A comparison of small-scale, pilot-scale and large-scale tests for predicting leaching behaviour of landfilled wastes

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Abstract

Landfills generate emissions over long periods, often longer than a lifetime. The longest lasting emission is leachate. In order to estimate the future requirements for leachate treatment, different kinds of leaching tests may be applied. In this paper, shaking leaching tests (SLT), landfill-simulator leaching tests and a field-cell leaching test performed with ash, municipal solid waste (MSW) and MSW + ash are evaluated. The tests are compared and the factors influencing leaching are identified and discussed. The factors are: liquid to solid (L/S) ratio, water withdrawal, recirculation rate, presence or absence of biological processes, size of particles, duration of experiment, temperature and pre-treatment of the waste. The presence of biological processes has the greatest impact on leaching and is the main reason why SLT is less useful for long-term predictions. The landfill simulator tests were found to be useful for several different kinds of predictions. However, they are not reliable for predicting the L/S required for reaching a certain concentration. The possibilities for reliable long-term predictions would be facilitated by a better knowledge of the influence of various factors on leaching. Such an increased knowledge would make it possible to enhance waste stabilisation in leaching tests as well as in full-scale landfills.

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1. Introduction

The composition of waste disposed at landfills changes over time. An ever-greater share of the material is not easily degradable. However, landfills constructed and filled in the past contain such materials. Those landfills are generating emissions and will probably do so for several decades or even centuries (Belevi and Baccini, 1989; Ehrig, 1988; Kylefors, 1997; Andreas, 2000).

Leachate is the longest lasting emission from landfills. Hence, it will determine the required time for emission treatment and control. In order to estimate the time required for leachate treatment, different methods are applied today, ranging from small-scale shaking tests and simulator tests to large-scale field tests. It is reasonable to assume that the more similar the test is to full-scale applications, the closer the results of the test will be to the future real emissions from landfills. Yet, regardless of the methods used, there will be differences between predicted and observed future behaviour. In order to design leaching tests that are reliable for long-term predictions, it is essential to know what factors influence the leaching and how they influence the leaching. Knowledge of these factors may also help to determine the appropriate means of enhancing waste stabilisation in landfills.

The following questions will be addressed in this paper:

- How is leaching influenced by different factors?
- What are the possibilities and limitations of various kinds of leaching tests as prediction tools?

In order to discuss these questions, literature references and results of leaching experiments are used. The result and discussion section will be structured according to the two questions.

2. Materials and methods

2.1. Data

Data were collected from several different projects and experiments (see Table 1). The chosen data are from projects with experiments performed with two
Different kinds of leaching tests for the same waste material. Another criterion for the chosen data was that the waste material should be commonly deposited in MSW landfills. The data cover both different scales of leaching as well as different kinds of waste (see Table 2).

Many different variables have been assessed in the different projects; not all are included in this study. The ones used in this paper are presented in Table 3. The metals chosen are examples of metals that differ in behaviour under anaerobic conditions. Chromium is an example of a metal that tends to form hydroxides, zinc is an example of a metal that preferably forms sulphides and nickel is an example of a metal that occurs in many different forms including organic complexes.

### 2.2. Waste materials

Three different kinds of waste were studied: ash, MSW + ash and MSW. The chemical composition of the different waste materials is presented in Table 4. More detailed description of the wastes can be found in the references given in Table 1. None of the ashes originated from MSW incineration.

The ash was a bottom ash from wood-chip incineration. It had relatively high organic content (VS content of 23% of TS), indicating low efficiency of the incineration process. This wood ash was used in shaking leaching tests (SLT) and simulator tests. The waste used in the two scales of tests came from the same sampling occasion.

### Table 1
Sources of data

<table>
<thead>
<tr>
<th>Project name</th>
<th>Kind of waste</th>
<th>Main reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Enzyme project</td>
<td>MSWa</td>
<td>Lagerkvist and Chen (1993)</td>
</tr>
<tr>
<td>GANSCA</td>
<td>Ashb</td>
<td>Bergman (1996)</td>
</tr>
<tr>
<td>East Germany</td>
<td>MSW + ashc</td>
<td>Andreas (2000)</td>
</tr>
<tr>
<td>Sunderbyn</td>
<td>MSW</td>
<td>Lagerkvist (1995)</td>
</tr>
</tbody>
</table>

a Municipal solid waste.

b Ash refers to bottom ash from wood-chip incineration.

c Ash refers to bottom ash from the burning of brown coal.

### Table 2
Tests performed with the different wastes

<table>
<thead>
<tr>
<th></th>
<th>Shaking leaching tests</th>
<th>Simulator</th>
<th>Test cell</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>One step</td>
<td>Sequential</td>
<td></td>
</tr>
<tr>
<td></td>
<td>L/S 4a</td>
<td>L/S 10b</td>
<td>ENAa</td>
</tr>
<tr>
<td>MSW</td>
<td>X</td>
<td>X</td>
<td></td>
</tr>
<tr>
<td>MSW + ash</td>
<td>X</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Ash</td>
<td>X</td>
<td>X</td>
<td>X</td>
</tr>
</tbody>
</table>

a Swedish standard method according to SNV (1985).

b German standard method according to DIN 38 414.

### Table 3
Analytical measures used for the different wastes

<table>
<thead>
<tr>
<th>Category of substance</th>
<th>Wood bottom ash</th>
<th>MSW + ash</th>
<th>MSW</th>
</tr>
</thead>
<tbody>
<tr>
<td>Salt</td>
<td>Conductivity</td>
<td>Chloride</td>
<td></td>
</tr>
<tr>
<td>Organic material</td>
<td>VS (COD)</td>
<td>COD, TOC</td>
<td></td>
</tr>
<tr>
<td>Nitrogen</td>
<td>N\textsubscript{Kj}</td>
<td>COD, TOC</td>
<td></td>
</tr>
<tr>
<td>Metals and sulphur</td>
<td>Cr, Ni, Zn, S</td>
<td>Cr, Ni, Zn</td>
<td>Cr, Ni, Zn</td>
</tr>
</tbody>
</table>

VS = volatile solids; COD = chemical oxygen demand; TOC = total organic carbon; N\textsubscript{Kj} = Kjeldahl nitrogen.

