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Geotechnical hazards associated with closed municipal solid waste landfill sites

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Abstract. As pressure for new infrastructure and development grows, it is inevitable that building projects will encounter some of the c20,000 closed former solid waste landfills in the UK, many of which will have accepted municipal solid wastes (MSW). Construction on or across these sites brings a special set of geohazards associated with the potential for large and difficult to predict settlements, gas (and odour) release or generation, contaminated leachate and the breach of containment systems and other environmental controls. The presentation will discuss these issues with reference to recent research into understanding and predicting settlements in municipal solid waste landfills; assessing the total, current and residual gas potential of biodegradable wastes; the role of the hydraulic regime in the flushing of contaminants from the waste and the quality of leachate; and the need or otherwise for the long term integrity of engineered barriers and controls.

1. Introduction
European waste management policy has for the past 20 years sought to move away from landfill in favour of waste avoidance and reduction, recycling and the recovery of energy from wastes. However, in Europe and much of the developed world, landfilling has been the primary means of municipal solid waste (MSW) disposal during more than 150 years of progressive industrialisation and population growth. As a result, there are estimated to be more than 20,000 closed former MSW landfills in the UK alone, containing ~1 billion tonnes of household waste and ~5 billion tonnes of Commercial and Industrial (C&I) type wastes.

The Environment Agency (EA) holds a GIS database of both current (authorised) and historic landfills that can be accessed via the “what’s in your backyard” part of their website. The database is a collation of information held by local authorities and the Environment Agency. Authorised landfills are sites that have a permit issued under the Pollution Prevention and Control Act 1999, or closed sites that no longer take waste but still have a waste management licence issued under Part II of the Environmental Protection Act 1990. Some waste management licences date back to disposal licences issued under the Control of Pollution Act 1974.
Historic (closed) landfill sites are those where there is no PPC permit or waste management licence currently in force. This includes (most) sites that existed before the waste licensing regime, and previously licensed sites where the licence has been revoked, ceased to exist or surrendered and a certificate of completion issued. From 1948 until the introduction of the waste licensing regime, regulation of landfills was undertaken through planning law. There were no controls on waste disposal before the Planning Act of 1948, and no formalised means of recording the location of sites.

The National Planning Policy Framework (NPPF) published in 2012 sets out Government planning policy for England and how this is expected to be applied to development. Paragraphs 120 to 122 of Section 11 of the NPPF, Conserving and enhancing the natural environment, concern contaminated land and state the following:

120. To prevent unacceptable risks from pollution and land instability, planning policies and decisions should ensure that new development is appropriate for its location. The effects (including cumulative effects) of pollution on health, the natural environment or general amenity, and the potential sensitivity of the area or proposed development to adverse effects from pollution, should be taken into account. Where a site is affected by contamination or land stability issues, responsibility for securing a safe development rests with the developer and/or landowner.

Legislative controls over authorised landfills and contaminated land are set out in Part II and Part IIA respectively of the Environmental Protection Act (1990) and a raft of Statutory Instruments and amendments thereafter. Within this Legislation responsibility for the adverse effects of contaminated land (appropriate person) also lie with the original person(s) who caused or knowingly permitted contamination at the site – usually taken to be the site operator.

It follows that any development over an old MSW landfill will require the involvement of a variety of parties including the original landowner and operator, the local authority and the EA (including as statutory consultees on any planning application), as well as the developer.

Many old landfills are no longer actively managed or monitored, and their state of degradation is uncertain. Many are likely to pose a residual potential pollution threat through the leakage of leachate into ground or surface waters or the escape of landfill gas into the ground or the atmosphere. Ongoing degradation of the waste will result in continuing settlement. Even if degradation is complete, the potential for creep or further load-induced settlement may be high. These processes and features represent possible geohazards for the construction of new infrastructure and buildings across a former MSW landfill, as summarised in Table 1.

