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Review:

Mechanisms of settlement in municipal solid waste landfills*

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Abstract: Settlement prediction of municipal solid waste (MSW) is a key issue in the design, construction, and post-closure development of landfills. As MSW is a highly heterogeneous material with compressible and degradable elements, the factors causing compression and settlement are complex. This paper explores the mechanisms and timescales for the settlement of landfilled MSW, and develops a rational taxonomy and framework of understanding. Conceptual models are reviewed and discussed, and the relative importance of different causes of settlement is assessed with reference to new, high quality, datasets and data from the literature.

Key words: Municipal solid waste (MSW); Compression; Settlement; Creep; Degradation
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1 Introduction

Municipal solid waste (MSW) is both compressible and degradable. When placed in a landfill, settlement continues over an extended period of time, and may reach 25%–50% of the initial fill height (Stearns, 1987). Settlement during active landfilling will increase the storage capacity, and have implications for landfill operation and management. Post-closure settlement may damage the cover (cap) and gas or leachate collection systems, and hinder reclamation. Thus, accurate predictions of waste settlement during and after filling are needed for the estimation of landfill capacity, the design of intermediate and final covers, the design of leachate and gas collection systems, and the design of landfill expansion or remediation and aftercare schemes (Hunte et al., 2007).

It is well-established that the settlement of MSW in a landfill arises from a variety of mechanisms that can take place on different timescales. However, the examination of the literature reveals a lack of consensus regarding nomenclature and in distinguishing between mechanisms and timescales.

The principal mechanisms of waste settlement described in the literature (Sowers, 1973; Rao et al., 1977; Edil et al., 1990; Wall and Zeiss, 1995; Powrie et al., 1999; Watts and Charles, 1999; El-Fadel and Khoury, 2000; Fei and Zekkos 2013; Siddiqui et al., 2013) are as follows:

1. Rearrangement of the solid matrix by sliding, reorientation or distortion of waste particles as vertical stresses are increased; either during compaction or as further material is deposited on top. This is

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analogous to the main mechanism of compression in conventional soil mechanics, in which it is assumed that the particles and the pore water are both incompressible, and the changes in total volume are always associated with changes in the void ratio as the particles distort and perhaps become rearranged by particle-on-particle slip. If the waste is unsaturated or highly permeable, settlement by this mechanism will occur effectively immediately on loading. If the waste is saturated and of low permeability, the rate of settlement will be limited by the rate at which voids can reduce in volume, and hence by the rate at which pore water can escape. The latter is the well-known process of consolidation in conventional soil mechanics.

2. Compression of the pore fluid. This is not normally considered in conventional saturated soil mechanics because the pore fluid (water) may reasonably be viewed as incompressible. However, it may be significant in unsaturated soils in which the pores contain air or an air/water mix, which cannot reasonably be viewed as incompressible.

3. Compression or crushing of waste particles. Again, this is not normally considered in conventional soil mechanics, but is potentially important in materials such as carbonate sands (Coop and Atkinson, 1993).

4. Breakage of particles as stresses are increased, or softening of particle contacts on wetting resulting in a loss of strength and/or structure. These mechanisms are not normally considered in conventional soil mechanics, although the potential importance of particle breakage is increasingly recognised (McDowell and Bolton, 1998).

5. Degradation, due to biological decomposition and physico-chemical processes such as corrosion and oxidation of the waste in the longer term. Degradable matter in MSW will be transformed into gas and leachate. If the remaining waste matrix can no longer resist the stress applied on it, settlement will result.

6. Conventional mechanical creep (i.e. continuing settlement at constant effective stress). Leonard et al. (2000) stated that creep is caused by: (i) erosion and sifting of finer materials into voids between large particles; (ii) material moving into voids as a result of degradation; (iii) continued plastic deformation.

7. Edil et al. (1990) identified ravelling (i.e. the gradual migration of finer particles into the larger voids) as a separate mechanism, but it is arguably the same as the first two mechanisms in mechanical creep and degradation settlement defined by Leonard et al. (2000).

A typical graph of settlement against time for a specimen of waste will generally show at least three distinct phases.

(a) An initial, immediate compression, probably resulting from the compression or expulsion of air initially present in the void space, or the compression or crushing of compressible particles (2 and possibly 1, 3, and 4, above).

(b) Settlement due to consolidation of the waste, at a rate governed by its hydraulic conductivity (which affects the rate at which water can flow from the saturated pores) and the stiffness of the overall structure (which affects the amount of compression that has to take place in response to a given increase in load): 1 and possibly 3 and 4, above. This is conventionally termed primary settlement.

(c) Continuing settlement at constant effective stress, after consolidation has ceased and the pore pressures have reached hydraulic equilibrium. This is due to degradation and creep (5 and 6, above), and is commonly referred to as secondary settlement.

This is reasonably consistent with the categories of time-dependence identified by Watts and Charles (1999), viz. immediate compression of an unsaturated fill, primary consolidation of saturated fill, and secondary consolidation (due to creep and bioconsolidation) of saturated fill. Watts and Charles (1999) attributed immediate compression to unsaturation: while the presence of air somewhere in the system is probably a prerequisite, it could be within the particles (making them compressible) rather than in the pore fluid (which makes the waste unsaturated). Their use of the term consolidation to describe ongoing settlements due to creep and degradation has a precedent in the soil mechanics literature but is perhaps unhelpful. The term bioconsolidation (Soler et al., 1995; Watts and Charles, 1999) seems almost as inappropriate because the effect it describes is distinct from conventional consolidation and may be due to chemical rather than biological agents.

While the descriptors primary, secondary, and tertiary settlements (for the time-related settlement regimes a, b, and c, respectively) might be more appropriate, different terms are now so well established that the designations immediate compression, the primary settlement or consolidation, and the secondary settlement will be adopted for the remainder of this paper.

2 Particle compressibility

The complex settlement mechanisms result from the particulate nature of wastes. According to the classification systems proposed by Kölsch (1995) and Dixon and Langer (2006), MSW is sorted into different material groups, and waste components in each group can be characterized by their shape, compressibility, and degradability. As shown in Table 1, waste components can be divided into 0D (small particles similar with soils), 1D (long particles with two short dimensions and one long), 2D (flat particles with two long dimensions and one short), and 3D (bulky waste) particles. The 1D and 2D particles are generally considered as fibrous materials which contribute to the reinforcement of waste structure. Paper, cardboard, plastics, and food waste are highly compressible as compared to other components. Food waste is highly degradable, followed by garden waste, paper, cardboard, leather, and textiles. In this section, the particle compressibility is discussed.

The potential compressibility of particles is one of the key factors distinguishing MSW from most soils. In general, particles can reduce in volume (compress or crush) as a result of a compression of the solid phase or of gas trapped within the solid phase. Based on the soil particle forces classified by Santamarina (2001), waste particle forces may be categorized as:

(i) Forces due to applied boundary stresses, transmitted along granular chains that form within the waste skeleton.

(ii) Particle-level forces including particle weight, buoyancy, and hydrodynamic forces: in normal flow regimes, hydrodynamic forces are usually small relative to external and body forces.

(iii) Contact-level forces including suction, electrical forces, and biochemical forces; again, usually these are small in comparison with external, body, and buoyancy forces.

Wastes are mixtures of various components (highly heterogenous, Table 1) and will experience long-term hydro-biochemical-mechanical coupled processes after filling. The compositions and properties of the solid, liquid, and gas within wastes will vary with time as well as the interactions among them. Waste particle forces are believed to evolve accordingly. For sake of simplicity, the solid phase within the waste particles might be expected to compress by an amount ΔV_{ps} as a result of the changes in total stress, $\Delta\sigma$ and the surrounding pore fluid pressure, Δu (which, in an unsaturated waste, could be the pore water pressure, u_w, or the pore gas pressure, u_g, or a combination of the two as appropriate):

$$\Delta V_{ps} = f_1(\Delta\sigma, \Delta u_w, \Delta u_g), \tag{1}$$

where V_{ps} (m^3) is the volume of the solid phase within the particles; and σ (kPa) is the total stress.

