# CHAPTER 22 - DISPOSAL

## 22.1 INTRODUCTION

Waste management policy is often expressed in the following sequence of declining priorities (see also Chapter 2):

1. Minimisation of waste production, hazard potential and energy consumption through substitution and cleaner technology measures

2. Recycling or utilisation

3. Incineration with energy recovery

4. Landfilling

Although ranked third and fourth in this sequence of priorities, incineration and landfilling are playing and will continue to play important roles in waste management in many parts of the world. The incineration process itself is obviously not a final waste treatment stage, and the various incineration residues must themselves be utilised or landfilled. In accordance with the sequence of priorities, utilisation of residues is, in principle, preferred to landfilling, provided this does not give rise to unacceptable environmental impacts or health hazards.

In practice, there are numerous factors which act as obstacles to utilising incinerator residues:

- existing regulations
- lack of economic incentives
- liability issues
- residue separation practices
- uncertainties concerning the functional properties of the residues
- uncertainties concerning the evaluation of the extent and acceptability of the environmental impacts and health hazards

Therefore, landfilling or storage are the predominant MSW incinerator residue management options presently available in some countries. For example, in the U.S.A., most of the incinerator residues currently produced are landfilled as combined ash, whereas in some European countries (e.g. Denmark, France, Germany, The Netherlands), significant quantities (40 to 60% or more) of the bottom ash from the incinerators are being utilised for road construction and similar purposes (see Chapter 21). APC system residues are landfilled in most countries, although in The Netherlands approximately 50% of the fly ash generated is used as a filler in asphalt. Incinerator residues may, in some cases, have to be treated prior to or during disposal in order to reduce the risk of unacceptable environmental impacts.

The major potential environmental impacts of concern in connection with disposal of incineration residues are those associated with the formation and release of leachate and with fugitive dust emissions. Fugitive dust problems generally occur only during the relatively short period of actual deposition in a landfill. The use of covered or closed transport containers and maintaining an adequate moisture content in the residues, are generally considered to be effective measures in controlling fugitive dust problems. The formation of leachate may, in contrast, constitute both a short and long-term problem which should be minimised through the application of a proper disposal strategy, and through appropriate design and operation of the landfill. Incinerator residue or ash disposal is therefore discussed primarily in terms of leachate production, management and fate in this chapter. The emphasis of the chapter is placed on the relationship between the leaching behaviour of the residues and the applied or underlying disposal strategies, and specific design and operations issues are merely outlined and not discussed in detail. Interested readers are referred to textbooks and manuals on these subjects.

## 22.2 CHARACTERISTICS OF INCINERATOR RESIDUE LANDFILL LEACHATES

### 22.2.1 Overview of Incinerator Residue Leachability

From a technical perspective, it is evident that the development of strategies for disposal of incinerator residues and management of the leachate should be based on extensive knowledge of both the short and long-term leaching behaviour of the particular residue streams in question. In this context, "short-term" may cover a time period of 25 to 50 or 100 years and "long-term" consequently represents the following several hundred to thousands of years. A reasonable amount of information is available on the short-term behaviour of most residues, and the evaluation of the short-term behaviour may to a large extent be based on the results of laboratory and pilot-scale leaching experiments and field observations.

The long-term behaviour of incinerator residues is less understood. Due to the lack of direct observations, the evaluation of the long-term behaviour is more complicated and requires a synthesis of information obtained from laboratory testing of fundamental leaching behaviour, leaching tests simulating long-term disposal conditions, field measurements, and hydrogeochemical modelling of mineral changes and speciation. The results of ongoing and future research in this field may be expected to reduce the substantial degree of uncertainty with which predictions of the long-term behaviour of incinerator residues are currently made.

Table 22.1 presents an overview of the maximum levels of concentrations of inorganic salts, trace elements and nonvolatile organic carbon (NVOC) observed in initial leachates from the major types of incinerator residues including:

- Bottom ash (usually including grate siftings and boiler ash)

- Fly ash and mixtures of fly ash and acid gas scrubbing residues from the semi-dry process and dry lime injection process, and
- A mixture of fly ash and sludge from treatment of the wastewater from the wet scrubbing process with lime and trimercaptotriazine, TMT (Reimann, 1987)

Table 22.1
Maximum Concentration Levels of Contaminants in Leachates from Various Incinerator Residues

| Typical maximum levels of concentration in leachate | bottom ash | fly ash and residues from dry and semi-dry APC processes | Mixture of fly ash and sludge from wet APC process |
|---|---|---|---|
| 10 - 100 g/l | | Na, K, Pb | $Cl^-$, Na, K |
| 1 - 10 g/l | $SO_4^{2-}$, $Cl^-$, Na, K, Ca | | $SO_4^{2-}$, Ca |
| 100 - 1000 mg/l | NVOC, $NH_4$-N | NVOC, $SO_4^{2-}$, Zn | |
| 10 - 100 mg/l | | Cu | |
| 1 - 10 mg/l | Cu, Mo, Pb | Cd, Cr | NVOC, Mo |
| 100 - 1000µg/l | Mn, Zn | As, Mo | |
| 10 - 100 µg/l | As, Cd, Ni, Se | | As, Cr, Zn |
| 1 - 10 µg/l | Cr, Hg, Sn | | Pb |
| < 1 µg/l | | Hg | Cd, Cu, Hg |

Hjelmar, 1991, 1992 and 1993; Thygesen et al., 1992; Hjelmar et al., 1993

The maximum concentrations shown in the table represent data from a number of laboratory leaching tests (mostly column leaching tests) and a few field investigations (Hjelmar, 1991, 1992 and 1993, Thygesen et al., 1992, Hjelmar et al., 1993).

The maximum concentrations occur in the initial leachate for most parameters (note, however, the sulphate curve in Figure 22.1), and most of the concentrations in Table 22.1 have been observed in fractions of leachate collected at or below liquid/solid or leachate/waste ratio (L/S) = 0.5 l/kg. (e.g. expressed as cubic metres of leachate produced per tonne of waste or residue deposited). For a particular disposal site, S will be constant and L will increase as the leachate is formed. An L/S scale may therefore be transformed to a time scale if the rate of percolation or flow through the site is known. It is often practical to express field data and experimentally determined data on leachate quality as a function of L/S for each particular system in question. Such data may subsequently be used (with care) in conjunction with additional information (e.g. on pH and redox conditions) to provide estimates of leachate quality as a function of time at a disposal site which contains waste/residues with similar properties and for which the rate of percolation of water is known. At low L/S values, the leaching of several contaminants, particularly trace elements, is solubility controlled and strongly influenced by the pH of the leachate (which in turn is governed by the major constituents of the incinerator residues and local conditions).