The presentation will discuss these issues with reference to:
- recent research into understanding and predicting settlements in municipal solid waste landfills
- the total, current and residual gas potential of biodegradable wastes
- the role of the hydraulic regime in the flushing of contaminants from the waste and the quality of leachate
- the point at which a landfill can be considered not to represent a potential environmental threat, hence the need or otherwise to maintain the long term integrity of engineered barriers and controls.
Table 1: Potential impacts of landfill processes on future site redevelopment

<table>
<thead>
<tr>
<th>Landfill process</th>
<th>Mechanisms</th>
<th>Potential impacts on site redevelopment</th>
</tr>
</thead>
<tbody>
<tr>
<td>Settlement</td>
<td>• Mechanical (self-weight stress)</td>
<td>Unlikely – this is a short term phenomenon</td>
</tr>
<tr>
<td></td>
<td>• Biodegradation</td>
<td>Could result in significant ongoing settlement if degradation is still ongoing</td>
</tr>
<tr>
<td></td>
<td>• Creep</td>
<td>Potentially an issue but decreases with log time</td>
</tr>
<tr>
<td>Settlement</td>
<td>• Mechanical (imposed additional stress)</td>
<td>Large settlements may occur as a result of structural loading owing to the generally low stiffness of wastes</td>
</tr>
<tr>
<td>Gas</td>
<td>• Biodegradation induced generation of CO₂ and CH₄</td>
<td>Odours. Potential release of gases, including H₂S.</td>
</tr>
<tr>
<td>Leachate</td>
<td>• Washout of recalcitrant contaminants and products of biodegradation</td>
<td>Biochemical effects on sub-surface structures and foundations. Leakage of leachate into groundwater (liner failure) or surface water (drainage failure leading to overtopping)</td>
</tr>
<tr>
<td>Passive engineered control features</td>
<td>• Cap, liner, drains, gas vents etc</td>
<td>May need to avoid penetrating cap and liner to preserve integrity if biodegradation is still ongoing and/or contamination potential through leachate escape remains. May need to maintain drainage to avoid overtopping (see above)</td>
</tr>
</tbody>
</table>

2. Landfill processes

It is well established that the pollution potential of a waste is reduced over time through the processes of biodegradation and flushing (e.g. DoE, 1986; Knox 1990). Biodegradation generates gases, principally methane and carbon dioxide. It also causes a loss of mass and volume, resulting in settlement of the waste. Some contaminants are not readily biodegradable, and must be removed from the landfill by leaching or flushing, i.e. the continued flow-through of relatively clean water to wash these contaminants out.

In principle, an MSW landfill could be operated in such a way as to remove the contaminant load through biodegradation and flushing to achieve a state of practical completion (in which the landfill is in hydraulic equilibrium with its surroundings, and the waste is considered to be of “final storage quality”, FSQ, in that it no longer represents a potential pollution threat) within a timescale of about a generation (~30 years; e.g. Harris et al., 1995). However, in the UK at least there is no financial or regulatory incentive to do this, and under current UK operational regimes it could take hundreds of years for a state of practical completion to be achieved (e.g. Knox, 1990; Hall et al., 2007). It follows that most closed landfills will still represent a potential pollution threat, which could be realised on redevelopment of the site through the disturbance of the waste and / or the engineered controls such as caps, liners and drains built to prevent the escape of pollutants from the waste during the operational and aftercare phases of the site.

Commonly accepted interpretations of the terms “Landfill completion” and “Final storage quality” are reflected in guidance published in the UK (Environment Agency; 2005, 2012), which requires that the flux of contaminants to the environment would still be acceptable assuming, inter alia:

• no active management;
• failure of all engineered containment;
• attainment of hydraulic equilibrium (i.e. water or leachate levels in the site have equilibrated with water fluxes into and out of the site in the absence of active management and failure of some or all of the engineered controls);
• no functioning gas or leachate management systems.

Biodegradation in landfills occurs primarily through the processes of anaerobic degradation, which produces landfill gas (LFG). The potential for aerobic degradation to occur is normally very limited. However, air injection (particularly in Germany) has been utilised as a final treatment stage in aged landfills. There is a close relationship between biodegradation and waste settlement.

3. Settlement

Settlement of biodegradable wastes may be considered in three stages, termed immediate, primary and secondary. Immediate settlement is due to the compression or expulsion of gas and/or the compression of particles. Primary settlement in a saturated waste is associated with the expulsion of liquid and the consolidation of the waste matrix. Secondary settlement results from mechanical creep and biodegradation.