An expression for the compression of gas trapped within a particle, ΔV_{pg}, as a result of an increase in its pressure of Δp was developed by Hudson et al. (2004):

Table 1 Waste component characteristics

Material group		Shape	Compressibility	Degradability
Organic	Paper/Cardboard	2D	High	Medium
	Food waste	3D	High	High
	Garden waste	2D, 3D	Medium	Medium
	Leather, textiles	1D, 2D	Medium	Medium
Inorganic	Plastics and rubber	1D, 2D	High	Low
	Metals, glass	3D	Low	Non
	Minerals	0D	Low	Non
Others*		0D	Low	Low

Note: * Indistinguishable components which are commonly very small and may be stuck together

$$\Delta V_{\mathrm{pg}} = V_{\mathrm{pg}} \frac{p_0 + 100}{(p + 100)^2} \Delta p, \tag{2}$$

where V_{pg} (m^3) is the initial volume of gas trapped within the particles; p (kPa) is the pressure of the gas within the particles; and p_0 (kPa) is a reference pressure at which the initial gas volume V_{pg} is measured. Hudson et al. (2004) expressed this relationship slightly differently in which they multiplied both sides of Eq. (2) by r_{gv} (i.e., the ratio V_{pg}/V, where V is the total volume of the element of waste containing the particles in which the volume of gas V_{pg} is trapped).

The relationship between the internal particle gas pressure p and the external stresses and pressures is complex, but following Hudson et al. (2004), it is reasonable to suppose that p could depend to some extent on each of the total stress, pore water pressure, and pore gas pressure, giving:

$$\Delta V_{\mathrm{pg}} = f_2(\Delta\sigma, \Delta u_{\mathrm{w}}, \Delta u_{\mathrm{g}}). \tag{3}$$

In most cases, the true change in the volume of solid material is likely to be negligible, with most of any reduction in particle volume arising from the compression or expulsion of gas initially enclosed within a vessel or sponge type object. However, in principle, volume changes in waste that arise from the compression of its particles will depend on some unknown combination of the total stress, pore water pressure, pore gas pressure, and other particle forces mentioned above. More generally, the same is likely to be true for changes in total volume that arise from a rearrangement of the structure of the waste as particles move relative to each other.

Currently, there is still a lack of particle level study on the volume change behaviours of wastes, and the effective stress law is generally applied. One of the key uncertainties and difficulties in the analysis of a medium with compressible particles and a compressible pore fluid is the definition of the effective stress σ' governing volume change in the sense that:

$$\frac{\Delta V}{V} = C\Delta\sigma', \tag{4}$$

where ΔV (m^3) is the total volume change; C (kPa^{-1}) is the overall compressibility; and $\Delta\sigma'$ (kPa) is the change in effective stress.

Skempton (1960) showed that, for an unsaturated porous medium with compressible particles, the effective stress controlling volume change may be written as:

$$\sigma' = \sigma - (1 - C_{\mathrm{s}} / C)[\chi u_{\mathrm{w}} + (1 - \chi) u_{\mathrm{g}}], \tag{5}$$

where, in addition to the parameters already defined, C_{s} (kPa^{-1}) is the compressibility of the particles; and χ is an experimentally-determined coefficient which depends on the degree of saturation, and varies from 1 (saturated) to 0 (dry).

For a saturated waste, χ=1 and Eq. (5) reduces to:

$$\sigma' = \sigma - (1 - C_{\mathrm{s}} / C) u_{\mathrm{w}} = \sigma - A u_{\mathrm{w}}, \tag{6}$$

where A=1–C_{s}/C.

This and some other potential impacts of particle compressibility on the concept of effective stress as applied to wastes were discussed by Powrie et al. (1999). On the basis of particle compressibilities measured in the large scale Pitsea compression cell (Beaven and Powrie, 1995) and following Skempton (1960), they suggested values of A in the range $0.19 \le A \le 0.57$. Interestingly, by using an effective stress defined according to Eq. (6) in the interpretation of triaxial test data on wastes, Shariatmadari et al. (2009) were able to improve significantly the compatibility between drained and undrained test data. Lü et al. (2017) found Eq. (6) could improve the prediction of pore water pressure in the consolidated undrained triaxial test data on wastes.

Consequences of Eqs. (1)–(6) are that: (1) volume changes in wastes may result from changes in the total stress σ, the pore water pressure u_{w}, and the pore gas pressure u_{g}; (2) even if the waste is saturated and the pore water is assumed to be incompressible, an immediate settlement may occur as a result of apparent particle compression.

3 Characterisation of settlement and strain

Conventionally, soil settlement is characterized by or related to a strain ε defined in simple terms as the change in specimen or stratum height divided by the initial height:

$$\varepsilon=\frac{\Delta h}{h},\tag{7}$$

where Δh (m) is the total specimen or stratum height change; and h (m^3) is the initial specimen or stratum height.

One potential difficulty with this for geological materials such as soils is that the definition of a state of zero strain is somewhat arbitrary. A further problem with wastes is that settlements can be large, so that there may be a significant difference, numerically, between the engineering strain (defined as the change in height over a finite increment of stress divided by the initial height, as above) and the true strain (defined as the integral of $\delta h/h$ over a series of infinitesimal increments δh). In interpreting data from laboratory tests on materials for which settlements can be a significant proportion of the original specimen height h, the dilemma is whether to re-set, at the start of each load increment, the initial height used in the determination of strain.

To overcome the need to select an arbitrary point as corresponding to zero strain, in soil mechanics it is customary to describe the volumetric state of the material by means of the void ratio e (defined as the volume of voids divided by the volume of solids) or specific volume v (defined as the total volume divided by the volume of solids). In conventional soil mechanics in which the particles may reasonably be assumed to be incompressible, the advantage of this is that the volume of solids V_s is constant. However, for a waste, V_s may change, or appear to change, because the particles are degradable or compressible. A reduction in the volume of solids ΔV_s may result in a reduction in total volume ΔV of anywhere between zero and ΔV_s, and possibly even more if the structure collapses further as a result of the loss of solid material.

Zhang et al. (2010), following Landva and Clark (1990), adopted an approach in which an appropriate proportion of the volume of solids is considered as incompressible, and a distinction is made between the voids between (inter-) and within (intra-) the solid phase. Some of the intra-voids are connected into the general void network, while some others are not. Intra-voids within a textile or a sponge would be connected to the general network, while air isolated and contained within a hollow particle would not (unless of course the particle was to break). This approach is consistent with that outlined earlier, with Eq. (1) relating to the compressibility of the solid phase (which is quite reasonably assumed to be negligible by Landva and Clark (1990) and Zhang et al. (2010)) and Eqs. (2) and (3) relating to the compression of the isolated (non-connected) intra-voids.

McDougall and Pyrah (2004) proposed a constitutive relationship between the change of solid volume ΔV_s and the induced change in void volume ΔV_v, of the form:

$$\Delta V_v=\Lambda\Delta V_s.\tag{8}$$

If the total volume remains constant, the void volume must increase by an amount ΔV_s to compensate; the void ratio will increase and $\Lambda=-1$. If the whole reduction in solid volume ΔV_s is reflected in a change in total volume by the same amount, the void volume will remain the same but the void ratio will increase, and $\Lambda=0$. If $\Lambda>0$, the void volume decreases but by less than the solid volume, so that the void ratio will increase. If $\Lambda=e$, the void ratio remains constant as the overall volume decreases. If $\Lambda>e$, the void ratio reduces because the loss of solid material leads to a proportionately greater reduction in void volume, perhaps as a result of structural collapse.

4 Immediate compression

What distinguishes immediate settlement from the primary consolidation is that it occurs when the waste contains air between or within the particles. Watts and Charles (1999) stated that for most practical purpose, the immediate compressibility of a waste can be described in total stress terms by the constrained modulus $E_{0,i}$, defined as:

$$E_{0,\mathrm{i}}=\frac{\delta\sigma_v}{\delta\varepsilon_v},\tag{9}$$

where $\delta\sigma_v$ (kPa) is the increment of vertical total stress; and $\delta\varepsilon_v$ is the corresponding increment of vertical strain.

