform decisions regarding installation of emission controls, and the timing of such installations.

4.4.4 Flux from sites grouped by age: up to 10 months

Figures 4-4 and 4-5 show a consistent pattern of increasing flux rate with increasing average age up to 9-10 months. These data are considered to represent a reliable indicator of the flux rates during this overall period, and the best fits for both a straight line and for a power regression line (constructed using the individual flux data sets rather than the averaged age group values) show good correlation values of $r^2 = 0.75$ and 0.97 respectively (Figure 4-5b).

The power best fit line for the period up to 10 months average age (Figure 4-5a) suggests that the flux rate may still be increasing significantly in the period after 10 months. A steady state average flux rate was expected to become evident reasonably soon after this period in the continued absence of gas controls because the deeper, highly methanogenic wastes would also be expected to be reaching steady state generation conditions. However the scale of average emissions measured after 10 months (average age, but 20 months elapsed time) is shown to have reached a value 10 times greater than the regulatory emission standard for temporarily capped sites (0.1 mg m^{-2} s^{-1}) and 1000 times greater than that for permanently capped sites (0.001 mg m^{-2} s^{-1}) (9).

4.4.5 Top surface versus side slope surface emission rates

The segregation of flux data from top surfaces (Figure 4-5b) and from cell side slopes (Figure 4-5c) (each excluding any landfill edge data) showed that, on average, the slope emission rates were 3 to 4 times the top rates. On individual sites the ratios varied considerably but the complexity of local site layouts did not permit a close assessment of the variations.

The top surface data showed that the monitored emission rates exceeded the regulatory emission standard for temporary caps (0.1 mg m^{-2} s^{-1}) after about 7-8 months, while the slopes would exceed the standard before 3 months. The top flux rate would usually be expected to apply to a much larger surface area than the slope flux rate.

The differential flux rates appear to reflect the classical landfill gas production curves (24) and reflect a greater lateral permeability of waste masses, most probably as a function of the waste layering and daily cover effects. The greater flux rates through side slopes are consistent with a landfill study in the USA (29) that showed that effective gas permeability of MSW is considerably greater laterally than vertically (lateral suction influences were found to be 7 to 8 times greater than the vertical).

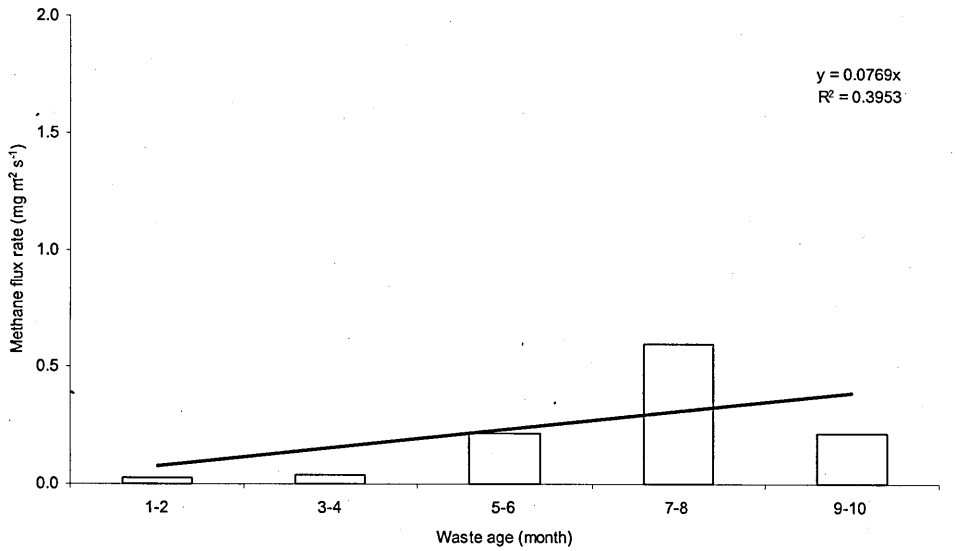

Figure 4-5a. Flux rates for top and slope surfaces with waste age up to 10 months

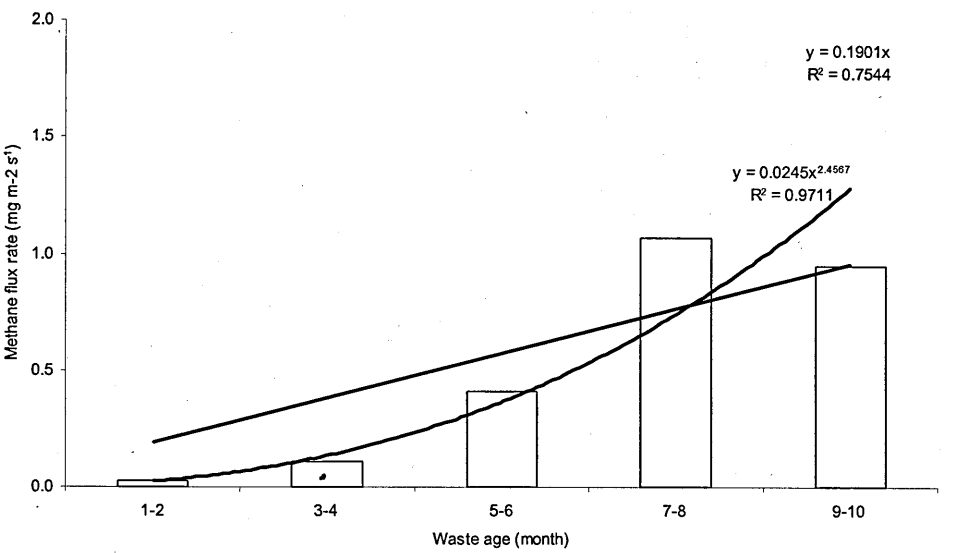

Figure 4-5b. Flux rates for top surfaces with waste age up to 10 months

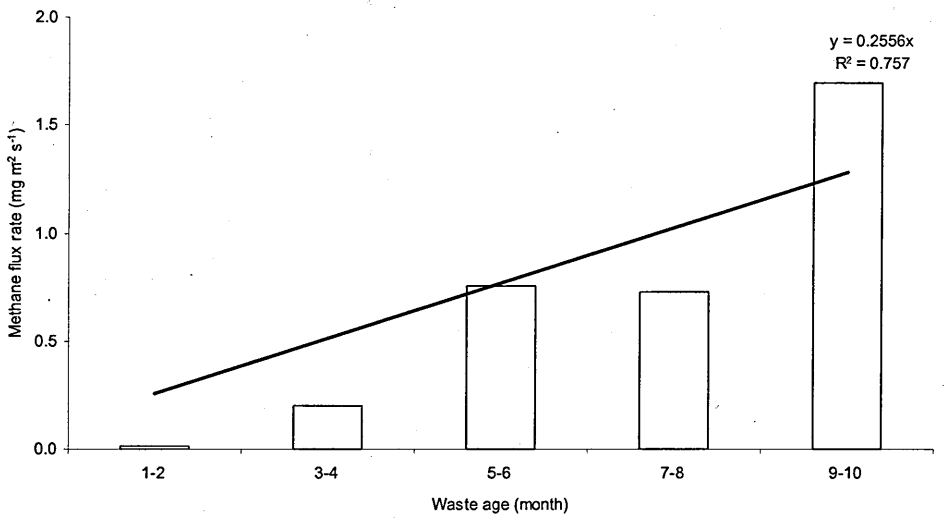

Figure 4-5c. Flux rates for slope surfaces with waste age up to 10 months

4.5 DISCUSSION

4.5.1 Methanogenesis

Data from perforated pipes showed a much more consistent pattern of reducing oxygen concentration over time compared to the probes. However in both cases oxygen depletion was consistently recorded after about 4 months. The oxygen reduction pattern in the three waste layers seemed more consistent in the shallower Layer 3 than in the two deeper layers. However, Layer 2 showed very low initial oxygen concentrations, and at the same time the concentrations of carbon dioxide and methane were both relatively elevated, phenomena that are not easily explained. No significant methane concentrations were recorded whilst oxygen levels were high.

Some high concentrations of carbon dioxide were noted almost immediately in the three layers. The methane concentrations increased progressively from about two months after waste placement/monitoring installations in the Layer 1 probes; this pattern was slower in the pipes. In Layer 2, however, there appeared to be a quicker start-up time for probes and pipes (generally < 2 months), while in Layer 3 the start-up time was somewhere between that of the other two layers.

If the time taken to reach >40% methane (v/v) is assessed, then the differences between the three layers did not appear to be significant, taking about 6 months after waste placement. The apparent lack of significant effect from rising gases could be simply related to the lesser vertical gas permeability of the wastes, as concluded from the flux data. The methane to carbon dioxide ratio increased to >1 about 5-6 months after commencement of waste filling.

Figure 4-2 shows the interpolated trends in gas concentrations for all probes and pipes. This is the first time that a real timescale has been ascribed to the different stages of the gas generation phases, in this case from the start of waste emplacement in a new landfill cell in August 2001.

The gas pressures measured in the probes were highly variable, albeit with low pressures recorded in all cases except for a period of about 4 months in Layer 1, some 10 months after waste placement. The high pressure for this period coincided with a significant increase in surface flux monitored on one occasion when atmospheric pressure was exceptionally low (978 millibars). However, the high pressure differential was not measured in Layers 2 and 3, layers that were expected to have responded to a low atmospheric pressure. Relative pressures were generally in the range of 0-2 millibars, with the majority of readings <0.5 millibars.

Monitoring showed that, almost from the outset, surface methane fluxes were measurable (0.06 mg m^{-2} s^{-1}) before the methane concentrations in any of the underlying wastes had reached 40% (v/v). However, flux measurements at the site were not measured at the same frequency as probes and pipes, so a full assessment of data is not possible.

Results strongly suggest that, whilst surface flux can be a function of both advection and a concentration gradient, the scale of contemporaneous methane flux measured is most likely to be based on advection of landfill gas.

The data clearly show that the methanogenic process commences effectively about 2 months after waste placement and can be well-established after about 6 months. The surface fluxing of methane has been shown to commence, and to be relatively significant, long before general methane concentrations in the waste mass reach >40% (v/v). This finding strongly suggests that early surface fluxing at least may be dominated by advection processes for both methane and carbon dioxide, rather than concentration gradients; carbon dioxide concentrations reached >60% (v/v) within 2-3 months.

One interpretation of the data is that gases from the lower waste layers do not have any significant effect on the gas regimes in the upper layers, an effect that was originally expected. This supports the conclusion that vertical gas permeability is relatively low within the wastes and that measurable surface flux emissions are not directly dependent on overall waste age.

4.5.2 Flux

The evaluation of the data exemplified the well-known complexity of gas generation and flux emissions from an operational landfill. Not only were there considerable spatial differences in results, but flux rates in areas near to adjacent capped waste cells were higher than positions further away. Whilst these findings indicate significant local changes in emission patterns, the database covering these aspects is not robust enough to fully quantify the observed effects of local capping and other possible influences.

Similarly, data taken from near the edges of three landfills, all of which had containment membrane barrier systems, showed emission rates several orders of magnitude higher than for

surface locations further away from the edge. This demonstrates a preferential pathway at the edges of a site such that the edge zone behaves like an open side slope, except that the gas emission zone is concentrated into a relatively narrow surface strip.

The emission rates from side slopes and edges were demonstrated to be on average 4 times higher than from top surfaces, a finding that is considered to reflect a greater lateral gas permeability of landfilled waste. Although the top surface flux rate might be relatively low, on most sites the top surface areas are much greater than the slope surface areas, and so could still represent the major flux volume for individual cells.

4.5.3 Environmental significance

An assessment of a series of hypothetical sites that had various filling rates, cell sizes and top/slope methane emission ratios, highlighted several factors that could influence the size and operational practices on future landfill cells in order to minimise emissions from the earlier period of landfilling.

A slower filling rate results in greater emission volumes for the same waste mass during the longer waste placement period. This reflects higher emission rates that develop with waste age but also the longer operational period of a cell, and the consequent delay in permanent capping and gas control. By month 30 (after commencement of waste placement) with a low filling rate, a large hypothetical cell could only be 50% full, but it would have emitted some 16 times as much gas as that from a high waste filling rate which the cell could have reached by month 10.

For small cells, the emission volumes for the different filling rates could be as little as half that emitted from a large cell, depending on the top/slope emission ratio. Small cells filled at a high rate could emit some 96% less methane than large cells that were filled slowly, assuming a fixed volume of waste. The doubling of the filling rate in a large cell would reduce emission volumes

by 75%, with a further reduction of more than 15% by trebling the filling rate. For small cells the relative reductions in flux emission volumes would be more than 85%. Small cells can have a disproportionately large slope surface area compared to the top surface so can exhibit high emission rates and high emission volumes.

From monitoring a series of *in situ* probes it was not apparent that the gas flux from lower waste levels affected the rate of methanogenesis in the upper, shallower layers. This observation reinforces the conclusion drawn from the surface flux measurements that the vertical gas permeability of the wastes is far lower than the horizontal permeability.

The overall database of surface flux measurements in this study has a considerable degree of randomness *e.g.* the types of sites monitored throughout the UK and the wide range of site and meteorological variables that can affect both gas generation and flux emissions at each of those sites. The monitoring locations were set across all areas of operational cells, including near (i) the centre of cells, (ii) the edges of cells, (iii) tops of cell slopes, (iv) capped cells and (v) on cell slopes. Thus there is high confidence in the analysis that, for landfills in general, the average rates of flux for the whole exposed waste area of a cell would be similar to that shown in Figure 3-4.

Landfill edges can have the highest rates of surface emission. This is consistent with the preferentially lateral gas migration potential within the wastes being amplified by the barrier and high permeability effects at the landfill edge, whether through specific design objectives such as leachate drainage layers, or simply through the consequence of heterogeneous wastes lying against a smooth surface.

The flux emission rates from top surfaces were found to increase relatively slowly with increasing waste age. This suggests that the influence of the deeper, more substantial gas regime is not so significant and that the flux rate might be equally influenced by near-surface waste layers. The generally greater areas of top surfaces, when compared with side slope areas, means

that the lower top surface emission rates could still represent the major emission from a waste cell.

The maximum flux rate appears to be reached about 20-24 months after commencement of waste placement. However, the reasons for the apparent reduction in emission rates after that time are not understood. Nonetheless, the relatively high average flux rates measured after about 12 months (actual time) would appear to justify consideration of active control measures being installed around this time.

Calculations for a hypothetical range of scenarios showed that smaller cells (plan area) with faster filling rates should result in considerably lower emission volumes than larger cells with slower filling rates. Such benefits will only be realised if temporary covers and gas abstraction systems are installed soon after completion of each cell. The results show that primary emission controls should focus on the edge zones of sites and on side slopes without compromising risk of air ingress. Further research is needed on the design, performance and adequacy of different gas control systems.

4.6 ACKNOWLEDGEMENTS

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4.7 SUPPORTING INFORMATION AVAILABLE

Worksheets of flux data analyses and methanogenesis gas data (3 layers). This material is available free of charge via the Internet at http://pubs.acs.org. A final research report is published separately (31).

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5. ESTIMATING POLLUTANT REMOVAL REQUIREMENTS FOR LANDFILLS IN THE UK: I. BENCHMARK STUDY AND CHARACTERISTICS OF WASTE TREATMENT TECHNOLOGIES

D. H. Hall[1], D. Drury[1], J. R. Gronow[2], A. Rosevear[3], S. J. T. Pollard[2] and R. Smith[2]*

[1] Golder Associates (UK) Ltd., Attenborough House, Browns Lane Business Park, Stanton-on-the-Wolds, Nottingham, NG12 5BL, UK

[2]Integrated Waste Management Centre, Sustainable Systems Department, School of Applied Sciences, Cranfield University, MK43 0AL, UK

[3] Environment Agency, Kings Meadow House, Kings Meadow Road, Reading, RG1 8DQ, UK.

* Corresponding author Tel +44 (0)1234 754963; Fax +44 (0)1234 376171; e-mail r.smith1@cranfield.ac.uk

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ABSTRACT

Introduction of the EU Landfill Directive is having a significant impact on waste management in the UK and in other member states that have relied on landfilling. This paper considers the length of the aftercare period required by the municipal solid waste streams that the UK will most probably generate following implementation of the Landfill Directive. Data were derived from literature to identify properties of residues from the most likely treatment processes and the probable management times these residues will require within the landfill

environment were then modelled. Results suggest that for chloride the relevant water quality standard (250 mg l^{-1}) will be achieved with a management period of 40 years and for lead (0.1 mg l^{-1}), 240 years. This has considerable implications for the sustainability of landfill and suggests that current timescales for aftercare of landfills may be inadequate.

Keywords: Equilibrium; completion; emissions; MSW

5.1 INTRODUCTION

In this work, presented here as the first of a series of three companion papers, we consider that landfill pollutant removal requirements are intrinsically linked to the concept of environmental equilibrium. Equilibrium is defined here as that state when emissions from a landfill site occur at a rate that allows sufficient natural attenuation in the surrounding environment to prevent environmental harm, so management is no longer required. To embody the principles of sustainability, equilibrium can only be achieved when the management period (post-closure when the site has ceased accepting waste for disposal, alternatively known as the aftercare period) is measured in decades rather than centuries. Throughout aftercare, the landfill licence or permit holder is required to take active measures to control pollution from the site. This involves monitoring, ensuring integrity of management and engineering systems such as leachate, landfill gas and restoration, and if necessary, taking corrective action until such time as licence or permit surrender is accepted by the regulator. In the UK, the Pollution Prevention Control (PPC) regime [1] requires an estimate of the time taken for a landfill to achieve completion, but guidance is lacking.

When assessing equilibrium there are a number of issues that must be considered. Firstly, it is expected that landfill liner performance will diminish with time and therefore the rate of

leachate or gas leakage may increase. Secondly long-term hydraulic performance of a landfill is dependent on the management and control of leachate levels. If a landfill operator ceases to manage their liabilities, financial provisions allow the regulatory authorities to take over management of the site. Financial provision is usually restricted to a 30-60 year period.

The setting of biodegradable municipal waste (BMW) diversion targets in the EU Landfill Directive [2] and the need to pre-treat waste prior to landfilling is bringing about considerable changes to the composition of wastes going to landfill. There are concerns relating to the sustainability of landfill in general and there is a growing recognition of the long timescales required to achieve equilibrium status; there is a possibility that the change in the nature of waste going to landfill may exacerbate this problem.

In the current study, treated wastes going to landfill were assessed using the number of years to achieve equilibrium status. A current-day landfill, designed and operated largely in compliance with today's UK guidelines was used as the benchmark. We have reviewed residue flows that can be anticipated from various waste pre-treatment and treatment processes with the aim of determining the revised properties of the waste residues that are destined for landfill. Factors that affect the biodegradable content, particle size or density of the material, or lead to the removal of metals etc. could influence the behaviour, both biological and chemical, of waste residues to an extent that landfill management options might not be the same (or even appropriate) for all waste streams. Ultimately, the success or otherwise of the options studied depend on both time and cost to achieve equilibrium status. Those techniques or combinations of technology and landfill management that can approach the definition of equilibrium were identified, and the ability to operate the scheme within the current legislative constraints

considered. The waste processes investigated are shown in Table 5.1 as variants of mechanical biological treatment (MBT), refuse derived fuel (RDF) and energy from waste (EfW).

Table 5.1. Overview of waste processes investigated.

Waste treatment process	Description
MBT product	Where mechanical biological treatment results in a product that should have been useable but is unable to be sold or used and must be disposed of to landfill.
MBT/C	MBT incorporating composting.
MBT/AD	MBT incorporating anaerobic digestion.
RDF co-incineration	Refuse derived fuel produced by mechanical sorting only.
RDF dedicated incineration	Refuse derived fuel produced by mechanical sorting only.
RDF – flock only	Where flock cannot be sold for incineration and is destined to landfill.
RDF/MBT	Where RDF results from a more sophisticated MBT process.
EfWMB	Energy from waste (mass burn).
EfWFB	Energy from waste (fluidised bed).
ATT	Advanced thermal treatment – pyrolysis/gasification.

This study has, as a consequence of data shortages, concentrated primarily on the inorganic components of leachate. Beyond understanding the origins of these compounds, little is known of their mass balance through various waste treatment processes. Data relating to the elemental composition of MSW is available for the primary fractions (*i.e.* paper, plastic, textiles, etc.) but we found few data indicating the elemental analyses of the various residues of treatment processes. Leachate data provide a means of integrating over a large mass of waste, but provide little insight into the total contaminant mass present where solubility limitations restrict the concentrations of some metals.

5.2 BENCHMARK STUDY: LEACHATE MODELLING

To provide a benchmark for the comparison of results from this study, leachate modelling was undertaken to determine the equilibrium status of a typical UK landfill designed and operated on a pre-Landfill Directive basis. The basic scenario was similar to that used for

the modelling undertaken to support the definition of European Waste Acceptance Criteria [3] and for this we utilised a published source term model [3, 4]. Figure 5.1 shows the basic hydrogeological scenario that forms the basis of the calculations.

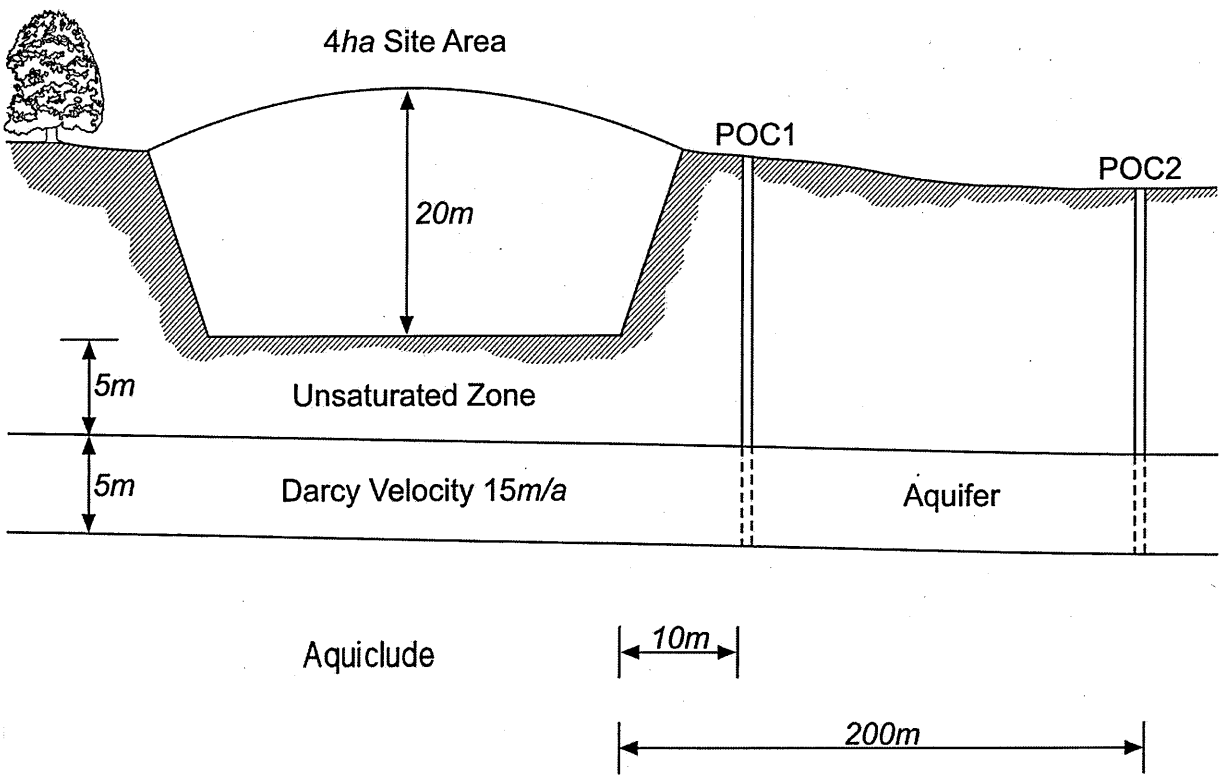


Figure 5.1 Hydrogeological scenario forming the basis of the calculations

The software used for this work was GoldSim [5], which has a probabilistic visual spreadsheet platform capable of transient modelling. Modifications were made to the basic scenarios in the GoldSim model used for the UK contributions to the setting of the leaching limit values given in the EU Decision [6] establishing criteria and procedures for the acceptance of waste at landfills. These modifications were carried out to reflect the work contained in the latest release of LandSim 2.5 [7]. They included the gradual unavoidable degradation of the liner and

cap (assuming HDPE construction) and the option to simulate the end of managed control (*e.g.* removing all management controls of leachate levels, recirculation and removal).

In order to provide a framework for the assessment, values were calculated for each species for which a leaching limit value was given in the EU Decision [8] and additionally for ammonium. The point of compliance for the species was the base of the unsaturated zone for List I substances [9], the edge of the landfill for List II substances and a point 200 m from the site boundary for highly mobile List II and non-listed substances. For each, a water quality standard or guideline was applied. These were either the relevant EU drinking water standard [10] or, where not available, the World Health Organisation drinking water guideline [11]. Also used was a kappa value for each species that served to describe the rate of concentration decline, and an appropriate individual contaminant/subsoil interaction (K_d) value for the liner and geosphere. Information about kappa and K_d values was taken from Hjelmar *et al.* (2001).

The model was run to determine the end of the aftercare period, *i.e.* when groundwater quality at the relevant point of compliance remained below the relevant water quality standard or guideline. This is a challenging modelling exercise, as emissions from the landfill are dynamic and need to be below certain emission criteria prior to the management system being switched off. The result of removal of management control will be an increase in leachate levels and a comparable increase in leakage rate. Hence, shortly after the removal of management control, there will be a period when the flux of contaminants from the site increases as a result of increased leakage. By treating the period of management control as a variable and running the model with this input represented as a logarithmic uniform probability distribution function varying between 3 and 2000 years, the appropriate time scale was determined for each contaminant.

5.3 RESULTS - BENCHMARK STUDY

The results are shown in Tables 5.2 and 5.3, which, with Figures 5.2 and 5.3, give an insight into the methodology. Achievement of equilibrium status for conventional landfills is likely to be controlled by a number of key species; chloride, lead, zinc and other metals. That is not to say that these controlling species will remain the same for other landfill types containing residues different from those in a typical current landfill used as the benchmark.

Table 5.2. Results of a preliminary benchmarking exercise for a current modern landfill

Contaminant	Point of compliance See note 2	Initial concentration (see note 3)	WQS	Years to achieve equilibrium	Comments
		$mg\ l^{-1}$	$mg\ l^{-1}$		
Antimony (Sb)	1				Not routinely measured
Arsenic (As)	1	0.013	0.01	<3	
Barium (Ba)	1				Not routinely measured
Cadmium (Cd)	US	0.01	0.005	<3	See note 1
Chromium (Cr)	1	0.18	0.050	<3	
Copper (Cu)	1	0.1	0.05	<3	
Mercury (Hg)	US	0.00009	0.001	<3	See note 1
Lead (Pb)	1	0.17	0.01	400	
Molybdenum (Mo)	1				Not routinely measured
Nickel (Ni)	1	0.24	0.02	<3	
Zinc (Zn)	1	5.09	0.1	1100-1300	Only applicable to the acetogenic phase of the landfill, therefore this is not a real issue.
Selenium (Se)	1				Not routinely measured
Fluoride (F)	2				Not routinely measured
Sulphate (SO_4)	2	263	250	<3	
Chloride (Cl)	2	1466	250	40-60	
Ammoniacal Nitrogen (NH_4)	1	495	0.5	<3	Assumed to biodegrade with a half life of 6 yrs

Note 1 – The water quality standard for List I substances has been used in the modelling. If the minimum reporting values [38], are used for these two species then the time period runs to in excess of 2000 years).
Note 2 – The point of compliance (POC) – US is the base of the unsaturated zone, point 1 is groundwater at the boundary of the site, and point 2 is groundwater at a distance of 200 m downstream.
Note 3 – Based on the mean values from the LandSim 2 defaults for non-List 1 substances.

Table 5.3. Results of benchmarking exercise for a current modern landfill using WAC leachate concentrations for hazardous waste going to a non-hazardous landfill.

Contaminant	Point of compliance See note 1	Initial concentration	WQS	Years to achieve equilibrium	Comments
		mg l^{-1}	mg l^{-1}		
Antimony (Sb)	1	0.15	0.005	>2000	Not routinely measured
Arsenic (As)	1	0.3	0.01	>2000	
Barium (Ba)	1	20	0.7	>2000	Not routinely measured
Cadmium (Cd)	US	0.3	0.005	>2000	
Chromium (Cr)	1	2.5	0.05	1100-1300	
Copper (Cu)	1	30	0.05	>2000	
Mercury (Hg)	US	0.03	0.001	>2000	
Lead (Pb)	1	3	0.01	>2000	
Molybdenum (Mo)	1	3.5	0.07	1300-1450	Not routinely measured
Nickel (Ni)	1	3	0.02	1500-2000	
Zinc (Zn)	1	15	0.1	>2000	
Selenium (Se)	1	0.2	0.01	930-1000	Not routinely measured
Fluoride (F)	2	40	1.5	1450-2000	Not routinely measured
Sulphate (SO_4)	2	7000	250	1300-1450	
Chloride (Cl)	2	8500	250	930-1000	
Ammoniacal Nitrogen (NH_4)	1	2000	0.39	1100	Assumed to biodegrade with a half life of 6 yrs

Note 1 - US is the base of the unsaturated zone, point 1 is groundwater at the boundary of the site, and point 2 is groundwater at a distance of 200 m downstream.