### Table 4
Composition of the different wastes (average values± standard deviation)

<table>
<thead>
<tr>
<th>Variable</th>
<th>Unit</th>
<th>MSW</th>
<th>MSW + ash</th>
<th>Ash</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fines &lt; 8 mm</td>
<td>%</td>
<td>69±8\textsuperscript{a}</td>
<td>320</td>
<td></td>
</tr>
<tr>
<td>TS</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>VS</td>
<td>g/kgTS</td>
<td>700\textsuperscript{b}</td>
<td>149±56</td>
<td>228</td>
</tr>
<tr>
<td>FS</td>
<td>g/kgTS</td>
<td>300</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cellulose</td>
<td>g/kg TS</td>
<td>347</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Hemicellulose</td>
<td>g/kg TS</td>
<td>77</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lignin</td>
<td>g/kg TS</td>
<td>170</td>
<td></td>
<td></td>
</tr>
<tr>
<td>C</td>
<td>g/kg TS</td>
<td>405</td>
<td>114±33</td>
<td></td>
</tr>
<tr>
<td>TOC</td>
<td>g/kg TS</td>
<td>78±32</td>
<td></td>
<td></td>
</tr>
<tr>
<td>H</td>
<td>g/kg TS</td>
<td>57</td>
<td></td>
<td></td>
</tr>
<tr>
<td>N</td>
<td>g/kg TS</td>
<td>9.5</td>
<td>2.6±0.9</td>
<td>1–4\textsuperscript{c}</td>
</tr>
<tr>
<td>S</td>
<td>g/kg TS</td>
<td>1.7</td>
<td>25±14</td>
<td>0.78</td>
</tr>
<tr>
<td>AI</td>
<td>g/kg TS</td>
<td>86</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Si</td>
<td>g/kg TS</td>
<td>145</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ca</td>
<td>g/kg TS</td>
<td>48±20\textsuperscript{d}</td>
<td>100</td>
<td></td>
</tr>
<tr>
<td>Fe</td>
<td>g/kg TS</td>
<td>51±17\textsuperscript{d}</td>
<td>29</td>
<td></td>
</tr>
<tr>
<td>K</td>
<td>g/kg TS</td>
<td>29</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mg</td>
<td>g/kg TS</td>
<td>7.0±3.2\textsuperscript{d}</td>
<td>10</td>
<td></td>
</tr>
<tr>
<td>Mn</td>
<td>g/kg TS</td>
<td>0.71±0.26\textsuperscript{d}</td>
<td>5.7</td>
<td></td>
</tr>
<tr>
<td>Na</td>
<td>g/kg TS</td>
<td>6.3</td>
<td></td>
<td></td>
</tr>
<tr>
<td>P</td>
<td>mg/kg TS</td>
<td>6.2</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ti</td>
<td>mg/kg TS</td>
<td>1445</td>
<td></td>
<td></td>
</tr>
<tr>
<td>As</td>
<td>mg/kg TS</td>
<td>3.27</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ba</td>
<td>mg/kg TS</td>
<td>1720</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Be</td>
<td>mg/kg TS</td>
<td>3.63</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cd</td>
<td>mg/kg TS</td>
<td>0.197</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Co</td>
<td>mg/kg TS</td>
<td>5.30</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cr</td>
<td>mg/kg TS</td>
<td>26.7</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cu</td>
<td>mg/kg TS</td>
<td>59</td>
<td></td>
<td></td>
</tr>
<tr>
<td>La</td>
<td>mg/kg TS</td>
<td>25.9</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mo</td>
<td>mg/kg TS</td>
<td>&lt;6.03</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ni</td>
<td>mg/kg TS</td>
<td>25.8</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pb</td>
<td>mg/kg TS</td>
<td>10.8</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sn</td>
<td>mg/kg TS</td>
<td>7.84</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sr</td>
<td>mg/kg TS</td>
<td>454</td>
<td></td>
<td></td>
</tr>
<tr>
<td>V</td>
<td>mg/kg TS</td>
<td>27.1</td>
<td></td>
<td></td>
</tr>
<tr>
<td>W</td>
<td>mg/kg TS</td>
<td>&lt;12</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Y</td>
<td>mg/kg TS</td>
<td>20.1</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Yb</td>
<td>mg/kg TS</td>
<td>2.51</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Zn</td>
<td>mg/kg TS</td>
<td>331</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Zr</td>
<td>mg/kg TS</td>
<td>117</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Hg</td>
<td>mg/kg TS</td>
<td>117</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

\textsuperscript{a} The three most recently generated wastes were not included.

\textsuperscript{b} Determined at 775 °C.

\textsuperscript{c} Estimated value.

\textsuperscript{d} Based on analyses of 10 out of 19 wastes.
The MSW consisted of household waste from the municipalities of Luleå, Piteå and Boden, all in Sweden. The waste can be regarded as typical Swedish household waste, i.e. rich in organic material. It was collected in October 1988 and used in simulator tests and in field tests. The waste was influenced neither by the wet autumn garden waste nor by the increase in dry matter that is normal at the end of the year.

The MSW + ash waste was derived from four landfills in former East Germany. It was collected at different depths and varied in age between 6 and 12 years. The ash accounts for a large portion of the waste and is predominantly bottom ash from the burning of brown coal. The MSW + ash waste was used in SLT and simulator tests. The two kinds of tests were performed with waste from the same sampling occasion.

2.3. Test procedure

What follows is a brief description of the different tests. For more detailed information, consult the references given in Table 1.

2.3.1. Shaking leaching tests

Shaking leaching tests (SLT) refer to short-term tests involving the rotation of bottles containing waste and deionised water. These tests include both one-step and sequential leaching tests.

The one-step leaching tests were performed at two different L/S ratios. In the test described by SNV (1985) 125 g TS was mixed with 500 ml of water, resulting in an L/S of 4. In the DIN 38414 test a waste mass of 100 g TS was added to a water volume of 1000 ml, resulting in an L/S of 10. In both cases, 21 bottles were used and rotated for 24 h.

A multistep leaching test was obtained by a modification, according to Andreas (2000), of the one-step leaching test S4 (DIN 38 414-S4). The water was withdrawn after each step (24 h) and replaced by the same amount of fresh deionised water until the concentration in the leachate reached the limits of analysis for the variables in question.

The ENA sequential test described by SNV (1985) consisted of a batch test in four steps in two different series. Both series started with the one-step test at L/S 4 as described above. In the first series the waste mass was exchanged while the water was retained, ending up at an L/S of 1. In the second series the waste mass was retained while the water was exchanged, leading to L/S ratios of 8, 12 and 16.

The tests with the ashes were run as single tests; i.e. without replicates. The tests with MSW + ash were performed with different strategies regarding replication, dominated by duplicates of the tests (seven wastes). Six of the wastes were performed in four parallel experiments, one waste in triplicate and five wastes without replicates.

2.3.2. Simulator tests

The landfill simulators were cylindrical steel containers of about 100 l. The simulators were filled with about 80 l of waste, which corresponds to 50–90 kg TS, depending on the kind of waste. Water was added to about the field capacity of the waste. Additional water was added and recirculated. Some of the recirculating water was exchanged at certain intervals. On these occasions, leachate samples were taken for analyses. Gas production and, to some extent, gas composition were also measured.

Two parallel reactors were run in the projects with ash and only MSW, respectively. The simulators with MSW were both acidogenic. The experiments with MSW + ash were performed in 19 different simulators. Two of those simulators were run with waste from the same sampling point and occasion. All the other simulators were experiments without replicates.