Watts and Charles (1999) reported a case study involving the use of a sand preload 4 m high to compress a waste fill 12 m deep prior to the construction of a road in Liverpool, UK from which they deduced a value of $E_{0,i}$=3 MPa for the untreated refuse. Values at other sites varied between 0.7 MPa and 6 MPa, with those at the low end of the range corresponding to recent domestic refuse with a bulk unit weight of rather less than 10 kN/m^3. Siddiqui et al. (2013) reported an immediate compression of about 18%–20% of the waste height just prior to load application for their mechanically-biologically treated (MBT) waste A1 and 21%–23% for MBT waste A2. The values of $E_{0,i}$ were in the range of 217–267 kPa for the two MBT wastes, indicating a relative lack of compaction and initial trapped gas (unsaturation). Wall and Zeiss (1995) and Liao (2006) reported that the immediate stiffness $E_{0,i}$ tended to reduce with increasing water content, organic waste content, and void ratio.

In many cases, immediate compression may be of limited practical importance except insofar as it increases the volumetric capacity of the landfill, as it will occur as waste is being deposited and is likely to pass unmeasured and unnoticed. It is perhaps for this reason that mechanical compression models developed by many researchers (Park and Lee, 1997, 2002; Marques et al., 2003; Hossain and Gabr, 2005; Hettiarachchi et al., 2007; Park et al., 2007; Chen et al., 2010; Bareither and Kwak, 2015) have been based on the primary and secondary settlements of wastes following (not during) placement, with immediate compression not being considered.

5 Primary settlement

5.1 Rate

Primary settlement arises from the consolidation of a saturated waste as water flows out from the pores in response to load-induced non-equilibrium pore pressure gradients. While it may be of limited relevance to unsaturated wastes, there will almost always be some parts of a landfill that are saturated and for which consolidation is important. It seems reasonable, at least in the first instance, to analyse the process using the approach developed by Terzaghi (1923) and Clayton et al. (1995) for conventional saturated soils. Consolidation is time-dependent, and the relationship between settlement and time may be non-dimensionalised using the parameters:

$$T = c_v t / d^2, \tag{10}$$

$$R = \rho / \rho_{ult}, \tag{11}$$

where T is the time factor; c_v (m^2/s) is the consolidation coefficient (c_v=kE_0/γ_w); k (m/s) is the hydraulic conductivity of the waste (measured in the principal direction of drainage); E_0 (kPa) is the 1D modulus in effective stress terms (Eq. (17)); γ_w (kN/m^3) is the unit weight of pore water; t (s) is the elapsed time; d (m) is the maximum drainage path length; R is the proportional settlement; ρ (m) is the current settlement at time t; and ρ_{ult} (m) is the ultimate settlement at the end of primary consolidation.

Primary consolidation is complete when the pore pressures are no longer changing. The drainage boundary conditions will govern the shape of the dimensionless settlement against time (R vs T) curve (Powrie, 2004). The application of this approach to MBT wastes is demonstrated by Siddiqui (2011) and Siddiqui et al. (2013). Xu et al. (2016) proposed a simplified analytical model for the 1D consolidation of saturated MSW by considering degradation-dependent compressibility.

If wastes are unsaturated, the time t_p for the completion of primary compression is generally short. Bjarngard and Edgers (1990) reported t_p within 2 d for field case histories. In an existing landfill subjected to surcharge loading, t_p is generally within the first 30 d after load application (Wall and Zeiss, 1995). Bareither et al. (2012) applied a first-order rate equation to settlements measured in laboratory and field 1D compression tests to determine t_p. Values of t_p were about 3–4 h for fresh MSW (F-MSW) under compression from 32 kPa to 64 kPa in the laboratory tests, and 12–16 d for fresh MSW (DTBE) at depths up to 6.85 m in the field tests. Xu et al. (2015) conducted long-term laboratory compression tests on fresh MSW specimens under different sustained constant external loads. Taking the inflexion point of the measured ε_v-lg$t$ curve to represent t_p, the value of t_p was estimated to be about 7 d for these wastes. Shi et al. (2016) suggested t_p of 8 h in laboratory compression tests on shredded fresh MSW.

5.2 Magnitude

For saturated soils, equilibrium states at the end of consolidation are conventionally presented as graphs of void ratio e or specific volume v against the natural logarithm (ln) or the logarithm to base 10 (lg) of the vertical effective stress σ'_v. The expectation is that the line joining equilibrium states in first compression will be linear in the e (or v): lgσ'_v (or lnσ'_v) plane, although some authors (Butterfield, 1979) have proposed that both axes should be logarithmic. The slope of the straight line joining equilibrium states defines a material parameter known as the compression index. If the vertical effective stress is plotted as lg, the compression index is conventionally given the symbol C_c. If natural logarithms are used, the equivalent symbol in the e (or v): lnσ'_v plane is λ. The corresponding parameters in unloading are C_s and κ, respectively. For the wastes, the compression index C_c is widely applied to estimate the primary compression.

$$C_c = \frac{-\Delta e}{\Delta(\lg \sigma'_v)} = \frac{e_0 - e}{\lg(\sigma'_v / \sigma'_{v0})}, \tag{12}$$

$$\lambda = \frac{-\Delta e}{\Delta(\ln \sigma'_v)} = \frac{e_0 - e}{\ln(\sigma'_v / \sigma'_{v0})}, \tag{13}$$

where σ'_{v0} (kPa) is the initial (pre-existing) vertical effective stress; and e_0 is the initial void ratio.

As mentioned above, Zhang et al. (2010) pointed out the difficulty of applying these equations to wastes, in which the volume of solids as conventionally defined is not constant. Following Landva and Clark (1990), they defined an incompressible volume of solids V_I and distinguish between the inter- and intra-particle void ratios e and f, according to Eqs. (14) and (15).

$$e = V_{\text{v-inter}} / V_I, \tag{14}$$

$$f = V_{\text{v-intra}} / V_I, \tag{15}$$

where $V_{\text{v-inter}}$ and $V_{\text{v-intra}}$ (m^3) are the volumes of inter- and intra-particle voids, respectively.

This is a rather major limitation to the use of Eqs. (12) and (13) to model primary compression of MSW (Sowers, 1973; Landva et al., 1984; Morris and Woods, 1990).

The vertical strain ε_v that takes place in response to an increase in vertical effective stress from σ'_{v0} to σ'_v from an initial void ratio e_0 is then given by

$$\begin{aligned} \varepsilon_v &= -\frac{\Delta h}{h_0} = \frac{-\Delta e}{1+e_0} = \frac{\lambda}{1+e_0} \ln\left(\frac{\sigma'_v}{\sigma'_{v0}}\right) \\ &= \frac{C_c}{1+e_0} \lg\left(\frac{\sigma'_v}{\sigma'_{v0}}\right) = C_c' \lg\left(\frac{\sigma'_v}{\sigma'_{v0}}\right), \end{aligned} \tag{16}$$

where C_c' is the modified primary compression index.

Perhaps the major advantage of using the modified primary compression index, C_c', in preference to C_c or λ, is that the definition of C_c' does not involve the void ratio and can be determined experimentally directly from measurements of strain. Thus, notwithstanding earlier remarks about the need for care in defining strain and a strain datum, it avoids the complication of defining and measuring the void ratio in a material with compressible particles, and both connected and non-connected intra-particle voids.

It is always possible to characterise an increment of vertical strain $\delta\varepsilon_v$ in response to an increment of vertical effective stress $\delta\sigma'_v$ by means of an equivalent 1D or constrained modulus E_0, where

$$E_0 = \delta\sigma'_v / \delta\varepsilon_v. \tag{17}$$

Comparison of Eq. (17) with Eqs. (12), (13), and (16) in differentiated form shows that:

$$E_0 = \frac{\sigma'_v(1+e_0)}{\lambda} = \frac{2.3\sigma'_v(1+e_0)}{C_c} = \frac{2.3\sigma'_v}{C_c'}. \tag{18}$$

Thus, unlike the various compression indices C_c, λ, and C_c', E_0 must be expected to vary with vertical stress. For this reason, it can only reasonably be taken as constant over a defined (and arguably small) range of stress and strain.

It is possible to identify a normal compression line (i.e. a unique relationship between vertical strain and vertical effective stress in the first loading) for wastes (Beaven and Powrie, 1995). However, in reality many wastes will be compacted to an uncertain degree by machines during placement in the landfill. The void ratio immediately following placement will reflect this degree of compaction, with greater compaction resulting in a larger initial density and smaller amounts and rates of settlement as further waste is deposited on top (El-Fadel and Khoury, 2000). One-dimensional mechanical compression tests on specimens prepared with different initial unit weights and void ratios by Machado et al. (2002) gave apparent primary compression indices C_c varying between 0.52 and 0.92, increasing with void ratio. However, the corresponding value of modified primary compression index C_c' was approximately constant (0.21); thus, the effect of different initial void ratios, as well as the need to define or determine this parameter, was removed.