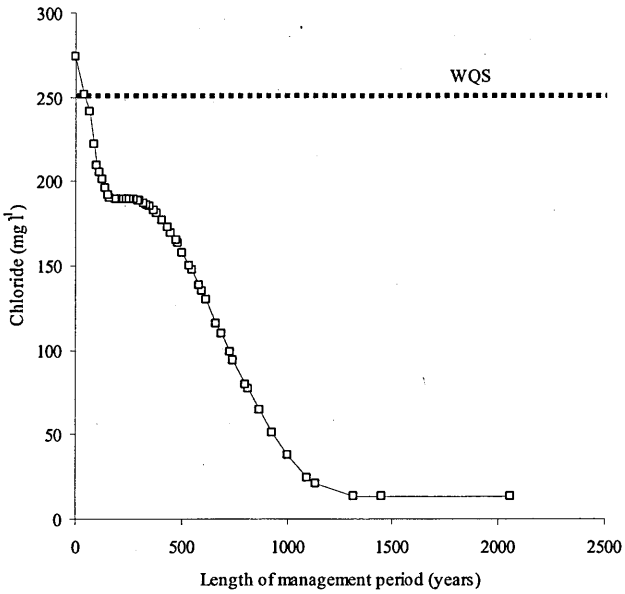


Figure 5.2. Maximum receptor concentration versus length of management time for chloride.

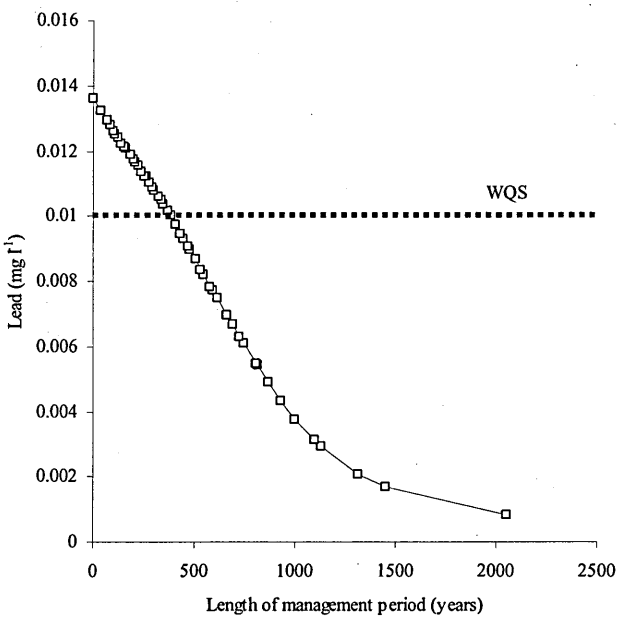


Figure 5.3. Maximum receptor concentration versus length of management time for lead

The benchmarking of a standard landfill showed that many of the contaminants achieve equilibrium status, with respect to leakage to groundwater, in a surprisingly short period. In this example, even ammoniacal nitrogen met the criterion within a relatively short period. However, other contaminants, notably lead, took a considerable amount of time to reach stabilisation. The leachate concentrations used as a starting point in this exercise are taken from the geometric mean values within LandSim, which in turn are based on published research [12].

A second scenario was run with the leachate concentrations set to the C_0 values prescribed for stable, non reactive hazardous waste going to a non-hazardous landfill. C_0 is the initial peak concentration of the contaminant in the leachate (mg l^{-1}) when subject to the standard upflow percolation test [13]. Results were significantly different with most species requiring an aftercare period in excess of 1000 years and 50% requiring a management period of greater than 2000 years (Table 5.3).

Figure 5.2 shows the relationship between receptor concentration and management time for chloride for a non-flushing landfill that has accepted predominantly raw MSW. It is clear that the relationship between the length of management time and the reduction of receptor concentrations is not linear.

Each point on the graph is the result of modelling a different management period using a logarithmic sampling scale. The relevant water quality standard (WQS) for chloride is 250 mg l^{-1} and this was achieved with a management period of 40 years (Table 5.2). It must be stressed that the leachate chloride concentration at this time (*i.e.* 1275 mg l^{-1} at 40 years) would not meet the WQS. However, the processes of natural attenuation and dilution result in compliance if the management of leachate ceases at this time. It must also be stressed that on the cessation of

leachate management there is an expectation that leachate treatment (or removal) ceases, leachate levels will rise, and leakage will increase in line with the increased leachate head. Furthermore, it should be noted that the concentration at the receptor did not reach 250 mg l^{-1} at 40 years. This maximum concentration occurred at 156 years, some 116 years after the management of the site ceased.

Figure 5.3 shows a similar relationship for lead. In this case, the aftercare period required to reach equilibrium status was approximately 400 years (Table 5.2). The leachate concentration at this time was 0.12 mg l^{-1} (twelve times the WQS). The actual time taken for the maximum groundwater concentration to be realised was 4000 years. There is therefore a large disjoint between the time when management of leachate could cease and the time when the maximum concentrations in groundwater will be realised.

5.4 NATURE AND CHARACTERISTICS OF TREATMENT TECHNOLOGY RESIDUES

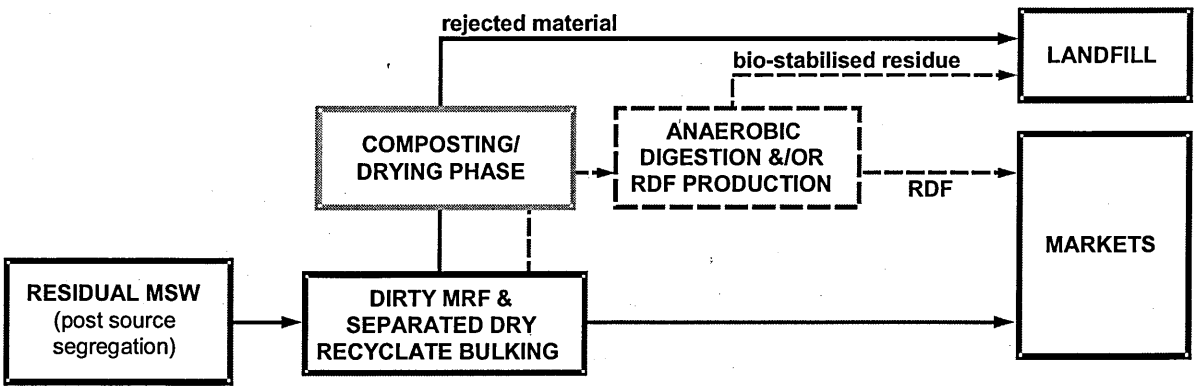
Almost all municipal waste authorities in the UK are involved with a certain amount of pre-treatment of MSW in the form of separate collection of some recyclables direct from households or from collection points. Some have increased this basic minimum diversion from landfill with the addition of materials recovery facilities (MRF) for more efficient removal of recyclables, or the provision of thermal and biological treatment facilities intended to be significant alternatives to landfill such as incineration or composting. For this study a number of representative treatment options were selected and the effect of these on the timescale for management of the landfilled residues was assessed.

Experience has shown that to achieve BMW reduction rates as high as those set by the Landfill Directive requires more than simply providing separate collection of recyclables [14]. Inevitably, therefore, the residues that will be landfilled in the future will be from a series of technologies and processes. In terms of process flows or combinations of the different technologies, a number of recent developments in the UK were reviewed and some UK disposal authorities' early waste strategies were studied to determine the most likely scenarios.

Waste sorting splits raw MSW into several waste streams; some of which are typically suitable for recycling. For the purposes of this study, waste sorting was reviewed as an integral part of the wider treatment process, rather than as a stand-alone technology. Consequently, little attempt was made to analyse the potential affects of residues obtained solely from waste sorting.

5.4.1 Materials recovery facilities

Materials Recovery Facilities (MRFs) are becoming more common throughout the UK and it is envisaged that they will remain an integral part of municipal waste recovery processes in the UK. Commonly paper, cardboard, plastics and metals suitable for recycling are derived as marketable baled materials following sorting and separation during MRF operations. Unsorted residues may be disposed of to landfill. MRFs can be categorised into clean and dirty variants. Clean MRFs process source-segregated material for recovery. Dirty MRFs (Figure 5.4) are simpler forms of MRF that process the entire collected (unsorted) waste stream. Whilst many dirty MRFs have been phased out, they still have a role to play.



DIRTY MRFs

All dirty MRFs operate on the residual waste streams tonnage levels for a given population will - at present - be higher than for dry co-mingled recyclates.
This will probably result in dirty MRFs scaled for rural communities.

Key

——— denotes existing practice
– – – denotes Landfill Directive compliant practice (which is illustrative of a form of Bio-Mechanical Treatment approach)

Figure 5.4. MBT system utilising a dirty MRF. [39]

5.4.2 Mechanical biological treatment

MBT is a generic term for a range of processes used to treat MSW (normally post source segregation) by means of a combination of mechanical separation and biological treatment. Although different technologies may be used, they have similar characteristics. These commonly comprise three stages: mechanical size reduction, the driving off of moisture and, finally, material separation to segregate output streams for different purposes.

Figure 5.4 illustrates the fate of each material extracted from the process. In this study, all the systems involving the elementary steps are referred to as MBT. MBT outputs incorporate

selected recyclable materials and a stabilised waste. Extraction of recyclables occurs both before and after biological processing of the residual waste.

It is widely believed that MBT pre-treatment of waste prior to landfilling reduces landfill emissions compared to untreated MSW. It is, however, difficult to track an accurate mass balance of trace contaminants through the system.

During the process there is a reduction in mass which has been mainly attributed to the decrease in water content and the degradation of organic material. Weight reductions typically range between 20% and 40% [15-18]. Volume reductions are thought to be a result of the mechanical stage, *e.g.* shredding. They are reported to be between 35% and 79%, largely depending on the degree of landfill diversion during the MBT process. Virtually all plants have means of removing ferrous and non-ferrous metals, and many also attempt to remove a reasonable proportion of the plastics that enter the process.

MBT residues typically achieve a higher emplaced density within landfills than standard MSW. A value of 1.3 t m^3 was reported by Binner [19]. Scheelhaase and Bidlingmaier [20] found during lysimeter tests that storage densities of 1.6 t m^3 were achievable. They also reported a notable decrease in permeability between feedstocks and MBT residues as a result of the increased density, high homogeneity, smaller grain sizes and the high proportions of 'earthy' components of the material following MBT procedures. Some authors report hydraulic conductivity ranges from $<1 \times 10^{-10}$ m s^{-1} (<25 mm, 50 weeks treatment) to $<1 \times 10^{-11}$ m s^{-1} (<12 mm, 5 weeks treatment). However, a more recent report [21] shows the relationship between MBT residue permeability and applied load. Results range from 3×10^{-5} m s^{-1} for waste under a load of 50 kN m^2 to 6×10^{-9} m s^{-1} with an applied load of 550 kN m^2. These values are almost

identical to those derived for raw MSW and as such indicate a comparable permeability. Care needs to be taken when comparing the results with raw MSW as the density of the two waste streams will be different.

Comparison of the organic matter content before and after the MBT process (using loss on ignition data) indicates degradation of between 50 and 70% by weight of organic dried solid matter. It should be noted that the composition of input material is a key-determining factor. Further intensive composting over a protracted period could reduce the weight of organic material (dried solid) to 12% of the original feedstock but this is unusual and would only result from highly selective waste streams. In most cases the loss on ignition value drops to 25 - 35 wt. % of organic dried solid matter after pre-treatment [20].

Through landfill simulation experiments, the influence of MBT pre-treatment has been compared to MSW in terms of leachate quality [15]. Available data demonstrate that for pre-treated waste the acidic phase (during which high strength leachate is produced from MSW) does not occur. Substantial reductions in long-term concentrations of Chemical Oxygen Demand (COD) and Total Nitrogen (as N) have been reported [22] compared to MSW, although the precise basis of these claims is uncertain. It is clear that leachate strengths are lower, but the composition of the non-degradable (hard) COD is unknown. Data are also available from several small-scale outdoor lysimeters which contained MSW and MBT residual wastes, *e.g.* Kabbe, 2000, reported in Robinson *et al.* [18]. Results for heavy metals content from both of these example lysimeters were similar.

Leaching tests undertaken on samples of MSW and MBT residues, solely for ammonia and TOC have been reported [17]. Concentrations of ammoniacal-N were found to be similar for

MSW and Mechanically Sorted Organic Residues (MSOR) of around 500 mg l^{-1}, but were notably lower for MBT waste, at about 150 mg l^{-1} [18].

The quality of MBT waste materials (and hence landfill leachate quality) will vary as a consequence of the extent of source-separation of the waste inputs (urban or rural source, seasonal collections), type of mechanical pre-treatment and type and duration of biological treatment [23].

Leachate quality data from full-scale landfill sites are also consistent with the loss of the acetogenic state in MBT waste landfills [19]. This is highlighted by comparatively high pH values during the first few years. Biochemical Oxygen Demand (BOD), Chemical Oxygen Demand (COD) and ammonium are all reported to be considerably lower in MBT pre-treated waste than in MSW landfills. The concentrations of some heavy metals are also reduced especially zinc, but it is recognised that the solubility of some metals (especially zinc) tend to reduce markedly at the onset of methanogenesis.

For organic compounds (*e.g.* mecoprop[3]), evidence suggests that effective composting processes are able to reduce the subsequent concentration in the leachate to below those in sites accepting MSW and MSOR. In practice, the extent of removal of mecoprop from leachates may be a good surrogate measure of the efficiency of the composting process itself, to which waste fractions have been subjected. It is noted however that the degree of composting achieved and the efficiency of individual composting processes cannot be determined by the duration of composting. A further leachate quality summary for leachates from landfills/test cells containing untreated MSOR and MSOR subjected to various composting regimes was obtained from Bone *et al.* [23].

[3] An acid herbicide that is widely used for agricultural, horticultural and domestic purposes.

Robinson *et al.* report a suggested leachate source term for landfills accepting predominantly MBT residues [18]. This represented their best estimate based on an extensive desk study and independent sampling from a number of European landfill sites.

5.4.3 MBT/composting

Many MBT systems were originally developed as compost plants. The intention was to convert raw MSW into quality compost, but there are few examples where the product has achieved the necessary quality, and large quantities are landfilled. The quality of the compost derived from MBT systems varies considerably and is largely dependent upon the quality of input material and process retention period. While the residue has been "composted" in the process, it still contains some of the contaminants present in the feedstock. The suitability of the resultant material for beneficial application is generally limited to low grade uses such as landfill daily cover, 'brownfield' restoration or forestry.

In the context of MSW in the UK, the term composting has usually been taken to mean the green waste shredding and outdoor windrowing that is undertaken by many municipal authorities. This study assumed that no significant residue from this process would be sent to landfill.

Changes in the composition of the organic fraction of municipal solid waste during the biodegradation process and the final waste products are strictly dependent on the process conditions. The reduction of carbon content due to biodegradation increases with process temperature from 20% at 20° C to about 40% at 37° - 42° C [24]. The fate of heavy metals and non-degradable species is less certain and they are likely to be unaffected by the process.

However, with the volume and mass reduction of the waste the concentration (but not the mass) of these contaminants is likely to increase.

Organic trace contaminants in conventional MSW include a number of species that are resistant to biodegradation in the anaerobic conditions that quickly become established in a modern MSW landfill. Species such as mecoprop, toluene and benzene do not undergo significant anaerobic degradation. While volatile species such as the BTEX (benzene, toluene, ethylbenzene and xylene) compounds leave the landfill via landfill gas and may be destroyed within combustion plant, non-volatile species such as mecoprop will only be removed by leaching. The removal of these species via aerobic degradation within the composting process is beneficial, should the material be landfilled.

5.4.4 MBT/Anaerobic digestion

The digestion of MSW is not common in the UK. The success of anaerobic digestion of MSW depends upon a high degree of pre-sorting and mechanical separation. Anaerobic digestion reduces organic waste to a relatively stable solid residue similar to compost. The process can be completed in 2-3 weeks. There is the risk that the digestate will contain high proportions of heavy metals, particularly when treating unsorted waste as demonstrated in Table 4.4. Anticipating this, it is necessary to carefully oversee the feedstock going to the reactor. The main products from AD are biogas, a liquid fraction and a solid residue. No data could be obtained on the mass balance of contaminants.

Table 5.4. Typical anaerobic digestate heavy metal concentrations from MSW. [40]

Parameter	Mixed collection Municipal solid waste mg kg^{-1} (dry basis)	Separated vegetable, fruit and garden waste mg kg^{-1} (dry basis)	Separated vegetable, fruit and garden waste and paper mg kg^{-1} (dry basis)
Cadmium	2	2	1
Zinc	1020	138	85
Copper	101	20	14
Lead	522	67	61
Nickel	42	25	7

Leachate quality data for landfills accepting AD residues are poorly documented. However, it is anticipated that leachates from those residues that have not been subjected to a post composting stage will be of a similar quality to a methanogenic leachate from a typical MSW site containing moderate levels of ammonia, residual hard COD, chloride concentrations comparable with MSW and the range of heavy metals typically found in MSW leachate.

For those sites that accept AD residues that have been composted following the AD stage, leachates are likely to be similar to leachates derived from MBT composted waste that is landfilled. Leachate concentrations from one site that had accepted AD composted material are taken from Robinson *et al.* [18]. The leachate was very similar to a methanogenic leachate albeit that the ammoniacal N concentration was less than 200 mg l^{-1}. There was a low, but detectable, concentration (0.47 μg l^{-1}) of mecoprop although there was a near absence of other identifiable trace organic contaminants.

5.4.5 Refuse derived fuel (RDF)

RDF (alternatively known as solid recovered fuel or SRF) has different meanings in different member states of the EU. It is well-established in Austria, Finland, Germany, Italy, the Netherlands and Sweden whilst in Belgium and the UK RDF production is still developing [25]. RDF generally encompasses a residue that is produced from waste with the intent of being traded and co-burnt in installations for power generation or in a manufacturing process where heat is required (*e.g.* cement production). The principal purpose for developing RDF from MSW is to arrive at a dry, high-calorific value (15->18 MJ/ kg [26]) product of improved homogeneity compared with raw MSW. In some cases, purpose-built incinerators have been developed to receive only RDF from MSW. The European standards organization (CEN) is classifying fuel according to net calorific value, chlorine and mercury content. However, the particle size, moisture content and fuel composition are also important.

There is a high likelihood that not all RDF will be utilised continuously in co-incineration plants. Historically, considerable amounts of RDF have been stored or landfilled for a variety of reasons such as lack of demand or poor quality. Hence, this study considered that one potential process flow will result in RDF being landfilled directly.

The composition of RDF from MSW will vary according to the origin of waste material and the sorting/separation process. This will in turn greatly influence the properties of RDF such as the calorific value. A typical composition for RDF from MSW originating from the UK is plastic (20%), paper/cardboard (58%), wood (5%), textile (15%), non-combustibles *e.g.* glass and metals (2%) [26].

The important characteristics for RDF as a fuel are the calorific value (20-23 MJ/kg for source-separated MSW; 13 MJ/kg for mixed MSW), water (10-35%), ash (10-16%), sulphur (0.2%) and chlorine (0.3-0.7%) contents [25]. These values are indicative and also vary according to the sources, the collection system (mixed or source separated) and the treatment applied (screening, sorting, grinding, drying etc.) [25]. RDF tends to affect the concentrations of cadmium, lead, copper and zinc when co-incinerated with other material. No specific information could be obtained on leachate quality from landfilling 100% RDF incineration residues.

Wet flock-type RDF is prepared by shredding, screening, magnetic separation, eddy current separation and possibly air classification to remove the non-combustible fraction (*e.g.* ferrous materials, glass and grit). The output tends to be dry and odour-free, with an enhanced calorific value, as both the moisture content and non-combustible fraction have been substantially reduced.

No specific information could be obtained from RDF/MBT where RDF results from a more sophisticated MBT process. However, if the material is incinerated, it is unlikely to differ significantly from RDF ash. If, because of lack of markets, it is not burnt, then it is likely to be similar to MBT residues that have been sent to landfill.

5.4.6 Mass burn incineration

In mass burn incineration the volume of waste is reduced by 90% and its weight by 75% [27]. Generally up to three different types of waste are produced: bottom ash, fly ash and air pollution control (APC) residues. Bottom ash arises from high temperature oxidative processes acting on the waste. Fly ash is the fine particulate matter (typically 1 to 500 μm) which is carried over from the combustion chamber and can be collected separately from other air pollution

control residues. APC residues result from the collection of other air pollutants from incinerator flue gases, especially acid gases, volatilised heavy metals and other micro pollutants. The three wastes have different compositions and will produce different emissions when landfilled. Fly ash and APC residues are unlikely to meet the hazardous waste acceptance criteria for acceptance at a landfill site, without treatment.

The Environment Agency reports that incinerator bottom ash (IBA) is approximately 25% to 30% by weight and 10% by volume of input, and APC residues are approximately 3% by weight of waste input [28]. These data depend on factors which include quantity, composition of waste burnt, any recycling schemes and the design and operation of the plant. MSW IBA will have a density of about 1.5 tonnes m^3.

Chemical analysis and characterisation of bottom ash and APC residues from three UK incinerators has been undertaken [18]. An example of the chemical composition of fresh bottom ash is presented in Polettini *et al.* [29].

Pre-treatment processes such as carbonation and acid treatment have been demonstrated to influence the pH of the waste and hence affect the waste emissions (via leachate). Williams [30] presented typical composition of bottom ash, fly ash, and APC residues from a dry/semi-dry system and a wet control system.

For leachate produced from incinerator bottom ash and APC residues, Robinson et al. [18] reported that 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. They reported that variations in ash leachates are more likely to occur as a result of site topography

and water regime than any variation in reactions occurring after landfilling. Leaching test data are therefore valued as a good guide to actual leaching quality. Concentration ranges for maximum levels observed in leaching tests on bottom ashes at liquid/solid ratios below 0.5 are provided by Hjelmar [31].

Inorganic components from a lysimeter study of bottom ash leached to a liquid/solid (L/S) ratio of 1.4 have been reported by Stegmann *et al.* [32]. Throughout this lysimeter study, chloride exhibited a washout pattern, falling to a L/S ratio of ~0.7 and then continued at a lower concentration for the remainder of the experiment. The pH remained high throughout due to the lime content, although sub-sampling at the end showed that lower pH values had developed in the upper (exposed) surface, indicating partial carbonation.

Sulphate concentrations were initially low, presumably because of very high calcium concentrations. Sulphate then fell further, before rising during the second half of the study, as carbonation began to remove calcium from solution. This is consistent with a long-term decrease in alkalinity. At the end of the test only 2% of the sulphate content had been leached.

Leachate quality data are available from a bottom ash landfill in Switzerland [33, 34]. L/S ratios at the time of the study were reported to be 0.2 – 0.25. Dry weather concentrations of Cl, SO_4 and Na were consistent with published eluate values for low L/S ratios [31], while Total Organic Carbon (TOC), K, Ca and most of the heavy metals were generally at lower concentrations than indicated in eluate concentrations.

A long-term dataset of leachate quality (1973 to 1998) is available for a PVC lined landfill [3]. The site accepted ~85% bottom ash and ~15% fly ash. It was noted by Robinson *et*

al. [18] that copper concentrations correlated strongly with dissolved organic carbon (DOC) indicating the presence of strong organic complexes. This is an issue raised by a number of researchers working with IBA leaching studies. It is possible that a number of other metals also exhibit similar relations with DOC, but to a lower degree. The mobility of copper as a complex is likely to be far higher than its mobility as a metal ion and could have implications for groundwater contamination from sites accepting IBA.

Bottom ash, while having virtually no dioxins, contained large numbers of other trace organics including halogenated aliphatic and aromatic hydrocarbons, BTEX compounds, alkenes and PAHs. Concentrations of trace organics were found to vary by up to three orders of magnitude in bottom ash from three different sites [18].

Carbonation is an important process that affects the physical and chemical nature of bottom ashes. It is formed by the reaction of atmospheric carbon dioxide with lime in the wastes, forming calcium carbonate. Carbonation of the bottom ash increases leachate concentrations of calcium and sulphate ions, but has no significant impact on the leaching of most other ions, DOC, organic nitrogen, and trace organics. However, it lowers pH significantly and the concentrations of certain heavy metals in the leachate by orders of magnitude, while some trace metals increase [18]. Insolubility of lead (in the short-term) can be ensured through accelerated aging of the fly ash by effective contact between wet fly ash and exhaust gas of the incineration plant [35].

5.4.7 Fluidised bed incineration

This type of thermal treatment (a simple modification to mass burn) involves the waste being used as a fuel, suspended by an updraft supply of air, and kept 'fluidised' on a base of

small inert particles such as sand or dolomite. Pre-processing essentially involves the removal of large items such as white goods and ferrous and non-ferrous metals and may be extended to include the recovery of other materials such as recyclables. However, it is noted that savings are made when compared with basic incineration systems because of the possibility of including lime in the combustor material to inhibit the production of acid pollutants. Furthermore, there is no need for the expensive bolt-on air pollution control systems typical of mass burn incinerators to clean up the flue gases. Fluidised bed incineration ensures a high level of waste destruction due to both the preparation of the waste and the method of combustion.

The incineration of MSW using a fluidised bed system is best achieved with some form of pre-screening and shredding, or the production of RDF pellets. Fluidised bed incinerators also have the potential for burning organic liquids, acid tars and sludges.

5.4.8 Pyrolysis

Pyrolysis is the indirect heating of material in the absence of oxygen such that the organic material is gasified and the resultant solid residues are inert and mainly contain carbon. Pyrolysis can be used to treat organic wastes, rendering the residues biologically inert while extracting a fuel as an energy source for later use. Organic waste is transformed to a medium calorific gas, liquid and a char fraction. These contain hydrocarbons (gas and oils/tars) and solid residue (char/pyrolysis coke) containing carbon, ash, glass and non-oxidised metals. The pyrolysis process occurs without the release of polluting dust containing dioxins and/or nitrogen oxides (combustible gases are released). These combustible gases can be used in any industrial application requiring heat or energy.

Studies have shown that increasing the temperature of pyrolysis decreases char production and increases oil and water production. Gas production increases between 300 and

420°C, but then stays constant. For flash pyrolysis at temperatures less than 600°C, the production of oil is very high; moreover, at temperatures above 700°C gas production is very high.

The Plasma Pyrolysis/Vitrification (PP/V) system produces a solid vitrified residue which presents a low leachability of pollutants and low toxicity levels in leachates [36]. A number of waste streams have been processed at the pyrolysis plant in Bristol including a batch of RDF. Subsequent analysis of the ash (char) is shown in Table 5.5 [37].

Table 5.5. Results of ash analysis for RDF. (EUS Laboratories Ltd, 22nd Feb. 2002)

Element	Solid mg kg^{-1}	Leachate μg kg^{-1}
Total organic carbon	455	Na
Total hydrocarbon	76	na
Cadmium	36	0.8
Thallium	<0.5	1.3
Mercury	<0.1	<0.1
Lead	8	8
Chromium	650	9
Copper	720	143
Manganese	830	12
Nickel	120	4
Arsenic	23	12
Antimony	48	4
Cobalt	12	8
Vanadium	16	9
Tin	870	10
Dioxin / furan	22.5 ng kg^{-1}	0.0063 ng l^{-1}

5.4.9 Gasification

Gasification is a similar process to pyrolysis but takes place with the addition of some oxygen as air or steam. The process produces a mixture of combustible gases (primarily methane, complex hydrocarbons, hydrogen and carbon monoxide), ash and a tar. The major environmental benefit of this process is that it retains pollutants (the sulphur, heavy metals etc.)

in the ash instead of the gas phase and prevents subsequent discharge to the atmosphere. The emissions from this technology may be lower than produced by conventional incineration and will require less flue gas treatment. There is little data available in the public domain regarding the contaminant concentration of the char. Initial indications [37] are that the char may have similar leaching properties to incinerator bottom ash.

5.5 CONCLUSIONS

During the last two decades the estimated timescale for the potential polluting life of a large modern landfill has increased from early estimates of 20-40 years to greater than 500-1000 years. Currently, technical and financial provision for post-closure liabilities is usually made for a 30-60 year period. Modelling results suggest that this period may be inadequate if measures are not taken to increase the rate of waste stabilisation.

Initial benchmark study results for pre-Landfill Directive landfills suggest that two key contaminants (lead and chloride) are likely to control the achievement of equilibrium status. A second scenario run with leachate concentrations set for stable non reactive hazardous waste going to a non-hazardous waste landfill (Landfill Directive) suggests that the post-closure management period is likely to be >1000 years for most contaminants with 50% of contaminants requiring >2000 years to achieve equilibrium. The predicted timescales of centuries rather than decades suggest that a reappraisal of the role of accelerated landfill stabilisation techniques such as aerobic and bioreactor landfilling is required. The benchmarking results presented here are a starting point and the next stage was to model the most likely options that may contribute towards meeting the Landfill Directive diversion and pre-treatment targets. This forms a second manuscript that presents the full modelling methodology and results.