2.3.3. Field tests

The data from field tests are from a pilot test cell for two-step anaerobic degradation in Sunderbyn, Sweden. The first step of the plant consisted of an acidogenic leaching cell and the second step of a methanogenic filter for the digestion of the acidogenic leachate. The first step, the leaching cell, contained 457 tonnes of MSW with a density of 0.65 tonnes/m³. The filling height of the waste was 2.8 m and the soil covering amounted to 1.3 m. The bottom area of the cell was 152 m² and the top area with waste was 354 m². A leachate recirculation system was installed in half of the cell. The other half acted as a reference cell. Bottom and surface lysimeters were installed. The degradation within the cell was acidogenic. The second step, the methanogenic filter, was barely three cubic metres in volume. The filter material consisted of two layers of gravel separated by a layer of shredded MSW. The second step is, however, not part of this study.

2.4. Calculations

2.4.1. L/S

L/S is the liquid over solid ratio. The following definitions have been used in the calculations of L/S:

\[ \text{L} = \text{water content of the waste at the start of the experiments} + \text{added water} + \text{infiltrated water} \quad [\text{kg}] \]

\[ \text{S} = \text{dry substance of the waste included in the experiments} \quad [\text{kg TS}] \]

Instead of the input amount of water to the system, i.e. the amount of added and infiltrated water, it may be possible to use the output amount of water, i.e. the amounts of collected leachate, in the calculations. Both variants have been applied earlier, and both have
advantages and disadvantages. Here, we used the amount of added and infiltrated water. The amount of collected leachate is here discussed in terms of the influence of water withdrawal.

2.4.2. Water withdrawal

Water withdrawal refers to the water that is taken out of the system. In order to compare the concentrations of the leachate in different leaching test systems, an adjustment was made presuming that no water had been removed. The concentration adjusted for the amount of substances taken out by leachate of the system, $C_{\text{adj},i}$, has been calculated according to:

$$
C_{\text{adj},i} = \left( \sum m_{\text{sample},(i-1)} + C_i \cdot V_{\text{water}} \right) / (V_{\text{water}} + \sum V_{\text{sample},(i-1)}) \text{ [mg/l]} 
$$

where $m_{\text{sample},(i-1)}$ = amount of substance withdrawn by water samples at the $(i-1)$ occasion [mg], $C_i$ = measured concentration in the $i$th sample [mg/l], $V_{\text{water}}$ = volume of free water in the reactor [l], $V_{\text{sample},(i-1)}$ = volume of withdrawn water at the $(i-1)$ occasion [l].

3. Results and discussion

3.1. How is leaching influenced by different factors?

The results from the different leaching tests show different results. The comparison between both shaking leaching tests (SLT) and simulator experiments (Figs. 1–3) and simulators and field test cell (Fig. 4) indicate discrepancies. The reasons for the different leaching results can be found in the design of the experiments. The factors that differed in levels between the tests were identified as:

- L/S ratio
- water exchange
- biological degradation
- recirculation/mixing
- duration of experiment
- size of particles
- temperature
- sample preparation/storage.

The levels of the factors in the single tests considered are compiled in Table 5. The general influence of the factors will be discussed in the following.

3.1.1. L/S ratio

The L/S ratio is a very important factor for leaching. A larger addition of water (increased L/S) improves the possibilities for potentially soluble material to dissolve.
<table>
<thead>
<tr>
<th>Factor</th>
<th>SLT (MSW + ash; ash)</th>
<th>Simulators (MSW + ash; ash)</th>
<th>Simulators (MSW)</th>
<th>Field test cell (MSW)</th>
</tr>
</thead>
<tbody>
<tr>
<td>L/S</td>
<td>1–16 (ash)</td>
<td>1.9–3.1 (ash)</td>
<td>1.4–5.8</td>
<td>1.0–3.5</td>
</tr>
<tr>
<td></td>
<td>10–63 (MSW + ash)</td>
<td>0.2–5.1 (MSW + ash)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Water withdrawal</td>
<td>≤ 500 ml/step (ash)</td>
<td>0.4 l weekly to monthly (ash)</td>
<td>0.2 l/sampling, 5–20 l occasionally (twice a month to every second month)</td>
<td>0.7 l/sampling (weekly over 3.5 years), less frequent during winter at the end. In addition a nine month period (corresponding to L/S 1.90 to 2.26) with 100–2400 l/week.</td>
</tr>
<tr>
<td></td>
<td>≤ 1300 ml/step (MSW + ash)</td>
<td>Normally 1–2 l/twice a week to every second week. Occasionally up to 20 l/sampling (MSW + ash)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Biological degradation</td>
<td>Probably not/undefined</td>
<td>Yes</td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Environment</td>
<td>Not defined, at least at start aerobic</td>
<td>Anaerobic. For ashes: aerobic start, but turns anaerobic</td>
<td>Anaerobic, acidogenic</td>
<td>Anaerobic, acidogenic</td>
</tr>
<tr>
<td>Size of particles</td>
<td>&lt; 15 mm (ash)</td>
<td>&lt; 15 mm (ash)</td>
<td>&lt; 10 mm</td>
<td>As arriving at the test cell, in plastic bags</td>
</tr>
<tr>
<td></td>
<td>&lt; 100 mm (MSW + ash)</td>
<td>&lt; 300 mm, most particles</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>&lt; 80 mm (MSW + ash)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sample preparation/storage</td>
<td>Drying at 105–110 °C, storage</td>
<td>None</td>
<td>Grinding, storage in freezer, mixing</td>
<td>None</td>
</tr>
<tr>
<td>Mixing/recirculation</td>
<td>Rotating bottles at 0.5 rpm (ash) or 1 rpm (MSW + ash)</td>
<td>Recirculation. In the order of 150 l kg TS⁻¹ year⁻¹</td>
<td>Recirculation of about 140 l kg TS⁻¹ year⁻¹</td>
<td>Recirculation of 0.6–1.8 l kg TS⁻¹ year⁻¹</td>
</tr>
<tr>
<td>Duration of experiment</td>
<td>24–96 h (ash)</td>
<td>403 days (ash)</td>
<td>604 days</td>
<td>2058 days (5.6 years)</td>
</tr>
<tr>
<td></td>
<td>24–168 h (MSW + ash)</td>
<td>260–980 days (MSW + ash)</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
The leaching of substances that are predominantly characterised by dissolution processes can therefore be expected to show clear trends when results are presented in relation to L/S.

### 3.1.2. Water exchange

The amount of water, or more accurately, the amount of substances taken out of the system by leachate, influences the concentration as well as the mass-flow development with L/S, since substances taken away from the system can no longer contribute to leaching. An exchange of water in several steps will lead to an exponentially decreasing concentration with L/S, if the equilibrium with the solid phase is neglected.