Data from 1D compression tests (cell diameter of 300 mm) on fresh waste specimens having the same composition (typical of Hangzhou, China in the early 21st Century, as shown in Table 2) but different initial void ratios are presented in Fig. 1. The initial void ratio is determined using the method suggested by Bear (1972) and probably corresponds to the ratio of the inter-plus connected intra-voids to the current solid volume. Values of primary compression index C_c vary with the initial void ratio, as indicated by the differences in slope in Fig. 1a. However, the graphs of vertical strain against $\lg\sigma'_v$ in Fig. 1b all have a similar slope after the first loading increment, indicating that the modified primary compression indices are substantially the same and, as found by Machado et al. (2002), independent of void ratio. This is confirmed by the values of C_c and C_c' plotted against the initial void ratio e_0 in Fig. 2.

The apparent dependence of the compression index, C_c, on void ratio and the apparent independence of C_c' are perhaps surprising, given that the first would be expected to be the more fundamental parameter. However, these results should be treated with caution, as either or both may be an artifact of the way that the void ratio has been determined and by inference defined, together with the neglect of the fact that the volume of solids, V_s, is not constant as assumed in Eq. (16).

Beaven and Powrie (1995) reported results from 1D compression tests carried out on a variety of wastes in a large scale (2 m diameter) purpose built cell. The compositions of the wastes tested are summarised in Table 3, and results are shown in Fig. 3 (e-$\lg\sigma'_v$) and Fig. 4 (E_0-σ'_v). Although immediate and primary compressions are usually most significant for

Table 2 Composition of fresh Hangzhou MSW tested to obtain the primary compression data (wet basis, %)

Component	Value	Component	Value
Food	67.7	Wood	0.6
Plastic and rubber	8.5	Cinders and dust	13.7
Paper	6.7	Glass and metals	1.5
Textiles	0.6	Others	0.7

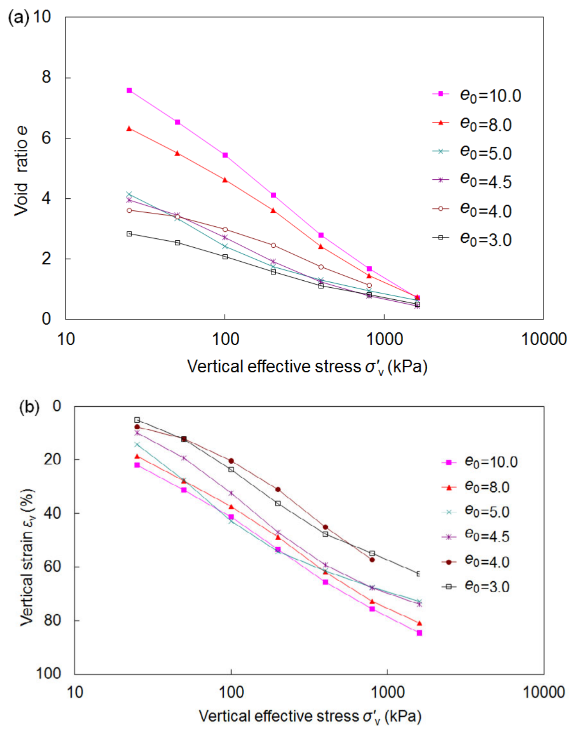

Fig. 1 Primary compression curves for fresh Hangzhou MSW with different initial void ratios: (a) e-$\lg\sigma'_v$; (b) ε_v-$\lg\sigma'_v$

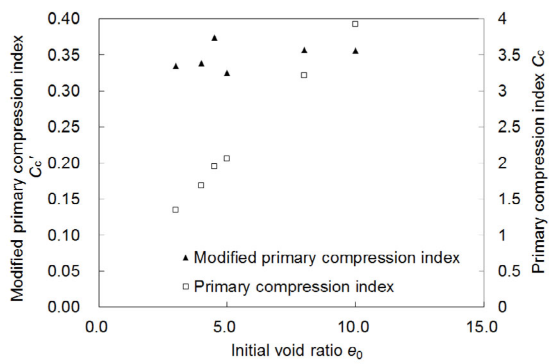

Fig. 2 Relationship between primary compression index/ modified primary compression index and initial void ratio (data from Fig. 1)

a fresh waste during and shortly after placement, there are circumstances in which these types of compression may be important for an aged waste, for example, when it is proposed to create additional capacity at a facility by placing fresh waste on top of an established landfill site (a "piggy back" landfill). The data reproduced in Fig. 3 indicate that aged and degraded waste (about 20 years old) is stiffer at a given vertical effective stress than raw or pulverised domestic waste. Fig. 4 confirms that the constrained modulus will generally increase with vertical effective stress, and cannot be considered constant.

Table 3 Compositions of wastes tested by Beaven and Powrie (1995) (dry basis, %)

Component	Waste category			
	DM2	DM3	PV1, PV2	AG1
Paper/Card	35.1	39.8	49.0	13.7
Plastic film	5.7	4.4	8.3	3.0
Dense plastics	4.4	6.4	7.8	1.8
Textiles	2.6	5.5	5.7	0.8
Miscellaneous combustibles	9.1	11.8	4.6	9.5
Miscellaneous non-combustibles	1.0	2.4	1.1	2.5
Glass	5.5	7.0	1.3	7.5
Putrescibles	22.0	13.2	6.5	23.3
Ferrous metals	7.0	3.2	9.0	3.9
Non-ferrous metals	1.5	1.2	1.6	0.1
Fines <10 mm	6.2	4.9	5.2	33.9

Note: DM2, DM3 are raw wastes, PV1, PV2 are pulverised wastes, and AG1 is aged wastes (about 20 years old) in UK

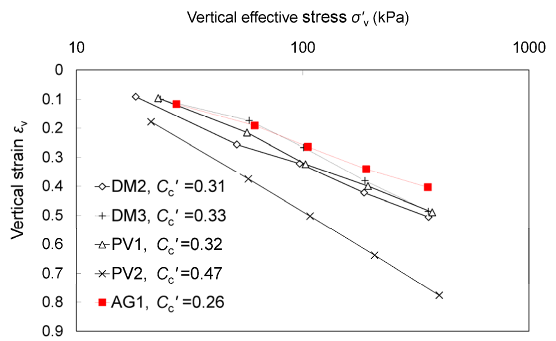

Fig. 3 ε_v vs lgσ'_v for different wastes, showing values of C_c' (data from (Beaven and Powrie, 1995))

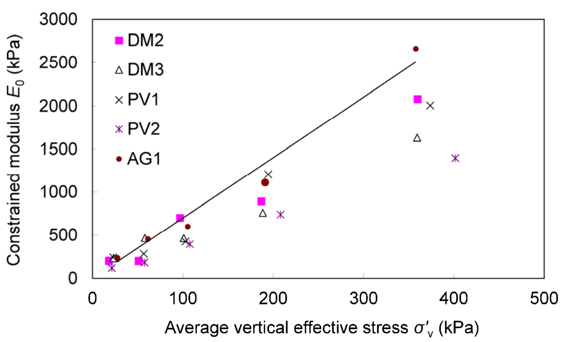

Fig. 4 E_0-σ'_v for different wastes (data from (Beaven and Powrie, 1995))

Bareither et al. (2012) reported a relationship between C_c' and vertical (total) stress. The value of C_c' was calculated incrementally with respect to the accumulated strain for each additional vertical stress increment. For fresh waste (F-MSW, F-R25, and DTBE) and decomposed waste (LD, MD, and HD), C_c' increased with vertical stress, up to a vertical stress of about 100 kPa. They suggested evaluating C_c' over either: (i) a broad stress range covering the expected field stresses, or (ii) a stress range where C_c' is approximately constant.

Values of modified primary compression index, C_c', taken from the literature, are summarised in Table 4.