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6. ESTIMATING POLLUTANT REMOVAL REQUIREMENTS FOR LANDFILLS IN THE UK: II. MODEL DEVELOPMENT

D. H. Hall[1], D. Drury[1], J. R. Gronow[2], A. Rosevear[3], S. J. T. Pollard[2] and R. Smith[2]*

[1] Golder Associates (UK) Ltd., Attenborough House, Browns Lane Business Park, Stanton-on-the-Wolds, NG12 5BL, UK

[2]Integrated Waste Management Centre, Sustainable Systems Department, School of Applied Sciences, Cranfield University, MK43 0AL, UK

[3] Environment Agency, Kings Meadow House, Kings Meadow Road, Reading, RG1 8DQ, UK.

* Corresponding author Tel +44 (0)1234 754963; Fax +44 (0)1234 376171; e-mail r.smith1@cranfield.ac.uk

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ABSTRACT

A modelling methodology using a leachate source term has been produced for estimating the timescales for achieving environmental equilibrium status for landfilled waste. Results are reported as the period of active management required for modelled scenarios of non-flushed and flushed sites for a range of pre-filling treatments. The base scenario against which results were evaluated was raw municipal solid waste (MSW) for which only cadmium failed to reach equilibrium. Flushed raw MSW met our criteria for stabilisation with active leachate management for 40 years, subject to each of the leachate species being present at or below their average UK concentrations. Stable non-reactive wastes, meeting EU waste acceptance criteria,

fared badly in the non-flushed scenario, with only two species stabilising after a management period within 1000 years and the majority requiring >2000 years of active leachate management. The flushing scenarios showed only a marginal improvement, with arsenic still persisting beyond 2000 years management even with an additional 500 mm y^{-1} of infiltration. The stabilisation time for mechanically sorted organic residues (without flushing) was high, and even with flushing, arsenic and chromium appeared to remain a problem. Two mechanical biological treatment (MBT) scenarios were examined, with medium and high intensity composting. Both were subjected to the non-flushing and flushing scenarios. The non-flushing case of both options fell short of the basic requirements of achieving equilibrium within decades. The intense composting option with minimal flushing appeared to create a scenario where equilibrium could be achieved. For incinerator bottom ash (raw and subjected to various treatments), antimony, copper, chloride and sulphate were the main controls on achieving equilibrium, irrespective of treatment type. Flushing at higher flushing rates (500 mm y^{-1}) failed to demonstrate a significant reduction in the management period required.

Keywords: Equilibrium; completion; emissions; MSW

6.1 INTRODUCTION

This work presents the second of a series of three companion papers. The first manuscript presented a leachate modelling benchmark study and review of treatment technologies [1] in relation to removing landfill pollutants and achieving environmental equilibrium status. Equilibrium is defined here as that state when emissions from a landfill site occur at a rate that allows sufficient natural attenuation in the surrounding environment to prevent environmental harm, so management is no longer required. To embody the principles of sustainability, equilibrium can only be achieved when the management period (post-closure when the site has

ceased accepting waste for disposal) is measured in decades rather than centuries. The hydrogeological scenario used as a basis for calculations is depicted in Figure 6.1.

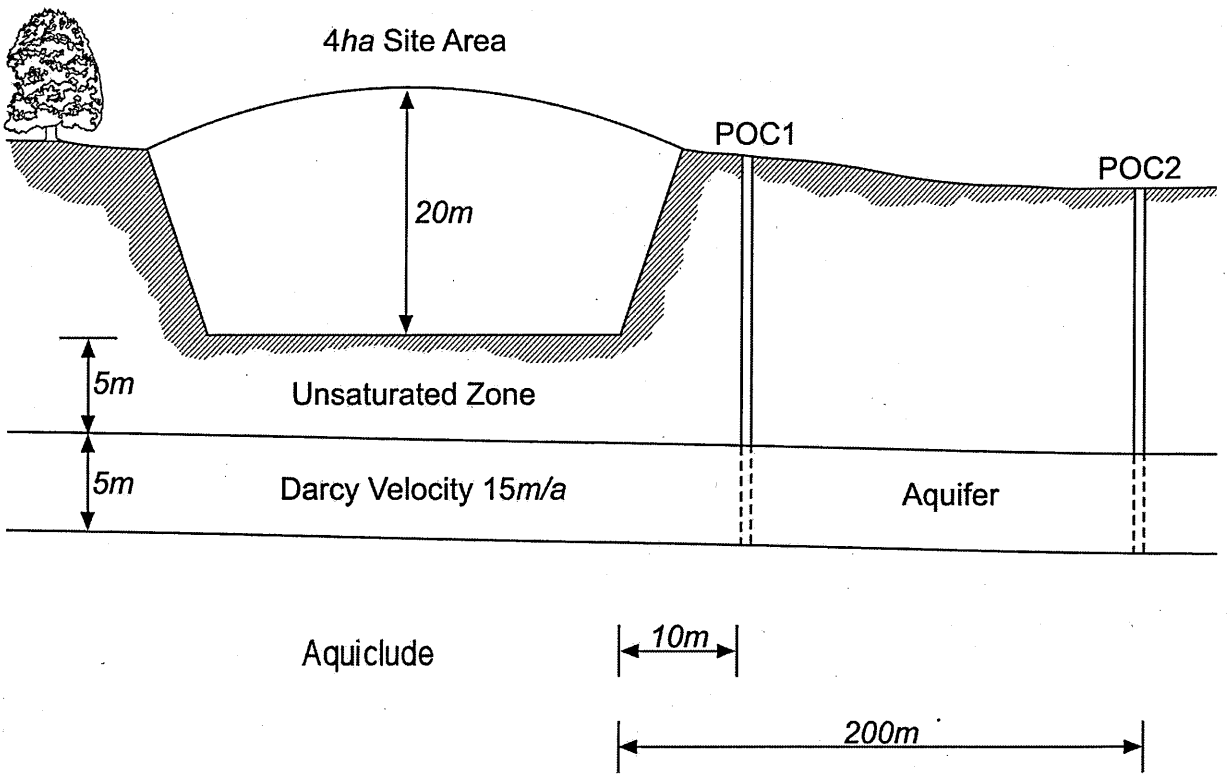

Figure 6.1 Hydrogeological scenario forming the basis of the calculations

6.2 MODELLING METHODOLOGY

6.2.1 Modelling approach and objectives

This section provides further information on the methodology; the details have been provided in the review of treatment technologies provided in the first manuscript. The post processing capabilities of GoldSim [2] allowed multivariate analysis of the results, which is essential in order to correlate the length of aftercare period with long-term groundwater impacts. Apart from the aftercare period, all other inputs to the model were set as single values only.

Essentially the aftercare period was defined as a variable and the model was run allowing this period to vary between 3 years and 2000 years. During each iteration (each using a different aftercare period) the maximum groundwater concentration for each of the contaminants modelled was recorded and then plotted against aftercare time. In this way, the management time period needed to achieve the water quality standard could be estimated. Any model run that resulted in a contaminant requiring a management period greater than 2000 years was simply recorded as >2000.

In the model it was assumed that leachate pumping at the site continued throughout the aftercare period maintaining leachate heads at 1 m. Once management ceases, leachate heads were allowed to vary. The methodology for this was based on the water balance model incorporated in LandSim 2.5 [3]. Leachate levels were expected to increase as a result of cap infiltration and cap deterioration despite the fact that the leakage rate was likely to increase markedly as a result of degradation of the liner system. No account was taken of the prospect of cap breakout of leachate as a result of a build-up of leachate head after the end of management control. It was assumed that this possibility was likely to be a further and semi-independent site-specific control on the duration of site management.

The European Landfill Directive [4] seeks to minimise leachate production by limiting rainfall infiltration and groundwater inflow. It has been shown that appreciable flushing rates are important for achieving landfill stabilisation [5]. Therefore, simulations were also run as flushing systems with 200 or 500 mm of additional infiltration, to determine the implications for aftercare periods. This recirculation ceases at the time management control ends. The receptors

for the various contaminants were selected as the downgradient boundary of the site for List II or non-listed substances [6] and the base of the unsaturated zone for List I substances.

Note that, on a site-specific basis, the water quality standards used may not be appropriate, as they may be too high (*e.g.* if the site is in a sensitive location) or too low (*e.g.* where background levels already exceed these values and the location is not sensitive). Also, certain species, such as zinc, are only found in high concentrations in leachates that are acetogenic; concentrations often fall below the relevant water quality standards once the waste becomes fully anaerobic. In this instance, the concentration was modelled at its higher aerobic concentration. It must be stressed that there will be situations where the leachate strengths will be higher than those used in the model, and where less attenuating capacity is available. However, the modelling was undertaken on the basis of mean leachate concentrations (not ranges) and is based on a 50th percentile result and not a 95th percentile (which would be the typical percentile used for a risk assessment). For each scenario and each contaminant, the modelling approach allowed for an estimation of the leachate strength when management control can end and groundwater impacts are acceptable (*i.e.* less than the Water Quality Standard (WQS)).

6.2.2 Model inputs

In all cases the landfill was assumed to be composite lined with a HDPE capping system. Infiltration into the open waste mass prior to capping was assumed to be 250 mm y^{-1} (taken over a ten year period) reducing to 50 mm y^{-1} on capping. Infiltration was then allowed to increase from 50 mm y^{-1} to 140 mm y^{-1} between 250 and 1000 years to simulate degradation of the cap. Single value inputs for the generic site modelled are provided in Tables 6.1 to 6.6 below.

Table 6.1. Values describing basal lining system.

Parameter – Flexible membrane liner	Parameter value
Area of pinholes	2.55 mm^2
Area of holes	52.5 mm^2
Area of tears	5050 mm^2
Number of pinholes	start = 0, end = 25
Number of holes	start = 0, end = 5
Number of tears	start = 0, end = 0.1
Contact coefficient for leakage calculations	0.68
Contact coefficient for calculation of radius of wetted area	0.435
Mineral component	
Clay permeability	1E-9 m s^{-1}
Mineral liner porosity	0.3
Mineral liner density	1800 kg m^3
Liner thickness	1 m

Table 6.2. Physical waste properties used in modelling.

Parameter	Parameter value
Waste density	Dependent on waste type.
Waste porosity	0.3
Waste field capacity	0.3

Table 6.3. Kappa values defining the rate of source term concentration decline.

Contaminant species	Kappa value (kg l^{-1})
Antimony (Sb)	0.11
Arsenic (As)	0.03
Barium (Ba)	0.15
Cadmium (Cd)	0.35
Chromium (Cr)	0.18
Copper (Cu)	0.57
Mercury (Hg)	0.05
Lead (Pb)	0.27
Molybdenum (Mo)	0.35
Nickel (Ni)	0.29
Zinc (Zn)	0.28
Selenium (Se)	0.38
Fluoride (F)	0.22
Sulphate (SO_4)	0.33
Chloride (Cl)	0.57
NH_4	0.59

Source: Data used for EU WAC modelling [10]

Table 6.4. Values describing unsaturated and saturated zones.

Parameter	Parameter value
Thickness of the unsaturated zone	5 m
Distance from edge of landfill to first saturated zone point of compliance (POC1)	10 m
Distance from edge of landfill to second saturated zone point of compliance (POC2)	200 m
Darcy flux	15 m yr^{-1}
Aquifer porosity	0.3
Aquifer thickness	5 m
Aquifer density	2000 kg m^3

Table 6.5. Values of contaminant species specific parameters.

Contaminant species	Liner, unsaturated zone and saturated zone partition coefficient (l kg^{-1})	Water Quality Standard (mg l^{-1})	Location of receptor where water quality standard must be achieved
Antimony (Sb)	5	0.005	Receptor 1
Arsenic (As)	50	0.01	Receptor 1
Barium (Ba)	2	0.7	Receptor 1
Cadmium (Cd)	20	0.005	Base of unsaturated zone
Chromium (Cr)	100	0.05	Receptor 1
Copper (Cu)	14	0.05	Receptor 1
Mercury (Hg)	100	0.001	Base of unsaturated zone
Lead (Pb)	50	0.01	Receptor 1
Molybdenum (Mo)	10	0.07	Receptor 1
Nickel (Ni)	50	0.02	Receptor 1
Zinc (Zn)	30	0.1	Receptor 1
Selenium (Se)	5	0.01	Receptor 1
Fluoride (F)	0	1.5	Receptor 2
Sulphate (SO_4)	0	250	Receptor 2
Chloride (Cl)	0	250	Receptor 2
NH_4	0.5	0.5	Receptor 2

Table 6.6. Initial leachate concentrations for MSW and allied waste streams (mg l^{-1}).

Waste Stream	MSOR	MBT	MBT	MSW	Stable non-reactive
Treatment		Intensive	Medium	Raw	None
Reference	[8]	[8]	[8]	LandSim [10]	WAC [11]
Species					
Sb	X	X	X	X	0.15
As	0.1	0.006	0.05	0.013	0.3
Ba	X	X	X	X	20
Cd	0.0005	0.003	0.02	0.01	0.3
Cr	5	0.1	0.3	0.18	2.5
Cu	0.5	0.2	0.35	0.1	30
Hg	0.0001	0.0001	0.0001	0.00009	0.03
Pb	0.05	0.04	0.3	0.17	3
Mo	X	X	X	X	3.5
Ni	0.5	0.1	0.4	0.24	3
Zn	0.5	0.2	1.5	5.09	15
Se	X	X	X	X	0.2
F	X	X	X	X	40
SO_4	400	500	3000	263	7000
Cl	6000	2000	6000	1466	8500
NH_4	4000	200	550	495	2000

Notes to Table

X - No reliable data from UK MSW sites or literature.

Data derived from LandSim 2.5.

MSOR - Mechanically Sorted Organic Residues.

MBT - Mechanical Biological Treatment.

MSW - Raw Municipal Solid Waste.

WAC - Waste Acceptance Criteria C_0 values.

Conservative retardation factors (identical to those used for the UK contribution to the derivation of EU Waste Acceptance Criteria (WAC) were used [7]. Ammoniacal nitrogen was not included in the WAC, so a typical value of 0.5 l kg^{-1} was chosen. It was assumed that biodegradation of ammoniacal nitrogen did not occur.

There is some uncertainty about the long term viability of kappa values (first order decay constants) as an input variable to the modelling that has been undertaken. Work is currently underway in the UK to derive additional waste characterisation information that will help to understand the short-term variability of kappa for some of the contaminants of interest. Only by looking at long-term leachate quality data from landfills where some realistic estimate of the liquid/solid (L/S) ratio can be made, will any real advance in our understanding of this variable be achieved. We consider that it remains, at this time, the best method of estimating long-term leaching behaviour, but it is unlikely to represent the perfect solution and further advances in this area should be made over coming years as more date become available.

6.2.3 Leachate source term

The main contaminants modelled were those that are included in the EU WAC, although the inclusion of ammoniacal nitrogen was necessary, as for some waste streams it will represent one of the key contaminants in relation to its concentration in leachate and its various water quality standards. The inclusion of ammoniacal nitrogen within the list of contaminants modelled required the derivation of a nominal (and certainly non-statutory) WAC for ammonia.

Data relating to initial leachate concentrations came from a variety of sources. For the Mechanical Biological Treatment (MBT) residues and incinerator bottom ash, data were derived from published research [8]. For those model runs relating to WAC values, C_0 was taken to be the initial flush from a standard column test equating to a L/S ratio of approximately 0.05 l kg^{-1} back calculated from the published WAC. Additional data were drawn from corporate knowledge and judgement.

The initial leachate concentrations used for the modelling of municipal solid waste (MSW) and treated MSW (or closely allied wastes) are shown in Table 6.6, and those for incinerator bottom ash (both raw and treated) are shown in Table 6.7. It should be noted that suitable leachate source terms were not identified for all of the ten processes discussed.

The column in Table 6.6 entitled 'Stable Non-Reactive' relates solely to the C_0 values derived from the WAC for that waste that could be placed in a non-hazardous landfill in a separate cell. The implied assumption was that the entire waste was deposited at the maximum concentration of each species. The likelihood of this occurring is very low, but given that individual species were being examined it did provide an insight into which species are likely to result in the need to extend aftercare periods from the processes included.

Table 6.7. Initial leachate concentrations for MSW incinerator ash (mg l^{-1}).

Waste stream	Incinerator bottom ash		
Treatment	Raw	Carbonated	Acid treated
Reference	[8]	[8]	[8]
Species			
Sb	0.025	0.1	0.2
As	0.001	0.001	0.001
Ba	1	0.1	0.25
Cd	0.01	0.01	0.01
Cr	0.01	0.2	0.03
Cu	5	5	10
Hg	0	0	0
Pb	5	0.005	0.015
Mo	0.3	0.4	0.5
Ni	0.075	0.05	0.05
Zn	0.001	0.001	0.002
Se	1	0.05	0.02
F	0	0	0
SO_4	500	2000	2000
Cl	1700	1700	1700
NH_4	10	10	15

6.2.4 Model results

Tables 6.8, 6.9 and 6.10 show a summary of the results of the modelling exercise simply indicating, for each waste stream, each landfill management option and each species with the number of years required to achieve equilibrium status. For each scenario the model was run using what might be regarded as a standard management option (*i.e.* the waste remains uncapped during the filling sequence and is then capped). In addition, a flushing scenario where infiltration is increased during the management period has been modelled. Whether this is achieved by irrigation beneath the cap, by not having a cap, by removing the cap, or via treated leachate recirculation is, to an extent, incidental for modelling purposes.

True equilibrium status for a landfill is only achieved after every contaminant has reached equilibrium status. The final row in each table picks up the longest period defined by any species within the landfill and therefore highlights the one that equilibrium status is dependent upon.

Table 6.8 examines raw MSW and a synthetic leachate derived to represent a site filled with waste at its maximum WAC for stable non-reactive waste. This is a slightly fictitious scenario as it is highly unlikely that wastes infilling a site would all equal the relevant WAC. However, it is conceivable that a process waste might be consistently close to the limit for one of the WAC species.

Table 6.8. Results of the modelling management time for MSW and stable non-reactive wastes.

Waste Type	Raw MSW	Raw MSW	Stable non-reactive	Stable non-reactive	Stable non-reactive
Treatment	None	None	None	None	None
Scenario	Basic	Additional 200 mm y^{-1} infiltration	Basic	Additional 200 mm y^{-1} infiltration	Additional 500 mm y^{-1} infiltration
Contaminant			Years to achieve equilibrium status		
Antimony (Sb)	X	X	>2000	1350	700
Arsenic (As)	<3	<3	>2000	>2000	>2000
Barium (Ba)	X	X	>2000	1050	490
Cadmium (Cd)	<3	<3	>2000	533	240
Chromium (Cr)	<3	<3	1200	1200	185
Copper (Cu)	<3	<3	>2000	500	219
Mercury (Hg)	<3	<3	>2000	>2000	1300
Lead (Pb)	400	40	>2000	750	350
Molybdenum (Mo)	X	X	1375	440	200
Nickel (Ni)	<3	<3	1750	533	240
Zinc (Zn)	<3	<3	>2000	670	300
Selenium (Se)	X	X	965	275	115
Fluoride (F)	X	X	1700	665	250
Sulphate (SO_4)	<3	<3	1375	390	150
Chloride (Cl)	40	4	965	200	75
Ammoniacal Nitrogen (NH_4)	<3	<3	1100	130	50
Maximum management period required in scenario	400	40	>2000	>2000	>2000

X - No reliable data from UK MSW sites or literature.

Table 6.9. Results of the modelling management time for mechanically and biologically treated wastes.

Waste Type Treatment Scenario	MSOR None Basic	MSOR None Additional 500 mm y^{-1} infiltration	MBT Medium Basic	MBT Medium Additional 200 mm y^{-1} infiltration	MBT Intense Basic	MBT Intense Additional 200 mm y^{-1} infiltration
Contaminant	Years to achieve equilibrium status					
Antimony (Sb)	X	X	X	X	X	X
Arsenic (As)	>2000	1100	<3	<3	<3	<3
Barium (Ba)	X	X	X	X	X	X
Cadmium (Cd)	<3	<3	41	<3	<3	<3
Chromium (Cr)	1600	300	<3	<3	<3	<3
Copper (Cu)	50	<3	<3	<3	<3	<3
Mercury (Hg)	<3	<3	<3	<3	<3	<3
Lead (Pb)	<3	<3	780	206	<3	<3
Molybdenum (Mo)	X	X	X	X	X	X
Nickel (Ni)	580	50	410	76	<3	<3
Zinc (Zn)	<3	<3	550	125	<3	<3
Selenium (Se)	X	X	X	X	X	X
Fluoride (F)	X	X	X	X	X	X
Sulphate (SO_4)	<3	<3	1050	184	<3	<3
Chloride (Cl)	900	70	900	157	367	40
Ammoniacal Nitrogen (NH_4)	1275	85	<3	<3	<3	<3
Maximum management period required in scenario	>2000	1100	1050	206	367	40

Notes

MSOR – Mechanically sorted organic residues (generally the fines)

MBT - Mechanical and Biological Treatment (Separation and composting)

Treatment – in this table it relates to the amount or intensity of the composting process.

X - No reliable data from UK sites or literature.

Table 6.10. Results of the modelling management time for incinerator bottom ash.

Waste Type	Raw bottom ash	Raw bottom ash	Bottom ash	Bottom ash	Bottom ash	Bottom ash
Treatment	None	None	Carbonated	Carbonated	Acid treated	Acid treated
Scenario	Basic	Additional 500 mm y^{-1} infiltration	Basic	Additional 500 mm y^{-1} infiltration	Basic	Additional 500 mm y^{-1} infiltration
Contaminant	Years to achieve equilibrium status					
Antimony (Sb)	1950	310	>2000	900	>2000	1150
Arsenic (As)	<3	<3	<3	<3	<3	<3
Barium (Ba)	<3	<3	<3	<3	<3	<3
Cadmium (Cd)	140	<3	150	<3	150	<3
Chromium (Cr)	<3	<3	<3	<3	<3	<3
Copper (Cu)	600	240	1750	240	2050	870
Mercury (Hg)	N/a	N/a	N/a	N/a	N/a	N/a
Lead (Pb)	2000	750	<3	<3	<3	<3
Molybdenum (Mo)	410	20	710	55	860	85
Nickel (Ni)	<3	<3	<3	<3	<3	<3
Zinc (Zn)	950	130	<3	<3	<3	<3
Selenium (Se)	<3	<3	<3	<3	<3	<3
Fluoride (F)	<3	<3	N/a	N/a	N/a	N/a
Sulphate (SO_4)	<3	<3	1190	75	1180	75
Chloride (Cl)	580	90	570	75	570	20
Ammoniacal Nitrogen (NH_4)	<3	<3	<3	<3	<3	<3
Maximum management period required in scenario	2000	750	>2000	900	>2000	1150

The raw MSW waste in the basic scenario (*i.e.* one where the waste is placed, capped and leachate generation minimised) formed the base case. Somewhat surprisingly this scenario contained only one contaminant that failed the general criteria of equilibrium status. It must be noted however that the compliance concentration for cadmium was taken as the drinking water standard and not the Minimum Reporting Value (MRV), albeit that the compliance point was taken as the base of the unsaturated zone. The MRV is the value that is normally applied to List 1 Substances to determine whether their presence is discernible or not, and is enshrined in the UK groundwater risk assessment methodology [9]. If the compliance water quality standard is

taken as the MRV, the time for cadmium to reach equilibrium status increases to slightly over 2000 years. The option of disposing of raw MSW to landfill is unlikely to remain as the Landfill Directive seeks to reduce the volume of biodegradable MSW being disposed of to landfill.

The flushed raw MSW met the criteria of stabilisation at 40 years, subject to each of the leachate species being present at or below their average UK concentrations. After this time it is interesting to note that landfill gas generation would also have ceased (or at least be below the point at which meaningful management could be applied), so the requirements of equilibrium status would have been met.

Stable non-reactive wastes meeting the WAC performed badly in the non-flushing scenario, with only two species stabilising within 1000 years and the majority taking in excess of 2000 years. The flushing scenarios showed only a marginal improvement, with arsenic still persisting beyond 2000 years with an additional 500 mm y^{-1} of infiltration.

Table 5.9 shows the results for mechanically and biologically treated waste. Mechanically Sorted Organic Residues (MSOR) generates a waste that is high in contaminants and has high ammonia loading. As such, its stabilisation time (without flushing) was high, and even with flushing, arsenic and chromium remained a problem. In this case cadmium did not appear to be an issue at either of its WQSs.

Two MBT cases were examined, one with medium intensity composting and one with highly intensive composting. Both were subjected to the non-flushing and flushing scenarios. The non-flushing case of both fell short of the basic requirements of equilibrium status within decades. However, the intense composting option with some minimal flushing appeared to

create a scenario where equilibrium status could be achieved. Cadmium met the MRV at around 400 years in the flushed scenario.

The final set of results (Table 6.10) relate to incinerator bottom ash (raw and subjected to various treatments). Antimony, copper, chloride and sulphate appeared to be the main controls in achieving equilibrium status of this waste stream, irrespective of the treatment type. Flushing at higher rates (500 mm y^{-1}) failed to make a significant reduction in the management period required. It may be that the source term used was selected with conservatism and that a greater familiarity with the material would generate lower mean values of the key contaminants. What is clear is that bottom ash on its own will remain a challenge.

6.2.5 Leachate source concentrations at time of equilibrium

One of the objectives of this work was to determine a test to assist in defining when waste has reached equilibrium status. Leachate quality itself should give an indication of the status of the waste, and it was originally thought that a series of leachate quality criteria could be developed that could be used to define equilibrium status (with respect to leachate). Leachate source concentrations when equilibrium status was attained (based on the results presented in Tables 6.8, 6.9, and 6.10) were extracted from the data and are presented in Tables 6.11, 6.12 and 6.13. It is clear that there is no single leachate value that dictates equilibrium. Degradation of the liner and capping systems causes considerable complexity of the relationship between leachate quality with time, transient groundwater quality, and equilibrium.

Table 6.11. Leachate concentrations at equilibrium status for MSW and stable non-reactive wastes.

Waste type	Raw MSW	Raw MSW	Stable non-reactive	Stable non-reactive	Stable non-reactive
Treatment	None	None	None	None	None
Scenario	Basic	Additional 200 mm y^{-1} infiltration	Basic	Additional 200 mm y^{-1} infiltration	Additional 500 mm y^{-1} infiltration
Contaminant	Leachate concentration when equilibrium status is attained				
Antimony (Sb)	X	X	AA	0.0082	0.01
Arsenic (As)	AB	AB	AA	AA	AA
Barium (Ba)	X	X	AA	1.1	1.6
Cadmium (Cd)	AB	AB	0.002	0.02	0.02
Chromium (Cr)	AB	AB	0.61	0.8	0.824
Copper (Cu)	AB	AB	AA	0.3	0.3
Mercury (Hg)	AB	AB	AA	AA	0.0022
Lead (Pb)	0.121	0.125	AA	0.1	0.109
Molybdenum (Mo)	X	X	0.153	0.3	0.309
Nickel (Ni)	AB	AB	0.051	2.4	0.248
Zinc (Zn)	AB	AB	AA	0.8	0.73
Selenium (Se)	X	X	0.0281	0.04	0.04
Fluoride (F)	X	X	1.8	3.5	5.28
Sulphate (SO_4)	AB	AB	365.5	860	1231.5
Chloride (Cl)	1224.2	1401.5	442.4	1230	1182.5
NH_4	AB	AB	49.5	533.9	504.4

Notes

AA = Groundwater concentration always exceeded WQS hence leachate concentration always above equilibrium status.

AB = Groundwater always below WQS hence leachate always below equilibrium status.

X - No reliable data from UK MSW sites or literature.

Table 6.12. Leachate concentrations at equilibrium status for mechanically and biologically treated wastes.

Waste type Treatment Scenario	MSOR None Basic	MSOR None Additional 500 mm y^{-1} infiltration	MBT Medium Basic	MBT Medium Additional 200 mm y^{-1} infiltration	MBT Intense Basic	MBT Intense Additional 200 mm y^{-1} infiltration
Contaminant	Leachate concentration when equilibrium status is attained					
Antimony (Sb)	X	X	X	X	X	X
Arsenic (As)	AA	0.0299	AB	AB	AB	AB
Barium (Ba)	X	X	X	X	X	X
Cadmium (Cd)	AB	AB	0.0183	AB	AB	AB
Chromium (Cr)	0.4	0.802	AB	AB	AB	AB
Copper (Cu)	0.417	AB	AB	AB	AB	AB
Mercury (Hg)	AB	AB	AB	AB	AB	AB
Lead (Pb)	AB	AB	0.112	0.126	AB	AB
Molybdenum (Mo)	X	X	X	X	X	X
Nickel (Ni)	0.255	0.254	0.256	0.267	AB	AB
Zinc (Zn)	AB	AB	0.799	0.859	AB	AB
Selenium (Se)	X	X	X	X	X	X
Fluoride (F)	X	X	X	X	X	X
Sulphate (SO_4)	AB	AB	418.6	1098.8	AB	AB
Chloride (Cl)	437.1	1139.8	433	1478.1	970.3	1381
NH_4	39.85	529.3	AB	AB	AB	AB

Notes

MSOR – Mechanically sorted organic residues (generally the fines).