The decrease in concentration due to water withdrawal is affected by the amount of water that is exchanged (same amount added and withdrawn) on each occasion. A less frequent but larger amount of water exchange generates a faster decrease in concentration with L/S than a more frequent exchange with smaller amounts of water. This is exemplified in Table 6 by the difference between the situations at 0.2 and 20% of the total water content exchanged at each instance of exchange. The situation at 0.2% could be considered normal in a landfill without recirculation in the Nordic climate zone. The hundred-fold increase in water withdrawal intensity (from 0.2 to 20% in each instance) leads to a halving of the concentrations at L/S 5. Another fact worth noticing is that the influence of water withdrawal on the concentration level increases with L/S. At L/S 1, there is little effect of water withdrawal. However, at L/S 5 there is an evident influence of water withdrawal with concentrations of less than 2% of those concentrations at ‘zero withdrawal’.

The decrease in concentration with L/S, due to the withdrawal of substances, is also affected by variations in the exchanged water amounts. A temporary increase in the exchanged amounts of water will generate a temporary decrease in concentration that does not fit the exponentially decreasing trend with L/S. It will also generate a long-lasting effect, i.e. a lower concentration level than without the temporary increase in exchanged amounts. As an example: there is about a 20% temporary decrease in concentration as an effect of an increase of exchanged amounts of water from 2 to 20%. When returning to the “normal situation” of 2% exchange, there will still be about 2% lower concentration than without the temporary increase in exchanged amounts. One explanation for the frequently observed fluctuations in field data might hence be found in an uneven water exchange.

### 3.1.3. Biological processes

The presence or absence of biological processes has a major influence on leachate quality. The biological processes influence environmental conditions like redox potential and pH, which are essential for the availability of material at biological degradation as well as for chemical processes.

It is not just the presence or absence of biological activity that influences leaching. The kind of biological activity also affects the leachate quality. At anaerobic degradation, two major degradation phases exist—the acidogenic and the methanogenic. The acidogenic phase is characterised by high organic content, often in the range of tens of g/l. The methanogenic leachate has a much lower organic content, up to a few g/l. The difference for nitrogen is not as apparent, but has been observed by Kylefors and Lagerkvist (1997). Several metals have also been observed to occur in higher concentrations under acidogenic conditions (Ecke, 1997). However, the concentration of some metals can under some circumstances be higher in methanogenic leachate (Kylefors and Lagerkvist, 1997). Finally, there are substances with no observed difference between the two phases. Examples of such substances are salts like chloride, potassium and sodium (Ehrig, 1989).

### 3.1.4. Recirculation and mixing

Recirculation and mixing are techniques that increase the contact between waste, water, microorganisms and nutrients. Mixing is most effective in so-called “dry” degradation systems, i.e. systems with high TS content. This can be concluded from Chen et al. (1995), who found that mixing enhances degradation most when the water content and the amounts of bacteria are limited. This means that without mixing, the moisture content or the supply of substrate and nutrients can have a limiting effect on the biological degradation.

Recirculation of leachate promotes biological activity by increasing and equalizing the moisture content, permitting a good contact between microbes, substrate and nutrients, and carrying away degradation products. The promoted biological activity affects the flow paths of the water. As degradation proceeds, it weakens the structure of the waste, channels within the waste will collapse (Reinhart and Townsend, 1998), and thus the water finds new pathways. Flow paths may also be reopened when gas production declines, as gas partially blocks the pores in the landfill. Other factors forcing the water to

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**Table 6**
The expected concentration at different L/S ratios and frequencies of water withdrawal

<table>
<thead>
<tr>
<th>L/S ratio</th>
<th>Frequency of water withdrawal (% of total water content, per occasion)</th>
<th>Percentage of conc. at ‘zero withdrawal’</th>
<th>Percentage of conc. at ‘0.2% water withdrawal/occasion’</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>0.2</td>
<td>94</td>
<td>–</td>
</tr>
<tr>
<td>1</td>
<td>20</td>
<td>89</td>
<td>96</td>
</tr>
<tr>
<td>5</td>
<td>0.2</td>
<td>1.7</td>
<td>–</td>
</tr>
<tr>
<td>5</td>
<td>20</td>
<td>0.9</td>
<td>50</td>
</tr>
</tbody>
</table>
find new flow paths are chemical precipitation and biological clogging in the pores of the waste body.

According to Townsend et al. (1996), laboratory studies, landfill lysimeters and controlled landfill cells have all demonstrated that increasing the moisture content and practicing leachate recycling have a positive effect on waste stabilisation. The degree of enhancement can be significant. In 1982 Klink and Ham (see Chugh et al., 1998) noted e.g. an increase in methane production rate of 25–50% in experiments with recirculation compared to those without.

The rate of recirculation is important for the enhancement of degradation. A larger volume of recirculated leachate will, according to the results of Chugh et al. (1998), promote the establishment of methanogenic conditions as well as increasing the methane production rate. They used a daily recirculation of 2, 10 and 30% of the initial volume of waste loaded in the reactor. However, it is unclear whether the increased methane production rate was due to the increased recirculation rate only or if it is an effect of the increased inoculation of methanogenic bacteria as well. During an initial period, the recirculating water was inoculated with methanogenic bacteria by circulation of the leachate over a reactor with stabilised waste (exhausted of its methane producing potential). The amount of water may hence not be the only reason for an increased degradability due to recirculation (in this case). The quality, including possible seeding, of the leachate may be important too. Contrary to this, Bogner and Spokas (1995) could not discern in laboratory experiments that the kind of water circulated had any impact. However, these experiments had an initial seeding that was probably sufficient to establish a good environment for degradation. In the case of a larger application, it may be useful to seed the waste volume with inoculated leachate, in order to enhance biological degradation.

3.1.5. Duration

It is essential that the experiment is long enough in duration to allow chemical equilibriums to be established and to enable the establishment and continuation of biological reactions.

The time required to establish chemical equilibriums depends largely on the water content and the size of the waste particles. Fällman and Aurell (1996) found that a larger particle size and a higher L/S ratio would increase the required time. According to Fällman (1997), a period of 24 h is enough to affect the conditions within the pores of small ash particles (< 125 μm) after a change in environmental conditions, like a decrease of pH. However, the leached amounts of pH-sensitive metals like lead, copper and iron were still increasing after 30 h of leaching (Fällman, 1997), indicating that 24 h is too short a time for leaching tests, even for waste material dominated by inorganic compounds.

A suitable duration of experiments with biological degradation depends largely on the technical system of the leaching test (design). The retention time of both the solids and the water is essential. In completely mixed systems, like the SLT, the retention time for the solids is just as long as the hydraulic retention time. Other test designs are based on the idea of prolonging the retention time for solids. This may be done by recirculation of the solids after sedimentation or by the use of filter media for immobilisation of the microorganisms. There have been observations of optimal hydraulic retention time for anaerobic reactors with immobilised microorganisms. The effect of the hydraulic retention time on those systems is, however, not easy to distinguish from the effect of the organic load. Fang and Yu (2000) claimed that the optimal hydraulic retention time under conditions of acidogenic leaching was 12 h. The concentrations of the degradation products were, however, still increasing after 24 h, but at a much slower rate than during the first 12 h. An optimal retention time of 12–24 h has also been observed in completely mixed systems for acidogenic degradation (Elefniotis and Oldham, 1994; Pavan et al., 2000).