Liao (2006) and Liao et al. (2007) reported some dependence of the modified primary compression index on the organic content of the waste, until this reaches 60%. It was attributed to the high compressibility of organic materials. Data on the mechanical properties of materials commonly found in wastes collated by Dixon and Langer (2006) confirm that organic matter, paper, and wood are relatively compressible (low compressive strength and low modulus of elasticity). The effect of particle compressibility on the bulk compressibility of a waste has already been mentioned.

Liao et al. (2007) found that the water content has little influence on the modified primary compression index. While these tests all started from the same initial void ratio, differences in void ratio were established immediately on loading–an effect attributed by Liao et al. (2007) to the differences in water contents. The modified primary compression indices obtained by Reddy et al. (2009) and Basha et al. (2016) from compression tests on fresh and borehole shredded waste with different initial water contents showed more scatter, but no consistent trend.

Zekkos et al. (2017) reported a decreasing trend of C_c' with increasing percentage of <20-mm material and dry unit weight γ_d. The waste composition, including the type of fibrous waste constituent, was also found to affect C_c'. For soil-waste mixed specimens compressed in the direction parallel to the waste constituent orientation, C_c' increased with the amount of plastic and paper. It was the greatest for soil-plastic mixtures, followed by soil-paper mixtures, and practically unaffected by the amount of wood fibers. They also found C_c' to depend on the relative orientation of the fibrous constituent long axis to the direction of loading. Bareither et al. (2012) also suggested that the effect of bulky waste constituents (e.g. paper, plastic, and wood) must be considered in evaluating C_c'.

Bareither et al. (2012) reported little difference in C_c' between fresh and decomposed specimens. This may be an artifact of the re-mixing process. Bareither et al. (2012) suggested that in reconstituting specimens in the laboratory compression cells, the remaining bulky waste constituents and organic solids were distributed so as to create a waste matrix of similar compressibility to fresh MSW. Tests on shredded waste reported by Hossain et al. (2003) indicated an increase in the primary compression index C_c with decomposition. However, as the initial void ratio data was not reported, the effect of decomposition on C_c' in this study was unknown. Data from Beaven and Powrie (1995), Bareither et al. (2012), and Xu et al. (2015) indicated that waste decomposed under a sustained load tends to have a smaller C_c' than the same MSW when fresh. This is probably a result of densification due to degradation-induced compression, and the loss of compressible solid components (Bareither et al., 2012).

Fig. 5 shows data from Chen et al. (2009) from tests on specimens of MSW obtained from boreholes at the Qizishan landfill, China. The data are divided into two broad groups corresponding to two different ranges of initial void ratio e_0: $0.5 \le e_0 \le 2.0$ and $2.0 \le e_0 \le 4.2$. Although the results display considerable scatter, within each group there is a general trend of decreasing C_c' with increasing waste age. The same trend is reported by Mokhtari et al. (2019).

Table 4 Values of modified primary compression index C_c' reported in the literature

Reference	Description of waste	Waste specimen	C_c'
Beaven and Powrie (1995)	Raw (DM2, DM3), pulverised (PV1, PV2) and aged (AG1, about 20 years old) wastes in UK	DM2, DM3	0.31–0.33
		PV1, PV2	0.32–0.47
		AG1	0.26
Liao (2006) and Liao et al. (2007)	Synthetic waste typical of cities in China, with a range of organic content (OC), tested dry	OC=0	0.08
		OC=30%	0.24
		OC=60%	0.30
		OC=80%	0.32
		OC=100%	0.31
Liao (2006) and Liao et al. (2007)	Synthetic waste typical of cities in China, tested at different initial water contents w	w=0	0.30
		w=30%	0.29
		w=50%	0.30
		w=70%	0.29
		w=132%	0.28
Reddy et al. (2009)	Fresh shredded waste from Orchard Hills landfill in USA	w=44%	0.28
		w=60%	0.25
		w=80%	0.33
		w=100%	0.24
Basha et al. (2016)	Fresh shredded waste from Orchard Hills landfill in USA	w=44%	0.039
		w=60%	0.033
		w=80%	0.042
		w=100%	0.027
	Borehole shredded waste from Orchard Hills landfill in USA	w=44%	0.027
		w=60%	0.028
		w=80%	0.028
		w=100%	0.038
Bareither et al. (2012)*	Reconstituted decomposed waste	Low decomposed	0.183–0.235
		Medium decomposed	0.173–0.257
		High decomposed	0.193–0.234
Bareither et al. (2012)	Fresh USA waste decomposed under sustained load	Fresh	0.27
		Decomposed	0.18
Xu et al. (2015)	Fresh Chinese waste decomposed under sustained load	Fresh	0.20
		Decomposed	0.10

Note: *The lower and upper limit values of C_c' are for the vertical stress ranges of 8–64 kPa and 12–400 kPa, respectively

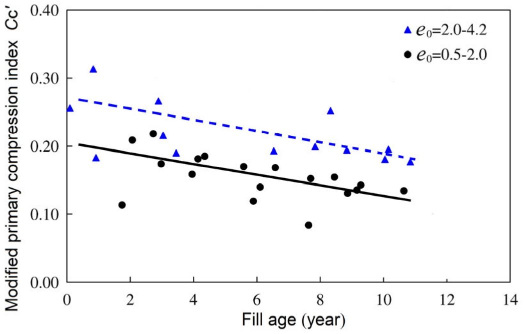

Fig. 5 Variation of C_c' with initial void ratio e_0 and fill age. Reprinted from (Chen et al., 2009), Copyright 2009, with permission from Elsevier

Estimation of C_c' from waste properties was discussed by Bareither et al. (2012) and Zekkos et al. (2017). A dimensionless compressibility index WCI related to the water content w_d (defined as the ratio of the mass of water to the mass of dry solids), dry unit weight γ_d, and organic waste content OW (again, by dry mass) of waste was proposed to estimate C_c' by Bareither et al. (2012). Zekkos et al. (2017) found that C_c' was better correlated with γ_d than the percentage of <20 mm material. However, there was considerable scatter in both studies indicating that further research on such relationships is required.

As shown in Table 4, the values of C_c' reported by Basha et al. (2016) are much smaller than those in other literature. This might result from the relatively small size (63 mm diameter) of the oedometer used in their study. Bareither et al. (2012) indicated that the wastes tested in 64 mm diameter cells had smaller C_c' values than those obtained in 100 mm and 305 mm diameter cells. In view of the possible effects of relative apparatus to waste particle size, the use of test cells >300 mm in diameter is suggested to evaluate waste compressibility (Bareither et al., 2012).

6 Secondary settlement

The two processes responsible for the secondary settlement, i.e. mechanical creep and degradation, are separable and need not be linked; it is therefore appropriate to consider them separately.

6.1 Mechanical creep

The secondary settlement due to mechanical creep is generally modelled using a simple equation of the form:

$$\mathrm{d}\varepsilon_c / \mathrm{d}t = \alpha_c / t, \tag{19}$$

where α_c is a constant; and ε_c is the creep strain.

Integrating Eq. (19) gives a linear relationship between creep strain and the natural logarithm of time:

$$\varepsilon_c = \alpha_c \ln \frac{t}{t_m}, \tag{20}$$

where t_m is a reference time for mechanical creep.

The relationship implied by Watts and Charles (1999) is $\varepsilon_c = C_\alpha \lg(t/t_m)$, where $\alpha_c = C_\alpha$ divided by ln(10)=2.303 to account for the difference between natural logarithms and logarithms to base 10.

Eq. (20) is a simplification of the form of equation used to describe creep in soils at constant effective stress (Mesri et al., 1981; Mitchell, 1993):

$$\varepsilon_c = G t_m \exp\left(\frac{\alpha q}{q_f}\right) \ln \frac{t}{t_m}, \tag{21}$$

where G and α are empirically-determined creep parameters; q (kPa) is the deviator stress; and q_f (kPa) is the deviator stress that would cause failure. A non-zero reference time is needed to avoid the calculation of infinite creep strains. The creep strain rate increases with deviator stress, and may also depend on the temperature although this is not explicit in Eq. (21).