MBT - Mechanical and Biological Treatment (Separation and composting).

Treatment – in this table it relates to the amount or intensity of the composting process.

AA - Groundwater concentration always exceeded WQS hence leachate concentration always above equilibrium status.

AB - Groundwater always below WQS hence leachate always below equilibrium status.

X - No reliable data from UK sites or literature.

Table 6.13. Leachate concentrations at equilibrium status for incinerator bottom ash.

Waste type	Raw bottom ash	Raw bottom ash	Bottom ash	Bottom ash	Bottom ash	Bottom ash
Treatment	None	None	Carbonated	Carbonated	Acid treated	Acid treated
Scenario	Basic	Additional 500 mm y^{-1} infiltration	Basic	Additional 500 mm y^{-1} infiltration	Basic	Additional 500 mm y^{-1} infiltration
Contaminant	Leachate concentration when equilibrium status is attained					
Antimony (Sb)	0.009	0.0113	AA	0.0092	AA	0.006
Arsenic (As)	AB	AB	AB	AB	AB	AB
Barium (Ba)	AB	AB	AB	AB	AB	AB
Cadmium (Cd)	0.0087	AB	0.0087	AB	0.0087	AB
Chromium (Cr)	AB	AB	AB	AB	AB	AB
Copper (Cu)	0.024	0.239	0.024	0.238	AB	0.212
Mercury (Hg)	X	X	X	X	X	X
Lead (Pb)	AA	0.0342	AB	AB	AB	AB
Molybdenum (Mo)	0.209	0.213	0.193	0.232	0.197	0.251
Nickel (Ni)	AB	AB	AB	AB	AB	AB
Zinc (Zn)	AB	AB	AB	AB	AB	AB
Selenium (Se)	AA	AA	0.025	0.034	AB	AB
Fluoride (F)	X	X	X	X	X	X
Sulphate (SO_4)	AB	AB	357	1052	357	1052
Chloride (Cl)	722.5	985	723	973.1	722.5	973.1
NH_4	AB	AB	AB	AB	AB	AB

Notes

AA - Groundwater concentration always exceeded WQS hence leachate concentration always above equilibrium status.

AB - Groundwater always below WQS hence leachate always below equilibrium status.

X - No reliable data from UK sites or literature.

Overall it would appear from the work undertaken that intensively composted MBT residues that undergo flushing within a landfill will provide one means of achieving equilibrium status. MBT is gaining in popularity in the UK and Europe. Reports that MBT residues have a low permeability do raise some concern. It would not be feasible to irrigate a material with a

permeability value of 1 x 10^{-10} m s^{-1} at a rate of more that 3 mm y^{-1}. A permeability value of greater than 6 x 10^{-9} m s^{-1} would be required to allow intensively composted MBT residues to meet the requirements of our definition of sustainable landfill (in relation to leachate).

The result of this is that rather than generating a specific time to achieve equilibrium status, the best that can be achieved is the definition of a broad time scale over which equilibrium status may be achieved. It is further cautioned that there remains some considerable uncertainty with these results and they must be regarded as tentative.

6.3 CONCLUSIONS

This work has demonstrated that there is no simple relationship between landfill leachate quality and equilibrium status. Equilibrium status is achievable for MBT treated waste with intensive composting, the residues from which are then landfilled. There is a wide range of reported literature values for the permeability of these waste streams and further work is required to assess their actual permeability which is likely to be the limiting factor for achieving equilibrium status. Further work is also needed on the composition and leaching potential of treated waste streams. Data for pyrolysis and gasification processes in particular is lacking, though these are not likely to become dominant strategic options for waste management in the foreseeable future. Data is also lacking for the anaerobic digestion of specific waste streams and there are a growing number of these facilities in the UK in common with Europe.

The achievement of equilibrium status for an entire landfill is site-specific and ultimately factors such as size of site and depth of waste are likely to be important. The behaviour of wastes at deep (>50m) landfills is uncertain and appropriate L/S ratios may not be achieved even with leachate recirculation. Robust leachate collection systems in such sites will be essential.

This requires an evaluation of landfill management practice at current landfills so as to promote the achievability of equilibrium. For pre-Landfill Directive sites there is likely to be a legacy of long active management periods taking centuries rather than decades. Some of these sites may ultimately fall under the contaminated land legislative regime. A third manuscript [9] considers the policy and operational implications of this work.

6.4 ACKNOWLEDGEMENTS

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7. ESTIMATING POLLUTANT REMOVAL REQUIREMENTS FOR LANDFILLS IN THE UK: III. POLICY ANALYSIS AND OPERATIONAL IMPLICATIONS

D. H. Hall[1], D. Drury[1], J. R. Gronow[2], S. J. T. Pollard[2] and R. Smith[2]*

[1] Golder Associates (UK) Ltd., Attenborough House, Browns Lane Business Park, Stanton-on-the-Wolds, NG12 5BL, UK

[2] Integrated Waste Management Centre, Sustainable Systems Department, School of Applied Sciences, Cranfield University, MK43 0AL, UK

* Corresponding author Tel +44 (0)1234 754963; Fax +44 (0)1234 376171; e-mail r.smith1@cranfield.ac.uk

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ABSTRACT

The policy analysis and management implications for achieving landfill equilibrium status within a sustainable timescale (decades rather than centuries) are presented based on modelled results reported previously. Until relatively recently, timescale estimates suggested that equilibrium or landfill completion could be achieved within 40-60 years *i.e.* the same order of magnitude as financial provision for aftercare. However results of modelling in this study (reported in previous paper) suggest that timescales may be considerably longer (many centuries in some instances) suggesting that financial provision may be inadequate. The role of the most promising and available waste treatment technologies and strategic waste management options in contributing towards achieving equilibrium status are discussed. Results suggest that a re-

examination of techniques for accelerating landfill stabilisation, including aerobic and bioreactor landfill, is warranted.

Keywords: Equilibrium; completion; emissions; MSW

7.1 INTRODUCTION

This work presents the final part of a series of three companion papers. The first manuscript presented a benchmark study of leachate modelling and reviewed treatment technologies likely to form the basis of meeting the biodegradable municipal waste diversion targets of the Landfill Directive [1] in England and Wales. The second manuscript presented model development [2]. Landfill pollutant removal is intrinsically linked with achieving environmental equilibrium status. Equilibrium is defined here as that state when emissions from a landfill site occur at a rate that allows sufficient natural attenuation in the surrounding environment to prevent environmental harm, so management is no longer required. To embody the principles of sustainability, equilibrium can only be achieved when the management period (post-closure when the site has ceased accepting waste for disposal) is measured in decades rather than centuries. Landfills must remain under management control until the surrender of a landfill permit or licence is accepted by the regulator. Surrender is a site-specific determination and it can only be accepted if a landfill has stabilised physically, chemically and biologically to such a degree that the undisturbed contents of the site are unlikely to pose a pollution risk in the landfill's environmental setting (landfill completion) [3].

7.2 POLICY AND OPERATIONAL IMPLICATIONS

7.2.1 Sustainable Landfill

Large-scale modern engineered landfill sites, developed for groundwater protection, are now a common aspect of the waste management system in many countries. Despite this, they are comparatively new, with their development only starting in the 1980s. Consequently such sites, that predominantly contain untreated wastes, are a long way from achieving landfill completion or equilibrium status. One measure of landfill sustainability, an aspect that the EU Landfill Directive [4] does not explicitly consider, is the timescale required to achieve landfill equilibrium. Consideration of landfill sustainability requires an understanding of complex landfill processes that are responsible for stabilisation. There are two main methods of achieving sustainable landfill; flushing *in situ* wastes or pre-treating waste to produce residues that meet set criteria prior to disposal. Current pre-treatment techniques do not produce residues capable of meeting stabilisation criteria [5].

Site-specific landfill leachate measurements provide some indication of landfill processes, but recent modelling is starting to provide an understanding of the factors that control leachate quality [6]. An increased understanding of waste streams that disproportionately affect leachate quality may suggest a change in management practice is required (*e.g.* waste segregation, diversion, development of additional pre-treatment techniques) so as to bring forward the achievement of equilibrium. This could ultimately increase the sustainability of landfill.

7.2.2 Strategic Waste Management Options

Whilst raw municipal solid wastes (MSW) managed within a site allowing a moderate amount of leachate flushing, may be close to achieving equilibrium status, the requirements of the Landfill Directive [4] will make this option unavailable for the majority of sites as pre-treatment is a prerequisite of the Directive. This result is contrary to predictions made in the early 1990s [7-9] when it was recognised that measures have to be taken to enhance landfill degradation processes else degradation was likely to take many centuries for landfills to meet trigger concentrations. Combusting raw MSW in incinerators will meet the waste diversion targets, but the effect of combustion and the concentration of non-combustible fractions would appear to make equilibrium status more difficult to achieve. Processing of ash (either artificially or naturally via carbonation) reduces the level of emissions of some heavy metals such as lead and zinc, but results in a significant increase in the sulphate emission.

Coupled with the problems of landfilling the ash, there is the issue of the disposal of air pollution control (APC) wastes and fly ash which may be classified as hazardous wastes. It is unlikely that APC waste from the majority of incinerators will meet the current hazardous waste acceptance criteria (WAC), and an additional waste pre-treatment will be needed for these wastes (or an alternate means of disposing of them needs to be found).

Treatment technologies such as mechanical biological treatment (MBT) followed by intensive composting may provide a means of getting close to the objectives. However, the hydraulic conductivity of MBT residues may make it difficult to recirculate the fluids or introduce irrigation within the landfill, meaning that they will remain at low liquid/solid (L/S) ratios for extended periods. This technology does also remain highly dependent upon the feed

stock entering the process, and there is little experience in the UK in managing variations in plant feed stock.

Results of this study (not reported here) supported the notion that it is leachate and not landfill gas (LFG) that will be the rate limiting process for achieving equilibrium status. In all cases the LFG emissions fell below a manageable production rate prior to leachate reaching the requirements of equilibrium status. In some cases this was marginal but in such scenarios, the management option that best suited the requirements of rapid stabilisation included moderate flushing of the landfill to wash leachable contaminants from the landfill and at the same time optimise the rate of gas generation (in order to degrade the organic carbon as soon as possible).

7.2.3 Flushing of Wastes

The waste management industry has discussed the concept of the flushing bioreactor for many years, and the majority of practitioners believe that it is fundamentally workable provided there are large volumes of water available. Flushing rates in excess of 2000 mm y^{-1} of equivalent infiltration have been reported [10] and such rates may be needed to achieve stabilisation (but perhaps not be achievable) for landfills in excess of 50 m deep. Some hydraulic properties of waste change dramatically with depth [11]. The scenarios that we examined here concentrated on relatively thin waste deposits (circa 20 m) but with considerably lower flushing rates. The modelling assumed that during the operational period there is infiltration of 250 mm y^{-1} from rainfall, falling to 50 mm y^{-1} upon capping of the site. For the flushing scenarios, there is an assumed further addition of 200 mm y^{-1} for the MBT, raw MSW and one of the stable non-reactive hazardous waste options, and 500 mm y^{-1} for mechanically sorted organic residues (MSOR), incinerator bottom ash (IBA) and the other stable non-reactive hazardous wastes

scenarios during the period of active management. The choice was dictated in part by the results of the non-flushed model results for each waste residue. There was an assumption that if the liquid being recirculated is leachate (rather than water) it should have undergone at least some basic treatment to remove high concentrations of organics and ammoniacal nitrogen. Sulphate and chloride removal would not normally be an issue for raw MSW or MBT residues as these leachates would be unlikely to reach solubility limits for these species, but could do so if concentrations were allowed to build up. The reality is that additional infiltration of the order of 200 mm y^{-1} would be readily achievable from run-off available from capped areas for the majority of sites in the UK. Additional infiltration amounting to 500 mm y^{-1} would be geographically more restrictive. Flushing as a means of accelerating the stabilisation of landfilled waste has been advocated for many years [10] and research continues to demonstrate that it can serve to shorten the aftercare period considerably. However, there is no real incentive for the waste industry to invest in such practice. It seems that the recent revision of financial provisions was a missed opportunity to provide that incentive. It might have been possible to introduce a two or three tiered system with lower amounts of money set aside at sites where accelerated stabilisation measures were installed.

In the case of IBA landfills, the leachate would be quite different from that found in the more organic sites that we are familiar with. Leachates from these landfills will be largely inorganic and contain primarily metal salts. Alternate leachate treatment technologies will be required and the recirculation of treated leachate could be more problematic since the removal of sulphate will be needed to avoid saturation of these salts.

In many respects, the easiest way to achieve higher flushing rates would be to postpone the installation of a cap until equilibrium status was achieved. However, for wastes that are

likely to require any form of landfill gas control, such an option is not viable. It may be an option for incinerator bottom ash (IBA) wastes as there is little evidence that the ash generates landfill gas, and odour should not be an issue. This might be a function of the site geometry as a deeper landfill will require a larger degree of flushing or a longer management time to achieve the required L/S ratio, but the depth of the site is unlikely to have a pronounced affect on the gas generation rate and the time to reach equilibrium status in respect of gas. The option of removing a cap at the end of the gassing phase of a landfill may remain, subject to the regulatory requirements for minimisation of leachate production.

Removal of the cap at this stage would be similar to maintaining a flushing rate at 250 mm y^{-1} and could then proceed as a flushed site for a further 40 years or so, at which time it would achieve equilibrium status. Thus, this management scenario would not require re-circulation of leachate, nor would it require collection and injection of water specifically to achieve the flushing rates. The resultant time to equilibrium status would be extended by around 30 years, but would be significantly shorter than if the cap were to remain in place.

Furthermore, the liner performance can be expected to be better during the first century of its operation as the degradation due to loss of antioxidants is unlikely to occur within this period [12]. One issue that would make this option less favourable is the disturbance of what would have been a restored landform for some 30 years or so.

The time disjoint between achieving what is modelled as equilibrium status and the maximum groundwater impact probably remains one of the major obstacles to the practical use of this work. It will require pragmatic regulation in order to issue a closure certificate for a landfill that still contains contaminants at 10 times the water quality standard. This could be

further compounded by the fact that most waste permits will require a specific leachate head to be maintained. In order to approach the sort of conditions that might occur when equilibrium status is approached, there may need to be a planned gradual increase in leachate level so that the site reaches some degree of hydraulic equilibrium as well as chemical equilibrium with its surroundings.

7.2.4 Leachate Strength at Equilibrium Status

It was envisaged that the research would indicate a leachate strength at which equilibrium status could be shown to have been achieved and, along with other tests and monitoring data, allow a completion certificate to be issued for a site. The work undertaken here showed that the relationship between leachate strength and equilibrium status is far from linear. Where a site meets our definition of equilibrium early on, then the concentrations of leachate within the site that satisfy equilibrium will be higher than where the site meets the requirements later. This is simply a function of the degradation of the liner and capping systems where there is an expectation that these systems will see degradation with advancing age. This is compounded by the fact that those sites that will need an extended aftercare period will also have to continue to manage leachate to a quality that is less polluting than one where the aftercare period is shortened due to selective processing or flushing.

It has been suggested that the Inert WAC C_0 values (initial eluate from the percolation test prCEN/TS 14405:2003) for inert waste sites might have shown an indication of the leachate quality at the point of equilibrium. It was therefore worth comparing the typical ranges obtained from this study (albeit that they varied due to time dependency issues) with the WAC values for

inert waste landfills. A comparison of the leachate concentrations and the inert C_0 values is shown in Table 7.1.

Table 7.1. Comparison of inert WAC C_0 values and leachate concentrations at equilibrium status.

Component	Raw MSW	MBT & MSOR	Incinerator bottom ash	C_0 (percolation test) from inert WAC
	mg l^{-1}	mg l^{-1}	mg l^{-1}	mg l^{-1}
As	0.03	n/a	n/a	0.06
Ba	n/a	n/a	n/a	4
Cd	0.01	0.018	0.008	0.02
Cr	0.8	0.8	n/a	0.1
Cu	0.25	0.4	0.2	0.6
Hg	0.002	n/a	n/a	0.002
Mo	0.3	n/a	0.2	0.2
Ni	0.2-2.5	0.25	n/a	0.1
Pb	0.08-0.13	0.13	0.03	0.15
Sb	0.008-0.01	n/a	0.01	0.01
Se	0.03-0.04	n/a	n/a	0.04
Zn	0.75-0.8	0.8	n/a	1.2
Chloride	960-1230	430-1480	720-980	450
Fluoride	3.5-5.4	n/a	n/a	2.5
Sulphate	860	400-1100	350-1050	1500
NH_4	n/a	40-530	n/a	n/a

n/a – not available, generally because the leachate was at equilibrium status within 3 yrs and would therefore be below the C_0 value. Ba, Se and fluoride were not modelled.

These results indicated that while there is a generalised relationship between the formal C_0 values for inert wastes and those derived from this study, the relationship is not always strong enough to be relied upon. This conclusion follows that of the time dependency issue; although there are a number of cases where the leachate concentrations need to fall well below the inert

WAC C_0 values for inert waste sites (*e.g.* copper, lead and chloride). This is in part due to the way in which the C_0 values were generated, being a combination of modelling, rounding up/down and a comparison with leach test data from common waste streams that are deemed to be inert.

7.2.5 Ammoniacal Nitrogen

It is clear that certain species drive the time periods required to achieve equilibrium status. Prior to this research it was thought that ammoniacal nitrogen would be a governing factor given its high concentration in raw MSW leachates and low environmental threshold. While the modelling undertaken did not indicate that ammoniacal nitrogen was a major problem, it is recognised that some MSW leachates contain considerably more ammoniacal nitrogen than those modelled. It is of some comfort that the MBT compost process removes a large amount of ammoniacal nitrogen and fixes the organic nitrogen in other less mobile forms.

While this is to an extent reassuring, there remain a significant number of current landfills within the UK where ammoniacal nitrogen will continue to be a notable issue for many years. In addition, the modelling in this study assumed a composite liner. Where clay liners are used, it is likely that a greater flux of ammoniacal nitrogen will result and that this may still be an issue that affects the attainment of equilibrium status. What is surprising is the lack of commercial development of the research that has shown benefits associated with the injection of nitrified leachate back into the waste mass. Work by Burton and Watson-Craik [13] has shown that the introduction of nitrate into waste results in the denitrification of the leachate (resulting in the formation of nitrogen gas) and a substantial reduction in the nitrogen load within the landfill.

Ammoniacal nitrogen is likely to be much more of a consideration with regard to surface waters in instances where specific landfill sites are at risk of spilling over (the bathtub effect). This can occur as a result of continued integrity of containment coupled with the waste being saturated. Our modelling thus far not included emissions to surface water, which could form a separate study.

7.2.6 Accelerating the Achievement of Equilibrium: Air Injection

In a similar vein work has been undertaken, particularly in Germany and Italy, relating to the injection of compressed air into landfills that are in their methanogenic stage. Conventional landfill disposal encourages waste decomposition processes that operate anaerobically. This study has reported that modelled predictions suggest that landfill completion may take centuries, based on modern highly engineered landfills. This raises the question of sustainability for both operational landfills and the landfill legacy *i.e.* sites that are no longer licensed or permitted, some of which pre-date regulation. Many of these latter sites are ex-local authority landfills that are often relatively shallow and are slowly degrading with considerable pollution potential remaining. This treatment affords an opportunity to accelerate the stabilisation process for organic waste aerobically and potentially return sites to market for redevelopment in a timely fashion.

Whilst the benefits of aerobic or semi-aerobic landfill have been clearly demonstrated, including full-scale studies in Spain and Canada, many practitioners are sceptical of the benefits and are concerned about the risks such as gaseous emissions and entry of oxygen into a landfill that could cause fires. Other uncertainty surrounds the optimum point at which to begin aeration and the effect of aeration on leachate parameters.

Cossu *et al.* [14] reported the full scale stabilisation of part of a landfill (for rail construction purposes) by the injection of compressed air over a 140 day period. A marked drop in methane generation occurred and presumably an aerobic composting process was established. The study did not report the affects on the leachate quality during this period, but it would be expected that there would be a reduction in ammoniacal nitrogen production and a fall in BOD and COD, accompanied by a reduction in TOC levels. Work by Purcell and Walker [15] showed that on a pilot scale, the forced aeration of green waste and shredded biodegradable MSW can achieve a relatively stable residue within 6 months (aerox process). While the organic indicators (BOD, COD and NH_4) all showed marked reductions from the control cells, unsurprisingly the total heavy metal content of the waste remained virtually unaffected. The removal of some of the organic material will result in a long-term lowering of the TOC content of the waste and a subsequent reduction in the capacity to generate organo-metal complexes that form readily with some metals (especially copper).

While the aerox process was designed primarily as a means of pre-treatment of organic wastes, the work by Cossu [14] showed that it is feasible to undertake forced aeration *in situ* on a landfill site scale. It may therefore be possible to speed up the stabilisation of wastes that are approaching the end of their methanogenic life and achieve equilibrium status at least for landfill gas. Certainly the application of forced air injection either at the beginning or the end of the landfilling process would remove certain trace organic species that remain problematical within biologically active landfills and pose a threat to groundwater quality. While a number of researchers are looking at this area of forced aeration of organic wastes, rate constants still remain to be generated so that these processes can be readily modelled and the effects predicted.

7.2.7 Controlling Inorganic Species

Depending upon the waste treatment and the applied technologies, different inorganic species appear to control the time taken to achieve equilibrium status. For conventional raw MSW, these are likely to be lead and chloride (with ammoniacal nitrogen at sites where biodegradation in the groundwater is unlikely, or above average concentrations exist). For MBT residues, these include lead, nickel, zinc, sulphate and chloride (but not ammoniacal nitrogen). For IBA the contaminants most likely to cause problems are antimony, cadmium, copper, molybdenum, zinc and chloride. As the ash undergoes carbonation, zinc and lead will cease to be an issue but will be replaced by sulphate.

Certain MSW waste streams contain high proportions of these contaminants and it might be worth considering whether the likely changes in waste management practices could result in an increased diversion of these waste streams from landfill. In the case of energy from waste mass burn (EfWMB), much of the organic and combustible fraction of the waste will be lost from the waste mass (primarily carbon in its many forms) although any metals in these fractions may well be retained. It is only when certain waste fractions are removed from a waste stream (as might happen in the sorting and mechanical treatment stage of MBT) that whole waste fractions will be removed from the final residue. Contaminants within the waste entering an EfWMB will be partitioned between the IBA and the APC and fly ash generated. Given the concentrating effects of combustion (in relation to non-combustible fractions) and the change in oxidation state that results in some metals becoming more leachable, it is not entirely surprising that IBA performs least well in the modelling undertaken.

Table 7.2 contains the percentage proportions of a selection of elemental contaminants in raw MSW. The figures take into account the composition of the waste fraction itself, and the relative proportion of that fraction within a typical household waste stream. It does not take into account the relative leachability of the contaminants from each waste fraction nor does it consider any changes to the leachability through waste treatment or combustion. However, Table 7.2 could be useful as a guide for managing a specific site *e.g.* if cadmium is a concern, removal of the dense plastic waste stream could be a priority.

Table 7.2. Percentage distribution of specific elements in waste fractions.

Element	As	Cd	Cl	Cr	Cu	F	Pb	Hg	Ni	Z
Paper and card	5.9	4.9	8.5	2.4	0.	**21.2**	1.8	5.3	4.1	1.
Plastic film	2.0	2.5	3.7	2.8	0.1	3.0	8.6	1.1	0.6	1.
Dense plastic	1.4	**39.9**	**44.3**	5.3	**20.7**	**14.9**	9.6	1.4	5.7	**14.**
Textiles	0.5	0.4	0.4	0.3	0.8	1.2	0.10	0.4	0.8	1.
Absorbent hygiene products	0.2	1.3	1.2	0.3	0.2	9.8	0.6	0.3	0.3	0.
Wood	0.9	0.2	0.2	0.5	0.0	0.0	0.5	0.0	1.2	0.
Combustibles	**14.6**	8.7	**21.3**	**16.9**	1.1	6.9	6.3	2.6	4.4	**15.**
Non-combustibles	**18.1**	7.7	2.6	6.5	0.3	**23.7**	**20.4**	2.0	9.2	4.
Organic	**13.3**	2.9	**11.6**	2.1	0.6	**13.4**	2.4	6.5	6.2	2.
Ferrous metal	**29.9**	**19.7**	0.0	**47.7**	1.9	0.0	**14.7**	**77.6**	**23.6**	8.
Non-ferrous metal	0.7	3.4	0.0	1.1	**20.8**	0.0	7.3	1.0	**11.6**	**13.**
Fine material <10mm	11.4	1.1	1.2	2.1	0.3	3.4	5.9	1.9	2.6	1.
Waste electrical & electronic equipment	0.0	0.0	5.0	**10.1**	6.8	2.6	5.9	0.0	7.7	6.
Hazardous household waste including batteries	1.2	7.2	0.0	2.0	**45.7**	0.0	**15.9**	0.0	**22.0**	**30.**
Batteries	0.3	1.3	0.0	0.3	16.3	0.0	5.0	0.0	5.4	8.
Clinical waste	0.4	0.1	0.1	0.1	0.0	0.6	0.6	0.2	0.2	0.
Paint/varnish	0.3	0.1	0.5	0.2	0.0	0.1	0.2	0.3	0.1	0.
Oil	0.2	0.1	0.3	0.1	0.0	0.1	0.1	0.2	0.1	0.
Garden herbicides & pesticides	0.3	0.1	0.5	0.2	0.0	0.1	0.2	0.3	0.1	0.

Note 1 – values in bold represent greater than 10% of the total contribution of each element.

Note 2 – Columns sum to 100% down to specific hazardous household waste. [18]

For MBT residues, we might expect that many of the contaminants associated with paper, plastics, ferrous and non-ferrous metals, and Waste Electrical & Electronic (WEE) goods will be removed. Given that for MBT issues remain with lead, nickel and zinc, it is of some concern that these elements are predominantly present within waste streams that should have been largely removed. With respect to lead, nearly 50% of the total lead in MSW should have been removed. For nickel the removal figure is 53%, and for zinc the figure is nearly 45% (see Table 7.2). Part of the reason why these contaminants remain a problem could be that much of the research on MBT residues comes from continental Europe, while the compositional data from MSW has been derived solely from UK waste streams. In addition, it must be remembered that MBT residues are likely to have a higher density than raw MSW and hence will contain a higher contaminant load within the same landfill volume. The fact that intensively composted MBT residues appear to have lower leachable metal content than less intensively composted MBT is not something that is readily understood, and may imply a fundamental difference in the overall performance of different MBT plants.

7.2.8 Organic Species

The understanding of the origins of some of the trace organics within leachate (which have not been modelled in this study) and landfill gas remains poor for many species. Certain trace organics such as mecoprop that is present in nearly all MSW landfills is probably one of the best understood, and is thought to originate from green waste (grass cuttings) and empty (or partially empty) herbicide containers. Mecoprop along with a number of other organic species

appears routinely in leachate samples primarily because they do not degrade anaerobically; and they are not volatile and hence do not partition into the landfill gas phase.

There is a very clear advantage in subjecting these contaminants (in the waste that they exist in) to an aerobic degradation period prior to placing them in to a landfill. Other contaminants, such as heavily chlorinated solvents, will degrade only in anaerobic conditions. However, these groups of compounds tend to be more volatile and will partition into the landfill gas phase.

7.2.9 Measuring Equilibrium Status

One of the objectives of this work was to define a method for assessing when a site can be regarded as having achieved equilibrium status. This research has shown that this is a complex issue and one that will, in part, be based on the specific design of a landfill and upon its local hydrogeological regime. Furthermore, the criteria used to judge the impact on groundwater will almost certainly not be the Drinking Water Standard (DWS) that has been used for the comparative modelling that has been reported in this document. Under the Water Framework Directive [16] it is likely that groundwater quality criteria will be developed for specific groundwater bodies or classes of groundwater body. Furthermore, the revised Groundwater Directive [17] may include a change in the way the discharge of Listed substances is regulated. These may change the way completion will be judged and may have a marked impact on the completion time or equilibrium status of a landfill.

Leachate quality and an assessment of the hydraulic performance of a site, coupled with long-term monitoring data from the site remains the only viable means of assessing completion

criteria at present. This will have to be addressed on a site-specific basis as it is clear that no uniform standard will work in each and every case.

For sites where flushing has been undertaken over the entire landfill area it is likely that much of the waste will have achieved a reasonably high L/S ratio, and while there are bound to be areas that have (due to short circuiting) received less flushing than others, on the whole, the residual emissions are likely to be acceptable. For sites where there has been less flushing, or flushing is restricted to certain areas of the site, then an investigation of the leaching properties of the waste may be warranted. The leach tests defined for the WAC characterisation testing will be appropriate tests, and the degree of variance between samples and comparison between the leach test data and existing leachate quality will provide an indication as to whether the whole site has reached equilibrium status or whether water flow through the waste has been channelled in very specific areas.