As shown in Chapter 16, incinerator residues exhibit systematic leaching patterns, and the leaching behaviour of several contaminants is controlled by such factors as pH, redox potential, ionic strength, complexing inorganic ions and organics, the presence of various minerals, etc. Once the relationship between the controlled contaminants and the controlling factors has been established, it becomes important to be able to predict how these controlling factors may develop within an incinerator residue disposal site. The estimation of contaminant release during disposal of incinerator residues is discussed in detail by Kosson et al. (1996).

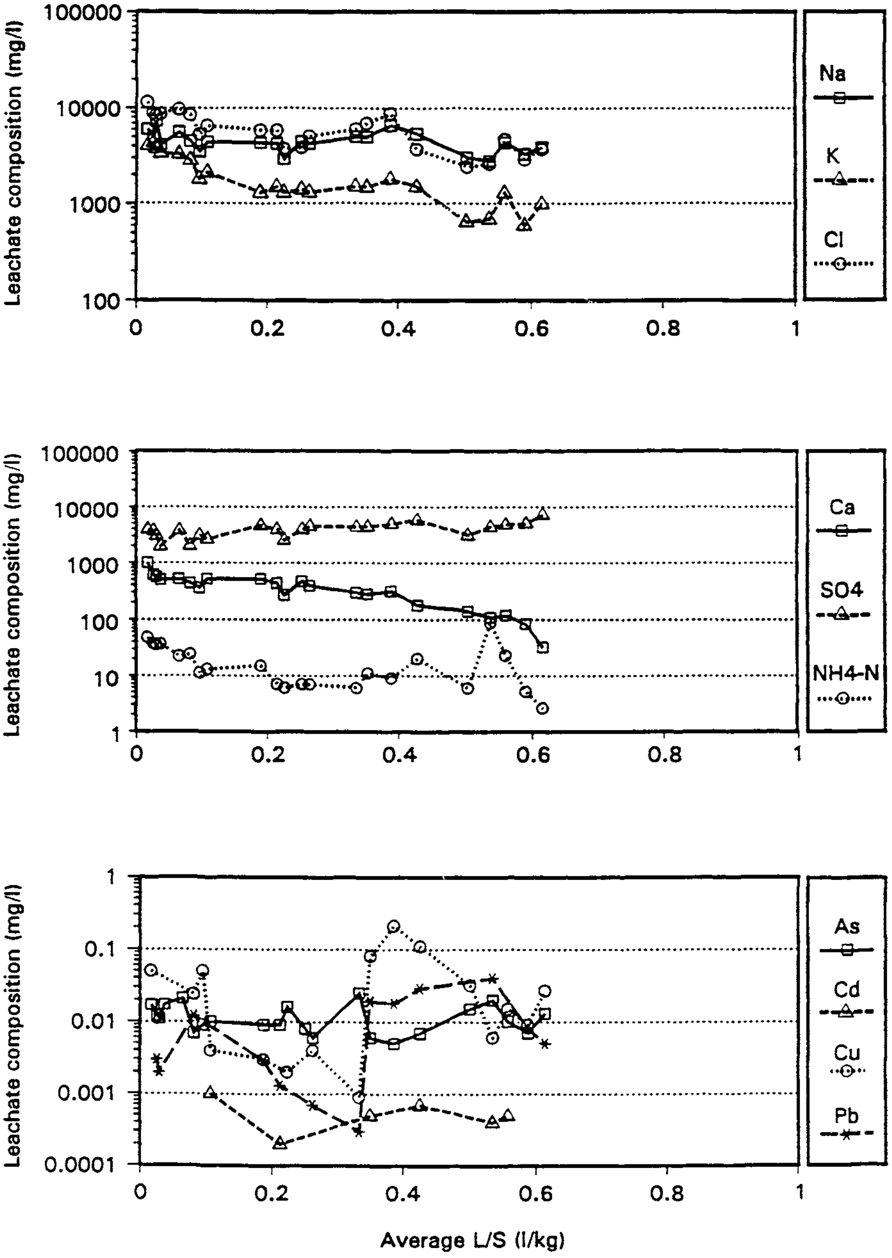
### 22.2.2 Bottom Ash Leachate

Table 22.2 shows the development over a period of 21 years of the quality of leachate from a Danish ash monofill, (Vestskoven), containing approximately 10,000 tonnes of ash (85% bottom ash, 15% fly ash). The data was derived from annual analyses of the leachate which is contained by a PVC bottom liner, conducted to a central pumping sump and pumped out at regular intervals. In 1994, a total of 7300 m$^3$ of leachate had been removed from the monofill, and the historical characteristics have been described extensively elsewhere (Hjelmar, 1987, 1989, 1991). The concentrations of some of the major components and trace elements in the leachate are shown as a function of L/S in Figure 22.1.

For bottom ash, the first leachate has a relatively high content of inorganic salts (chloride, sulphate, sodium, potassium, calcium). The content of dissolved organic matter (measured as NVOC) and ammonia are both associated with the residual uncombusted organic material and may vary considerably. The pH is usually slightly to strongly alkaline, depending on the degree of carbonation.

The redox potential is low (reducing) due to microbiological degradation of the residual organic material. The concentrations of trace elements in the leachate are low due to the reducing environment (most of them form relatively insoluble sulphides) and a solubility limiting pH regime. The content of uncombusted organic material in incinerator bottom ash may be one of the key factors in controlling the hydrogeochemical conditions and hence the behaviour of trace elements within a bottom ash landfill. It may be a delicate balance, i.e., sufficient organic material should be present to support a certain level of microbiological activity, but the organic and degradation products should not be too high as to dominate the composition of the leachate. In order to minimise the mobility of trace elements, the amount of acidic degradation products must not be significant in comparison to the acid neutralisation capacity of the bottom ash. Dissolved organic material, per se, is an undesired contaminant, and it has further been shown to be associated with increased mobility of Cu in bottom ash (van der Sloot et al., 1992).