For well-controlled laboratory tests, the value of the reference time t_m will depend on the interpretation of the data used to obtain the other creep model parameters. In (Bareither et al., 2013), the value of t_m was set equal to t_p, as determined using the method of Bareither et al. (2012). The inflexion point in the measured ε_v-lg$t$ curve (with curve fitting) was also used to represent t_m in some studies (Zekkos et al., 2017). For field measurements, the value of t_m might not be so obvious yet it may be crucial to the determination of creep settlements. For example, the creep-induced secondary compression after 100 years calculated using Eq. (20) for a given value of α_c and t_m=1 d is about 1.5 times that for t_m=1 month. Sharma and De (2007) suggested that in practice, the value of t_m for calculating creep settlement may be taken as 3–4 months, on the basis that this is the timescale over which the primary settlement usually occurs. This approach should be treated with some caution and on a case-by-case basis: Eq. (10) shows that the time for the primary (consolidation) settlement will depend on the landfill depth, and it may in reality be difficult to distinguish between various types of settlement as they occur in the field especially on an operational landfill site.

One of the problems in interpreting the long-term settlement data reported in the literature is that secondary settlement due to mechanical creep and degradation are usually not distinguished. This could be a feature of the experimental approach, the interpretation of the data, or both. Ivanova et al. (2008) reported laboratory tests on specimens of MSW carried out in consolidating anaerobic reactors (CARs). The effects of creep and degradation were separated through the operation of a "control" reactor in which degradation was prevented by the addition of acetic acid. Fig. 6 shows data of settlement against lg$t$ measured in the three CARs. Full details of these experiments are given by Ivanova et al. (2008). CAR3 was the control reactor, in which degradation was successfully inhibited for the first 350 d or so of operation. This is evidenced by the gas generation curves shown in Fig. 7.

The slope of the initial graph of settlement against lg$t$ for CAR3 was used to determine the creep parameter C_α. Assuming that creep effects are the same in CAR1 and CAR2 as they are in CAR3, the subtraction of the creep settlements from CAR1 and CAR2 enables the degradation settlement to be eliminated (Fig. 8). Siddiqui et al. (2013) adopted the same method as Ivanova et al. (2008) to separate the effects of creep and degradation in their studies of MBT waste.

Table 5 summaries values of creep index α_c reported in the literature from experiments in which degradation was deliberately inhibited, considered not to be occurring or otherwise eliminated from the data e.g. by means of a model. These values range from 0.3% to 4.9%.

Park and Lee (2002) reported that the value of the creep index α_c is comparatively insensitive to the overburden pressure and waste density, but it is not clear how representative their data were (see the note in Table 5). The limited data from Ivanova et al. (2008) tend to support this, and any effect could be one of density rather than overburden stress per se. Similar

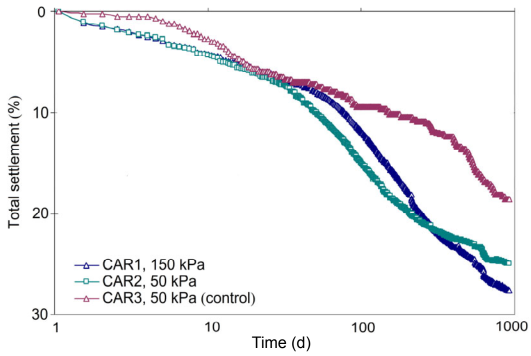

Fig. 6 Settlement against lg(time) observed in consolidating anaerobic reactor experiments. Reprinted from Ivanova et al. (2008), Copyright 2008, with permission from ICE Publishing

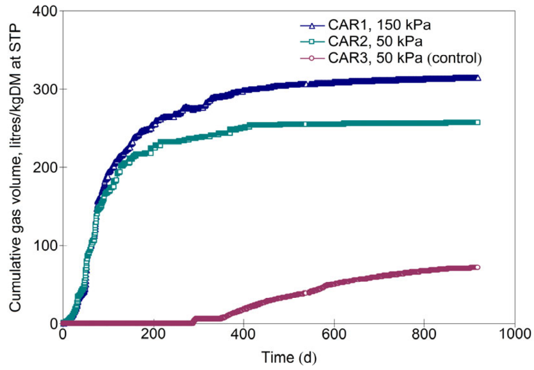

Fig. 7 Cumulative gas production against time observed in consolidating anaerobic reactor experiments. Reprinted from Ivanova et al. (2008), Copyright 2008, with permission from ICE Publishing

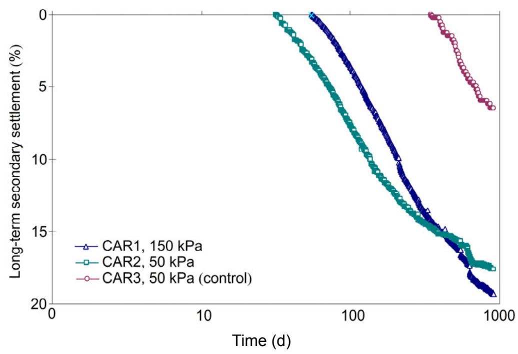

Fig. 8 Estimated degradation-induced settlement against lg(time) (creep settlements removed) observed in consolidating anaerobic reactor experiments. Reprinted from Ivanova et al. (2008), Copyright 2008, with permission from ICE Publishing

Table 5 Values of creep parameter α_c from experiments in which degradation was inhibited or accounted for empirically or using a model in the literature

Reference	Description of experiments			α_c (%)
Zacharof and Coumoulos (2001)	Control cell of Mountain View landfill project was analyzed for creep by splitting total settlement into two curves distinguished by slopes			0.608–1.13
Park and Lee (2002)	Determined creep parameters by fitting their own model to previous studies. Park and Lee (2002) argued that decomposition did not affect these tests as they lasted only a few days, in which case it is uncertain how representative the measured creep was			1.042–1.602 (fresh waste); 0.347–0.434 (old waste)
Hossain et al. (2003)	Waste collected from a transfer station was shredded, and then was used to obtain specimens in different states of decomposition		Specimens B1–B4 were saturated with 6% acetic acid to prevent decomposition during compression	0.651–1.302
			Specimens C1–C4 were neither seeded nor neutralized (while leachate was recirculated) to retard the onset of methane production	0.868
Benson et al. (2007)	Settlement monitoring at landfill sites: creep was distinguished from degradation-induced settlement by slope of settlement-lg$t$ curve			1.13–1.26
Ivanova et al. (2008)	Fresh waste obtained from White's Pit Landfill, UK. Degradation was successfully inhibited in CAR3 for the first 350 d which is evidenced by the measured gas generation			2.18
Chen et al. (2010)	Artificial specimens were of typical waste composition in Hangzhou, China (Table 2). Degradation was inhibited			0.308–0.521
Siddiqui et al. (2013)	Degradation was inhibited for the entire duration of the tests[a]		Aerobically degraded MBT	0.69–0.91
			Anaerobically-aerobically degradated MBT	0.65–0.78
Bareither et al. (2013)	Mechanical creep prior to the onset of degradation	Fresh MSW[b]	Fresh MSW	1.56–1.65
			Fresh MSW passing the 25-mm screen	0.87–0.95
			Fresh MSW retained on the 25-mm screen	1.82–2.26
		Fresh MSW[c]	Fresh MSW	2.08–2.43
		Reconstituted decomposed waste[d]	Low decomposed	0.91–1.30
			Medium decomposed	1.43–1.86
			High decomposed	1.26–1.56
Shi et al. (2016)	Degradation was inhibited for the staged compression tests (12.5–1600 kPa) on shredded fresh MSW			3.4–4.9

Note: [a]The lower and upper limit values of α_c are for the vertical stress of 150 kPa and 50 kPa, respectively. [b]The cell diameter is 305 mm. The lower and upper limit values of α_c are for the vertical stress of 400 kPa and 64 kPa, respectively. [c]The cell diameter is 2440 mm. Data obtained from the settlement plates at different depths. [d]The cell diameter is 305 mm. The lower and upper limit values of α_c are for the vertical stress of 64 kPa and 400 kPa, respectively

α_c were obtained by Bareither et al. (2013) for wastes under 64 kPa and 400 kPa. Chen et al. (2010) and Zekkos et al. (2017) reported a decreasing trend of α_c with increasing density or unit weight.

Zekkos et al. (2017) reported the decrease of α_c with the increase of the percentage of <20-mm material. For soil-waste mixed specimens compressed in the direction parallel to the waste constituent orientation, they found that α_c increased with the increase of the amount of plastic and paper, and was the greatest for soil-paper mixtures, followed by soil-plastic mixtures, and then wood.