The fact remains that it is easy to measure leachate concentrations. What is needed is a means of measuring the flux of contaminants migrating from the site. Only when we are in a position to measure the flux can we properly judge the equilibrium status in a way that does not require a great deal of conservatism.

7.3 CONCLUSIONS

This study has examined the municipal solid waste streams that the UK will most probably generate following the implementation of the Landfill Directive requirements to reduce biodegradable waste from going to landfill and the requirements for waste pre-treatment. The properties of the residues undergoing the most likely processes were investigated via a literature

search and an assessment made via modelling of the probable management times these residues would require within a landfill environment. A re-evaluation of these properties in light of emerging knowledge, for example on pre-treatment aspects, may show whether such aspects bring us any closer to equilibrium status or further from it.

Each process investigated generates a residue that will need to be landfilled and there is a likelihood that at times the products generated by these processes (*e.g.* MBT compost, or refuse derived fuel (RDF) material) may be out of specification, or in quantities greater than the demand, and hence these may also need to be landfilled.

On the whole those processes that involve the combustion of wastes are shown to lengthen the period of time taken for the landfill to reach equilibrium status. This is partially due to the higher densities of the wastes resulting from combustion and hence the higher amount of metals and salts that can be disposed off within the same landfill void space. However, the same argument cannot be applied to intensively composted MBT residues which, despite their higher density, give the best performance waste residues considered. Interestingly, these MBT residues that result from intense composting perform as well as, but no better than, raw MSW except that they give the same performance while possessing a higher density, and therefore represent a better use of the void space within a landfill.

Leaching processes can be accelerated by the flushing of landfills with either recirculated, but treated, leachate, or the infiltration of additional volumes of water. Other processes that show promising results at full scale were discussed, including landfill aeration and recirculation with nitrified leachate.

The modelling was restricted to those processes that are readily modelled (even if the model parameters are uncertain). There remain a number of landfill processes and management practices (such as nitrate injection and air injection) that are the subject of interest world-wide but as yet have not been developed to the extent that the processes have rate constants. Air injection in particular, shows promising results in reducing the emissions from landfill with marked improvement in leachate quality. It is likely that modelling the effectiveness of these processes will be forthcoming in the very near future. Further work is required to examine more critically the sensitivity to baseline assumptions, such as C_o, hydrogeological settings and kappa values. This can only happen once more data values are available.

While this study does not provide the answers to how the landfill aftercare period may be shortened sufficiently to attain sustainable development criteria *i.e.* achieving equilibrium within decades rather than centuries, it does take a step in the right direction and identifies where significant progress can be made towards achieving sustainable landfill and equilibrium status. It is clear that until such time as landfill sites achieve this status, risk assessment will be required. A sensitivity analysis of the baseline assumptions and data underpinning the modelling work is required. This was beyond the scope of our study but it could lead to the development of equilibrium criteria. A full technical and economic reappraisal of techniques that demonstrably accelerate waste decomposition processes is justified on the basis of modelled results presented in this study.

7.4 ACKNOWLEDGEMENTS

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8. PEER REVIEW, OVERALL CONCLUSIONS AND RECOMMENDATIONS

During the publication process, journal paper peer reviewers (**Chapters 4, 5, 6 and 7** in this thesis) and examiners (two external and one internal) raised a number of comments. There were editorial and substantive comments and it is the latter ones that are summarised here with author response. This chapter also contains integrating conclusions and recommendations, a need identified by the examiners.

Since Chapter 3 was written, the proposed regulatory emission standards for trace components, gas engines, flares and surface emissions have been finalised (see discussion in Chapter 1 and Environment Agency references).

The status of manuscripts was discussed and citations are as follows:

Submitted (Chapter 4)

Barry D.L., Gregory R.G., Harries C., Gronow J.R. and **Smith R.** (2007). Onset of methanogenesis and surface methane fluxes from uncapped, operational landfills in the UK. *Environmental Science and Technology*. Revision submitted 30th March 2007.

Published (Chapters 5-7)

Hall D.H., Drury D., Gronow J.R., Rosevear A., Pollard S.J.T. and **Smith R.** (2006). Estimating pollutant removal requirements for landfills in the UK: I. Benchmark study and characteristics of waste treatment technologies. *Environmental Technology*, 27, (12), 1309-1321.

Hall D.H., Drury D., Gronow J.R., Rosevear A., Pollard S.J.T. and **Smith R.** (2006). Estimating pollutant removal requirements for landfills in the UK: II. Model development. *Environmental Technology*, 27, (12), 1323-1333.

Hall D.H., Drury D., Gronow J.R., Rosevear A., Pollard S.J.T. and **Smith R.** (2007). Estimating pollutant removal requirements for landfills in the UK: III. Policy analysis and operational implications. *Environmental Technology*, 28, (1), 25-32.

8.1 ADDRESSING ISSUES RAISED BY REFEREES AND EXAMINERS

The examiners requested a section that identifies the personal contribution that the thesis author made to individual co-authored chapters (journal and conference papers). This was discussed at length with the examiners and is summarised in section 1.11.

Modelled timescales to achieve environmental equilibrium for UK landfills was discussed in the context of the major UK "Brown Book" study on dilute and disperse (also known as natural attenuation) landfills reported in 1978. This is discussed in section 8.3.

The definition of landfill sustainability was discussed with the external examiners in light of "no unacceptable burdens" rather than "no emissions" being passed on to the next generation (30 years). A wider commentary on this is provided in Chapter 1, section 1.3.

Chapter 2 refers to further work that is reported else where and additional references are now provided (Zacharof and Butler 2003; Knox 2005; Knox *et al.* 2005).

Summary of comments and response - Chapter 4

Peer reviewers recognised that this is a "significant contribution to a very relevant environmental issue" and they found the paper to be "very important to ES&T readers and that the paper may be published subject to satisfactory revision".

The main issue raised was regarding presentation of data which has been substantially revised both within the paper and the supporting information that would be published on the internet. The reviewers sought clarification to substantiate occasional statements with data. This has been addressed and a table has been added to the paper to provide details of individual landfill site characteristics *e.g.* age and depth of waste, cover material, number of monitoring points etc. Reference is made to a greater body of work (Environment Agency 2004g).

It is now stated that the landfill areas studied did not have landfill gas collection systems installed and that the monitoring installations for the methanogenesis study were only installed at one of the 21 landfill sites.

Additional information on the flux box design *e.g.* volume (0.12 m^3) and surface area (0.4 m^2) are provided.

One reviewer sought additional information on the funnel technique. Reference to the funnel technique was part of a lengthy exercise to select the flux box design and has now been removed as being outside the remit of the paper, focusing instead on presenting the design that was used.

Summary of comments and response - Chapters 5, 6 and 7

Chapter 5

A concern was raised as to the extent that LandSim data reflect sufficiently modern full scale MSW landfill leachate. LandSim is underpinned by various data from UK modern engineered landfills *e.g.* based on the work of Robinson & Knox on leachate composition which was updated to reflect Pollution Inventory data (includes many list 1 substances), so this reflects the best data available (see references).

One reviewer suggested that C_0 values are compared with leachate concentrations. However, it is not clear what comparison is actually made. The model has simplified the source-term - pH & Eh changes are not modelled, C_0 has simply been degraded.

Figure 1 (Chapter 5) is based on a release mechanism that is questionable in the assumption that there is one mechanism, which is relevant over the entire modelled time frame. What about preferential flow, stagnant zones releasing Cl to the infiltrating water, thus prolonging the time frame of higher levels? Yes we are simplifying. The kappa value for Cl takes this into account (slower). Much more real data for kappa values is needed. This is a recognised limitation.

The benchmark study includes a figure to show readers the hydrogeological situation that forms the basis of the calculations. This wasn't described until Paper 2 (Chapter 6) and the same figure now appears there.

Zinc is listed here as a key contaminant but Table 5.2 acknowledges that this is only when using the concentrations found in acetogenic leachate and therefore "not a real issue", so it is no longer highlighted as such in the conclusions.

Explanation of the terms MRF and dirty MRF is now provided. Dirty MRFs may be suitable for rural communities (suggested in **Figure 5.4**) where the volumes of material and other practicalities associated with requiring individual households to voluntarily segregate recyclables from non-recyclable material are particularly challenging.

Table 5.2: Initial concentrations given for Cr, Cu, Ni and Zn were different than those given in Table 6.7.

Chapter 6

Dissolved organic matter (DOC) was identified as being absent from the parameter list. Though an important parameter DOC was not modelled because the compound is unknown and therefore the fate and transport properties are not yet established.

Table 6.3: The kappa values were considered to be generally high indicating substantial mobility. These data were derived for construction materials with an emphasis on alternative materials and they may not reflect behaviour of waste mixes in landfill. However, these are values used by the modelling subgroup (TAC) for developing EU Waste Acceptance Criteria and are therefore the best available data.

One peer reviewer was under the impression that biodegradation for ammoniacal nitrogen was not included. Degradation of ammoniacal nitrogen is discussed early on in the paper - see also Environment Agency (2003) Review of ammonium attenuation in soil and groundwater, NGWCLC Report NC/02/49.

In the tables the number of decimal places sometimes gave an impression of precision, which was not there. Data is now modified to reflect some of the uncertainty in the modelling.

Chapter 7

Reviewers were critical that only two options were listed for attaining sustainability: flushing and pre-treatment. This is strongly focussed at reactive organic wastes as they represent most current UK landfills.

Some key messages were identified that needed to come across with greater clarity, in particular: the need for further work to examine more critically the sensitivity to baseline assumptions (C_o, hydro settings, kappa, K_d); and the need to re-appraise pre-treatment techniques to show whether they bring us closer to equilibrium status or further away from it.

The modelling has so far been concerned solely with emissions to groundwater. Discussion is now included on the bathtub effect (the filling up of engineered landfills) and consequent risks to surface water. This requires a separate modelling exercise before policy implications can be definitive.

8.2 OVERALL INTEGRATING CONCLUSIONS

1. Liquid and air injection have shown significant enhancement in landfill gas production rates; air injection in particular gave rapid and prolonged gas enhancement **(Chapter 2)**.

This suggests that there is a need to revisit the role of accelerated waste stabilisation techniques in achieving earlier landfill completion (see also **Chapters 5, 6 and 7**).

2. Regulatory emission standards and methods have been developed for landfill gas trace components, engines, flares and landfill surfaces (**Chapter 3**) to advance understanding and provide greater environmental control of both raw gas emissions and those from combustion.
3. Landfills are shown to be unsustainable waste management methods in their present form when considering the predicted timescales associated with achieving environmental equilibrium status for selected contaminants (often many decades).
4. The timescales associated with the onset of methanogenesis in landfilled waste have been quantified at a UK landfill site (**Chapter 4**) and data suggest that gas control systems should be installed in operational cells within 6 months of initial waste emplacement (assuming that the waste is predominantly MSW).
5. By identifying the proportions of selected elemental contaminants present in raw MSW (**Chapter 7, Table 7.2**), contaminants likely to be of concern for a particular waste treatment processes *e.g.* MBT can be identified and potentially removed at source.

8.3 RECOMMENDATIONS

The "Brown Book" study (Department of Environment 1978) of nineteen landfill sites, selected as being representative of the main geological types found in the UK, concluded that "attenuation mechanisms are available in the landfill and underlying strata, which are extremely beneficial if used with discretion". Furthermore, "substantial reductions of TOC which were often noted were sufficient to allow any residual tail to be attenuated to background concentrations in groundwater". There is therefore a need to reappraise our approach to landfill

philosophy perhaps by revisiting some of those "Brown book" landfill sites to ascertain the degree of intrinsic bioremediation and other attenuation processes achieved as they may be more powerful than is calculated. This has implications for landfills operating on both attenuation (pre-Landfill Directive) and an engineered containment basis.

The role of accelerated stabilisation techniques and their potential benefits in reducing the timescales associated with achieving rapid waste stabilisation (within 30 years) needs further work at the field scale. In particular the role of controlled air injection (aerobic or semi-aerobic landfill) needs to be demonstrated at the full site scale, including detailed cost benefit analysis.

A further examination of modelled timescales for achieving environmental equilibrium for landfill should be made once additional data is available *e.g.* kappa values and properties of waste residues from treatment processes. This modelling could include a sensitivity analysis of baseline assumptions and data underpinning the modelling work.

The actual timescales associated with the exponential decay phase of landfill gas production (see section 1.2) need to be established through reliable monitoring to validate gas production models and reduce their uncertainty.

Emissions data derived from landfill surfaces and gas management systems need to be captured and assessed on a national basis to identify the temporal and spatial variability for example in trace gas components.

The relative contribution of emissions from landfill gas needs to be assessed in the context of Local Air Quality objectives *e.g.* NO_x emissions from gas engines versus transport.

The potential synergistic and additive effects of landfill gas trace components needs to be considered in order to understand potential health and environmental impacts.

For landfill operators and their consultants, the design of landfill gas management systems needs to take account of the timescales for the onset of methanogenesis (**Chapter 4** and **Appendix C**) and for the preferential lateral movement of gas within waste.

APPENDICES

Proceedings Sardinia 1999, Seventh International Landfill Symposium, S. Margherita di Pula, Cagliari, Sardinia, 4-8 October 1999, Vol.II, 499-506.

APPENDIX A: GUIDANCE ON BEST PRACTICE FLARING OF LANDFILL GAS IN THE UK

R.D.EDEN * AND R.SMITH **

* *Organics Ltd, The Barclay Centre, University of Warwick Science Park, Coventry CV4 7EZ E-mail: comms@organics.co.uk*

** *Environment Agency, Block 1 Government Buildings, Burghill Road, Westbury-on-Trym, Bristol BS10 6BF E-mail: richard.smith@environment-agency.gov.uk*

SUMMARY: Regulation of landfill gas in the UK has historically been focused on the need to control migration and flares have been operated in the absence of any emissions standard. In recognition of this, the Environment Agency's National Landfill Gas Group commissioned a technical review of emissions from different types of landfill gas flare. This work has resulted in a document entitled "Interim Internal Technical Guidance for Best Practice Flaring of Landfill Gas". The current paper summarises some of the key aspects of this guidance.

A.1 BACKGROUND

A.1.1 The development of the Environment Agency's position

Regulation of landfill gas in the UK has historically been focused on the need to control migration. Flares have been operated in the absence of an emission standard. In recognition of the lack of flare emissions monitoring and the absence of a UK standard, research was

commissioned by the Department of the Environment (Frost *et al*, 1996) in the form of a technical review of emissions from different types of landfill gas flares. The Controlled Waste Management R&D Programme of the Department of the Environment transferred to the Environment Agency and became the Waste Regulation and Management Research Programme on the Agency's creation in April 1996.

The Environment Agency's National Landfill Gas Group commissioned work to develop this further and has released internal guidance based on the initial review, predominantly for use by its waste regulators. The aim of this guidance is for the Agency to move towards landfill gas emission control on a site-specific, risk assessment basis in preparation for the EU Landfill Directive.

It was estimated in 1994 that approximately 20% of total UK methane emissions could be attributed to landfill gas. This is believed to represent between 1 and 2 million tonnes of methane per annum (Environment Agency, 1998). Increased control of landfill gas emissions is therefore an essential part of the UK Climate Impacts Programme to reduce methane emissions in accordance with the Kyoto Protocol. It is expected that the contribution of methane emissions from landfill will continue to decrease as an increasing number of modern landfill sites will be required to have effective full-site landfill gas collection systems.

Large-scale passive venting of landfill gas has been a standard measure to relieve gas pressure and control migration in the UK, arguably without due consideration of the atmosphere. Large-scale passive venting of landfill gas is no longer an acceptable landfill gas disposal or adequate control option. In instances where a flare is not self-sustaining, the Agency is encouraging gas collection combined with the use of alternative techniques, predominantly methane oxidation, which can be an efficient, effective, low cost control option.

A.1.2 Background to landfill gas flaring in the UK

Techniques for the combustion of landfill gas have undergone many changes over the last fifteen years. From the initial "pipe-flares" of the early 1980s, where vertical tubes were simply forced into the surface of a site and the emerging landfill gas lit with a burning oily rag, the technology employed has advanced significantly.

Higher standards demanded for the landfilling of wastes, including the need to control emissions, have added impetus for the flaring of landfill gas, where undertaken, to be carried out in an acceptable manner.

With a typical landfill gas flow rate of one thousand cubic metres per hour, the mass of methane that may be released to the atmosphere could be approximately three thousand tonnes per annum, dependant upon the percentage of methane. Worse still is the mass of certain other pollutants that may be released into the atmosphere. On specific sites in the United Kingdom the following typical rates of substance release (based upon 1,000 cubic metres per hour of landfill gas) have been encountered:

- Aromatic hydrocarbon: 5 tonnes per year
- Alcohols: 45 tonnes per year
- Halogenated compounds: 12 tonnes per year

In the mid to late 1980s it became apparent that open flares, burning with simple diffusion flames, might create new difficulties by the release of novel airborne emissions. Open flares were shown, for example, to allow a significant percentage of gas to pass through the combustion zone without being ignited. One study by the Atomic Energy Authority at Harwell measured fifteen-percent methane in the exhaust of one flare.

A further difficulty arises with the products of incomplete combustion. Certain halogenated hydrocarbons, for example, when subjected to a cooling reaction around the edges of a flame

may act as precursors to the formation of a variety of novel compounds including dioxins and furans.

A.2 STRUCTURE OF ENVIRONMENT AGENCY POLICY

The Environment Agency has recently adopted a general overarching landfill gas policy, with an associated landfill gas flaring policy. These policies have been introduced to allow the Agency to carry out its statutory duty to protect human health and the environment through the licensing of landfill sites. Arguably, this cannot be achieved without applying risk assessment principles and associated flare monitoring. This is a regulatory approach consistent with that used for other combustion processes that the Agency regulates.

The general landfill gas policy states that gaseous emissions from licensed landfill sites will be regulated by the Environment Agency according to site-specific risk in order to minimise the impact on health, the local environment and global atmosphere.

The recommendations that embody the landfill gas flaring policy that the Environment Agency has adopted are described in section 6 of this paper. Future best practice guidance and associated policy statements are anticipated from the Environment Agency regarding landfill gas utilisation and methane oxidation.

A.3 CURRENT AND FUTURE ENVIRONMENT AGENCY LANDFILL GAS R&D

A modular landfill gas risk assessment tool that models the human health and environmental impacts of landfill gas has recently been developed for the Environment Agency. This model complements an existing suite of risk assessment models on landfill design and groundwater protection (Landsim) and land contamination (Consim). The model will help the Agency and other relevant organisations compare, with increased confidence, the relative risks associated

with different landfill management options. It also provides a framework that can contribute to the assessment and valuation of the inventory of burdens associated with landfilling wastes.

In order to reduce the uncertainty surrounding the assessment of methane emissions from different categories of UK landfill sites, the Environment Agency commissioned and has published an R&D technical report that verified estimates by appropriate field measurements (Environment Agency, 1998). This project demonstrated that emissions of landfill methane vary according to quantity, composition, age and depth of the waste, site area, site-filling regime, type of cap and/or cover and the efficiency of gas collection systems (if installed). A subsequent project is now developing landfill methane measurement protocols to enable the regulation of methane emissions from landfills under the range of operational parameters likely to be found in the UK (Environment Agency, 1999b).

Currently in the UK there are no legislative requirements or regulatory standards for the emission of combustion products from landfill gas energy recovery plant. The Department of the Environment commissioned an R&D Project to monitor gaseous and particulate emissions from landfill gas energy recovery plant, the aim of which was also to design an emissions monitoring protocol for subsequent use. The final report is in preparation and is expected to contribute to a further guidance document (Environment Agency, 1999d). To avoid a potential conflict when prescribing an emission standard for utilisation plant it will be crucial to take account of the wider environmental and energy benefits which using LFG as fuel can provide. Emissions dispersion is also likely to be an important factor.

Building on previous research, an R&D contract was awarded to assess artificial enhancement of the microbiological oxidation of methane with the aim of providing an additional strategy for controlling methane emissions at UK landfill sites. This study is ongoing and is due to report in December 1999. The aim is to produce guidance on the application of selected landfill site cover materials to achieving high rates of methane oxidation when applied

as a surface layer for landfill restoration. This will be particularly useful in circumstances where a flare cannot be sustained due to insufficient gas yields.

The measurement of gas potential is an R&D project that was commissioned to provide the ability to assess the potential of existing landfill sites to produce and emit landfill gas. This is of fundamental importance in terms of Section 39 of the Environmental Protection Act 1990 when a landfill licence holder makes a licence surrender application. The application will only be accepted if the Environment Agency is satisfied that the site is unlikely to cause harm to human health or the environment. Guidance will be provided on the use of biological methane potential tests to assess gas pollution potential. This study is also due to report in December 1999.

Future landfill gas R&D, planned to commence in the next 1-2 years includes:

- a review of the Waste Management Paper on landfill gas;
- a national atmospheric methane emissions reduction strategy;
- the effect of gas clean-up and best practice guidance on emissions from landfill gas energy recovery plant ;
- the development of landfill gas flare monitoring protocols;
- best practice guidance on methane oxidation

A.4 THE TECHNOLOGY OF LANDFILL GAS FLARES

A.4.1 Combustion requirements

Combustion is still considered by many authorities to be a 'black art', in the sense that the complex reactions that occur between ignition and complete combustion are still a matter of some uncertainty. Design remains a largely empirical process relying upon the accumulated experience of combustion engineers.

A flame results from the rapid combination of oxygen and fuel with the release of heat. So called perfect combustion is obtained by mixing and burning exactly the right proportions of fuel and oxygen so that nothing is left over. If more oxygen than theoretically required is supplied the mixture is 'lean' and the flame is said to be "oxidising". This results in a flame that is short and clear. The excess oxygen plays no part in the reaction but it assists towards good mixing and an efficient burn. The excess air also cools the combustion process. If too little oxygen is supplied the mixture will be "rich" in fuel and the flame "reducing". This results in a flame that tends to be longer and sometimes smoky. A "rich" mixture will result in incomplete combustion with the resultant emission of unburned hydrocarbons.

Good combustion requires a proper proportioning of fuel and air, thorough mixing of the fuel and air, and initial and sustained ignition of the mixture. It is worth noting at this point that the use of excessive amounts of air may cool some parts of the combustion zone below the ignition temperature so that some of the fuel goes out unburned. Thus it is possible to have incomplete combustion and unburned hydrocarbon emission with lean mixtures as well as with rich mixtures.

In the specific case of landfill gas flares the minimum requirement is as follows.

i. Combustion must occur with excess air, sufficient to ensure complete combustion but not so much that the flame temperature is cooled below the point at which effective destruction and removal of toxic trace gases is obtained

ii. The temperature in the combustion chamber should be as uniform as possible across the measured retention volume. Excessive temperatures will assist the formation of oxides of nitrogen (NO_x) and low temperature spots within the combustion chamber will lead to partial combustion. Cooled surfaces must be avoided to prevent the formation of products of partial combustion, particularly the conditions that might lead to the formation of dioxins and furans

iii. The temperature should remain as near constant as practical for different thermal loadings of the combustion chamber. This implies that there should be some form of control over the combustion air or recycled exhaust gases.

iv. Mixing should be optimised to balance the requirement for thorough mixing with the above.

v. The retention time at the specified temperature must be adequate to deal with the trace components in the landfill gas.

A.4.2 Current technology

Current technology employed in the UK and overseas can broadly be placed into three categories.

i. The first, and the most basic, is an open flame flare that employs diffusion flame technology. This involves air and gas mixing after the gas has passed through the burner port. This type of flare results in a long diffusion flame with high luminosity. No control is attempted over combustion air, other than in a rudimentary form by controlling exit velocities.

ii. The next level of sophistication is another type of open flame flare where some air is entrained with the fuel gas prior to the gas exiting the burner port. This controls the combustion to a greater degree and results in higher flame temperatures with lower luminosity flames. Such flames are characterised by no, or little, yellow tipping.

iii. Enclosed, or "ground", flares are used to hide the flame and to reduce heat loss to the environment from the flame zone. This generally results in higher combustion temperatures, dependant upon how the combustion air is controlled. Enclosed flares can be based either upon aerated or non-aerated flames. Combustion strategies employed by some manufacturers, for example, use turbulent diffusion flames to produce high temperatures over extended retention times. In this category the combustion chamber temperature and the

period over which the gas is held at this reference temperature is the key to its specification. The Environment Agency recommendation is that a minimum retention time of 0.3 seconds at 1000°C is the minimum that is acceptable.

iv. Advanced strategy burners employ various techniques such as exhaust gas recycling, combustion air staging and fuel gas staging to achieve higher quality emission standards.

A.4.3 Definitions employed by the Environment Agency

There exists a large number of trade names and technical jargon to describe different types of flares. To clarify these definitions the Environment Agency has adopted the terms "open flares" and "enclosed flares" to provide structure to their internal guidelines.

"Open flares" are those which burn landfill gas as open 'Bunsen-burner' type flames, though a windshield is normally fitted to prevent the flame from being blown out. The flame is generally visible to the naked eye, particularly at night. Because of the rapid diffusion of gases, open flares are virtually impossible to monitor with any confidence or degree of consistency.

"Enclosed flares" are those in which the landfill gas is burned in a vertical, cylindrical or rectilinear enclosure. Some means of combustion control is normally provided and the enclosure is generally insulated to reduce heat loss and allow operation at higher temperatures. Due to a higher residence time within the flare, combustion is more complete than that in the associated open flares, having the effect of reducing flare emissions particularly with regard to trace gas components. Emissions monitoring from enclosed flares is more readily achievable.

A.5 HEALTH IMPACTS

A.5.1 Environmental Impacts

Landfill gas and the exhaust from landfill gas flares may have a wide range of impacts. Where improperly managed, their impacts may include, as well as the basic concern of explosion and fire, odour nuisance, harm to flora and fauna, noise pollution, photochemical air pollution, acidic precipitation, stratospheric ozone depletion and global warming.

A.5.2 Air dilution

As an indication of the type of long-term impacts that may be encountered it is useful to consider dilution factors around a typical flare installation. The results of a long-term air dispersion study for two typical installations in the United Kingdom showed that long-term dilution factors exceed 10,000, except within the close vicinity of the flare, and very rapidly reach 50,000. In terms of typical trace gases emissions from flares this implies that in the long-term risks to health and the environment will be small. The output was based upon mathematical modelling of atmospheric conditions.

By contrast, however, a study carried out in Germany (Gerhardt, 1993) estimated the degree of dilution of gas leaving a landfill site at night. It was found that landfill gas plumes were travelling a distance of up to 1 km with very little dilution whatsoever. This disparity clearly indicates the need to treat such information with caution.

A.6 RECOMMENDATIONS CONTAINED WITHIN THE INTERIM INTERNAL TECHNICAL GUIDANCE FOR BEST PRACTICE FLARING OF LANDFILL GAS

Against the above background of increasing awareness about the issues involved in the flaring of landfill gas, and based upon substantive advice from the National Environmental Technology Centre (NETCEN), the following recommendations have been agreed as applicable to the question of assessing landfill gas flaring systems.

Recommendation No.1. No more open flares should be installed on UK landfills except for test and emergency purposes, and then only for limited periods of not greater than six months.

There are a number of specific problems with open flares that militate against their continued use. There is ample evidence to indicate that the conditions within an open flare are favourable to the formation of by-products of incomplete combustion. These include a wide range of compounds and include dioxins and furans. Carbon monoxide is a good indicator of incomplete combustion.

Recommendation No.2. Existing waste management licenses at landfill sites should be modified so as to effect the replacement of open flares with enclosed flares (or techniques offering equivalent performance) over a period of five years, particularly at sites that:

- **produce large amounts of landfill gas; and/or**
- **are close to population centres or other areas of environmental importance.**

The period of five years, starting from January 1st 1999, has been chosen to permit adequate time for this standard to be adopted. It is specifically acknowledged that other technologies may exist or be developed that will dispose of landfill gas in an acceptable manner. It is not the intention of these guidelines to be exclusive of such alternatives.

Recommendation No.3. The combustion air supply should be controlled so as to achieve a minimum of 1,000oC and 0.3 seconds retention time at this temperature whatever the landfill gas composition and throughput (within expected design limits).

In an open flare the presence of cool zones at the flame's periphery results in incomplete combustion and, therefore, less heat release than is theoretically possible; radiative and convective heat losses are also substantial and uncontrollable.

Precise calculation of retention time is difficult to achieve. Retention time calculations should be assessed on the basis of satisfactory compliance with the intention of holding combustion gases at the design temperature for an adequately long period of time. Further research work is being carried out to define a method for obtaining a more accurate measure of this variable.