The above was demonstrated by data from tests on raw MSW and mechanically biologically treated (MBT) wastes in consolidating anaerobic reactors (CARs) reported by Ivanova et al. (2008) and Siddiqui et al. (2012, 2013). Specimens of wastes were held under a constant vertical stress of 50 kPa or 150 kPa and the progress of degradation, gassing and settlement monitored. The data included control tests in which degradation was prevented, which enabled settlements due to mechanical creep to be separated from those due to biodegradation.

Terzaghi’s classical theory of consolidation was successfully applied to the analysis of primary settlement by Siddiqui et al. (2013), although this mechanism may be of limited direct relevance to an unsaturated waste in the field.

If simple models for estimating medium and long term settlements are to be applied successfully, mechanical creep must be considered separately from biodegradation as the mechanisms are different and may proceed at different rates under different operational conditions or environmental conditions of (for example) water content, leachate recirculation or temperature. Also, the ultimate amount of biodegradation settlement ($\varepsilon_{bt}$) is finite, whereas creep continues in log time.

In simple terms, creep strains $\varepsilon_c$ may be characterised by a linear (in log time) relationship of the form:

$$\varepsilon_c = C_{axc} \log_{10}(t/t_1)$$

where $C_{axc}$ is a coefficient of creep settlement, $t$ is the time following the application of stress and $t_1$ is a reference time. The value of $C_{axc}$ reduces with increasing waste bulk density but is substantially insensitive to vertical stress, at least over the range 50-150 kPa.

Biodegradation settlements $\varepsilon_b$ can be characterised using a relationship based on first order reaction kinetics,

$$\varepsilon_b = \varepsilon_{bt}(1 - e^{kt})$$

where $k$ is the reaction rate coefficient.
where $\varepsilon_{bt}$ is the total eventual settlement due to biodegradation, $k_b$ is a biodegradation rate constant and $t'$ is the elapsed time since the start of degradation. This approach is similar in concept to the Tier 2 methods of the IPCC for estimating GHG emissions from landfills (IPCC 2000).

Alternatively, more fundamental but complex models that fully couple degradation, gas and liquid transport, settlement and mass transfer between phases such as LDAT (White et al., 2004, 2013, 2014; or McDougall, 2007) may be adopted. The advantage of such models is that they can take into account the potential impact of an infrastructure development on settlements not only in terms of new loadings but also in terms of waste degradation. This may be important, especially if any redevelopment results in the ingress of water, which could cause biodegradation in parts of the site that were previously dormant. If remediation of the landfill is a necessary part of redevelopment, this will probably involve further biological stabilisation of the landfill. The use of fully coupled models can then aid predictions of the impact of the remediation on settlement.

“It is not possible to characterise the engineering properties of waste fully, because of its heterogeneous nature, but it is important that its basic behaviour is understood and that likely ranges of the key engineering properties are known.” (Dixon et al 2006). Evidence of the variability in the engineering properties of landfills is provided by results of cone penetration testing (CPT). For example, McKnight et al (2015) reported highly variable data compared to typical CPT use in soil.

For developments that involve the building of new infrastructure on top of old landfills there is a need to consider the impact of load and how the load will be carried, for example by shallow or piled foundations.

The Environment Agency have produced guidance on Piling and Penetrative Ground Improvement Methods on Land Affected by Contamination: Guidance on Pollution Prevention (EA, 2001). This is general guidance and does not cover MSW landfills in particular. However, the guidance discusses potential environmental hazards associated with piling through contaminated soils, and how sites may be improved. It focuses on the effects of piling on any controlled waters, especially groundwater, with brief consideration of gas and human health issues.

“Where there is an option, concrete raft foundations are the preferred means of foundation on contaminated sites” (EA, 2001) This may be a useful general principle for contaminated land, but in landfills the application of substantial additional surface loads may result in significant primary settlement. If the waste is already fully or partly saturated this may cause unacceptable rises in leachate levels.

Knowledge of the void ratio and the one dimensional constrained modulus of MSW waste allows the impact of this type of loading on leachate levels to be assessed.

There are limited data on the one dimensional (constrained) modulus of MSW waste (e.g. Hudson et al 2004, Gomes et al 2014 and McDougall 2008). However, in general constrained modulus values, lie within the range of compression ratios of 0.08 to 0.41. Compression ratio is the negative of the slope of a graph of vertical strain against the logarithm of vertical stress, and is a useful measure that normalises constrained moduli against vertical stress.