Figure 22.1 Composition of Leachate from a Danish Incinerator and Fly Ash Monofill as a Function of L/S over a Period of 20 Years



Hjelmar, 1995

Table 22.2
Overview of a Time Series of Leachate Quality Measurements from a 21 Year Old Danish Bottom and Fly Ash Monofill

| Parameter | Unit | Variation 1973-94 (24 observations) | Average Values | |
|---|---|---|---|---|
| | | | 1973/74 | 1991/92 |
| pH | - | 8.3 - 10.5 | 8.8-10.1 | 8.9-10.2 |
| Alkalinity | meqv/l | 1.4 - 9.3 | 2.5 | 7.4 |
| Redox potential, $E_h$ | mV | -10 - -290 | -66 | - |
| Conductivity | mS/m | 1300 - 3900 | 3100 | 1900 |
| $BOD_5$ (from 1981) | mg/l | <2 - 26 | - | 2 |
| Sulphate | mg/l | 2000 - 7200 | 3100 | 6100 |
| Chloride | mg/l | 1600 - 11400 | 9300 | 3300 |
| Ammonia-N | mg/l | 1.5 - 87 | 39 | 3.9 |
| Na | mg/l | 2600 - 7300 | 5600 | 3600 |
| K | mg/l | 600 - 4300 | 3900 | 800 |
| Ca | mg/l | 32 - 1000 | 670 | 58 |
| As | mg/l | 0.004 - 0.025 | 0.014 | 0.010 |
| Cd | mg/l | <0.0001 - 0.001 | <0.003 | <0.0002 |
| Cr | mg/l | <0.001 - 0.08 | 0.03 | <0.002 |
| Cu | mg/l | <0.0005 - 0.21 | 0.013 | 0.018 |
| Fe | mg/l | <0.01 - 1.6 | 0.21 | 0.055 |
| Hg | mg/l | <0.00005 - 0.003 | 0.00008 | 0.0004 |
| Pb | mg/l | <0.0005 - 0.04 | 0.0013 | 0.007 |
| Zn | mg/l | <0.01 - 0.59 | 0.05 | 0.09 |
| Ionic strength | gmol/l | 0.18 - 0.48 | 0.41 | 0.23 |
| Accumulated L/S | l/kg | 0.017 - 0.730 | 0.027 | 0.602 |

The content of fly ash is estimated at 15%

There is a tendency for the concentrations of most salts in the leachate to decrease as the leaching progresses (Table 22.2 and Figure 22.1), however, sulphate concentrations actually increase due to the decrease in calcium concentrations. The pH remained between 8.5 and 10.5 (the lowest values probably reflect non-optimal sampling conditions) throughout the period. Reducing conditions have been maintained and the concentrations of trace elements have remained low during the 21 years of observation at this particular site. The low short-term concentration levels of most trace elements have been confirmed by observations at other ash landfills

(Hjelmar, 1989). In Table 22.3, the total amounts of some major and trace components leached from the Danish landfill site over the 21 year period from 1973 to 1994 are presented and compared to the estimated total amounts of those components initially present in the landfilled ash. The data indicate that approximately 40% of the chloride, 30% of the sulphate, 9% of the sodium, 6% of the potassium and 0.3% of the calcium have been leached, whereas the amounts of trace elements leached are negligible compared to the total amounts present.

Table 22.3
Total Amounts of Major and Trace Components Leached from a 21 Year Old Danish Incinerator Ash Monofill Compared to the Estimated Total Contents of these Components

| Parameter | Unit | Amount leached in 21 years (L/S = 0.73 l/kg) | Estimated total content in the landfilled ash |
|---|---|---|---|
| Chloride | tonnes | 35 | 90 |
| Sulphate | tonnes | 32 | 110 |
| Na | tonnes | 31 | 340 |
| K | tonnes | 10 | 180 |
| Ca | tonnes | 2.1 | 700 |
| As | kg | 0.093 | 270 |
| Cd | kg | <0.008 | 400 |
| Cu | kg | 0.25 | 39000 |
| Pb | kg | 0.065 | 31000 |

### 22.2.3 Fly Ash and Acid Gas Scrubbing Residue Leachate

An indication of the leaching properties of fly ash and acid gas scrubbing residues may be obtained from the results of laboratory studies (3 fly ash samples, 2 dry, 2 semi-dry APC residues and one wet process product, (Hjelmar, 1992 and 1993)). Field observations and large scale lysimeter leaching tests have subsequently confirmed these findings (Andersen and Boll, 1994) (See Chapter 11 and Chapter 16). The amounts of some major and trace components leached from the above mentioned residues in combined column and batch leaching tests (L/S = 0-25 l/kg) are shown in Table 22.4. The compositions of the first and last fractions of leachate collected in the laboratory leaching experiments on the dry/semi-dry and wet scrubbing process residues are shown in Table 22.5.

The results indicated that the acid gas scrubbing residues from the semi-dry and dry processes contain 20 to 35% (w/w) readily soluble material. Most of the soluble material consists primarily of chlorides and hydroxides of calcium, sodium and potassium, which appear in the first few fractions of leachate (L/S = 0 to 2 l/kg). The most leachable metals/trace elements are lead (2.3 to 65%(w/w) of the total content) and molybdenum (9 to 19% (w/w) of the total content). All other trace elements were less than 4% (w/w) leachable under these circumstances, and most of them far less than 1% (w/w) leachable. The high leachability of lead is caused primarily by chloride complexation, but the amphoteric behaviour of lead also plays an important role at pH values above 10. No releases of mercury, nickel and tin were observed. Concentrations of soluble matter are very high in the first leachate fractions from the dry and semi-dry products but level off to moderate and low values in later fractions (see Table 22.5). The leachability of the fly ash resembles that of the dry and semi-dry acid gas scrubbing residues, although the leaching of Ca and the alkalinity of the leachate is much smaller for the fly ash. The pH of the fly ash leachate is generally lower than that of the dry/semi-dry residue leachate which, under certain circumstances, may cause some changes in leachability of trace metals such as Cd, Pb, Zn, etc.