Recommendation No.4. To ensure that flare systems are operating correctly they should not exceed the following emission concentrations when referred to Normal Temperature and Pressure (NTP=0oC and 1013 mbar) and 3% oxygen:

Carbon monoxide (CO) - 50 mg/m3

Oxides of nitrogen (NO_x) - 150 mg/m3

Unburned hydrocarbons - 10 mg/m3

Complete combustion may not be achieved in flares operating outside their design conditions, and partially burned fuel may show up as carbon (smoke, soot, particulates) and/or intermediate reaction products such as CO. Incomplete combustion may result from:

- lack of oxygen caused by poor mixing of fuel and air or an overall air deficiency;
- cooling of the flame by, for example, radiation or its impingement on cold surfaces;
- inadequate time at high temperature for the complete oxidation of carbon - the limiting factor for gases being the oxidation of CO to CO_2.

Recommendation No.5. Inlet gas concentrations should be analysed to determine if there are any incombustible substances contained within the inlet gas that may require to be removed prior to entering the flare stack.

Certain substances, such as chlorines and fluorines, are not destroyed thermally at the temperatures found with landfill gas flares. In large quantities such substances may lead to difficulties in their own right.

Recommendation No.6. Enclosure design should:

- **permit an homogenous temperature distribution across the combustion chamber;**
- **be lined with refractory material on the interior**
- **contain the flame within it; and**
- **be maintained in an effective manner**

Recommendation No.7. Operators of landfill sites should undertake - or commission - an environmental assessment of the emissions from existing and proposed flares which:

- **should use either measured or reported emissions data, flow rate data and local meteorological data;**
- **consider the impacts of the dispersed emissions in the vicinity;**
- **determine whether flaring is required for migration control and/or pollution prevention; and**
- **be approved in writing by the Environment Agency.**

Given the highly site-specific nature of landfill sites and the pollutants emitted from landfill sites, it is recommended that an environmental assessment be carried out to ensure that a proposed or existing flare will meet with the environmental criteria for which it is designed. The Environment Agency should approve in writing the findings of such studies.

Recommendation No.8. Flares should be positioned and sized so that potential health and environmental impacts are minimised.

The optimum location for a flare will come out of the environmental assessment recommended above. This should be such as to minimise potential health and environmental impacts.

Recommendation No.9. Guidance is given in the guidelines as to the level of monitoring that should be recommended by Environment Agency officers.

Recommendation No.10. Flares should be maintained in accordance with the manufacturers' recommendations. Full records should be available for inspection.

Recommendation No.11. All results obtained by flare-system managers should be the subject of a formal review. Such reviews must accompany results and reports when communicated to Environment Agency officers.

A.7 CONCLUSIONS

The introduction of a UK best-practice flaring standard has many benefits to the local and global environment and in minimising potential human health risks. The associated Environment Agency's landfill gas policies represent a paradigm shift towards landfill gas emissions control with the clear message that large-scale passive venting of landfill gas can no longer be considered as an effective control option. The design of open flares makes emissions monitoring that has any degree of accuracy difficult and the replacement of such flares will ensure that current UK best practice is maintained and becomes widespread. It will also provide the confidence that enclosed flares can be monitored, assessed, optimised and correctly maintained.

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APPENDIX B: DEVELOPMENT OF A BIOCHEMICAL METHANE POTENTIAL (BMP) TEST AND APPLICATION TO TESTING OF MUNICIPAL SOLID WASTE SAMPLES

C. R. HARRIES*, C. J. CROSS** AND R. SMITH°

* *Waste Technical Services Ltd., 10 Burlington Terrace, Cardiff, UK. CF5 1GG*

** *HARP International Ltd., Gellihirion Industrial Estate, Pontypridd, UK. CF37 5SX*

° *Environment Agency, Head Office, Block 1, Government Buildings, Burghill Road, Westbury-on-Trym, Bristol, UK. BS10 6BF*

SUMMARY: Biochemical methane potential tests may be used to predict the magnitude of methane emissions produced by wastes or organic materials decomposing under anaerobic conditions. The test results may be used to determine the size of any environmental impact or energy production which may be related to anaerobic decomposition whether in landfill conditions, specific reactors or in the wider environment. The different elements of the BMP test procedure are considered, from sample preparation, through incubation conditions to methane quantification and an experimental procedure presented. Experimentally determined results are presented for MSW samples and selected solid wastes. Interferences from nitrate and sulphate are demonstrated, as is the effect of sample drying method.

B.1 INTRODUCTION

Determination of the quantity of methane or biogas from anaerobically decomposing organic matter may be of interest to scientists involved in environmental impact studies, energy production or treatment process control activities. Laboratory tests for the quantification of the methane produced have become known as Biochemical Methane Potential (BMP) tests and have been used in research laboratories and by specialist practitioners for many years. However, they have not become as widely used as the aerobic equivalent test, the Biochemical Oxygen Demand (BOD) test.

The reason for the slow adoption of BMP tests is probably due to a number of factors, including the long timescale of the test, specialist laboratory techniques involved and the uncertainty over interpretation of the results. Analysts may often prefer a 'short cut' and analyse samples for carbon, cellulose or volatile solids and then make inferences about the quantity of biogas that may be generated under anaerobic conditions.

The BMP test may be considered to be the anaerobic equivalent of the BOD test. In the BOD test, the quantity of oxygen consumed during the aerobic decomposition of organic matter is measured while, in the BMP test, it is the quantity of methane produced during the anaerobic decomposition of organic matter that is measured. Both tests are bioassays in which a sample is incubated in culture medium with a seed of appropriate bacteria under optimised conditions.

BMP tests use conditions in which the temperature, provision of nutrients and bacteria are more or less optimised for microbial methanogenesis. As such, it is the total amount of methane produced in the test that is of interest and not the rate of production. The rate of methane production under the BMP test conditions may bear little relationship to the rate observed from the sampled materials decomposing in less optimal conditions such as in a landfill site.

Methods used in specialist laboratories have often been designed for the assessment of toxicity or degradability of substances that may enter the sewage treatment system or for wastes that may be considered for treatment by anaerobic digestion. Methods have been published by Wang *et al*, (1994), CEN (1998), Pagga and Beimborn, (1993), Shelton and Tiedje (1984) and Owen *et al*, (1979).

This paper describes the development of a BMP test method aimed specifically at municipal solid waste (MSW) samples and in particular those from older landfills, where there is a need to determine the potential for future methane emissions and the need for their control.

B.2 METHOD DEVELOPMENT

An initial consideration of BMP test procedures was undertaken and the following elements were identified as the principal parts that would need to be specified in any test procedure.

- Sample preparation, including sample size, particle size, and drying method.
- Sample quantity and scale of test
- Culture medium definition including nutrient provision, buffering capacity and use of reductants
- Incubation conditions including temperature and test period
- The use of an inoculum including source and quantity
- The method of gas composition analysis and gas volume quantification

The following sections consider the elements of any BMP test procedure and give the reasons for our preferred approach.

B.2.1 Sample preparation

Sample preparation includes the steps of sampling from the source waste mass or landfill, drying and grinding, if used, and sub-sampling. The primary sampling in the field remains one of the most difficult areas of waste research. The quantity of material that constitutes a representative sample is likely to be large with landfilled MSW but will vary according to waste composition and the collection and tipping practices employed. It is also probable that the variability declines with age and the state of decomposition. Lohani and Ko (1988), indicated a sample mass of 93.75 kg should be used for sampling MSW at source, but in most cases it comes down to what can be physically and economically handled in the field and at the laboratory. The sampling method may also have a significant effect on variability as some drilling techniques cause considerable fragmentation of landfilled waste materials.

We favour drying, grinding and sub-sampling of waste samples which allows greatly reduced variability between sub-samples used in the actual tests and so the whole test procedure can be reduced in size. This facilitates replication.

Oven drying at 105°C was the preferred method because of time, the fact that the method was aimed at old landfill samples and that this type of drying facility is universally available. The effects of other drying methods on BMP test results for specific wastes are described later. If high concentrations of volatile organic compounds are present then alternative drying methods and possibly the addition of alkali to immobilise volatile acids may be considered.

Our tests indicated no significant difference in the rate of the methanogenesis whether the MSW sample was ground to pass through a 1mm, 3 mm or 5mm screen, though the variability between sub-samples is likely to increase with increasing mesh size. The rate of decomposition was observed by the use of scaled up tests to continuously monitor gas volume production as described in the section on incubation conditions. In practice it proved to make little difference in sample preparation time so all samples were ground to pass through a 1 mm mesh as a standard procedure. This was achieved using a Cross-beater mill (Retsch Model SK1).

Approximately 10kg wet weight of primary sample was routinely oven-dried and ground to <1mm particle size.

B.2.2 Sample quantity and test scale

Considerations when choosing the scale of the test are the quantity of gas that will be produced, the ease of replication, the requirement for inoculum, the availability of suitable test vessels and the variability of the ground sample at the proposed sub-sample size.

For reasons of minimising the labour element of the test, we decided that the tests should be set up, left for the entire incubation period and the gas removed and quantified at the end of this period only. No intermediate measurements would be made. To allow this, the quantity of sample employed must be selected in relation to the size of the test vessel and its headspace such that there is no risk of vessels bursting during the tests.

Our method uses 0.5 g of ground sample in 50 ml of culture medium contained in a butyl-stoppered, 160ml glass serum vial. At this rate of sample addition, there is little risk of the vials bursting even with highly degradable sample types.

B.2.3 Culture medium

The primary requirements of the culture medium are that it contains a good balance of primary nutrients relative to the potential carbon present in the sample to give a favourable C:N:P ratio. It should also contain all necessary trace elements and growth factors for optimum methanogenesis. Good pH buffering capacity is beneficial to resist acid destabilisation resulting from more rapid organic acid formation by the fermentative bacteria than their conversion to biogas. The medium we have used was developed from that used by Wang *et al*, (1994). In comparative trials against other media we found that this had the best resistance to pH reduction when titrated with an acetic acid solution. However, it will be de-stabilised if 0.5g of readily

fermentable organic matter such as glucose is used, resulting in severe pH drop and no methanogenesis. The sample quantity must be reduced to 0.25 or 0.1g for substances like this to avoid this problem. A pH check at the end of the test can be used to check if this has occurred as well as the gas composition analysis.

The use of reductants in the culture medium should create the reduced conditions suitable for methanogenesis without causing any inhibitory effects. Although sodium sulphide or cysteine hydrochloride have been used in many methods we have favoured amorphous ferrous sulphide which will give good reduction without high sulphide concentrations which may be inhibitory to methanogens. This was produced according to the method of Brock and O'Dea (1977) but using ferrous chloride ($FeCl_2.4H_2O$) instead of ferrous ammonium sulphate. $CaCl_2.2H_2O$ was added to the vials containing the stock FeS suspension rather than into the main medium as it tended to cause precipitation of other constituents. The composition of the culture medium used in our method is defined in Table AB-1.

The main culture medium was heated to boiling and cooled under a nitrogen stream to remove dissolved oxygen then dispensed into serum vials. Vials were flushed with nitrogen, allowed to cool to 35°C, the FeS suspension added by injection through the stopper and then once the pink resazurin colour had cleared, the inoculum was added to each vial by injection. Excess vial pressures were then vented to leave a nitrogen headspace at prevailing atmospheric pressure.

Table AB-1. Composition of BMP test medium

Part A - Main medium Component	Quantity/litre of complete medium* (g unless otherwise stated)
NH_4Cl	1.3
$K_2HPO_4.3H_2O$	2.7
NaH_2PO_4	1.43
$NaHCO_3$	4.2
$MgCl_2.6H_2O$	0.3
$FeCl_2.4H_2O$	0.1
Resazurin (0.1% solution)	1 ml
Tryptose	0.1
Yeast Extract	0.1
Mercapto-ethane sulphonic acid (MES)	0.05
Trace Element Solution (Part C below)	10 mls
Deionised Water	to 960 mls
Part B - Additions per vial (each containing 48 mls of above medium)	
Seed culture	1 ml
FeS / $CaCl_2$ suspension (containing nominal 8.8g.l^{-1} FeS and 10 g.l^{-1} $CaCl_2.2H_2O$)	1 ml
Refuse or Waste sample	0.5g
Part C – Trace element solution contains the following constituents (g/l) $AlCl_3.6H_2O$, 0.04, $CoCl_2.6H_2O$, 0.2, $CuCl_2.2H_2O$, 0.02, H_3BO_3, 0.03, $MnCl_2.4H_2O$, 0.1, $Na_2SeO_3.5H_2O$, 0.03, $Na_2WO_4.2H_2O$, 0.03, $(NH_4)_6Mo_7O_{24}.4H_2O$, 0.02, $NiCl_2.6H_2O$, 0.02, $ZnCl_2$, 0.03.	

Notes

* quantities indicated are g/l of complete BMP medium based on final volume after addition of seed and reducing agent to individual vials. 960 mls Part A volume allows for subsequent addition of 2 mls/vial. *i.e.* for 20 vials total volume used is 960 mls of Part A medium plus 20 mls seed culture plus 20 mls reducing agent = 1 litre total.

B.2.4 Incubation conditions

The incubation conditions should achieve rapid methanogenesis. This requires constant mesophilic or thermophilic conditions. We have used mesophilic conditions (35°C) as these can be readily created through use of a temperature-controlled room in which staff can work. This is not realistic in the thermophilic range. In addition, the mesophilic anaerobic

environment is more common in nature than the thermophilic one which could mean the test results are more relevant. Thermophilic inocula are scarce though could be cultivated.

The incubation period should ensure a very high level of anaerobic degradation such that 95% or more of the potential biogas is produced in the test period. We have used a standard test period of 3 months though many samples produce virtually all the gas within 2 months.

To determine the necessary incubation period we undertook scaled up tests using the BMP test method detailed here but with all quantities increased 10 times to give a 500 ml test culture volume and using 5g dry weight of sample. Scale up in this manner enabled continuous monitoring of gas volume production from the culture which was contained in a 1 litre conical flask. Gas volume was recorded by a P181 low flow gas meter (Triton Electronics). Figure AB-1 shows the gas production from a scaled up BMP test of this type.

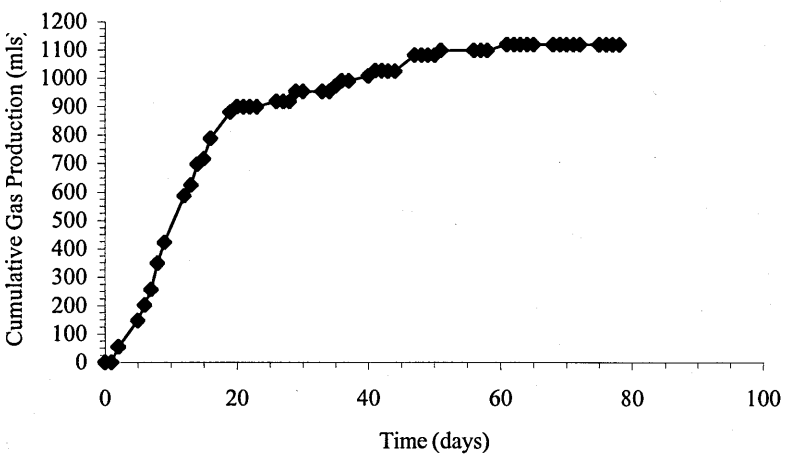


Figure AB-1. Gas production from scaled-up BMP test using sample of dried ground (< 1mm) 'wet pulverised' MSW and laboratory maintained seed culture as inoculum

B.2.5 Inoculum

It is essential that all tests contain the necessary metabolic capability in the form of the bacteria present, to achieve total methanogenesis. As an oven dried sample will be virtually sterile, an inoculum or seed should be added. This may be an 'off-the-shelf' seed such as a methanogenic leachate or anaerobically digested sewage sludge (ADSS) or a specially prepared inoculum. We favoured a laboratory maintained seed, as ADSS and leachate may be variable in methanogenic activity and also in the quantity of degradable organic matter they contain and hence their intrinsic gas potential. A laboratory-maintained seed, if maintained under constant conditions should provide a far more consistent inoculum, with high activity and low intrinsic gas potential.

We have maintained a laboratory 'seed culture' for several years and this has been fed on a constant basis with a simulated MSW medium. The culture is maintained in a continuously stirred anaerobic fermenter of 2 litres capacity fed semi-continuously with 100 mls per day of the refuse medium (200 mls Monday and Friday, none Saturday and Sunday). The medium is made of a mineral base with refuse solids selected from the domestic waste stream to include only the degradable items in approximate proportion to their occurrence in MSW. The gas production from this fermenter has proven to be remarkably constant as has its intrinsic gas potential when used as a BMP test inoculum. Table AB-2 details the composition of the medium used to maintain the BMP seed culture. This showed a remarkably steady biogas production of about 1000 $mls.d^{-1}$ over a period of approximately 3.5 years.

The seed culture was fed at the end of each working day to minimise the intrinsic gas potential of the seed prior to use. For use, 100 ml sub-samples of the mixed liquor, drawn from the culture were transferred to sealed centrifuge tubes and lightly centrifuged (5,000g for 10 mins.) before use of the supernatant to inoculate BMP test vials. This procedure enabled the inoculum to be injected into the vials via the butyl rubber stoppers without clogging of needles.

Removal of solids in this manner also helps to reduce the intrinsic gas potential of the inoculum. Only 1 ml of inoculum per vial (2%) was used in order to minimise the 'blank' gas production.

B.2.6 Gas analysis and quantification

Apart from the weight of sample used, other measurements to be recorded are the gas volume produced over the incubation period and the methane composition of the gas. It is necessary to know the volume of the test vessel headspace and the volume of excess gas produced. We have tried various methods to quantify the excess gas such as pressure measurement, upturned burettes and syringe measurement. We believe the syringe method in which gas is removed by syringe via a 3-way valve connected to an electronic manometer is the simplest method. The syringe plunger is drawn out until the gas pressure reaches barometric pressure and the volume is recorded from the syringe graduations. The arrangement is shown in Figure AB-2. This method was found to give the most repeatable results when compared to other methods. A correction is made for the barometric pressure which is recorded at the time of gas quantification to enable conversion to standard temperature and pressure.

Blank BMP tests are carried out alongside each batch of test vials to determine the methane production from the test inoculum and culture medium constituents. This blank value is subtracted from the test results to obtain the methane production from the sample alone.

The methane content of the gas is best determined by gas chromatography as this can be carried out on small quantities of gas with a high level of accuracy.

We used a two-column GC analysis with argon carrier gas, thermal conductivity detector and 0.5 ml sample loop to determine hydrogen, oxygen, nitrogen, methane and carbon dioxide. Columns were a 5 m Poropak N and 1 m molecular sieve. The advantage of detecting the five gases in the TCD method is that they potentially indicate problems of acid de-stabilisation or leaks indicating air ingress.

BMP calculation

The calculation of the BMP value is based on the measurements of the barometric pressure and gas volumes and composition recorded at the end of the incubation period where

b = the barometric pressure at the time of gas removal and quantification (mbars)

m = the methane content of the excess gas removed from the BMP vials (% by volume)

V_1 = the volume of the excess gas recorded at the time of removal at pressure b and 35°C (308°K) (mls)

V_2 = the volume of the headspace in the serum vials (mls). Measured to be 105 mls in the nominal 160 ml serum vials suggested. (Total vial volume = 155 mls - 50 mls liquid volume (culture medium + seed + reducing agent) = 105 mls headspace).

To calculate the BMP value at standard temperature and pressure (1 atmosphere, 273°K) the following equation is used.

The BMP value in mls methane/vial = $((V_1 + V_2) \times (m/100)) \times (273/308) \times (b/1013)$

The factors 273/308 and b/1013 are the corrections to standard temperature and pressure respectively.

Table AB-2. Composition of MSW medium used to feed laboratory BMP seed culture

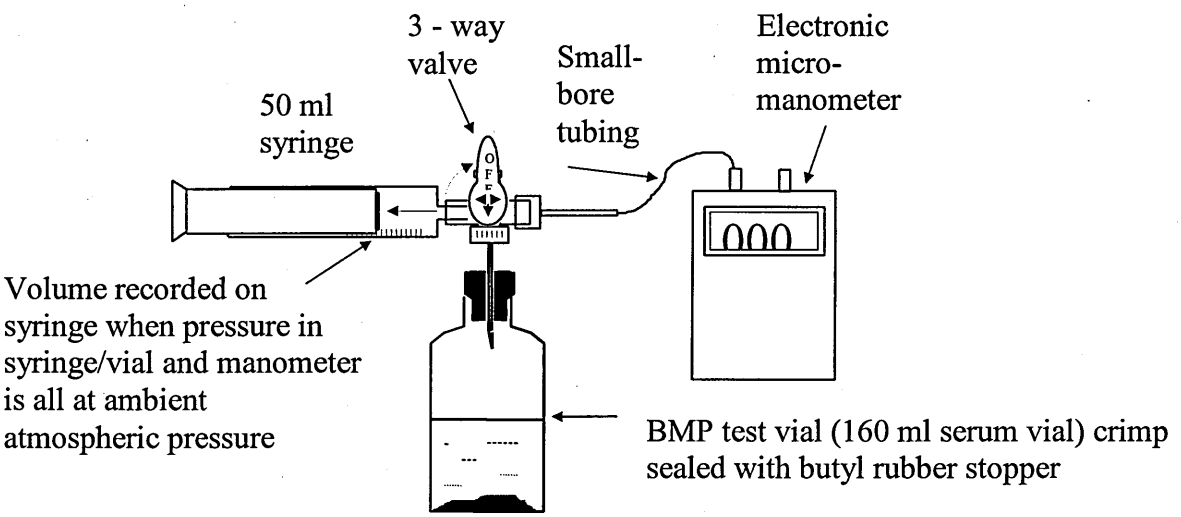
Part A - Complete medium recipe		
Component		Amount (g / litre)
$K_2HPO_4.3H_20$		3
KH_2PO_4		1
NH_4Cl		4
$MgSO_4.7H_20$		0.6
NaOH		1.5
$FeCl_2.4H_20$		0.1
$CaCl_2.2H_20$		0.15
Tryptose		0.1
Yeast extract		0.1
Dried ground (<1 mm) refuse (composition detailed in part B)		35.1
Tap water		to 1 litre
Part B - Composition of dried ground refuse		
Waste type		Amount (g / litre)
Paper products (72% of total)	Corrugated cardboard (20%)	5.04
	Writing / printing papers (25%)	6.3
	Packaging cards / boards (25%)	6.3
	Tissue papers (10%)	2.52
	Newspapers (20%)	5.04
Putrescibles (11% of total)		3.85
Garden waste (4% of total)	Lawn mowings (50%)	0.7
	Prunings (50%)	0.7
Textiles (7% of total)	Wool (45%)	1.1
	Cotton (45%)	1.1
	Silk (10%)	0.25
Wood (6% of total)	Softwood (50%)	1.1
	Hardwood (50%)	1.1
Total		35.1
The pH of the complete medium should be above 7, but if below this it should be adjusted to 7 by the addition of additional NaOH.		

The BMP values are calculated for each of five replicate vials for each test and the five 'blank' vials for the same batch. The mean values for the groups of five replicates are then used to calculate the sample BMP as follows.

The sample BMP value in m^3 methane.tonne dry $weight^{-1}$ (or $mls.g^{-1}$) = (Mean BMP of test vials - Mean BMP of blank vials) x 2

The factor of x 2 at the end arises from the use of 0.5g of sample per test. This will be x 4 if 0.25g sample/vial is used. The deduction of the blank value corrects for any contribution to the methane production from the seed and organic constituents of the BMP test medium. The blank value also gives an indication of the activity of the seed and acts as an internal standard. In our experience the blank value when using the laboratory prepared seed culture detailed in Section 2.5 is very consistent, amounting to 2.97-4.01 mls methane/vial.

Figure AB-2. Arrangement used for quantification and removal of biogas from BMP test vials



Values may be reported as m^3 methane.tonne dry $weight^{-1}$ or converted to a wet weight basis if sample water content has been determined. The detection limit of the basic technique as described is about 2 m^3 methane/tonne dry weight. However this can be reduced by using more sample per test vial if it is known that the samples have a very low BMP value. If this is done, adjustments to the headspace volume used in the calculation should be made to account for the extra volume of the sample.

For some soils we have used 5g sample per vial to give a detection limit of approximately 0.2 m^3 methane.tonne dry $weight^{-1}$.

B.3 INTERFERENCES

Potential interferences may occur with the BMP test procedure when high concentrations of certain compounds are present in the test sample. This may be due to inhibition of the methanogenic bacteria or stimulation of other anaerobic decomposition pathways such as nitrate or sulphate reduction.

We carried out tests to assess the potential effects of high sulphate and nitrate concentrations in samples on the BMP results obtained. These are shown in Table AB-3.

The results obtained using the BMP test with standard substrates indicated that interferences occurred at sulphate contents above 0.005g SO_4 per vial (100 mg $SO_4.l^{-1}$) and nitrate contents above 0.005g NO_3 per vial (100 mg $NO_3.l^{-1}$). The interference appeared progressive with sulphate perhaps indicating competition between sulphate reducing bacteria and methanogens but abrupt with nitrate indicating an inhibitory threshold.

Table AB-3. Effect of nitrate and sulphate on BMP test results - Standard substrates used were 1.Ashless cellulose floc (Whatman) and 2. Microcrystalline cellulose (Aldrich Chemicals)

Addition per vial	BMP value (m^3 CH_4.tonne dry weight^{-1})
0.5g Floc[1]	332
0.5g Floc+0.0005gSO_4 as $MgSO_4.7H_2O$	333
0.5g Floc+0.005gSO_4 as $MgSO_4.7H_2O$	331
0.5g Floc+0.05gSO_4 as $MgSO_4.7H_2O$	172
0.5g Floc+0.1gSO_4 as $MgSO_4.7H_2O$	138
0.5g Floc+ $MgCl_2.6H_2O$ as Mg control at highest Mg level used with SO_4 additions	258
0.25g microcrystalline cellulose[2]	335
0.25g microcrystalline cellulose+0.0005g NO_3 as KNO_3	338
0.25g microcrystalline cellulose+0.005g NO_3 as KNO_3	331
0.25g microcrystalline cellulose+0.05g NO_3 as KNO_3	-16 (<blank value)
0.25g microcrystalline cellulose+0.1g NO_3 as KNO_3	-20 (<blank value)
0.25g microcrystalline cellulose+KCl as K control at highest K level used with NO_3 additions	330

B.4 SPECIFIC TEST RESULTS

During the course of the development of the BMP method described in this paper, a number of different samples have been tested using the standard procedure. These included specific waste types, sorted waste fractions from the municipal waste stream and specific test materials dried in different ways. Tables AB-4, AB-5 and AB-6 detail some of the results obtained from these tests.

Table AB-4. BMP test results obtained for specific materials isolated from the municipal waste stream at source

Material	BMP value (m^3 CH_4.tonne dry $weight^{-1}$)	Material	BMP value (m^3 CH_4.tonne dry $weight^{-1}$)
Newspapers	93	Hedge prunings	91
Lawn mowings	235	Cotton	225
Tissue paper	249	Wool	62
Corrugated cardboard	183	Silk	0
Hardwood	158	Packaging card	209
Softwood	14	Writing paper	236

Table AB-5. BMP test results obtained for sorted fractions of the municipal waste stream, separated after collection.

Results from two separate waste sorting exercises, 1 and 2, carried out on separate dates.

Sorted waste fraction	BMP value (m^3 CH_4.tonne dry $weight^{-1}$)	Sorted waste fraction	BMP value (m^3 CH_4.tonne dry $weight^{-1}$)
Textiles (1)	142	Fines (<10 mm) (1)	84
Textiles (2)	94	Fines (<10 mm) (2)	7
Paper (1)	127	Putrescibles (1) inc. garden	94
Paper (2)	167	Putrescibles (2) inc. garden	16
Wood	28		

Table AB-6. The effect of different drying methods on the BMP values obtained from certain materials arising in the putrescibles fraction of MSW (as supermarket purchased, except grass)

Sample type	BMP (m^3 CH_4/tonne dry weight)			
	FD	50°C	105°C	Microwave
Flour	336	335	320	
Bread	319	297	326	313
Potatoes	311	309	274	260
Vegetable oil	8	7	6	
Grass	244	255	231	
Cabbage	253	271	180	
Chicken	265	281	222	

FD= freeze dried, 50°C = oven-dried at 50°C, 105°C = oven-dried at 105°C, microwaved = dried in microwave oven. 0.25g sample used for flour, bread, potatoes and vegetable oil, all others 0.5g.

The test results indicate that this BMP test can be used to compare effectively the methane potential of selected components of the waste stream as well as mixed MSW samples. Analysis of sorted waste fractions indicated considerable variability between similar fractions obtained at different times. The marked difference between the fines and putrescibles results was believed to relate to increased garden waste in sample 2 including soil which entered the fines fraction.

Different drying methods appear to influence the test results in some cases and this effect is believed to be primarily relevant to unprocessed foodstuffs or fresh vegetable matter.

Surprisingly low results were noted for the vegetable oil and softwood samples perhaps indicating a mixing difficulty with oils and possibly an influence of resins in some woods.

B.5 CONCLUSIONS

This paper has considered the various steps in a BMP test methodology and described the authors' favoured approach with details of a method used extensively to test landfilled MSW samples, sorted waste fractions or specific components of the waste stream. Potential problems have been identified with samples containing high levels of sulphate and nitrate and the drying methods applied to fresh foodstuffs and vegetable matter can influence the results. These problems are only expected to be relevant when testing specific industrial waste samples or highly specific sorted waste fractions rather than mixed MSW samples such as from landfills.