There is no obvious influence of age on compressibility; the largest influence relates to the nature of the landfilled material. Greater increases in leachate levels will be seen in landfills with a characteristic compression ratios at the top of this range.
EA (2001) considers various pollution scenarios associated with piling in contaminated lands within the framework of source–Pathway–Receptor risk assessments. Two of their scenarios are highly relevant to MSW landfills. The first covers the case where piling goes through the contaminated land and into underlying strata to provide the load bearing capacity required for the infrastructure development. A hydrogeological risk assessment would be needed, particularly if the landfill is located in a source protection zone or overlies a major or minor aquifer. Many historic landfills are unlikely to have any as-built engineered containment, and many sites in old sand and gravel exaction pits are in deposits that have the characteristics of minor aquifers. The geo-hazards associated with this situation are discussed in the section on Leachate and Groundwater Quality.

The second potential pollution scenario relates to the creation of preferential pathways through a low permeability surface layer, allowing migration of landfill gas and odours to the surface. The geo-hazards associated with this are covered in the section on Gassing.

4. Leachate and groundwater quality

The database associated with What’s in your backyard holds general information (where known) on the waste types accepted at a site. For authorised sites, the nature of the licence held is recorded (e.g. Licence type A5 : Landfill taking non-biodegradeable wastes). For historic sites, information on whether a particular category of waste (e.g. Industrial waste, Commercial waste, Household waste, Liquid /sludge etc) was landfilled is held.

Historically, many landfills accepted a range of industrial/ chemical liquid and solid wastes that would not be licensed today. Consequently the development of an old landfill site would need to include a comprehensive organic and inorganic leachate analysis to characterise potential pollutants and toxins. Landfills that accepted predominantly MSW will, in comparison, contain or generate less aggressive leachate. However, leachates from many old MSW landfills will nevertheless contain species not suitable for direct discharge to controlled waters.

Many contaminants found within wastes do not biodegrade and can only be removed through the liquid phase. The mechanism of removal in landfills is invariably through the generation (and subsequent abstraction / treatment) of leachate. Landfill completion does not necessarily require that all contaminants are removed from the waste; the key point is rather that their rate of release can be attenuated by the receiving environment, at the point of completion and all times into the future.

There has been a considerable amount of research into the release of contaminants from wastes in the liquid phase. Various standard waste leaching protocols have been developed in Europe and America in support of waste acceptance criteria tests; and a large database of test results drawn up (e.g. LeachXS; van der Sloot and Kosson, 2012) including a detailed appraisal of the geochemistry of the leaching process. Leach test results are often expressed as a function of the liquid to solid (LS) ratio, which is the volume of liquid leached (in litres) from the mass of dry solids (in kg) of an element of waste. In general terms, the higher the LS ratio achieved, the lower the leached concentration of a contaminant.

A secondary control on the rate of release of contaminants from landfilled waste comes from the heterogeneity of the waste at a variety of scales. Although there appears to be a relatively simple relationship between the mass of contaminants leached and the LS ratio, an obvious inference from this is that if there has been no flow of water through waste there will have been no leaching of contaminants. Numerous studies of contaminant transport processes...
in waste (e.g. Rosqvist and Destouni 2000; Beaven et al., 2003; Fellner and Brunner, 2010) have noted a dual porosity effect. This characterises flow in waste as occurring through a number of discrete mobile pathways around blocks of waste within which no advective flow occurs. Contaminants are removed from the blocks by a process of diffusion into the flow in the mobile pathways. The rate of diffusion of contaminants out of a block follows an inverse square law with the block size. Barker (1985) characterised dual porosity systems in terms of block geometry functions, with a characteristic diffusion time of a block related to its average size. The characteristic diffusion time for an ‘immobile’ 1 cm cube of waste is approximately 4 days, whereas for a 1 m cube of waste it is approximately 110 years (Beaven and Barker, 2010). The importance of this is that the size of the immobile blocks effectively controls the rate of release of contaminants. If in defining FSQ we are concerned about the rate of release, then a site containing effectively larger immobile blocks of waste may achieve FSQ more quickly than a site with the waste in smaller blocks.