Table 22.4
Total Amounts of some Major and Trace Components of Fly Ash and APC System Residues Leached in Combined Column and Batch Leaching Tests for L/S = 0-25 l/kg

| Parameter | Unit | Fly ash | Residues from dry and semi-dry APC processes | Residue from wet scrubbing process mixed with fly ash |
|---|---|---|---|---|
| TDS | g/kg | 210 - 230 | 290 - 380 | 140 |
| Sulphate | g/kg | 13 - 41 | 0.47 - 3.1 | 32 |
| Chloride | g/kg | 89 - 106 | 120 - 200 | 56 |
| Na | g/kg | 23 - 30 | 12 - 17 | 21 |
| K | g/kg | 28 - 50 | 17 - 29 | 21 |
| Ca | g/kg | 9.1 - 21 | 61 - 109 | 17 |
| As | mg/kg | 0.10 - 0.19 | < 0.02 - 0.04 | 0.28 |
| Cd | mg/kg | 0.05 - 35 | 0.03 - 0.44 | < 0.0006 |
| Cr | mg/kg | 0.05 - 0.12 | < 0.6 - 2.4 | 0.52 |
| Cu | mg/kg | 0.03 - 0.31 | 0.13 - 22 | < 0.004 |
| Hg | mg/kg | < 0.01 | < 0.003 | < 0.001 |
| Mo | mg/kg | 7.1 - 14 | 1.1 - 3.0 | 12 |
| Ni | mg/kg | < 0.01 | < 0.2 | < 0.2 |
| Pb | mg/kg | 17 - 370 | 220 - 3400 | < 0.001 |
| Sn | mg/kg | < 1 | < 0.4 | < 3 |
| Zn | mg/kg | < 2 - 580 | 45 - 340 | < 0.2 |
| NVOC | mg/kg | 14 - 46 | 71 - 780 | 78 |

Hjelmar, 1992 and 1993

Table 22.5
Results of Column and Batch Leaching Tests on Residues from the Dry and Semi-Dry Processes (Ranges for 4 Products) and Wet Scrubber Product (Single Determination)

| Parameter | Unit | Residues from Dry and Semi-dry Processes | | | Residue from Wet Process (+FA) |
|---|---|---|---|---|---|
| *First leachate fraction (L/S = 0.0 - 0.2 l/kg)* | | | | | |
| pH | - | 9.8 | - | 10.2 | 9.0 |
| TDS | mg/l | 430000 | - | 520000 | 116000 |
| Alkalinity | meqv/l | 87 | - | 540 | 0.96 |
| Chloride | mg/l | 1900000 | - | 310000 | 65000 |
| Sulphate | mg/l | 150 | - | 430 | 2100 |
| Ca | mg/l | 110000 | - | 160000 | 5800 |
| Na | mg/l | 7500 | - | 37000 | 23000 |
| K | mg/l | 19000 | - | 66000 | 23000 |
| Cd | mg/l | 0.099 | - | 1.9 | 0.006 |
| Cr* | mg/l | 0.48 | - | 2.3 | <0.003 |
| Cu | mg/l | 0.17 | - | 37 | 0.0009 |
| Hg | mg/l | | <0.003 | | <0.0005 |
| Mo* | mg/l | 0.31 | - | 0.61 | 7.9 |
| Pb | mg/l | 2.2 | - | 11000 | 0.0019 |
| Zn | mg/l | 0.02 | - | 730 | 0.02 |
| NVOC | mg/l | 24 | - | 640 | 8 |
| *Last leachate fraction (L/S = 5.0 - 25 l/kg)* | | | | | |
| pH | - | 12.1 | - | 12.5 | 10.6 |
| TDS | mg/l | 770 | - | 1900 | 1600 |
| Alkalinity | meqv/l | 13 | - | 42 | 2.0 |
| Chloride | mg/l | 20 | - | 36 | 2.8 |
| Sulphate | mg/l | 2.5 | - | 40 | 1100 |
| Ca | mg/l | 290 | - | 840 | 460 |
| Na | mg/l | 8 | - | 24 | 15 |
| K | mg/l | 10 | - | 41 | 19 |
| Cd | mg/l | <0.00002 | - | 0.00006 | <0.00002 |
| Cr | mg/l | <0.004 | - | 0.004 | 0.023 |
| Cu | mg/l | 0.002 | - | 0.007 | <0.0005 |
| Hg | mg/l | | na | | na |
| Mo | mg/l | 0.017 | - | 0.075 | 0.027 |
| Pb | mg/l | 4.3 | - | 8.6 | <0.0001 |
| Zn | mg/l | 0.48 | - | 1.5 | <0.02 |
| NVOC | mg/l | 1.9 | - | 4.0 | 2.8 |

* - Combined fraction L/S = 0.0 - 1.0 l/kg analysed. na - Not Analysed
Comparison of the First (L/S = 0.0 - 0.2 l/kg) and the Last (L/S = 5.0 - 25 l/kg) Leachate Fractions
Hjelmar, 1992

The residue from the wet scrubbing process was shown to have a content of readily soluble material which is considerably smaller than that of the dry and semi-dry residues. The major ions in the initial leachate fractions are chloride, sodium, potassium, and to a lesser extent calcium. Sulphate from the wet scrubbing process is present in appreciably higher concentrations than in the leachates from the dry/semi-dry products. With molybdenum as the only exception, the concentrations of trace elements/heavy metals are low in all leachate fractions, probably due to the presence of the organic sulphide TMT. The long-term stability and effectiveness of this compound are not known. The highest concentration of lead measured in the leachate from the residue from wet condensing process was 1000 times lower than the lowest concentration measured in the initial leachates from the dry and semi-dry residues.

### 22.2.4 Combined Ash Leachate

In the US, it is common practice to manage and landfill incinerator residues as mixture of bottom ash and scrubber residue (combined ash). Some of the short-term effects on the leachate quality of adding the highly soluble residues from the semi-dry APC process to the bottom ash prior to disposal are seen in Table 22.6 which presents leachate quality data from a combined ash monofill (the Woodburn Landfill), in Oregon, US (Cambotti and Roffman, 1993). A comparison of Table 22.6 with Table 22.2 shows that the concentrations of soluble inorganic salts, particularly chloride and calcium from the acid gas cleaning residues, are substantially higher in the combined ash leachate than in the bottom ash (and fly ash) leachate. The higher concentrations of Cd (up to 0.6 mg/l as compared to a maximum value of 0.025 mg/l at Vestskoven) which have been observed at the Woodburn Landfill are consistent with the lower pH and the high (complexing) chloride concentrations. The lower pH at the Woodburn Landfill may be caused by organic acids produced by biodegradation of residual unburnt material in the bottom ash which may possibly be higher at Woodburn than at Vestskoven. This would also be consistent with the relatively high levels of iron observed in the leachate from the Woodburn Landfill. Leachate composition data from three other combined ash landfills in the US (Kosson, 1995) confirm that the parameter ranges shown in Table 22.6 are typical of combined ash leachate.