3.1.6. Size of particles

Smaller particle size will lead to larger surface area, which promotes the leaching of both organic and inorganic material.

Palmowski and Muller (2000) studied the influence of particle size of different organic wastes on anaerobic degradability. They found that the digestion time is shortened for all the studied wastes, ranging from mixed vegetables and meat to hay, when the particle size is reduced. They compared waste fractions of about 2 cm in diameter with ground material in the order of micro-to millimetres. The effect was most apparent for harder degradable material, in this case hay and leaves. They also noticed increased degradability in the harder degradable material when the particle size was smaller. The increase was in the order of 15–20% and was registered by an increased amount of collected gas. Muller et al. (1998) also found an increased degradability of 10–20% as a result of disintegration, in this case of sewage sludge. Reinhardt and Ham (1973) and Tittlebaum (1982) found significantly higher organic leachate content from shredded MSW compared to non-shredded MSW. This indicates that dissolved organic material in the leachate had accumulated due to the increased availability of the shredded waste. Reinhardt and Ham (1973) noted that, in spite of the higher initial organic concentrations, test cells with shredded waste (5–16 cm) became methanogenic faster than cells with unprocessed waste. Test cells with unprocessed waste remained acidogenic through the whole study, which lasted several years.
3.1.7. Temperature

Temperature affects chemical equilibriums as well as biological activity.

Solubility generally increases with temperature. The effect of the temperature is compound-dependent. A solubility increase of about 10%, when the temperature rises from 20 to 30 °C, can be considered a maximum increase, based on information from temperature-dependent curves for several different salts presented by Petrucci (1985).

It is well known that temperature influences the rate of biological reactions (van’t Hoff’s rule). Over limited temperature ranges, the rate of biological reactions would be expected to double for each 10 °C increase. The acidogenic degradation, however, seems to be less temperature sensitive. Pavan et al. (2000) were unable to note any improvement in either the yield or the kinetics when the temperature of the acidogenic step was increased from mesophilic to thermophilic conditions. Ecke (1997) studied the influence of different factors on the acidogenic degradation at a plant for two-step anaerobic degradation. No increase in TS content of the leachate in the acidogenic step was noted as the temperature increased from 25 to 35 °C.

3.1.8. Sample preparation

Sample preparation includes grinding, mixing, drying, changing temperature, biological pretreatment, etc. Most of these activities influence the size of particles and/or the possibilities for biological activity and solubility, which were discussed above.

3.2. What are the possibilities and limitations of various kinds of leaching tests as prediction tools?

In order to address this question, results of leaching tests of different kinds of waste will be discussed. After a comparison between leaching tests in various scales, the use of L/S as a tool to cross-compare results from various scales will be discussed as well as the predictions that may be possible.

3.2.1. Comparison of SLT and simulators

The leaching patterns of SLT and simulators are not similar (see Figs. 1–3), indicating that either L/S is not useful as a predictive tool or that one or more other factors influence leaching to a greater extent than the amount of added water.

Likely concentration trends for organic material and salts from the SLT and simulator tests are presented in Fig. 5. The appearance of the trends is based on a compilation of information from the tests with ash as well as the tests with MSW + ash, in combination with mass flow considerations:

- The ash experiments that have results at low L/S for both SLT and simulators show much higher concentrations at low L/S for conductivity, TS and VS in the simulators than in the SLT (see example in Fig. 1).
- The decreasing trend of the ash SLT experiments was rather flat and covers quite a broad interval of L/S (see Fig. 1).
- The MSW + ash experiments that have simulator results at low L/S and SLT results above L/S 10 show generally higher concentrations of COD, TOC and chloride at L/S 10 (SLT results) than what could be predicted from the trends of the simulator results. This can be seen in Table 7, as all experiments show SLT results above, or in some cases about equal to, the simulator results. Fig. 2 shows an example of the discrepancy between SLT and simulators for the category SLT > sim of Table 7.
- A certain amount of the substances in the solid waste matrix will be leached. A high concentration can then not prevail indefinitely and the concentration will hence decrease at higher L/S.

Two observations can be made from Fig. 5: first, the higher concentration that is reached in simulators compared to the SLT at low L/S ratios and secondly, the somewhat higher concentrations of SLT compared to simulators in the longer term (at higher L/S ratios). This difference in behaviour between the two tests is especially noticeable for chloride, since chloride is normally considered a conservative substance, and can be expected to

<table>
<thead>
<tr>
<th>Chloride</th>
<th>COD</th>
<th>TOC</th>
<th>Nitrogen</th>
</tr>
</thead>
<tbody>
<tr>
<td>SLT &gt; sim</td>
<td>6</td>
<td>9</td>
<td>8</td>
</tr>
<tr>
<td>SLT &gt; sim</td>
<td>8</td>
<td>8</td>
<td>8</td>
</tr>
<tr>
<td>SLT ≥ sim</td>
<td>3</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>SLT &lt; sim</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
</tbody>
</table>

Table 7: The numbers of experimentsa that show different degrees of fit between the trends of simulators and the results of SLT.

a In total, 17 experiments performed with MSW + ash.

b SLT show higher values than the trend of the simulators at several L/S values. SLT show higher value than the trend of the simulators mainly at L/S 10 (example in Fig. 2). SLT and the trend of the simulators could fit together. SLT show lower values than the trend of the simulators.
be mainly dependent on solubility reactions. The differences in behaviour will generate quite different results for the predictions of maximum concentration levels as well as the leachate composition in the long term.

For substances other than organic material and salts, the principal leaching trends of Fig. 5 may not be applied. Based on the MSW + ash experiments, it seems as if the nitrogen concentration level may be about the same for both kinds of experiments at L/S 10, as indicated by the larger number of experiments that show a fairly good fit for nitrogen between simulator trends and SLT (see Table 7). Some experiments show lower values from the SLT than what can be predicted from the simulators due to the presence or absence of biological processes (see discussion in Section 3.2.2 about factor influence).

The leaching pattern of metals is based on data from the experiments with ashes. The concentrations of nickel and chromium from the SLT and simulator experiments at low L/S ratios are at best about equal, but the concentrations are generally lower from simulators than from SLT. Zinc and sulphur show constantly lower concentrations from simulators than from SLT (see Fig. 3). The reason for the different behaviour of zinc in comparison with nickel and chromium might be found in the preferences of the metals to form sulphide complexes. Zinc has a greater tendency to form sulphide complexes than the other metals. All metals studied were generally detected at lower concentrations from simulators than from SLT, indicating that when mass flows from the different experiments are considered, there will be large discrepancies between SLT and simulators.