Park and Lee (2002) reported a decrease in α_c with age, from 1.042–1.602 for fresh waste to 0.347–0.434 for 15–30 years old waste. In contrast, Hossain et al. (2003) found α_c to be independent of the state of waste decomposition, while Ivanova et al. (2008) found no difference in α_c as the waste degraded. In (Bareither et al., 2013), for the reconstituted decomposed waste with different degrees of degradation, α_c showed no clear relationship with decomposition. However, the data of all the fresh and decomposed wastes showed a decrease of α_c with decreasing ratio of cellulose plus hemicelluloses to lignin.

For the prediction of α_c or C_α, Bareither et al. (2013) reported a relationship between C_α and WCI. Laboratory studies of Zekkos et al. (2017) indicated that the ratio of C_α to C_c' for MSW is between 0.01 and 0.04. The data (cell diameter of 305 mm) of Bareither et al. (2012, 2013) indicated that the ratios of C_α (creep measured before the onset of degradation) to C_c' for the fresh and the reconstituted decomposed wastes are within a range of 0.08–0.24 and 0.11–0.19, respectively.

6.2 Degradation

Many authors, remarking on the similarity of the time-dependent nature of creep and degradation-induced settlements, amalgamate the effects of creep and degradation into a single equation of the same form as Eq. (22) or (23). This is intellectually unsatisfactory, because the rate of degradation will depend on factors such as the water content of the waste and the leachate management regime, which would not affect the rate of creep in the same way or to the same extent. The apparent success of this approach when applied to real landfill sites is probably a result of the fact that the operational regime and the type of waste were similar at various sites considered.

Watts and Charles (1999) described the degradation-induced strain using a relationship of the form:

$$\varepsilon_b = \alpha_b \ln \frac{t}{t_b} = C_b \lg \frac{t}{t_b}, \tag{22}$$

where ε_b is the degradation-induced compression strain at a time t after the start of degradation; t_b is a reference time (generally considered as the onset time of degradation) for the degradation-induced compression; and $\alpha_b = C_b/2.303$.

Eq. (22) is similar in form to the expression given in Eq. (20) for mechanical creep, but allows the two phenomena to be considered independently. A remaining drawback is the inherent assumption that degradation-induced settlement has no limit, whereas eventually it will substantially cease.

Other, perhaps more satisfactory models for longer-term degradation-induced waste settlement have been developed on the basis of the first-order degradation kinetics (Edgers et al., 1992; Wall and Zeiss, 1995; Park and Lee, 1997, 2002; Marques et al., 2003; Liu et al., 2004; Hettiarachchi et al., 2005; Park et al., 2007; Elagroudy et al., 2008; Bareither et al., 2013; Xu et al., 2015). These generally take the form:

$$\varepsilon_b = \varepsilon_{bt}[1 - \exp^{-k_b(t-t_b)}], \tag{23}$$

where ε_{bt} is the total strain potentially resulting from degradation; and k_b is a degradation rate constant.

The degradation rate constant for settlement modeling is related to the decomposition rate of degradable matter in wastes, but does not necessarily take the same value or form for different waste types. This is illustrated by the considerable variation shown in Fig. 9. Elagroudy et al. (2008) found Eq. (23) to be applicable to the secondary compression of wastes with different compositions, with or without the addition of sewage sludge or enzymes, in different operational conditions; however, it is not clear whether they accounted separately for creep and biodegradation.

One disadvantage of Eq. (23) is that it is necessary to estimate the potential total degradation-induced secondary compression strain, ε_{bt}. The value of ε_{bt} will increase with increasing degradable matter content (Wardwell and Nelson, 1981). Siddiqui et al. (2013) found that ε_{bt} was well related to the methane generation potential. Values of k_b and ε_{bt} reported in the literature for various real and simulated wastes are summarised in Table 6.

In reality, the rate of degradation will depend on the way in which a landfill is managed and operated. In a MSW landfill in which the waste remains relatively dry, stabilization may take several hundred years (McBean et al., 1995; Hall et al., 2004). In a bioreactor landfill, stabilization could occur within 10 years or less (Reddy and Bogner, 2003). Assuming that stabilisation corresponds to 99% of the potential degradation settlement having occurred, inserting a range of stabilisation times of 10 to 1000 years into Eq. (23) implies values of the degradation rate constant k_b between 10^{-5} d^{-1} and 10^{-3} d^{-1}. The latter is reasonably consistent with the published values summarised in Table 6, which are for experimental conditions favourable to degradation (e.g. the waste is shredded, mixed with sewage sludge, seeded with enzymes or nutrients; the leachate is recirculated; the landfill gas is extracted).

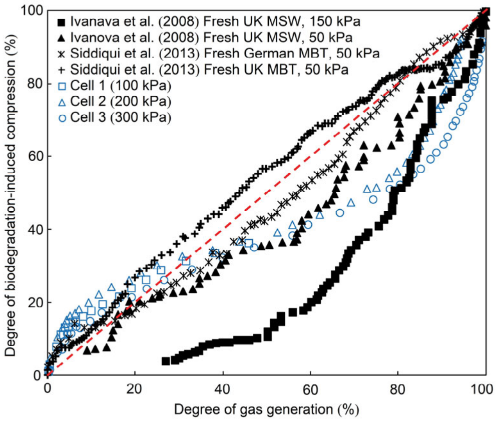

Fig. 9 Relationship between the degrees of degradation-induced settlement and gas generation. Reprinted from Xu et al. (2015), Copyright 2015, with permission from SAGE

7 More advanced models coupling degradation and settlement

The approach of decoupling and using simple models to describe mechanical and degradation settlements described above is advocated by Gourc et al. (2010), who also investigated parameter values for wastes of different degradable contents and operational regimes. An evaluation of MSW settlement model performance and applicability was conducted by Bareither and Kwak (2015) based on the analysis of two field-scale datasets. They found that using the uncoupled approach involved the lowest number of total and optimized model parameters and gave statistically high performance for all applications. However, when using these models, the uncertainty of the governing factors remains due to the application of different initial conditions, boundary conditions, and experimental procedures. Therefore, most researchers suggest simulating the field conditions in the experiments in order to obtain a better determination of model parameters.

Table 6 Values of degradation rate k_b and total degradation-induced compression strain ε_{bt} in the literature

Reference	Description of experiments		k_b (d^{-1})	ε_{bt} (%)
Rao et al. (1977)[a]	Fresh MSW	An experimental cell (3 m× 3 m square, 1.5 m high)	1.0×10^{-3}	11
Gandolla et al. (1992)[a]	Fresh MSW	Shredded, mixed with sewage sludge	1.0×10^{-3}–1.1×10^{-3}	21
Marques et al. (2003)	Fresh MSW	A test fill covers an area of about 8400 m^2 and is about 4 m high	9.51×10^{-4}–1.10×10^{-3}; average: 1.14×10^{-3}	13–22; average: 16
Ivanova (2007)	Fresh MSW, CAR1	Vertical stress 150 kPa	3.7×10^{-3}	17
	Fresh MSW, CAR2	Vertical stress 50 kPa	6.1×10^{-3}	13
	Aged MSW, CAR4	Vertical stress 50 kPa	7.3×10^{-3}	5
Bareither et al. (2013)[b]	Fresh MSW	Cell diameter 305 mm; vertical stress 64 kPa	1.2×10^{-2}–1.6×10^{-2}	12–16
	Fresh MSW [c]	Cell diameter 2440 mm; vertical stress 20–67 kPa	2.3×10^{-3}–2.9×10^{-3}	18–21
	Fresh MSW passing the 25-mm screen	Cell diameter 305 mm; vertical stress 64 kPa	1.4×10^{-3}–1.3×10^{-2}	14–44
	Fresh MSW retained on the 25-mm screen	Cell diameter 305 mm; vertical stress 64 kPa	5.8×10^{-3}–6.5×10^{-3}	19–23
	Reconstituted decomposed waste	High decomposed cell diameter 305 mm; vertical stress 64 kPa	3.5×10^{-3}	4.9
Siddiqui et al. (2013)	MBT waste		2.03×10^{-2}	1.7
	Raw MSW	CAR1 at 150 kPa	4.4×10^{-3}	13.8
		CAR2 at 50 kPa	4.9×10^{-3}	11.2
Xu et al. (2015)	Fresh MSW	Vertical stress 100–400 kPa	5×10^{-3}–1.1×10^{-2}	10.9–16.9