## 22.3 DISPOSAL STRATEGIES

Due to the potential leaching of contaminants, landfilling of incinerator residues such as bottom ash, grate siftings, boiler ash, economiser ash, fly ash and acid gas cleaning residues or combined ash may have long-term consequences for the surrounding environment. It is therefore important that the disposal solutions chosen for these residues are sustainable in terms of environmental impact, maintenance requirements and energy consumption. This may be achieved only through careful consideration of potential disposal strategies.

Table 22.6
Results of Analysis of Leachate from a Combined Ash Monofill, The Woodburn Landfill, in Oregon, USA over a Period of 5 Years

| Parameter | Unit | Range of variation 1988 - 1993 |
|---|---|---|
| pH | - | 5.7 - 7.5 |
| TDS | mg/l | 14000 - 73000 |
| TOC | mg/l | 4 - 110 |
| Sulphate | mg/l | 80 - 1500 |
| Chloride | mg/l | 7700 - 50000 |
| Ammonia N | mg/l | < DL - 35 |
| Na | mg/l | 3000 - 9300 |
| K | mg/l | 520 - 6900 |
| Ca | mg/l | 1300 - 16000 |
| As | mg/l | < DL - 0.4 |
| Cd | mg/l | < DL - 0.6 |
| Cr | mg/l | < DL - 0.03 |
| Cu | mg/l | < DL - 0.6 |
| Fe | mg/l | < DL - 32 |
| Hg | mg/l | < DL |
| Pb | mg/l | < DL - 0.14 |
| Zn | mg/l | < DL - 1.6 |

TDS : Total dissolved solids
TOC : Total organic carbon
DL : Detection limit
Cambotti and Roffman, 1993

### 22.3.1 General Philosophy

The primary objective of landfilling as a waste management technique is to remove from general circulation materials/products that are no longer useful in any respect, and preferably to do this in a manner which eventually returns the basic constituents of the waste to the ecological cycle at levels similar to natural geologic materials.

A second and equally important objective of landfilling of waste is to ensure that the landfilled waste does not cause any unacceptable short or long-term impacts on the environment or on human health. This should preferably be done in a sustainable manner, i.e. without excessive and/or prolonged maintenance or operation requirements, and without a need for long-term care. These objectives can be met by disposal strategies which observe the following generally accepted principles:

- Landfills should be designed to minimise the required lifetime of active environmental protection systems (i.e. systems requiring operation or maintenance)

- Any disposal strategy should consider the intrinsic properties of the waste, and the potential health risks associated with a given disposal strategy

- Landfill design, operation and siting should be adapted to the admitted waste in such a manner that long-term emissions of leachate (and gas) become or remain environmentally acceptable.

- Any strategy for waste disposal at a landfill site must include consideration of the ultimate fate of the leachate and the residues remaining in the disposal site, as well as derived effects of disposal and leachate management.

In the following, each of these principles are discussed in relation to landfilling of waste in general and landfilling of incinerator residues in particular.

From a philosophical perspective, it could be argued that all expenses and resources required to ensure reliable and sustainable disposal must also be acceptable in terms of public health, long-term environmental impact, and energy consumption, and should be covered by those who enjoy the goods and produce the waste. Therefore, encapsulation/total containment strategies may not be considered adequate to meet these requirements. Since these only postpone the potential impacts until some time in the future when the encapsulation fails. For most waste types, the implementation of a strategy which reduces the contamination potential of the waste to a safe level, either prior to landfilling or gradually during the initial period after landfilling would be ideal. Alternatively, storage for potential recovery of resources sometime in the future could also be considered.

### Lifetime of Active Systems

Both active environmental protection systems (i.e. systems for leachate collection/removal, transport and treatment of leachate which require maintenance and input of energy) and passive environmental protection systems (i.e. systems such as geologically stable low permeable top covers and barriers and surface drains that do not require maintenance or operation) must have expected lifetimes that are long enough to ensure that they perform as intended. For a strategy based on restricting percolation of infiltrated precipitation through mineral wastes, such as most well combusted incinerator residues, this could in some cases require system lifetimes of several hundred years or more. It would be preferable (but for many types of wastes, not yet feasible) if no active environmental protection systems were required to function beyond a period of e.g. 30-50 years, after which the landfill could be left alone without any risk of unacceptable environmental impacts. A postclosure care period of 30-50 years may be a realistic target, both in terms of legal aspects and expected lifetime of technical systems, and would also be in agreement with the concept that each generation should care for its own waste.

## Waste Properties

There may be fundamental differences between leachates produced by different types of waste. For example, leachate from relatively stable mineral waste (e.g. well combusted bottom ash) behaves very differently from that produced by more reactive, biodegradable types of waste (e.g. raw MSW such as domestic waste, garden refuse). These differences should be reflected in the disposal and leachate management strategy chosen for each type of waste. Types of waste that are incompatible in terms of disposal strategy should be directed to different categories of landfills.

The normal course of landfill degradation of MSW dominated by organic, biodegradable waste may be described as an initial aerobic phase followed by the anaerobic acetogenic and methanogenic phases during which the organic compounds in the waste are broken down to simpler molecules such as short chain carboxylic acids and amino acids and eventually to ammonia, hydrogen, carbon dioxide and methane (Knox, 1992). The remaining, more persistent organic compounds may eventually be transformed into humic substances. These degradation processes normally occur in so-called "sanitary" landfills which have traditionally been used for the disposal of MSW. During the first several years after landfilling, i.e. during the aerobic and acetogenic phases and well into the methanogenic phase, the leachate generated will contain relatively high concentrations of organic compounds. Therefore, it makes sense, both from an environmental and a technical perspective, to collect the leachate and subject it to biological and perhaps chemical treatment prior to discharge, at least during the initial stages of waste degradation. This is a common practice for "sanitary" landfills. The biological wastewater treatment effectively removes a substantial part of the readily degradable organic contaminants from the leachate, and the degradation/mineralisation processes within the landfill will gradually reduce the pollution potential of the landfilled waste.