B.6 ACKNOWLEDGEMENTS

The authors are grateful for the support of the Environment Agency who funded this study which was undertaken at Minton, Treharne and Davies Ltd., Cardiff, UK. However, the opinions expressed in this paper are those of the authors and do not necessarily reflect those of the UK Environment Agency or Government. We are also grateful to Shanks Waste Services provided the sorted waste fractions and funded the analysis detailed in Table A2-5.

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Proceedings Sardinia 2003, Ninth International Landfill Symposium, S. Margherita di Pula, Cagliari, Sardinia, 6-10 October 2003, Paper 417.

APPENDIX C: METHANE PRODUCTION, EMISSION AND CONTROL DURING MSW LANDFILLING

D.L. BARRY*, R. SMITH**, R.G. GREGORY *** & C. HARRIES****

* *Atkins Environment, Epsom, UK*

** *Environment Agency, Risk & Forecasting, Reading, UK*

*** *Land Quality Management, University of Nottingham, UK*

**** *Waste Technical Services, Cardiff, UK*

SUMMARY: Until quite recently the major emphasis on landfill gas control has related to 'post-completion' of landfill cells or phases, rather than during active waste disposal operations. Increasingly, odour control needs have changed that pattern through *ad hoc* capping and gas collection systems but, to date, there has been little quantitative information available on the scale of methane emissions from the commencement of waste deposition. This applied research project involved extensive surface flux emission monitoring carried out on 21 operational UK landfills. Main findings were that (a) the onset of methanogenesis appears to occur within about 2 months of waste placement and is well-established, with methane at least 40% by volume, after about 6 months; and (b) surface methane flux emissions were detectable within about 1 month after waste placement. Additionally, surface flux data showed that (c) the emission rates from waste side slopes were much greater than from top surfaces, and (d) areas close to landfill edges can have the highest emission rates. Both of the latter findings confirm the high lateral (as

opposed to vertical) permeability of landfilled wastes. Moreover, emission rates from top surfaces of waste do not appear to increase significantly with age. The average surface flux rate appeared to peak around 20-24 months (following initial waste placement) at about 1mg. $m^{-2}.s^{-1}$, but this result could be influenced by the particular conditions at the relevant sites. Nonetheless, the flux rate is some 100-times greater than the proposed UK limit for emissions from temporarily capped sites. The main zones where effective reductions can be made in emissions are the waste side slopes and landfill edges. Controls in such areas should be based on horizontal rather than vertical collection systems, reflecting the greater lateral permeability of wastes; such systems would also be more compatible with on-going disposal operations by virtue of minimal disruption to working practices. These control networks could be integrated in due course with permanent gas collection systems for energy recovery.

C.1 PROJECT BACKGROUND

As part of the UK Climate Impacts Programme, the UK Government is committed to reducing the overall emissions of greenhouse gases in accordance with internationally agreed targets. The Environment Agency is developing a strategy for emissions-based regulation of landfill gas in order to minimise global impacts of methane and local impacts on health, environment and amenity. This strategy includes guidance on the proposed introduction of a surface methane emissions protocol for permanently or temporarily capped landfills (Environment Agency, 2002).

C.2 INTRODUCTION

Historically, the pre-completion stages of waste landfilling in cells have not usually been subject to any active gas controls, except where local odour controls have been required to reduce impacts on sensitive neighbouring areas. This situation is changing, partly due to

implementation of the Landfill Directive which requires increased collection and treatment of landfill gas and changes in the waste composition that may result in significant changes both in the generation and constituent components of landfill gas.

This paper follows a preliminary paper presented at the 2001 Sardinia conference (Barry *et al* 2001) and reports a major applied research project addressing methane emission during the period following waste placement, focusing on the results of extensive surface flux monitoring carried out on 21 operational UK sites. It highlights the scale of emissions that can occur in the period before 'normal' gas controls are introduced following the completion and capping of particular landfill phases (which usually comprise several cells and take several years). This paper also highlights the findings relating to the time taken for the onset of methanogenesis and when surface methane fluxes become detectable. Lastly the paper addresses the practical options for the targeted control of such surface emissions prior to the introduction of the longer-term post-completion gas abstraction systems used mainly for energy recovery purposes.

The project was funded jointly by two approved UK environmental bodies, Biffaward and *shanks first* fund, in collaboration with the Environment Agency for England and Wales. The authors kindly acknowledge the permission of the project funders to present this paper. The opinions expressed are those of the authors and do not necessarily reflect those of the funding bodies. An R&D Technical Report detailing the full study will be available shortly.

C.3 SITES MONITORED

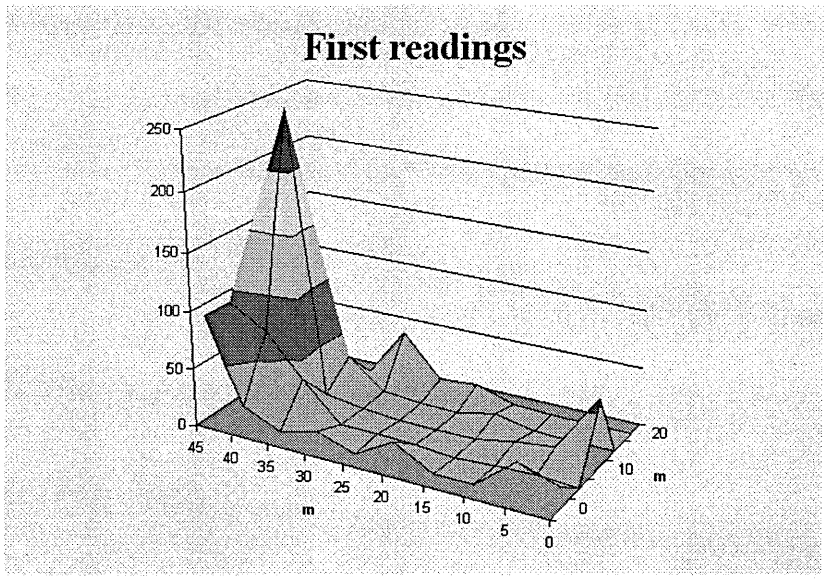
Flux was monitored at 21 sites (on 32 occasions), with repeat visits made to seven sites, resulting in a total of nearly 650 sets of data. (A further 140 data sets were collected but were discarded for various reasons due to potential unreliability.) The sites varied in age and waste composition, and were situated in various parts of the UK that had different meteorological conditions. The MSW content of the sites varied from about 55% to 100%, and the waste depths

at the time of monitoring ranged from about 5m to 40m. The age of the wastes (as defined by the 'oldest' waste in the cell) at the time of monitoring ranged up to about 28 months. Overall, monitoring positions were usually set on some geometric pattern that covered either the top (or horizontal) waste surface (360 positions), or the waste side slope surface (about 240 positions), or, in some instances, near the edge of the landfill (about 50 positions). The emphasis on measuring side slope emissions was based on the early project finding that the emission rates from such slopes appeared much higher than from the corresponding top surfaces. Also, industry experience highlighted that some landfill edges had very high emission rates.

C.4 MEASURED FLUX EMISSION RATES

C.4.1 Introduction

As discussed in the 2001 paper (Barry *et al* 2001), taking flux measurements on operational sites with daily cover is fundamentally more difficult than on sites with final/temporary capping due to (a) very uneven surfaces by virtue of the inherent nature of the temporary waste cover, and (b) the far greater scale of temporal and spatial variations in fluxing conditions (Figure AC-1). Accordingly, any measurement method can be seen as being more relative than absolute in its accuracy.



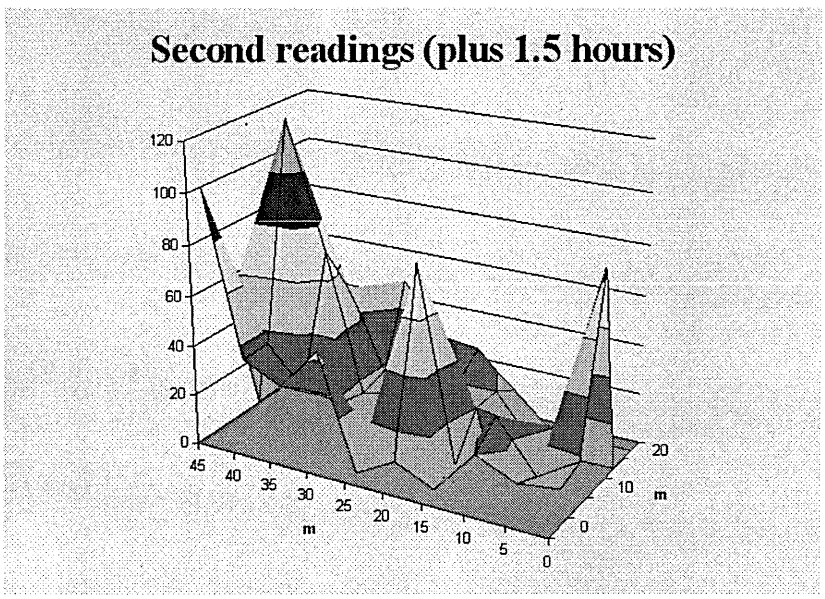


Figure AC-1. Example of spatial and temporal variations in methane surface concentrations*

* *Concentrations measured just above the waste surface (ppm)*

C.4.2 Main findings

Collected flux data were examined statistically with respect to many factors and the most significant correlation was found to relate to waste age, though this also has a close relationship with waste depth. Figure AC-2 shows the basic flux rates measured for the 'top' and 'slope'

surfaces of all the individual landfill sites, with sites grouped in age sequence (age, in this case, represents the 'average' age of the waste *e.g.* 10 months represents a waste mass that varies between 0 and 20 months old). The data show that even with sites of apparently similar ages, there are significant differences in their measured emission rates.

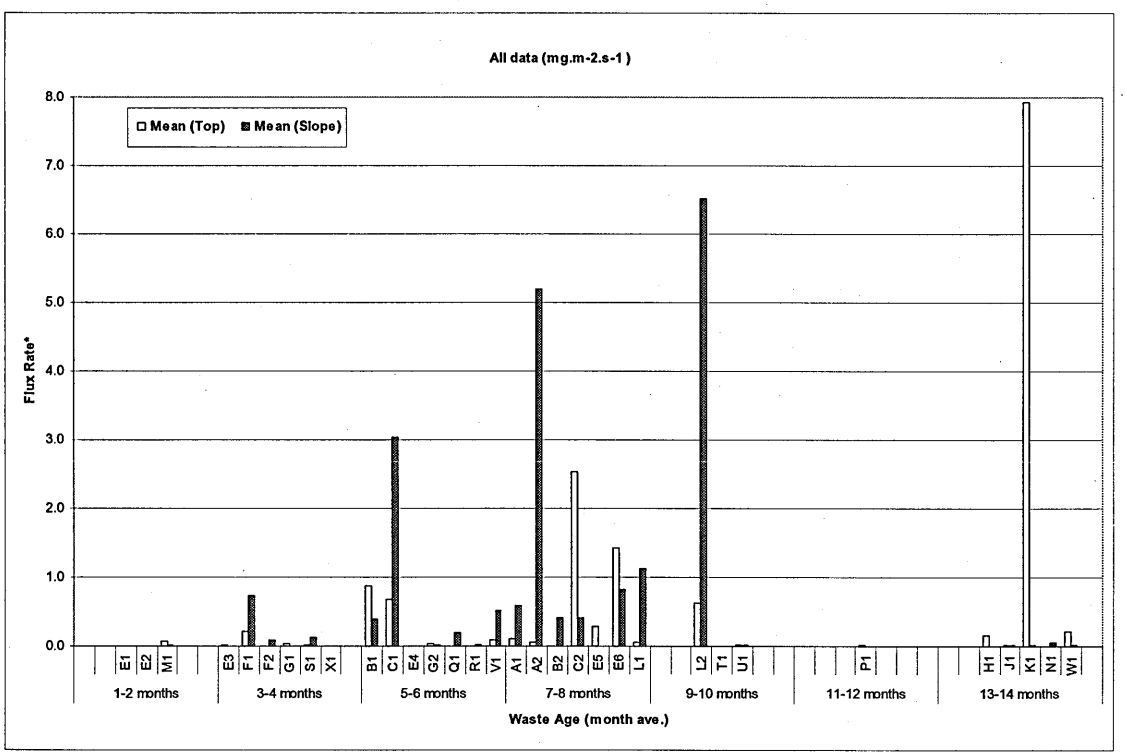


Figure AC-2. Individual site surface flux rates (averaged) for top surfaces and slopes

Figure AC-3 shows the same basic information but with the sites condensed to 'age groups' (of 2 months 'average' spans).

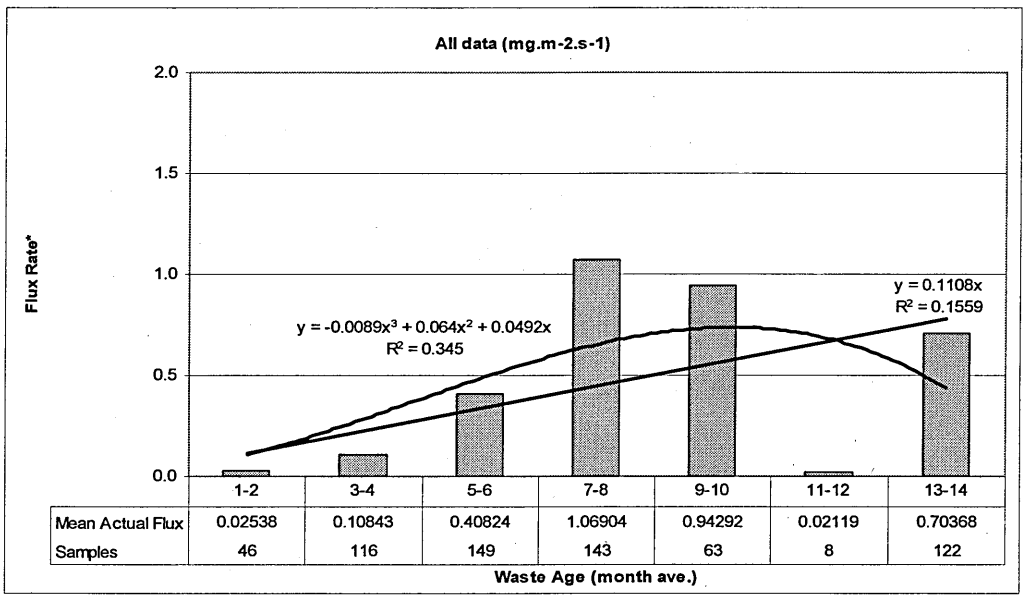


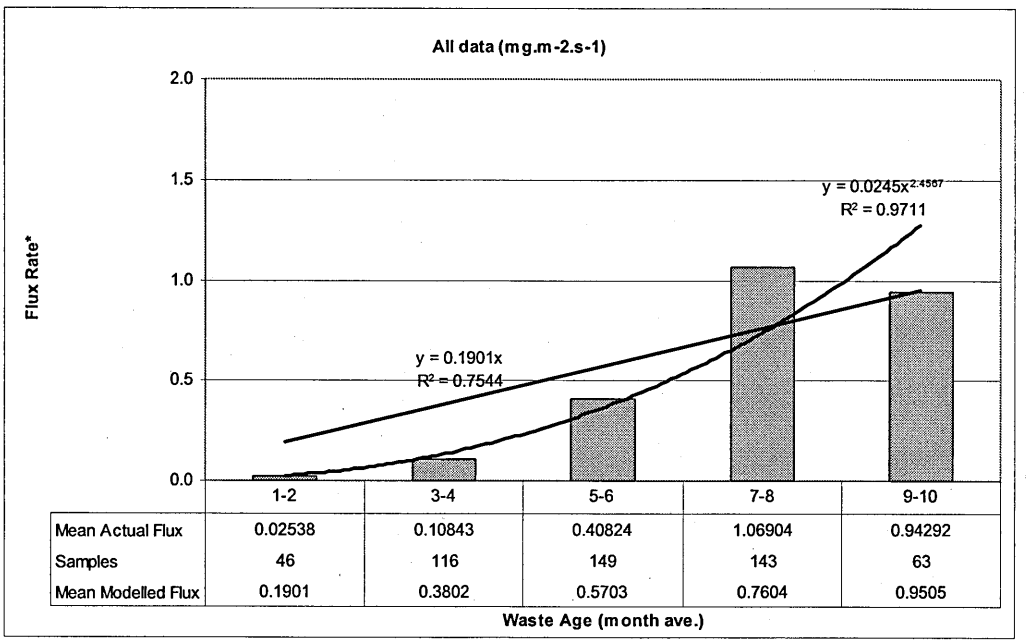
Figure AC-3. Mean surface flux rates for sites grouped by age to 14 months

The results show that the overall flux rate from sites older than about 10 months 'average' (*i.e.* more than 20 months since waste placement commenced) has slowed down and even reduced by the time the waste is some 14 months old on average (*i.e.* 28 months after placement). Indeed, as can be seen from Figure AC-3, the flux rate in the 13-14 month age group (average) would be much reduced if the data from the single most gassing site monitored on the project was discarded (*i.e.* Site K1 in Figure AC-2). However, the number of data sets (122) in that 13-14 month age group represents about 20% of the total database, and so must be considered significant. On the other hand, the effective lack of data for the 11-12 month age group complicates the interpretation of the flux pattern from the wastes older than about 10 months (average).

The cubic regression 'best fit' line shown on Figure AC-3 shows a good correlation with the flux/age data, while a similar best fit line showed a comparably good correlation with the flux/depth data. Thus, taking both the apparent maximum flux rates from the age and depth assessments, and the known relationship between age and depth for the monitored sites, it was

concluded from the monitoring data that the time at which the maximum (average) flux is reached is likely to be about 20-24 months after commencement of waste placement.

Although the reason for the apparent reduction in surface emission rate after this time is not known, this conclusion does not affect the findings of this project which is primarily aimed at establishing the emissions during the earlier part of the landfill life. Thus, the emission assessment can be concentrated on the more explicit flux patterns measured from the 'younger' wastes, *i.e.* up to 10 months (average) age. Indeed, when the flux data for this period are examined (Figure AC-4) it can be seen that a 'best fit' line with a very high correlation (R^2 = 0.9711) can be defined.



	1-2	3-4	5-6	7-8	9-10
Mean Actual Flux	0.02538	0.10843	0.40824	1.06904	0.94292
Samples	46	116	149	143	63
Mean Modelled Flux	0.1901	0.3802	0.5703	0.7604	0.9505

Figure AC-4. Mean surface flux rates for sites grouped age to 10 months

C.4.3 Differential emission rates

An evaluation of the data amplified the well-known complexity of gas generation and emissions from an operational landfill. Not only were there considerable spatial differences between

monitoring positions but it was also noted that flux rates near to adjacent capped waste cells were higher than positions further away. Whilst such findings indicate changes in emission patterns, the database covering these aspects is not robust enough to deduce a meaningful quantification of the observed effect.

Similarly, data taken from near the edges of three landfills (all of which had containment membrane barrier systems) showed emission rates several orders higher than for surface locations further away from the edge. Thus, the conclusion was drawn that there is a preferential pathway at the edges of a site such that the 'edge' behaves like an open side slope except that it concentrates the flux into a relatively narrow surface strip. It was considered inappropriate (due to the relatively small data base) to suggest the width of such a zone from the database collected.

The emission rates from side slopes (and edges) were demonstrated to be considerably higher than for top surfaces, a finding that is considered to reflect greater lateral gas permeability. Thus, although the top surface flux rate might be relatively low, on most sites the top surface is much greater than the slope surfaces. However, that ratio changes with increasing waste height, as addressed below. Figure AC-5 shows the principal zones of surface emission on a typical landfill.

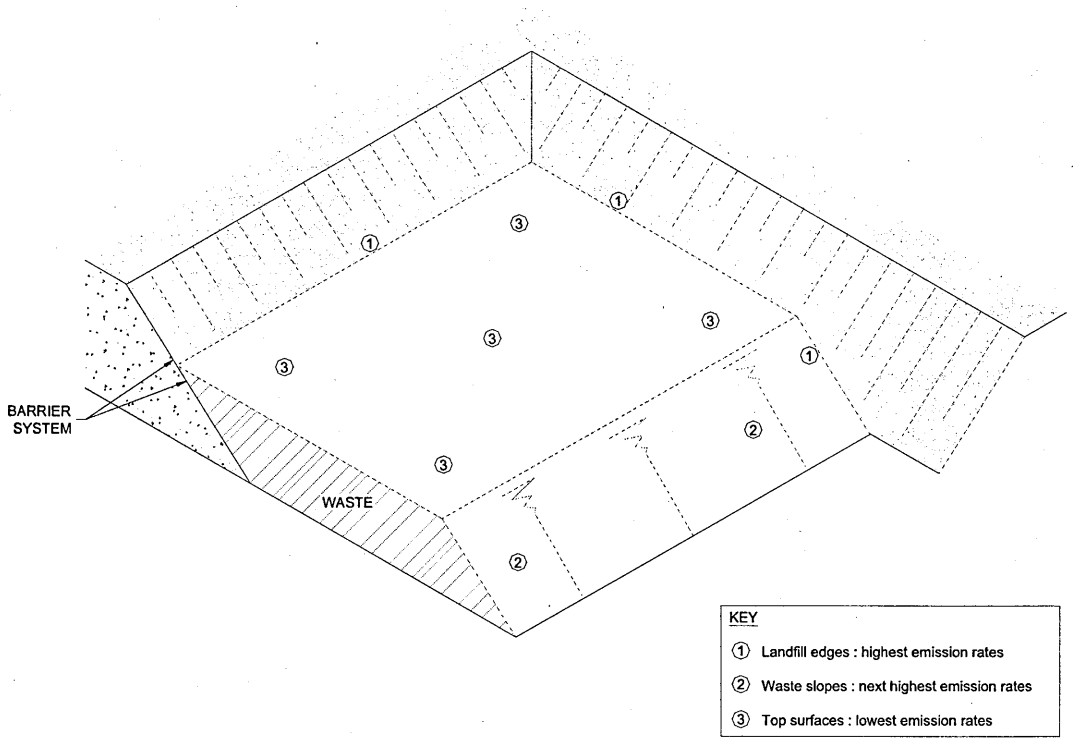


Figure AC-5. Main surface emission zones

C.5 OVERALL EMISSIONS

The emission rates from operational sites were found to be highly variable and ranged over a spectrum covering some 9 orders of magnitude. Converting all these data into an overall emission volume for a particular operational site (which, by definition, varies constantly in age and shape) is very complex. However, in order to assess the potential overall scale of emission volumes from typical landfills, a series of hypothetical landfill scenarios was assessed involving waste cell areas from 1ha to 4ha, and some 25m deep. The assessment related to the placement

of 500,000m^3 waste at different filling rates, with different flux emission ratios between top and slope surfaces that varied from 1:2 to 1:10.

This simplified assessment of total flux emissions, showed that, for the chosen landfill geometry, (i) slope surfaces could amount to about 50% or more of the total surface emissions, and (ii) placing waste in smaller cells can result in considerably lower overall flux than when larger cells are used. This apparent benefit from using smaller cells was shown to decrease with an increase in the slope/top emission ratio, a change that amplifies the importance of slope surface areas in overall flux emissions. Also, as might be expected, faster filling rates can significantly reduce the total flux emissions simply because it can be assumed that gas controls can be introduced at a comparatively early date, *i.e.* soon after completion of waste placement.

To reiterate, it is recognised that methane emission assessment through surface flux monitoring is likely to underestimate significantly the actual scale of emissions. Accordingly, it is considered that the monitoring data, which show clearly that not only are there high average emission rates but that the emission rates from slopes and landfill edges can be very high, confirm that a considerable proportion of gas has escaped to atmosphere before 'conventional' gas controls have been installed.

C.6 ONSET OF METHANOGENESIS

The time period for onset of methanogenesis was assessed through monitoring the gas regime at one particular site over a 16-month period. This involved the installation of ten probes and two long perforated pipes at three different layers as waste disposal operations progressed. Concentration data for oxygen, carbon dioxide and methane were measured at about monthly intervals. Surface flux measurements were also taken on six occasions (the data from which were included in the main flux database discussed earlier).

Figure AC-6 shows how the concentrations of the three main gas components varied over time and followed the 'classic' patterns (note that the concentration lines were forced through the respective 'origins' by introducing 'dummy data'). Although, due to operational difficulties, some of the data from the 'lower' layer of monitoring points were ultimately considered unreliable, there was a clear pattern that the oxygen levels became depleted and methane/carbon dioxide ratio increased to >1 after about 5-6 months. Further, some measurable surface methane flux rates (6.21×10^{-3} $mg.m^{-2}.s^{-1}$, or about 6 times the proposed Environment Agency emission standard for completed landfills) were recorded within about 1 month of waste placement. It is considered that this scale of surface methane fluxing confirms that the driving mechanisms for surface flux involve both advection and diffusion processes; in other words, the carbon dioxide (>50%) dominated early gas appears to be 'carrying' the methane.

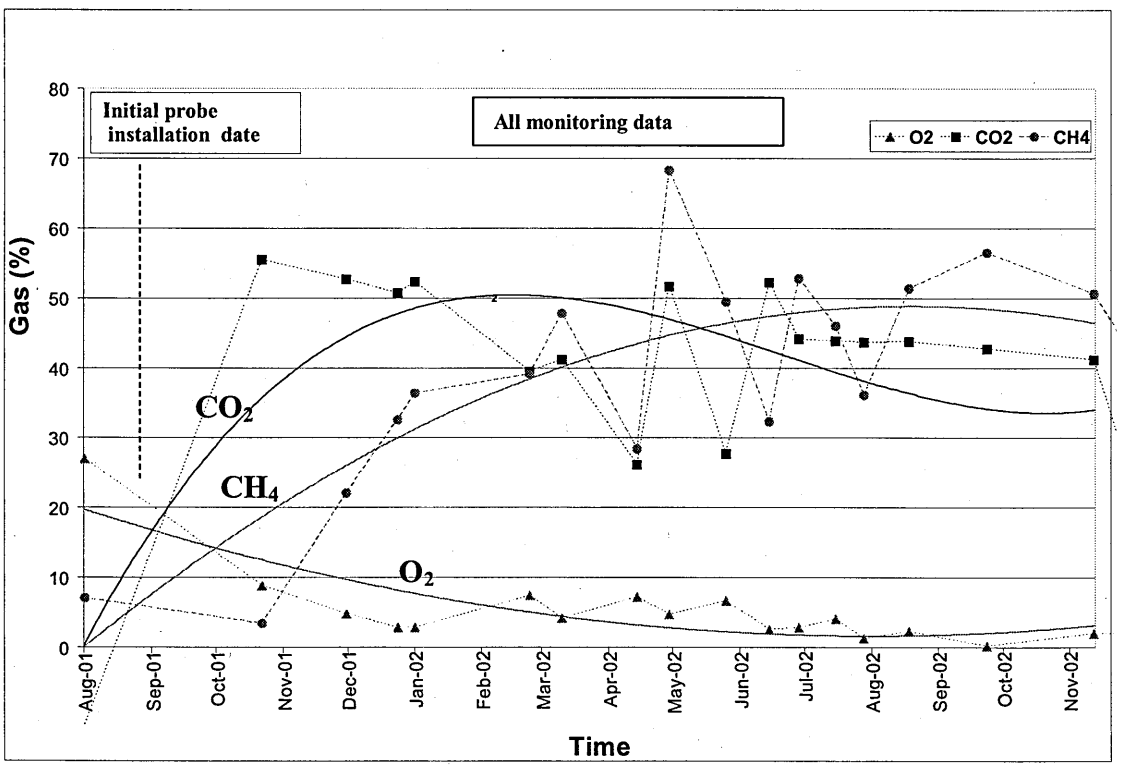


Figure AC-6. Onset of methanogenesis: gas concentration profiles in probes with associated trendlines

C.7 GAS CONTROLS DURING WASTE DISPOSAL OPERATIONS

C.7.1 General current practice

As indicated earlier, during the past few years gas emission controls for active waste disposal areas have been installed largely for odour abatement purposes. In all cases, however, there are no established best practice guidelines and so current systems have evolved largely by trial and error, using knowledge gained from conventional gas abstraction and collection systems. In terms of adapting existing 'odour control' systems, the most likely option is considered to be a horizontal system, possibly connected to a vertical system, whether now or at some future date when the relevant part of the landfill site has been completed. This design choice does not pose any significant operational difficulties (in contrast with vertical wells) and is also less prone to physical damage.

The main types of horizontal systems used to date involve either perforated pipes or high permeability gas pathways, or both, and are usually constructed in a trench excavated into a waste layer. Pipes are normally laid in 'rock-filled' trenches in order to support and protect the pipe; the backfill material also acts as a gas pathway. Further, this backfill medium can be also be particularly important in reducing the risks of the pipework becoming 'water-logged' by perched leachate conditions, a factor that is fundamental to the effectiveness of the abstraction system. It is recognised that while larger 'high permeability' trenches provide a higher efficiency of gas collection, these trenches can also consume 'expensive' void space (quite apart from the materials and operational costs for their installation).