Note: [a]From Park and Lee (2002); [b]Includes both mechanical creep and degradation-induced compression; [c]Data obtained from the settlement plates at different depths

In recent years, however, a number of more fundamental models coupling the settlement of a waste to its rate of degradation have been developed. Principal amongst these are those described by Haarstrick et al. (2001), Machado et al. (2002, 2008), White et al. (2004), McDougall (2007), and Lobo et al. (2008). A review of these models, and their efficacy in modelling the behaviours of the CAR experiments reported by Beaven et al. (2008) and Ivanova et al. (2008), was presented by Beaven (2008). As shown in Fig. 10, these hydro-biochemical-mechanical coupled models consider the links between the micro-level and macro-level responses. The micro-level responses generally include (i) consumption or generation of solid, liquid, and gas phases due to degradation; (ii) solid adsorption/desorption, solute transport, and multi-component gas transport; (iii) phase change between solid, liquid, and gas phases (see the dashed curved lines in Fig. 10). Evolution of particle forces (e.g. causing diagenesis, coagulation or flocculation) may also be a factor. The macro-level responses (i.e., solid skeleton deformation, overall pore liquid flow, and overall pore gas flow) are the results of the initial and boundary conditions and the micro-level responses.

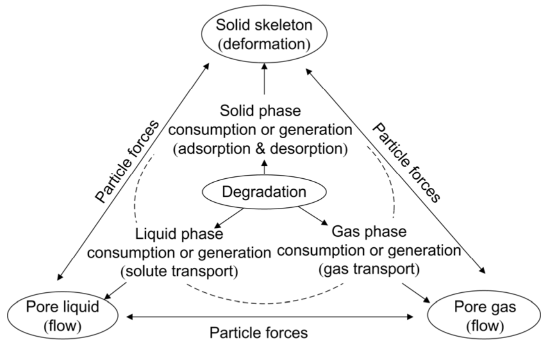

Fig. 10 Processes of mass change and transport in wastes (modified from (Chen, 2014))

Such a coupled approach offers the potential for a major step forward in our ability to model and predict the behaviour of a landfilled waste, by linking settlement and gas production rates to waste composition, water content, and operational regime. Such models can be used, for example, to estimate the increase in settlement and leachate levels within a landfill in response to the deposition of further waste on top ("piggy back" landfilling); to estimate the remaining gas potential of a waste to assess the economic viability of installing a gas extraction system; to investigate the ability of the waste to store leachate; or to predict the response of the waste to a change in operational regime. However, obtaining the parameters needed to run such models can be problematic. Though the coupled models consider the micro-level degradation of wastes, the kinetics describing the degradation processes and their effects on settlement are still generally back-analyzed from macro-scale testing results of the laboratory or field experiments. The coupled models are not considered further in this paper, but are discussed by Beaven (2008), McDougall (2008), and Reddy et al. (2017).

8 Conclusions

Settlements of landfilled wastes may arise from a variety of mechanisms not normally considered significant in conventional, saturated soil mechanics. These include the effects of particle and pore fluid compression. Application of the concepts of conventional soil mechanics is further hampered by the difficulty of defining and measuring an appropriate void ratio. This arises because (i) owing to the effects of particle compression and degradation, the volume of solids is unlikely to be constant; (ii) in addition to the voids between the particles (termed inter-voids), the voids within the particles (intra-voids)–only some of which will be connected to the inter-void network–must be considered. Pragmatically and practically, this difficulty can be circumvented by characterising the settlement behaviour of a waste by means of the strain rather than the void ratio. Any interpretation of compression data for wastes based on the assumption that the volume of solids remains constant should be treated with caution.

Immediate settlement (compression) of a landfilled waste may arise due to compression of the particles and/or compression or expulsion of gas from the pores. Medium- and long-term settlements may arise as a result of consolidation (termed primary consolidation or primary settlement), and mechanical creep and/or biodegradation (together termed secondary settlement). While the effects of creep and degradation in terms of settlement against time might be superficially similar, the mechanisms are quite distinct;

a sound understanding of long-term waste settlements requires that they are considered separately.

Immediate compression may be characterised by means of a total stress constrained modulus, $E_{0,i}=\delta\sigma_v/\delta\varepsilon_v$. In reality immediate compression may occur as waste is being deposited and thus may pass un-noticed. In cases where immediate compression has been monitored, reported values of $E_{0,i}$ range between 0.2 MPa and 6 MPa, with lower stiffnesses generally associated with lower density, and increasing water and organic contents.

The primary settlement results from the consolidation of a near-saturated waste and may be characterised by means of the consolidation coefficient, $c_v=kE_0/\gamma_w$, and E_0 will generally increase with vertical effective stress, waste age, and waste density. Over a wider range of stress, the amount of primary compression $\Delta\varepsilon_v$ in response to an increase in vertical effective stress $\Delta\sigma'_v$ may be characterised directly by the modified primary compression index, $C_c'=\Delta\varepsilon_v/\Delta(\lg\sigma'_v)$. This avoids the need to define and quantify the relevant void ratio. Data from the literature indicate a range of C_c' of generally 0.20 to 0.47 for wastes having a significant organic component.

The secondary settlement due to creep may be characterised by an equation of the form $\varepsilon_c=\alpha_c\ln(t/t_m)$. It is not always clear whether in the literature the effects of creep have been adequately separated from those of degradation, and some discrepancy may arise from differences in the value of the reference time t_m. Setting at least the second of these aside, reported values of creep parameter α_c range from 0.3% to 4.9% for loose, fresh waste. There is no real evidence of any dependence of the creep parameter on stress, but there may be a density effect, with denser wastes being less susceptible to creep.

The secondary settlement due to degradation may be characterised by an equation of the form $\varepsilon_b=\varepsilon_{bt}[1-\exp(-k_b(t-t_b))]$. The amount of degradation-induced settlement will depend on the proportion of degradable material present, while the rate of degradation will depend on the way in which the landfill is managed and operated. Values reported in the literature for laboratory tests on fresh MSW in which the effects of creep and degradation-induced settlement have been separated generally range from 11% to 22% for ε_{bt}, and 9.5×10^{-4} to 1.1×10^{-2} d^{-1} for k_b. In real landfills that are operated in relatively dry conditions, full degradation could take some hundreds of years, implying k_b as low as 10^{-5} d^{-1}.

The methods on which this paper has focussed are pragmatic and empirical, but coupled with appropriate scale laboratory tests offer the potential to predict medium and long-term landfill settlements due to consolidation, degradation, and creep. A developing and more fundamental approach is to couple the waste settlements, degradation, liquid flow, and gassing. Such models are versatile and show great promise, although obtaining the required parameters is more challenging.

Contributors

William POWRIE and Xiao-bing XU designed the research and processed the corresponding data. David RICHARDS, Liang-tong ZHAN, and Yun-min CHEN contributed to the development of the ideas proposed. William POWRIE and Xiao-bing XU revised and edited the final version.

Conflict of interest

William POWRIE, Xiao-bing XU, David RICHARDS, Liang-tong ZHAN, and Yun-min CHEN declare that they have no conflict of interest.

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中文概要

题　目： 城市固体废弃物填埋场沉降机理

目　的： 帮助填埋场工程方面的学者和工程师更全面地了解城市固体废弃物的沉降机理。

创新点： 揭示填埋场城市固体废弃物的沉降机理，明确沉降计算模型的参数确定方法和不确定因素。

结　论： 1. 填埋场城市固体废弃物的沉降机理不同于传统土体，这主要是固相颗粒的可压缩性、可降解性以及孔隙流体的可压缩性不同导致的。2. 可以将沉降划分为瞬时压缩、主压缩、蠕变次压缩和降解次压缩，并分别进行计算；这些计算模型虽然带有经验性，但具有很好的实用性。3. 能更好反映沉降机理的模型需要耦合填埋场城市固体废弃物的降解、压缩和液气运移。4. 这些耦合模型能分析更复杂的工况，但其模型参数的确定更具挑战性。

关键词： 城市固体废弃物；降解；沉降；压缩；蠕变