It is generally assumed (but rarely verified) that traditionally landfilled MSW as well as several other wastes will become harmless in a relatively short-time, and that a landfill therefore may be safely abandoned and forgotten after a period of, e.g. 30 or 50 years. However, neither the criteria for determining whether abandoning a site is safe, nor the length of time needed to reach this point are generally well defined or known. Both depend strongly on the exact nature of the waste and on local hydrologic conditions, the landfill and the surrounding environment. Based on a number of assumptions, Belevi and Baccini (1989) have calculated that it may take 500 - 1700 years before the content of organic C in the leachate from a traditional "sanitary" landfill has been reduced to a level of 20 mg/l. They have also calculated, that it may take 55 - 80 years for the concentration of $NH_3$+$NH_4^+$ to fall to 5 mg/l, 100 - 700 years for P to fall to 0.4 mg/l and 100 - 150 years for $Cl^-$ to fall to 100 mg/l. In relation to groundwater and surface water protection, it is often the concentrations of ammonia which remain high over a considerable period of time that are of major concern.

Inorganic waste types are not subject to biological degradation and mineralisation processes, although they may be influenced by biological activity in the residual organic matter or co-disposed organic waste. The relevant processes (Chapter 13), include chemical reactions such as hydration, carbonation, reduction, oxidation, dissolution/leaching, precipitation, etc. Like biodegradation, most if not all of these reactions will only occur in the presence of water. In addition, they are influenced by such factors as pH, redox potential, the amount of water percolating or leachate generated (e.g. expressed in terms of the liquid/solid ration, L/S), temperature, the presence of complexing or chelating organics and by time. In the long-term, diagenetic changes in ash mineralogy will occur. The time horizon for these processes may be very long, from decades to perhaps several thousands of years. The processes which are significant in relation to the risk of emission from landfilled inorganic waste and contamination of the environment may, in some cases, proceed within a few decades and strongly reduce the risk of further contamination from the landfill. In other cases, it will be uncertain whether or not even prolonged exposure to leaching has reduced the risk of contamination and whether or not the contamination may not actually increase with time, e.g. if the local conditions changed.

In a recent study (Hjelmar et al., 1995), a very simplified estimation indicated that for a 12 m high landfill and an assumed rate of infiltration/production of leachate of 200 mm/year, minimum periods of approximately 300 years and 100 years might be required for landfilled MSW and some inorganic waste types (e.g. bottom ash), respectively, to reach "final storage quality", i.e. a condition which allows the site to be safely abandoned without active environmental protection systems. An increased rate of infiltration may shorten these time periods, whereas a decreased rate of infiltration may lengthen them.

Certain types of inorganic or mineral wastes will generate leachates which initially have a relatively high concentration of inorganic salts and a moderate to low content of trace elements. As the leaching progresses with time, both the salt content and the concentration of trace elements may gradually decrease to very low values. The content of organic substances in the leachate is often very low. The application of traditional "sanitary" landfilling techniques, i.e. installation of (multiple) bottom liners, collection and subsequent treatment of leachate at a biological wastewater treatment plant, to mineral wastes exhibiting such properties would generally not constitute an optimum or a sustainable solution for a number of reasons. First, the potential period of leaching may easily exceed the projected lifetime of the liners and the leachate collection system, and pumping the leachate for treatment (particularly biological) are likely to be both energy consuming and ineffective. For such wastes, a controlled contaminant release management strategy may in some cases be more appropriate. This strategy implies that the transfer of contaminants from the landfilled material into the surrounding environment is limited to an acceptable level. This may be accomplished by controlling the quality and/or quantity of the leachate which is generated and is subsequently released without being collected and treated.

The disposal strategies applicable to organic and inorganic waste types are obviously very different and generally incompatible, both in the short and long-term. Co-disposal of organic and inorganic types of waste is therefore generally not advisable. Different types of incinerator residues may also exhibit very significant differences in behaviour when landfilled, which means that it may often be advantageous to handle and landfill each type separately.

## Adaptation of Landfill Design, Operation and Siting to Strategy and Waste Types

Since the avoidance of prolonged postclosure care is one of the objectives of landfilling, it may be necessary to design and operate some categories of landfills in two stages:

- An initial, relatively short and very active stage during which the contamination potential of the waste is reduced to an acceptable level,
- A subsequent, passive long-term stage during which the contamination potential remains at or below an acceptable level.

During the active stage, mitigating measures may be taken to enhance the processes which reduce the contamination potential of the waste. These measures may include accelerated leaching due to increased infiltration of precipitation or use of irrigation systems. Active environmental protection systems such as leachate containment, collection and treatment systems must remain fully operative during this period. When the landfill changes from the active first stage to the passive second stage, the active environmental protection systems are no longer required, and they must cease to function or undergo changes to comply with a controlled contaminant release if this is warranted by the changes in leachate composition.

Landfilling of organically dominated waste types must generally be operated in two stages; an initial mineralisation stage with active environmental protection systems and a subsequent second stage with passive environmental protection systems. The required duration of the active mineralisation stage is uncertain but may easily be several decades.

Landfilling of inorganic, mineral wastes may be operated in one or two stages. For mineral wastes with a high content of soluble contaminants (like some APC residues), a two-stage landfill operation may be necessary: An initial stage based on (possibly enhanced) leaching of contaminants with active environmental protection systems followed by a second stage, based on controlled contaminant release and requiring only passive environmental protection systems. For mineral wastes containing only limited amounts of soluble contaminants, particularly trace elements, one-stage landfill operation based on controlled contaminant release with only passive environmental protection systems may be sufficient. For some mineral wastes which do not initially qualify for one-stage landfill operation, treatment prior to disposal may present an alternative to the first active stage and render one-stage landfilling based on controlled contaminant release feasible. The treatment could consist of extraction of soluble contaminants from the waste and/or stabilisation with a binder. Extraction would reduce the total pollution potential of the waste, whereas proper stabilisation would also reduce the rate of release of contaminants substantially.

When the disposal strategy is based on controlled contaminant release, a sufficiently slow rate of transfer of contaminants from the landfill to the surrounding environment may be achieved either by ensuring that the concentrations of potential contaminants in the leachate are sufficiently low or by restricting the rate of generation and emission of leachate to an acceptable (low) level. Properly designed measures aimed at controlling the composition of the leachate (e.g. extraction and/or stabilisation) are generally secure but they also tend to be relatively complicated and energy consuming. Passive environmental protection measures (e.g., geologically stable top covers and surface drainage systems) which are aimed only at restricting the quantity of leachate produced are technically much simpler, but are dependent on long-term durability and functionality of the systems. However, for passive systems, long lifetime expectancies are not unrealistic.