3.2.2. Factors influencing leaching results (SLT vs. simulators)

3.2.2.1. Biology and water withdrawal. The absence or presence of biological activity is the factor that probably influences leaching the most, especially for substances converted in biological degradation. Chloride is, however, a conservative substance and biological reactions are not expected to influence leaching. Yet, as mentioned above, higher chloride concentrations were observed at L/S 10 for the SLT than predicted from the simulators. Hence, factors other than the absence or presence of biological activity influence leaching. One such factor is the withdrawal of substances by the collected leachate. The measured concentrations were adjusted for the substances taken out of the leaching system (on sampling occasions, etc.), according to Eq. (1). From the MSW + ash experiments, it can be concluded that the concentrations of the simulators and the SLT fit better for organic material and chloride after the adjustments. This is shown in Table 8. Chloride fits well in most cases, which supports the assumption that chloride is a conservative substance that is not affected by other factors to any appreciable extent. Still, four out of 17 wastes showed a large discrepancy between the chloride concentration trends of SLT and simulators. Three of those wastes show results of parallel experiments with either a large spread in SLT data (some resulting in a good fit between SLT and simulators and others not) or questionable analytical results on single occasions. This makes it difficult to classify the connection of the tests for those wastes. For the fourth waste, no evident explanation could be found for the discrepancy. The results of the ash experiments (all studied variables) still show large discrepancies in behaviour between SLT and simulators. This is probably due to influence of biological activity in the simulator, but absence of such activity in the SLT.

Organic material and nitrogen can be expected to be largely influenced by biological processes, since these substances are included in compounds used as energy sources as well as substrates for microorganisms. An increased biological activity would imply an increased release of organic material and nitrogen. The results of the MSW + ash experiments indicate higher concentrations of organic material from SLT than from simulators (see Table 8), contrary to what could be expected. Nitrogen, however, generally appears in higher concentrations from the simulators (after adjustments of water withdrawal). The difference in behaviour between organic material and nitrogen can be sought for in gas emissions. Nitrogen is almost entirely emitted by leachate, since nitrogen gas emissions under constant anaerobic conditions may be neglected. Organic material, however, is largely emitted in the gas phase. When the gas emissions are taken into consideration, the summarised organic emission of simulators will be higher than shown by the SLT. The influence of biological gas emissions is also valid for ashes. The simulators with ashes showed a relatively long initial period with relatively high VS content in the leachate (though

<table>
<thead>
<tr>
<th></th>
<th>Cl</th>
<th>Cl</th>
<th>COD</th>
<th>COD</th>
<th>TOC</th>
<th>TOC</th>
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<td>adj</td>
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<tr>
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<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>SLT &gt;&gt; sim^b</td>
<td>6</td>
<td>4</td>
<td>9</td>
<td>5</td>
<td>8</td>
<td>7</td>
<td>1</td>
<td>0</td>
</tr>
<tr>
<td>SLT &gt; sim^b</td>
<td>8</td>
<td>2</td>
<td>8</td>
<td>5</td>
<td>8</td>
<td>6</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>SLT &lt;= sim^b</td>
<td>3</td>
<td>9</td>
<td>0</td>
<td>7</td>
<td>1</td>
<td>5</td>
<td>8</td>
<td>2</td>
</tr>
<tr>
<td>SLT &lt; sim^b</td>
<td>0</td>
<td>2</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>2</td>
<td>15</td>
</tr>
</tbody>
</table>

^a In total, 17 experiments performed with MSW + ash.
^b >> SLT show higher values than the trend of the simulators at several L/S values. >, SLT show higher value than the trend of the simulators mainly at L/S 10 (example in Fig. 2). <=, SLT and the trend of the simulators could fit together. <, SLT show lower values than the trend of the simulators.
it is uncertain whether this was due to solubility or biological degradation), and no methane production. However, at days 120 and 330, respectively, a methanogenic period started. At this point, the leachate content of VS decreases and methane gas production became more intensive (see Fig. 6). The establishment of methanogenic conditions happens within a very short period of L/S. So, considering the summarised emissions in both gas and leachate, the emissions of organic material as well as nitrogen will be higher from simulators than from SLT. Consequently, there will be a considerable difference regarding nitrogen mass flow by leachate when comparing tests with and without biological activity. The mass flow of organic material by leachate is very much dependent on the efficiency of the gas formation, as can be seen for the simulators with wood bottom ash (Fig. 6).

3.2.2.2. Duration. The duration of each leaching step of the SLT experiments was 24 h. This time was too short for both chemical equilibrium to establish between very small ash particles and leachate and the establishment of a microbiologically adapted community. Processes could be enhanced by the addition of a suitable inoculum.

3.2.2.3. Temperature. The temperature differed about 10 °C between SLT and simulators, and could be responsible for about 10% difference in concentration level.

3.2.2.4. Mixing and size of particles. The mixing rate and the size of the particles most probably played a minor role in these experiments. There were no recognizable difference between SLT and simulators with respect to the size of the particles. Mixing and recirculation, respectively, were considered in the tests and performed at fair intensity.

3.2.3. Comparison of simulators and field test cell

When comparing the simulator results with the field test cell results, it is possible to see similarities in leaching behaviour, even though the leaching of the field test cell passed different phases. Initially, the field test cell showed a phase with increasing leachate concentrations; after reaching a maximum, concentrations started to decrease, see Figs. 4, 7 and 8. The slope of the decrease was close to the decreasing slope of the simulators, as can be seen for conductivity and TS in Figs. 4 and 7 respectively, even though the concentration of the simulators was much higher (it is not possible to make the slope comparison for other variables, due to lack of data). The similarity in the pattern of decrease indicates that similar processes occurred in the two tests. It further supports the hypothesis that a larger leaching test will generate results closer to field conditions, since the simulators showed similarities in leaching pattern with the field test cell while the SLT did not. The test cell is considered to be large enough to mirror full-scale leaching.

The decreasing trend of the field test was interrupted. The conductivity in the leachate of the field test cell increases and was about doubled at L/S 3, see Fig. 4. The reason for this as well as the discrepancies in concentration levels will be discussed in the following chapters.

3.2.4. Factors influencing leaching results (simulators vs. field test)

3.2.4.1. Recirculation. In this study, the recirculation rate of the simulators was about 13% of the initial waste volume per day. Chugh et al. (1998) noted an increased rate of degradation with increased recirculation. The maximum recirculation rate was 30% of the initial waste
volume per day, which illustrates an option of increased loading for the experiments within this study.

The recirculation rate of the field test cell was two orders of magnitude less than in the simulators. Recirculation may hence be one explanation as to why the leachate concentration of the field test cell was lower than that of the simulators (see Figs. 4, 7 and 8). Recirculation may also explain the doubling in conductivity of the field test cell at about L/S 3, see Fig. 4. The increase in conductivity was preceded by a longer period (1.5 years) of operation with only recirculation (no sampling, water addition or withdrawal and no other measures). Recirculation promotes biological activity, which weakens the structure of the waste and causes channels within the waste to collapse. Thus, the water finds new pathways. Waste that was not initially involved in the leaching can be incorporated.