The typical vertical frequency of such systems is about every 5-8m (or about every 2 to 3 waste lifts), with pipes/pathways placed at 20-25m intervals. This differential horizontal/vertical spacing reflects the different permeability values in the wastes. This factor was established both

in this project and in US studies (Lofy, 1996) which suggest that suction influence is 7-8 times greater in the horizontal direction than in the vertical direction.

Whatever the overall design layout, the final 10-15m section of the pipe/pathway system approaching an open waste face usually comprises a solid pipe so as to minimise the risk of air ingress to the wastes, and the consequent risk of inducing combustion through enhanced oxidation. This risk is seen as being very significant and so quite a conservative stance is taken in the design and operation of such systems.

In some instances the control system involves a combination of both horizontal and vertical pipe networks. One site that employs this method has constructed 150m long x 1m x 1m trenches, dug into fresh waste at 6m vertical spacing. The trenches, which were laid out in a regular grid 40m pattern, were filled with building rubble and car tyres prior to being capped with clay.

C.7.2 Potential future design principles

Taking account of the study findings and recognising potential implications for operational practices, it is considered that the most cost-effective controls are likely to focus on emissions from the waste side slopes and from the landfill edges (Figure AC-7). While both of these areas might be relatively modest in comparison with the top surfaces, the emission rates can dictate that they represent the most significant volume of emissions.

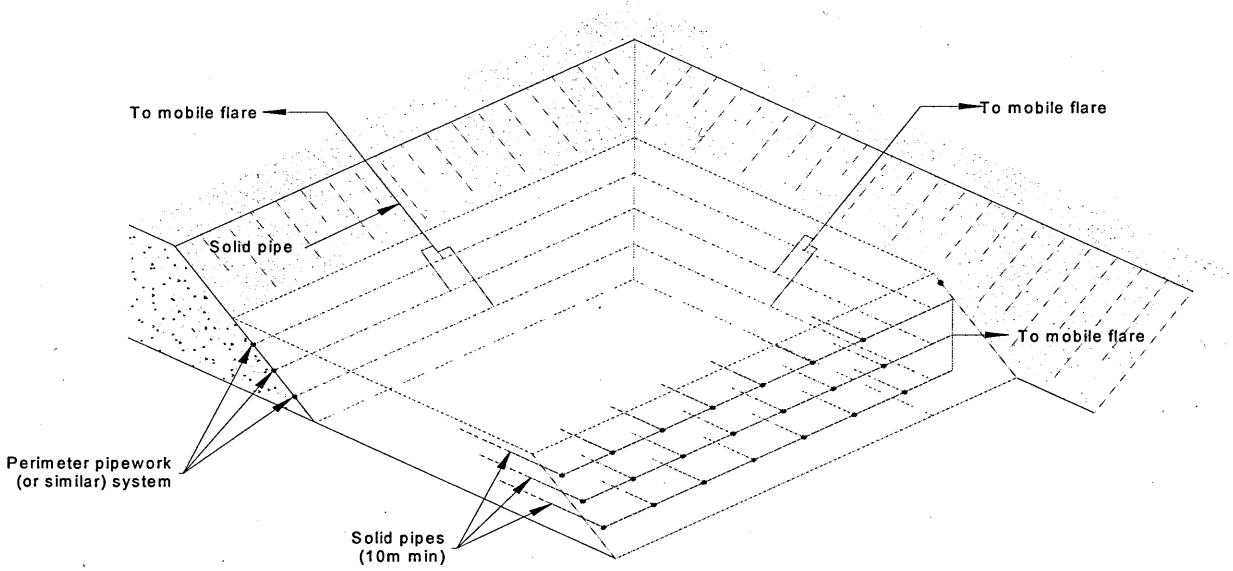


Figure AC-7. Potential operational cell design for optimising methane emission control

In the case of the side slopes, the emphasis should be on 'intercepting' the gas before it reaches the slope and so, in theory at least, the gas collection system need only extend into the waste only, say, 20-30m (with the first 10m being of solid pipe). The benefits of extending it further are unlikely to be significant because, whatever the length, the suction will be most effective close to the point of application. For cell edges, the gas control measure can be in the form of a perimeter perforated pipe/pathway, connected to the surface by solid pipe at regular intervals (say 50m). In both cases the suction should not be applied until at least 5m of waste has been placed on the relevant pipe network. For edge controls, the deeper pipe system will probably continue to be necessary with the increasing waste height because the volumes of gas will inevitably increase.

It is considered that any attempt to control top surface emissions is unlikely to be cost-effective. This is because the monitoring data suggest that such emissions can be as much

related to the shallower wastes as to the deeper wastes and controlling shallow gas sources carries a risk of induced combustion.

C.8 CONCLUSIONS

Field data has reinforced the extreme complexity in gas generation and surface emissions rates, rates that can also be affected by a wide range of variables, both physical (such as daily cover) and meteorological (such as rainfall and pressure). The potential effect of each variable was not discernible although an attempt was made to assess if data from apparently similar sites showed any correlations with prevailing conditions. Despite the range of complexities, the collected data clearly showed a progressive increase in the overall surface flux rates and that this increase was principally manifest on side slopes and landfill edges, with the top surface showing relatively little increase with age. This finding highlights the areas where the greatest benefits can be gained from the introduction of gas controls, bearing in mind, however, that the lower flux rates can apply to proportionately high areas of an active site.

The project confirmed that methanogenesis effects can be evident after only 1-2 months and that surface methane flux can be detected in a similar timescale, a process that is considered to reflect both advection and diffusion mechanisms. However, from monitoring the series of *in situ* probes it was not apparent that the gas flux from lower waste levels affected the rate of methanogenesis in the upper, shallower waste layers. This reinforces the conclusion, already established elsewhere (Lofy, 1996), that lateral gas permeability is far greater than vertical permeability in a landfill.

The maximum flux rate appears to be reached about 20-24 months after commencement of waste placement, with a reduction in rate after that point. However, the data should be seen as

being an underestimate of the actual flux rates and so the time taken to reach a specific flux rate is likely to be less than that suggested by the data.

The relatively high rates of average flux after about 12 months (actual time), which were measured at some 40-times the proposed UK standard for temporarily capped sites would appear to justify the need to activate control measures after such a period. By focussing any controls on the key emission areas, the system can be optimised whilst having a minimal effect on operational activities and being readily incorporated into long-term systems.

C.9 REFERENCES

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APPENDIX D: COST BENEFIT ANALYSIS FOR ENCLOSED LANDFILL GAS FLARING

ASSESSMENT OF LIKELY COSTS AND BENEFITS

Proposal: *ENVIRONMENT AGENCY LANDFILL GAS FLARING POLICY EAP/LFG/002*

Proposed Policy:

The Agency requires:

(a) that no more 'open' flares shall be installed at licensed landfill sites, except for experimental or emergency purposes

(b) that all currently operational landfill gas flares operated as 'open' flares at licensed landfill sites shall be replaced progressively with 'enclosed' flares, or non-combustion techniques offering equivalent performance, by 31st December 2003 (the Agency will prioritise sites which produce large amounts of gas or pose a significant risk to the local environment).

(c) that all existing 'enclosed' flares operating at licensed landfill sites shall demonstrate operational performance required to meet the prescribed emission standard.

Baseline: Business as Usual

Assessment of likely benefits and costs of proposed approach to:

(note: does not need to be quantified in £s, although if available, this information should be included)

The Environment:

The proposed policy will have both direct and indirect environmental benefits. It is believed that more controlled combustion conditions within enclosed flares as opposed to open flares allow greater destruction of trace compounds with a potential to cause local and near-field air pollution and health impacts. Enclosed flares may offer some visual impacts benefits as there is no visible flame, but open and enclosed flares are not considered to differ significantly in terms of noise, landscape impacts and planning issues.

The indirect benefits, which can be realised through improved monitoring of emissions from landfill gas flaring, will be significant. The design of open flares prevents monitoring of emissions from gas flaring. Enclosed flares provide conditions under which emissions from flaring can be monitored, providing the data required for the understanding, control and regulation of potential pollution from landfill gas flaring.

Society:

Better information on emissions from landfill gas flaring should benefit all actors- industry, government and the agency, NGOs and other groups through informing measures to reduce potential environmental and health impacts from flaring emissions. Improved information

should be available to address the concerns of people in the vicinity of landfill sites with flaring on emissions of compounds with potential or perceived health and environmental impacts.

The Operator:

For operators this policy ensures that their competitors are applying best practice in the flaring of landfill gas. The monitoring data will provide operators with information to improve the siting of landfill sites, to control potential pollutants and for communication with the public.

The additional capital costs for the installation of an enclosed flare as opposed to an open flare are in the range of £46,000 to £65,000 per flare, to meet the minimum performance standards required by the Agency (Table 1). These additional costs are assumed to be the same for new installation and retrofitting.

Table 1: Capital cost of 'Open' and 'Enclosed' Landfill Gas Flares:

Capacity - landfill gas m^3/hour	Cost of Open Flare £	Cost of Enclosed Flare £	Additional cost of installing an enclosed flare
250-300	16000	62000	46000
500-600	20000	67000	47000
1000	23000	78000	55000
1500	29000	94000	65000

Source: Based on manufacturers' quotes, 1999

Note: Costs for enclosed flares meeting Agency's requirements for combustion performance: minimum retention time of 0.3 seconds at 1000°C

Costs include equipment, delivery and installation, but not complex control technology.

The policy applies to new landfill sites and those landfills with flares installed. Landfill gas flaring is not feasible for all landfill sites. Flares tend to be installed in those sites with large capacities and with sufficient expected revenue streams to cover the costs of installation and operation, with the aim of achieving the greatest environmental benefit in terms of reducing the impacts of landfill gas emissions without jeopardising operators' viability and ability to meet other existing environmental requirements.

Of the 3500 or so landfill sites in England and Wales, it is estimated that around 350 have landfill gas flares in operation. It is estimated that around 15% of these flares are 'enclosed'.

Under the proposed policy, about 300 sites will be required to replace existing open flares with enclosed flares by the end of 2003. The costs to operators will include the capital costs of the new equipment (Table 1), but also any costs associated with replacing existing flares before the end of their expected life. At present, there is no information compiled on the age and expected replacement timetable for existing landfill gas flares. On the basis of consultation with flare manufacturers and service providers, the expected lifetime of a flare is estimated at around 5 years. It has been assumed for the purposes of this simple assessment that existing flares will be replaced as they reach the end of their 'normal life'.

It has also been assumed for simplicity that existing flares will be replaced at an even rate of 60 flares per year from 1999 up to the end of 2003. This implies additional costs for the replacement of existing flares of the order of £3.2 million per annum for the 5 years up to

December 2003, or a total capital cost of around £14 million in 1999 prices (using a discount rate of 6%).

The cost for new landfills will depend on the number of new sites per annum. If it is assumed that between 5 and 10 new landfills are opened each year, the additional costs installing enclosed flares in place of open flares for new landfills will be in the order of £0.25 million to £0.55 million per annum.

Operators will also have monitoring costs, with a maximum estimated cost of £10,000-15,000 per annum per site for a full analytical suite under routine monitoring (where no problems are encountered with emissions).

The manpower requirements for the operation of enclosed flares do not differ significantly from those for open flares. The technology, design and input requirements are also similar.

The Agency:

The proposed policy is required to allow the Agency to apply its duty to protect human health and the environment through the licensing of landfill sites. The policy for the installation of enclosed flares will allow the Agency to monitor emissions from landfill gas flares, to allow control of emissions from landfill gas flares to be improved towards the level achieved for other combustion processes. This will allow the agency to determine whether and where reductions in pollutant emissions from flares are required, and contribute to improved risk-based regulation. It will also enable the Agency to move towards greater consistency in licensing.

Improved information on emissions from landfill gas flaring should enable the Agency to respond to public concerns about emissions from flares and improve the Agency's credibility in addressing potential and perceived impacts of trace components in emissions from flares. Concerns about dioxins are expected to become a significant issue following the publication of the USEPA report on dioxins (In particular, data from monitoring of enclosed flares will allow better outputs from modelling of health impacts associated with landfills using HELGA).

The risk of appeal by operators is low, as the use of enclosed flares is generally considered to be best practice within the waste industry. In addition, the policy is targeted at new sites and those which already have flaring.

The Agency may be required to modify licenses for existing landfill sites. The Agency will also need to develop and disseminate monitoring protocols, and allocate resources to the analysis of data from landfill gas emission monitoring.

Alternative approaches considered:

Alternative Option 1: The Baseline- Business As Usual

Certain operators would apply best practice and install closed flares at new sites, and replace open flares at the end of their life with enclosed flares. Others would continue to install cheaper open flares. Reliable monitoring of emissions from flares would not be possible.

Alternative Option 2: Replacement within 3 years

Require all new landfills to install enclosed flares, and the replacement of all open flares by the end of December 2001.

This would require more operators to replace open flares with enclosed flares before the end of their normal life, imposing additional costs on operators.

Alternative Option 3: Replacement within 10 years

Require all new landfills to install enclosed flares and the replacement of all open flares by the end of December 2008.

This would result in no additional cost to operators due to replacement of technologies before the end of their natural life. It would delay the expected benefits from monitoring of landfill gas flaring emissions in terms of potential for emissions control and better risk-based regulation, and prevent the Agency from addressing problems in regulatory consistency and credibility. The Agency may be regarded as being too lenient with operators.

Reason for choosing preferred option- Replacement within 5 years:

The proposed option has been selected due to its significant expected benefits in terms of improved monitoring and control of emissions from landfill gas flaring which will allow benefits for the environment, the public, especially communities in the vicinity of landfill gas flares and the agency (more detailed summary of main benefits). The proposed option is considered to

minimise the costs to operators by allowing replacement of flares in a timescale which is expected to allow natural replacement of open flares with enclosed flares at the end of their expected life, and avoids imposing 'excessive' costs on operators by focusing on those sites with the largest emissions and sufficient revenues for installation of flares.

Table 2: Summary of likely costs and benefits

Likely Benefits	Likely Costs
Improved monitoring and control of emissions from landfill gas flares Expected lower emissions of trace compounds and associated environmental and health impacts. Better information expected for all actors to inform measures to reduce potential environmental and health impacts from landfill gas flaring emissions, for better siting of landfills. Enabling improved control of landfill gas emissions by agency, risk based regulation. Enabling the agency to improve consistency in licensing of landfills. Greater credibility for Agency in addressing concerns of public regarding landfill gas flaring emissions Expected better information to address the concerns of those in the vicinity of landfill gas flares Operators ensure that competitors are applying best practice	Additional capital cost for operators of around £14 million for replacement of existing open flares with enclosed flares. Additional capital costs of around £0.25-£0.55 million for fitting new landfills with enclosed rather than open flares Additional monitoring costs for operators of £10,000 to 15,000 per annum per site. Costs of developing and disseminating monitoring protocols, analysing emissions data and amending licenses for Agency.

Consideration of options with further potential for sustainable development benefits?

Requiring the use of landfill gas for energy recovery has been considered, but is not proposed at this moment. At present, it is believed that combustion engines for energy recovery may have a poorer combustion performance. This could result in the formation of trace compounds. Further investigation of emissions from energy recovery technologies is being investigated. Flares also allow better dispersion of emissions than combustion engines, reducing the potential impacts on near-field communities.

Degree of residual risk:

Low: There is a risk that the Agency will not have sufficient resources to develop monitoring protocols and train inspectors to meet need to collect and analyse data on emissions from flares.

Further assessment of likely costs and benefits required?

Residual risk to the agency is low. No further assessment of likely costs and benefits is required.

Assessment undertaken by: Jan Gronow, Landfill Policy Manager

Richard Smith, R&D MSO (Waste Programme) & Member of National Landfill Gas Group

Vicky Pollard, Business Economist, NCRAOA

Date: 14 January 1999

APPENDIX E: ENVIRONMENT AGENCY GENERAL LANDFILL GAS POLICY

ENVIRONMENT AGENCY POLICY

Policy Number: EAP/LFG/001 **Landfill Gas - General**

Policy Statement:

Gaseous emissions from licensed landfill sites will be regulated by the Environment Agency according to site specific risk to minimise the impact on health, the local environment and global atmosphere.

ENVIRONMENT AGENCY EXPLANATORY NOTE

Policy Number: EAP/LFG/001 **Landfill Gas - General**

Background:

The UK ratified the Framework Convention on Climate Change in 1993. The target was to return UK methane emissions to their 1990 levels by 2000. In Kyoto in 1997 it was agreed to reduce annual emissions of methane to an average of 5% below 1990 levels over the period 2008-2012. Increased control of landfill gas emissions is an essential part in the UK Climate Impacts Programme to reduce methane emissions.

Article 9 of the Framework Directive on waste 75/442/EEC (as amended) requires waste disposal operations to be licensed. Those disposal operations have been transposed into National Law by Schedule 4 of the Waste Management Licensing Regulations 1994 (as amended).

Guidance is contained in Waste Management Papers 4 'Licensing of Waste Management Facilities' (DOE, 1994) and 26B 'Landfill Design, Construction and Operational Practice' (DOE, 1995). Non-statutory guidance on monitoring and control of landfill gas is contained in Waste Management Paper No.27 'Landfill Gas' (Department of the Environment, 2nd Edition, 1991). The clear intent of these documents is that flaring is preferred over passive venting and that the effect on air quality and the global climate by the release of greenhouse gases should be minimised. Gas emissions from landfill sites are also within the scope of the EC Landfill Directive, which is likely to be adopted during 1999, and implemented by 2001.

Agency directors agreed a ten-point action plan for 1998/99, which was selected from the corporate plan key performance targets for 1998-2001. Two of these actions which involve climate change and air quality are relevant to this policy and encourage the installation of methane control and recovery systems with associated enclosed flaring to improve combustion and reduce odours. In the context of low methane emissions, biological methane oxidation is promoted by passing the gas through appropriate media to reduce its global warming potential.

To ensure consistency of approach the Agency's National Waste Group formed a series of subgroups including the Scientific & Technical Working Group (S&TWG). The Landfill Gas Task & Finish Group, a subgroup of S&TWG is charged with producing Agency guidance on landfill gas issues with reference to underpinning legislation.

Clarification:

Landfill gas is an end product of the degradation of biodegradable wastes in landfill sites. Typically it is a mixture of up to 65% methane and 35% carbon dioxide, plus trace concentrations of a range of organic gases and vapours (up to 350 have been identified).

Methane is an active greenhouse gas, which has a global warming potential 24.5 times greater than carbon dioxide. Landfill gas can also be toxic, explosive, asphyxiating and highly odorous, making it a potential threat to human health and amenity.

A recent article in *The Lancet* published details of a European wide study investigating congenital birth anomalies in close proximity to landfill sites. This study and others have been widely publicised and has brought human health issues from landfill sites to the forefront of the public arena leading to increased scrutiny of regulatory and operational practices.

Policy Author: Ian Cowie, Environmental Protection Team Leader and Chair of Landfill Gas Task & Finish Group (NE Region, Ridings Area)

Policy Sponsor: Steve Lee, Head of Waste Function	**Version:** Draft 1.2 19-Jan-1999

Policy Number: EAP/LFG/001 Landfill Gas – General (Continued)

Desired outcome:

To ensure a nationally consistent approach to the fulfilment of the Agency's statutory duty in respect of gaseous emissions from licensed landfill sites.

Audience:

Internal - area environment planning and protection staff, regional waste staff and EPNS staff.
External - waste management licence holders and their agents, consultants, landfill gas equipment manufacturers, national and local government and the general public.

References:

EC Framework Directive on Waste 75/442/EEC (as amended by 91/156/EEC)
EC Proposed Landfill Directive (COM (97)105 Final)
The Waste Management Licensing Regulations 1994
Waste Management Papers 27, 4 and 26B
Planning Policy Guidance Note (PPG)10
Licence Process Handbook
Landfill Gas Licensing Issues - Interim Internal Agency Guidance, Version 4, November 1998
Interim Internal Technical Guidance for Best Practice Flaring of Landfill Gas, Document Reference No. LFG2, Version 2.0, January 1999

Policy Author: Ian Cowie, Environmental Protection Team Leader and Chair of Landfill Gas Task & Finish Group (NE Region, Ridings Area)

Policy Sponsor: Steve Lee, Head of Waste Function	**Version:** Draft 1.2 19-Jan-1999

POLICY IMPLEMENTATION PLAN

Policy Numbers: EAP/LFG/001	**Landfill Gas - General**
1. Who are the target audiences?	Internal - area environment planning and protection staff, regional waste technical and scientific staff. External - waste management licence holders and their agents, consultants, landfill gas equipment manufacturers, national and local government and the general public.
2. What do they need to know?	National awareness and understanding of Agency landfill gas policy and technical guidance.
3. When do they need to know it?	As soon as practicable.
4. How will they be told?	Via the Licence Process Handbook and associated items of technical guidance.
5. Who will tell them?	Those responsible for training staff on licensing and enforcement and the use of the Licence Process Handbook.
6. What supporting material must be developed?	In collaboration with EPNS: Interim internal technical guidance for best practice flaring of landfill gas - attached and about to be published and go for full external consultation. Interim internal technical guidance for best practice landfill gas utilisation (based on R&D Technical Report - final version pending). Health effects from landfill gas (HELGA) risk assessment model, shortly to be finalised (R&D contract). Agency guidance will need to be developed. Landfill methane measurement protocols (commenced Oct-98 as R&D contract). Interim internal technical guidance for best practice methane oxidation (based on previous R&D). Cost benefit analysis procedures for landfill gas control options and emissions monitoring protocols (an external R&D Group is working on this). Review of Waste Management Paper No.27 (R&D Project, Form A submitted for approval January 1999).
7. What resources are required?	Resources will be required for development of associated guidance only, in liaison with EPNS.
8. Monitoring of Progress Method(s): Success Criteria:	 Through auditing of landfill licences and associated working documents. Achieving regional targets for climate change and air quality under the 10-point action plan. Contribution to achieving the UK plan for the reduction of greenhouse gas emissions to atmosphere. Phasing out of operational 'open' flares at licensed waste management facilities.

Date(s) Undertaken:	Compliance with emission standard. Open flares phased out by 31 December 2003.
Comments:	10-point action plan monthly with annual auditing of landfill sites. This is an overarching policy statement, which will be implemented by a series of associated policies.
9. Authorisation	Policy Sponsor Sign: Name: Steve Lee Title: Head of Waste Policy Date: 20January 1999
	Chair Approval Body
10. Review Date	September 2000
11. Version: Draft 1.2 19-Jan-99	

APPENDIX F: ENVIRONMENT AGENCY LANDFILL GAS FLARING POLICY

ENVIRONMENT AGENCY POLICY

Policy Number: EAP/LFG/002 **Landfill Gas Flaring**

Policy Statement:

The Environment Agency requires:

(a) that no more 'open' flares shall be installed at licensed landfill sites, except for experimental or emergency purposes;

(b) that all currently operational landfill gas flares operated as 'open' flares at licensed landfill sites shall be replaced progressively with 'enclosed' flares, or non-combustion techniques offering equivalent performance, by 31st December 2003 (the Agency will prioritise sites which produce large amounts of gas or pose a significant risk to the local environment);

(c) that all existing 'enclosed' flares operating at licensed landfill sites shall demonstrate operational performance required to meet the prescribed emission standard.

POLICY IMPLEMENTATION PLAN

Policy Numbers: EAP/LFG/002	**Landfill Gas Flaring**
1. Who are the target audiences?	Internal - area environment planning and protection staff, regional waste technical and scientific staff. External - waste management licence holders and their agents, consultants, landfill gas equipment manufacturers, national and local government and the general public.
2. What do they need to know?	National awareness and understanding of the Agency's landfill gas flaring technical guidance.
3. When do they need to know it?	As soon as practicable.
4. How will they be told?	Through the issue of this policy and associated technical guidance with effective dissemination and press release. Through the Licence Process Handbook. Planned publication of journal article in IWM Proceedings. National Landfill Gas Conference (24 March 1999) 'Landfill Gas – from Research to Policy'.
5. Who will tell them?	Those responsible for training staff on licensing and enforcement and the use of the licence process handbook.
6. What supporting material must be developed?	Minor revisions of the attached interim internal technical guidance for best practice flaring of landfill gas in light of comments received during external consultation.
7. What resources are required?	1. In the first instance there is no additional resource requirement. 2. It would be advisable to train staff on the monitoring of flares in order to effectively and efficiently assess monitoring data and landfill gas flare performance. 3. Liaison is occurring with the National Compliance Assessment Service with a view to further developing monitoring protocols for flares. Training is to be co-ordinated with training and implementation of the National Licensing Process Handbook. The estimated maximum resource requirement is 1-2 persons per area for one day, depending on the agreed level of service.
8. Monitoring of Progress Method(s):	Through auditing of landfill licences. Achieving regional targets for climate change and air quality under

	the 10-point action plan.
Success Criteria:	Phasing out of operational 'open' flares at licensed waste management facilities. Compliance with and effective regulation against the proposed emission standards. Open flares phased out by 31 December 2003.
Date(s) Undertaken:	10-point action plan monthly with annual auditing of landfill sites.
Comments:	This is the first of a series of guidance documents associated with the attached LFG Policy EAP/LFG/001.
9. Authorisation	Policy Sponsor — Chair Approval Body Sign: Name: Steve Lee Title: Head of Waste Function Date: 20 January 1999
10. Review Date	September 2000 — 11. Version: Draft 1.2 19-Jan-99

APPENDIX G: ENVIRONMENT AGENCY WASTE R&D PROGRAMME - RELATED PROJECTS UNDERPINNING LANDFILL GAS REGULATION

Measurement of gas potential for landfilled material

Role: Project Manager

Detailed methodology used in a technique for the determination of the quantity of methane that can be produced by the anaerobic microbial decomposition of organic matter under controlled conditions (Biochemical methane potential tests). Developed as a tool to assist in the determination of the condition of landfill completion status.

Monitoring accelerated stabilisation at Brogborough landfill

Role: Project Manager

Continued landfill gas monitoring at the Brogborough test cells. The cells were developed at an operational landfill to demonstrate landfill gas enhancement via six cells of differing waste types and filling techniques. Results to date have provided the largest and the longest validated set of data from any field-scale landfill study worldwide. More importantly, this work is the first study to progress to the stage of a maximum observed landfill gas yield at the field level, demonstrating the timescales involved.

Final storage quality at the Brogborough landfill test cells

Role: Project Manager

By definition, the concept of final storage quality or environmental equilibrium status accepts that landfills will not achieve 100% degradation and that the controlled release of substances into the surrounding environment can be acceptable if equilibrium status can be achieved.

Samples extracted from the Brogborough test cells were used to quantify the level of stabilisation achieved over the past 14 years by comparing current waste characteristics with published data for the characteristics of MSW that has been pre-treated by composting and anaerobic digestion. The work considers what might be achieved, through a range of site management techniques, if a further 16 years had been available (giving a total 30-year life *i.e.* the notion of achieving completion in one generation).

Definition of environmental equilibrium status for landfill

Role: Project Manager

A re-examination of what is meant by landfilling according to sustainable development principles and how it might be achieved. Scopes environmental equilibrium and assesses the implications of different timescales for achieving it. Links to FSQ at Brogborough.

Final characterisation of the Brogborough landfill test cells

Role: Project Manager

This study excavated the test cells and carried out a final characterisation study of the wastes before they were buried. Specific objectives were to make a final comparison of landfill gas trace component data across cells; assess settlement; determine final leachate quality; recover and characterise waste samples representative of horizons in the 6 cells physically, biochemically and microbiologically; examine landfill gas abstraction wells to assess the level and properties of accumulated scale and enhance understanding of well development and blinding; examine and characterise the clays forming some of the bunds including measurement of hydraulic conductivity; estimate the extent to which the wastes within the Brogborough test cells have reached final storage quality; and excavate to the base of at least one cell in order to assess the integrity of the basal layer and leachate collection system.

Estimating biodegradable municipal solid waste diversion from landfill

Role: Project Manager

The Environment Agency is required to monitor the diversion of biodegradable municipal waste (BMW) from landfill. Reliable methods are needed to measure the biodegradability of municipal waste, both as mixed municipal waste and as individually separated fractions. An evaluation of several methods was carried out using a variety of organic materials typically found in municipal solid waste. The assessment considered biological and non-biological methods to determine which provides the best-fit surrogate measurement for relative waste biodegradability. A novel method (cellulase hydrolysis) was partially developed and is being developed further under contract to Defra.

Minimising methane emissions from landfill research in collaboration with Shanksfirst & Biffa First Fund.

Role: Project Manager

Prior to this research, there was little quantitative information available on the scale of emissions from the operational phase of landfills, nor on the timescale for the onset of methanogenesis. This work quantified the relative and absolute volumes of gas emissions during the active working life of landfills before final capping and active gas abstraction systems are set in place, by assessing emissions from working surfaces and the associated side slopes. Quantitative measurements were carried out at 21 UK landfills, having established the most appropriate flux measurement techniques. Implications for gas control strategies were assessed.