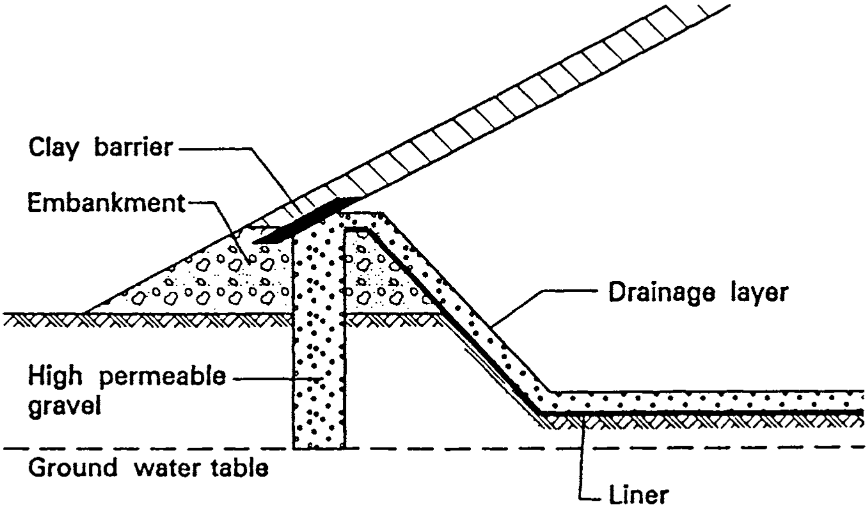
The transformations of active environmental protection systems into passive systems are required when a landfill based on two-stage operation passes from the active initial stage, which may depend on collection and treatment of leachate, to the final, passive stage which may depend solely on a controlled rate of release of relatively benign leachate. In this case, an impermeable bottom liner apparently changes status from being an instrument of one strategy to become an obstacle to another strategy. One way to avoid this is shown in Figure 22.2. Drainage layers are built into the sides of the landfill, sufficiently high up to keep them inactive by maintaining a low level of leachate on top of the liner during the active stage of landfill operation. Once the removal of leachate is discontinued, the level will rise and the drainage layers will provide conduits for dispersion of leachate into an acceptable receiving environment (Johannessen et al., 1993).

Proper siting is a crucial part of any disposal strategy, particularly if controlled release or passive discharge is envisaged from the start or during a later period of the existence of the landfill. Siting may strongly influence the criteria for acceptability of leachate in the surroundings. No disposal site should be placed on top of or immediately upstream of a valuable, sensitive aquifer or adjacent to a sensitive surface water body. However, if a saline (initial) leachate is expected, a location of a landfill near a wastewater conveyance or at the coast of a receptive marine environment would appear suitable (if available) both in case of controlled contaminant release and leachate collection, treatment and discharge.

### Ultimate Fate of the Leachate

Regardless of whether the leachate is collected, treated and discharged, or it is allowed to disperse into the surrounding environment, an energy and resource consumption analysis should always be conducted. This analysis should also include an environmental impact assessment covering the entire pathway of the leachate (until it is indistinguishable from the surroundings) for the entire leachate production period.

Figure 22.2 Example of Design which Allows the Transformation of an Active Environmental Protection System into a Passive System



Johannessen et al., 1993

Derived effects such as the environmental impact of producing the energy necessary to pump and treat the leachate for the prescribed period of time should be accounted for, and environmental impacts caused by, but remote from, the landfill (e.g., at the outfall from a wastewater treatment plant which receives the leachate) should also be considered. Any environmental protection measures at a landfill should therefore be subjected to a lifecycle analysis prior to implementation. This may help prevent the application of landfill solutions which are based on suboptimisation in space or time.

### 22.3.2 General Disposal Strategies

Disposal strategies may be categorised according to the prescribed management and intended fate of the leachate. Some of the specific strategies which may be relevant or have been applied to the disposal of incinerator residues include:

- Total containment or “entombment” (dry storage)
- Containment and collection of leachate
- Controlled contaminant release
- Unrestricted contaminant release

Total containment and containment with leachate collection generally require active environmental protection systems, whereas the controlled contaminant release and unrestricted contaminant release strategies may require only passive environmental protection systems. Active environmental protection measures may be necessary during a first stage of landfilling in a number of cases but only strategies based on passive systems are sustainable in the long-term. The four leachate management strategy scenarios are presented schematically in Figure 22.3.

### Total Containment or Entombment

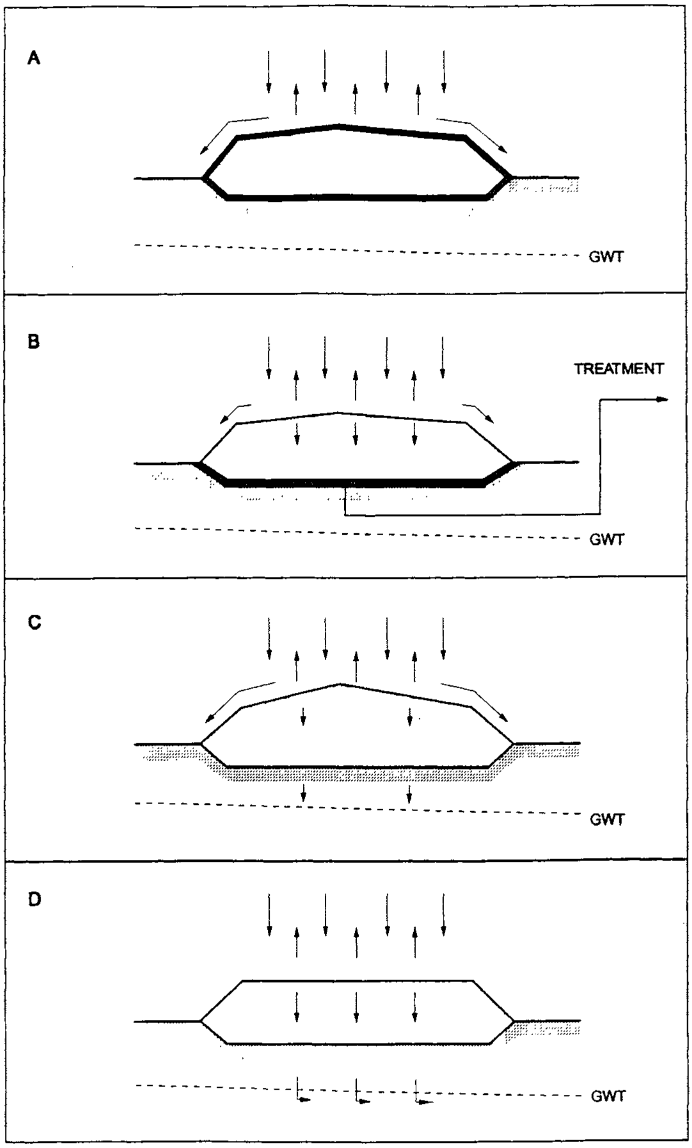
Total containment of incinerator residues will prevent any infiltration and percolation of water and, consequently, any generation and emission of a leachate provided the containment system remains intact (see Figure 22.3, A). The main weakness of this strategy is that the landfilled residues and hence the potential risk to the environment may remain virtually unchanged for a very long period, until the containment system finally fails and an uncontrolled plume of leachate may be released.