3.2.4.2. Particle size. Particle size is of minor importance for the explanation of the differences in leaching behaviour between simulators and the field test cell. The fact that the waste was wrapped in plastic bags in the field test cell influenced leaching far more than the waste particle size. The plastic bags were efficient barriers for the water flow in the waste body. The bags were torn to some extent during compaction, but they were still able to force the flow of water and to prevent water from coming in contact with waste. The effect of the plastic bags is considered one reason why the concentration level is lower from the test cell than from the simulators (see Figs. 4, 7 and 8). The part of the waste that did not come in contact with the water was roughly estimated by calculative increase of the L/S ratio of the test cell. It is about half of the waste volume at a maximum. At a doubling of L/S, the trends for conductivity of both tests showed a fit (see Fig. 9). For TS the fit between the decreasing concentration trends of simulators and the test cell (with doubled L/S ratio) show a larger discrepancy than the conductivity trends of Fig. 9. The TS concentrations of the test cell are then slightly higher than from the simulators. The rough estimation that half of the waste material was not involved in the leaching process was hence somewhat high.

3.2.4.3. Temperature. The temperature difference between field and simulators was in the order of 15–20 °C. For most biological systems, this would imply a rate difference of a factor of 1.5–2. The concentration differed by a factor of about 2 for TS (Fig. 7), conductivity (Fig. 4) and organic material. The difference for nitrogen was slightly less (Fig. 8). However, no appreciable differences between simulator and field test cell are likely to be caused by temperature differences, since the temperature was found to have little effect on the acidogenic degradation (Pavan et al., 2000; Ecke, 1997). The difference in concentration of the different metals further supports the hypothesis that it is not simply an effect of temperature. The metal concentrations differed considerably depending on the kind of metal. There was no difference for chromium. The differences for nickel and zinc were larger and amounted to factors of 4 and 6, respectively, with higher concentrations in the simulators.

3.2.4.4. Withdrawn water. The influence of withdrawn substances was small for the test cell as well as the simulator experiments. For the test cell, there was only a difference between measured and adjusted concentrations in the order of 5–10% at L/S 2.5. This was probably due to the low L/S ratio of the experiments. At those low L/S there were no appreciable differences between measured and adjusted concentrations for the simulator experiments, which means that the water withdrawal has little effect on the comparison of leaching results at low L/S. The withdrawal of substances can, however, be an influential factor at higher L/S ratios, i.e. in the longer term. This is indicated by the examples presented in Table 6.

3.2.4.5. Duration. Simulators, test cells and landfills can be regarded as large filters. They differ from most other filter configurations because both microorganisms and substrate are present in the filter. Information about an optimal hydraulic retention time in such systems has not been found, but the observed 12–24 h of acidogenic degradation might be regarded as a minimum, due to the more complex substrate composition in landfills.

3.2.5. L/S as a tool for comparisons

In order to be able to compare the leaching results of various kinds of leaching tests, a tool other than time is required. It is difficult to extrapolate time in days for a lab test to months or years under field conditions, as the conditions of the test and the field situation may differ substantially, especially with respect to the amount of water added to the waste. By using time as the sole tool of comparison, other factors influencing leaching are not taken into consideration. It might be applicable in studies of specific landfills, but not in general. The L/S ratio is a more objective tool, since it considers both the
amount of solid waste as well as the amount of water added to the waste and it can be related to time, as the amount of infiltrated water is time-dependent. The use of L/S has proven to be useful in the comparisons made between different leaching tests. The good correlation of chloride between simulator trends and SLT from the MSW + ash experiments, after adjustments of withdrawn leachate (see Table 8), is an example of data that verify this. The L/S ratio, however, has limitations. For example, the L/S ratio does not consider kinetics or changes in environmental conditions, which limits the possibilities for predictions (this will be further discussed in the following section). Other factors may also influence the leaching behaviour, as discussed earlier. All in all, the use of L/S makes it easier to compare tests and field leaching than the use of time only.

3.2.6. Predictions

The factor dominating the development of leaching is the absence or presence of biological processes. For predictions of leachate quality, it is essential to include and promote biological reactions in the leaching tests. The short-term shaking tests (SLT) evaluated here did not promote biological activity and cannot be considered a good tool for predictions of leachate quality in the long term. Some single constituents, e.g. conservative substances like chloride, may be predicted, but the overall composition is impossible to predict. The difference in leachate composition is most evident for nitrogen and metals. Nitrogen shows a higher mass flow in the liquid phase from leaching tests with biological activity, as indicated in Fig. 10, when estimating the infinite leachate mass flow by the different leaching methods. It may be noted that nitrogen is the only substance of the ones presented in Fig. 10 that shows a higher mass flow by simulator test than by SLT. Metals were not included in Fig. 10, but they belong to the substances with a mass flow that may be considerably lower in tests with biological activity, at least under methanogenic conditions with sulphur-containing waste, than without such activity. The estimated mass flows of metals from the wood bottom ash in the simulators were only between 10 and 45% of the mass flows by SLT when considering the metals nickel, chromium and zinc. The presence of biological processes also influences the content of organic material to a large extent, yet the concentration of organic material in the liquid phase is very much influenced by the kind and efficiency of the degradation process; e.g. under methanogenic conditions a large portion of the organic matter will be emitted over the gas phase.

Regardless of how optimally a test is performed, the results cannot predict future observations in the field perfectly. It may never be possible to simulate such a large and inhomogeneous system such as a landfill correctly, and this is even less likely in the case of landfill operations that are usually far from optimal. Yet, the simulator tests can be useful for predictions of:

- the lowest L/S required for waste stabilisation, the calculation will be based on mass-flow calculations, so it is essential to measure added water, leachate flow (taken out of the system) as well as leachate concentrations;
- the highest possible leachate concentration that can be expected for substances closely related to biological degradation;
- leaching trends that are likely to occur in the future reflecting the processes within the waste.

The prediction of required L/S to reach a certain level of concentration (e.g. discharge limits) is associated with great uncertainties. This is due to the higher concentration level of the simulators in comparison with the field and the fact that the conditions in the landfill may change over time. The concentration level of the studied simulators and the field test cell differed by a factor of about 2 in concentration at the same L/S. This would imply that the predicted L/S must be at least doubled in comparison with the simulator results. The increase in concentration observed for the field test cell (doubling) is not unlikely to happen in full-scale landfills as well. Thus, a safety factor of 2 would be prudent and the predicted L/S to reach a discharge limit based on simulator leaching would then be associated with an uncertainty by at least a factor of 4.