Beaven et al. (2014) discussed the relevance of hydraulic equilibrium to the technical debate on aftercare. “During aftercare, active management and the functioning of engineered controls will result in an imposed hydraulic equilibrium, which, in many sites, will mean the majority of the waste is unsaturated. As active management (e.g. leachate pumping) is discontinued and engineered controls (e.g. the cap and/or liner) deteriorate or fail, then a new hydraulic equilibrium will be established that in many cases may involve the slow filling of the site with leachate. For completion to occur, the regulator must be satisfied that future fluxes to the environment will be acceptable under a range of hydraulic equilibrium situations.” Extending this reasoning to the dual porosity concept, we could imagine a large block size that in unsaturated conditions has led to low fluxes of contaminants to leachate collection systems. If leachate pumping were then stopped and leachate levels rose, new saturated flow paths could access different parts of the site with a different block geometry, leading to different fluxes of contaminants.

5. Landfill gas and odours
The anaerobic degradation of organic material in landfill will yield landfill gas (LFG) comprising approximately 60% CH\textsubscript{4} and 40% CO\textsubscript{2}. To be explosive, landfill gas has to be in a confined space and mixed with oxygen as methane is flammable only when the concentration in air is between 5 and 15 percent.

The lateral migration of gas from municipal solid waste landfill sites is recognised as a significant hazard (e.g. Christophersen et al 2001). It resulted in an LFG explosion at Loscoe in 1986, which destroyed a bungalow located within 20m of a landfill.

In landfills with LFG control systems overpumping is the primary mechanism by which air may be drawn into the landfill. However, most old landfills will not have LFG control measures installed, in which case barometric pumping is the primary mechanism of air ingress (Young, 1990).

The total gassing potential of an MSW landfill may be estimated from any of a variety of tests including loss on ignition, biological methane potential (BMP) and the ratio of cellulose and hemicellulose to lignin, (C+H)/N, in the waste as determined by the Fibrecap test (Zheng et al, 2007). Siddiqui et al. (2013) reported a close correlation between the biodegradation induced settlement and the cumulative volume of gas produced (at SPT) for both raw MSW and MBT waste, and hence between the potential ultimate biodegradation induced settlement and loss on ignition, BMP and (C+H)/L. Many old landfills retain a gas
potential from cellulose and hemi-cellulose possibly as high as \( \sim 75 \text{m}^3/\text{tonne} \) (Knox et al., 2011), but nevertheless have low gas generation rates. There are two main reasons for this. The first may relate to a lack of moisture in parts of the waste essential for biodegradation. The second is that lignin shielding restricts the access of bacterial exo-cellular enzymes to much of the degradable content (e.g. Van Soest, 1994). Lignin degradation occurs mainly under aerobic conditions.

Consequently, any development on an old landfill would need to consider the implications for landfill gas generation and control. Introducing more water and/or air could increase waste degradation leading to landfill settlement and potentially more LFG production. Any development that restricted the ability for landfill gas to be released to the atmosphere may increase the potential for off-site lateral migrations; this too would need to be accounted for.

Odours from landfill sites are predominantly associated with landfill gas, with 12 trace components contributing to odours (EA 2002). Hydrogen sulphide is a major contributor and has been detected in landfill gas in concentrations up to 12,000 ppm, although more typical concentrations in landfill gas are \( \sim 100 \) ppm. At these concentrations \( \text{H}_2\text{S} \) cause olfactory paralysis (desensitisation), which is a concern because \( \text{H}_2\text{S} \) is fatal to humans at concentrations of \( < 500 \) ppm. The HSE short and long term exposure limits for \( \text{H}_2\text{S} \) are 5 ppm and 10 ppm respectively. A main cause of high concentration of \( \text{H}_2\text{S} \) in landfills is linked to the disposal of gypsum (sulphate) based industrial wastes, which is now banned.

### 6. Summary and conclusions

1. Old MSW landfills can be considered as a sub-set of contaminated land, and in terms of their re-development the legislative controls are the same.
2. The main environmental geohazards arising from old MSW landfills relate to the potential for water pollution, and hazards linked to landfill gas including risk of explosions and the possibility of hazardous concentrations of \( \text{H}_2\text{S} \). Any new development across an old landfill will need to assess its impact on the above, including the potential to reactivate degradation and gassing through a change in the landfill hydraulic regime.
3. While landfill settlement does not in itself constitute a hazard to the surrounding environment, its impact on the development infrastructure may be significant. Models exist to help predict the magnitude of this settlement which will be of help to designers.

### 7. Acknowledgments

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### 8. References