The simulator experiments can be used for the mentioned predictions if degradation is proceeding in the same phase. A transition between degradation phases, especially the point in time at which it will occur, is hard to predict. When the amount of organic material is depleted and anaerobic degradation is no longer favoured, a transition to other biological systems, like the sulphide oxidising system, may occur in both landfills and leaching tests.

The applicability of leaching tests for predictions would probably be better if the stabilisation of waste was enhanced in both leaching tests and landfills. Moving towards an optimised test, with respect to waste
stabilisation, would mean performing the test under the following conditions.

- A high addition of water to the waste. The rate of water addition is limited by kinetics. Under acidogenic conditions, the minimum hydraulic retention time ought to be in the range 12–24 h. Under methanogenic conditions, longer times, which are not quantified, are probably necessary.
- A high recirculation rate. Above a daily recirculation of 30% of the initial waste volume, if technically feasible.
- A high water withdrawal. As for the water additions, kinetics influences the maximum rate.
- A small size of waste particles. A smaller particle size will generate enhanced degradation. Too small a particle size can limit the stability and the hydraulic conductivity and, hence, the possibilities to optimise water addition, water withdrawal and leachate recirculation. A particle size in the range of 10 cm is a suggested upper limit for an optimal enhanced stabilisation.
- Increased temperature. The temperature, at least under methanogenic degradation conditions, will influence the degradation rate, with higher rates at higher temperature (within certain limits). It is possible that a thermophile temperature will generate an enhanced degradation compared to mesophilic or lower temperature range. However, thermophilic conditions are harder to control and are unlikely to occur in the field. Therefore, a mesophilic temperature can be regarded as a maximum temperature and should be kept stable.

Performing leaching tests in an enhanced way would make it possible to generate results of leaching tests that cover a wide L/S interval and also consider biological activity. Such information is lacking today, as simulator tests only cover a short L/S interval. The predictions of today are based on prolonged (mathematically extrapolated) trends of simulator leaching. Actual data that covers a broad L/S interval would make the predictions more reliable.

Without enhanced waste stabilisation, sudden changes in concentration under field conditions are more likely to happen, probably caused by the water finding new pathways, resulting in more waste being subject to leaching. This was seen in the field test cell as the conductivity increased at L/S 3 (see Fig. 4). Such an increase cannot be predicted.

The following considerations are important with respect to limitations of the enhancement of waste stabilisation.

- An increased addition of water and rate of circulation cannot be done without considering the kinetics, i.e., the retention time of the leachate must be sufficient for reactions to occur. A probable minimum hydraulic retention time is in the range of 12–24 h, which in “normal” landfills is exceeded by far.
- A waste-particle size that is too small can give rise to a decrease in permeability that might limit the rate of recirculation. Assarsson et al. (1994) found that the hydraulic conductivity of putrescible waste is very dependent on the density of the waste and that grinding of the waste will make it even more sensitive to changes in density. The hydraulic conductivity varied between $10^{-5}$ m/s at a loose compaction to $10^{-8}$ m/s at a dense compaction.
- An increased water addition in combination with small-sized waste particles can have a detrimental effect on geotechnical landfill stability.
- Possible technical limitations in relation to the addition or recirculation of sufficient water.
- The possibility of collecting the leachate. This is a problem, mainly in existing landfills that were constructed without any liners or drainage systems for leachate collection. There is no apparent or easy solution to this problem. Either it must be solved, or one must suppose that landfills emit pollutants at a rate that natural systems can accommodate. Otherwise, future generations will be left with a pollution problem.
- Use of daily cover. The daily cover can limit and steer the water flow within the landfill.

4. Conclusions

The liquid to solid ratio (L/S) is a better tool for comparisons between different tests and real landfills than the time factor, but the L/S alone has limited application. Other factors influence the leaching. Those factors have been defined as:

- Water withdrawal
- Biological processes
- Recirculation or mixing
- Duration
- Particle size
- Temperature
- Sample preparation.

Leaching tests that do not promote biological activity, like shaking leaching tests (SLT), cannot be considered good tools for the prediction of leachate quality from
MSW landfills over the long term. The leaching pattern in landfills is markedly influenced by biological activity, which is not reflected by the SLT. Still, some single constituents, e.g. conservative substances like chloride, are not influenced to any significant degree by biological activity and can be predicted.

Landfill simulator tests can be used to predict:

- highest expected concentration of various leachate analytes such as organic material, nitrogen and conductivity;
- concentration development;
- lowest L/S ratio to stabilise the waste (i.e. a certain percentage of the amount in the waste has been removed and the pollution potential has decreased) with respect to organic material, nitrogen and salts; it is likely that this is also valid for metals.

Landfill simulator tests are not very useful for predicting the L/S ratio required to reach a certain concentration. The uncertainty in the predictions is associated with a factor of at least 4.

The applicability of leaching tests is mainly limited by the fact that the landfills have not been operated at optimal conditions. It is likely that some elements of the waste in landfills do not contribute to the leaching due to, e.g. uneven water distribution and barriers such as plastic bags. This was indicated by lower concentrations from the field test cell than from simulators and a sudden increase in electrical conductivity of the field test leachate after a longer period of recirculation.

The applicability of leaching tests for predictions is based on enhanced stabilisation in both leaching tests and at landfills. To increase the predictive reliability of the leaching tests one should focus on decreasing the duration of the stabilisation process in tests and at landfills, see Table 9. Kinetic aspects, geotechnical landfill stability issues, as well as practical considerations limit the degree of increase or decrease, as indicated in Table 9. In addition, the design of the leachate collection system influences the possibility of applying enhanced stabilisation.

Applying a modified landfill simulator test in accordance with information in Table 9 would make it possible to generate results of leaching tests that cover a wide L/S interval and consider biological activity, making leaching predictions more reliable.

Table 9
Suggested changes of factor levels in order to enhance waste stabilisation

<table>
<thead>
<tr>
<th>Factor</th>
<th>Changed direction compared with situation of today</th>
<th>Suggested suitable level</th>
</tr>
</thead>
<tbody>
<tr>
<td>Water addition</td>
<td>Increase</td>
<td>Limited by kinetics. Min HRT(^a) of 24 h at acidogenic degradation. At methanogenic conditions longer HRT</td>
</tr>
<tr>
<td>Water withdrawal</td>
<td>Increase</td>
<td>See water additions</td>
</tr>
<tr>
<td>Leachate recirculation</td>
<td>Increase</td>
<td>Minimum a daily 30% of initial waste volume</td>
</tr>
<tr>
<td>Particle size</td>
<td>Decrease</td>
<td>Upper limit about 10 cm</td>
</tr>
<tr>
<td>Temperature</td>
<td>Increase (methanogenic conditions)</td>
<td>Mesophilic range, keep stable</td>
</tr>
<tr>
<td>Sample preparation</td>
<td>Additions of inoculum with leachate (initially)</td>
<td>Additions of inoculum with leachate (initially)</td>
</tr>
</tbody>
</table>

\(^a\) HRT: hydraulic retention time.

References