This is particularly true for residue landfills equipped with (multiple) impermeable, artificial bottom and top liners and relying on zero discharge of leachate. The environmental risk is less pronounced for storage of untreated residues (such as air pollution control (APC) residues) in old salt mines which is practised in some parts of Germany. Underground storage is, however, a relatively costly solution which is not generally available. A total containment strategy may be acceptable for temporary, short-term storage of residues, but it should always include plans or development of plans for appropriate final disposal or utilisation of the residues.

### Containment and Collection of Leachate

A strategy based on containment with leachate collection and treatment is the standard method of designing and operating a sanitary or MSW landfill (see Figure 22.3, B). The leachate generated is contained by an impermeable or low permeable bottom liner, recovered and normally subjected to treatment prior to discharge to a body of water. The rate of leachate formation may be reduced by an impermeable top cover layer. Although commonly used, this strategy may not be optimal for incinerator residues since it requires maintenance and operation for a period of time exceeding the expected lifetime of the environmental protection systems. Also, effective treatment of the leachate is likely to be difficult and energy consuming. Although the strategy may not be a long-term solution to landfilling of incinerator residues, containment and collection of leachate may be applied as a first stage of operation, for example, in connection with measures designed to enhance leaching. However, eventually it

Figure 22.3 Disposal Strategies Categorised in Terms of Leachate Management and Fate



A - Total Containment
C - Controlled Contaminant Release

B - Containment and Collection + Treatment of leachate
D - Unrestricted Contaminant Release

GWT - Groundwater Table

should be replaced by a second stage of operation, based on a more long-term oriented strategy.

### Controlled Contaminant Release

The controlled contaminant release strategy implies that the release and discharge of contaminants are maintained at an acceptable level by controlling the quantity and quality of the leachate generated within the landfill (see Figure 22.3, C). The leachate is allowed to discharge into the surroundings or to be passively discharged to a wastewater conveyance as it is formed. An assessment must always be carried out to ensure that the impact of the emitted leachate on the environment is acceptable. Both the quantity and quality of leachate depend upon the characteristics of the waste, the design and operation of the landfill and the climatic conditions. Treatment of the waste may reduce both the contamination potential and the permeability. Installation of geologically stable, sloped top covers with surface drainage systems could ensure a very low rate of infiltration of precipitation and, consequently, a very low rate of release of contaminants from a disposal site in the short and long-term. This concept requires proper siting. Since contaminants are being removed from the landfilled material, a continuous reduction of the contamination potential will occur. The controlled contaminant release strategy may represent a sustainable and therefore preferable long-term solution to disposal of residues. For some residues, a final controlled contaminant release stage of operation should be preceded by a short-term, active stage of operation based on a different strategy, e.g. enhanced leaching and containment and collection of leachate. This strategy currently is applied to disposed residues in some countries.

### Unrestricted Leaching

An unrestricted leaching strategy may simply be described as a landfill scenario where no precautions at all are taken to prevent or reduce the generation and emission of leachate (see Figure 22.3, D). This strategy represents in most respects the opposite of the total containment strategy. The environmental impact will depend on the leaching characteristics of the landfilled waste as well as on local physical and climatic conditions and the vulnerability of the surrounding environment. Since the strategy implies a total lack of control, it is generally unacceptable for landfilling of incinerator residues except in cases where the leachate has reached ambient quality.

## 22.4 DESIGN AND OPERATIONS ISSUES

Although the disposal strategies employed for landfilling of MSW incinerator residues may (or should) differ somewhat from those employed for traditional landfilling of raw MSW, most of the general considerations concerning design criteria and operational conditions of landfilling are quite similar in both cases. A detailed discussion of design and operations issues is, as mentioned previously, beyond the scope of this treatise.

The reader is referred to the appropriate textbooks and manuals on this subject which is often covered in great detail by national or regional legislation and guidelines. The following is merely a brief listing of some of the more important issues that must be dealt with before, during and after the landfilling of incinerator residues (and most other wastes).

### 22.4.1 Siting

Siting criteria are frequently utilised in the process of determining the location of disposal facilities. The type of disposal strategy being used (i.e. uncontrolled leaching, containment and leachate collection, controlled contaminant release, encapsulation/total containment) and the type of residue requiring disposal influence the siting criteria. Frequently criteria can include:

- Geotechnical Issues (slope, stability, seismic activity, soil bearing capacity)
- Hydrogeological and Environmental Issues (overburden soil type, overburden/bedrock interactions, groundwater direction and velocity, travel time, separation from potential groundwater receptors, background groundwater quality and vulnerability, separation from potential surface water receptors, background surface water quality and vulnerability)
- Civil Engineering Issues (traffic control, property abutment, site workability, site stormwater management).

### 22.4.2 Liners and Leachate Collection Systems

Landfills for which a leachate containment and collection strategy is chosen must be equipped with bottom lining and leachate collection systems. Leachate containment may be achieved using mineral liners (natural and/or artificial), synthetic liners or composite liners. Regulations or guidelines may specify hydraulic conductivity, liner thickness, type, materials, multiple liner systems, etc., often depending on landfill category (the type of waste received) and/or the vulnerability of the environment. Leachate collection systems typically consist of a layer of coarse gravel, containing drainage pipes leading to collection sumps. The allowable head of leachate on the bottom liner is typically restricted (e.g. to 0.3 to 1.0 m). Sections of a landfill which are segregated in terms of waste types accepted should have separate leachate collection systems. The malfunction of drainage systems may occur due to clogging, and one separate system should not cover too large an area of the landfill. There is only limited data available on the long-term performance of liners (particularly synthetic liners), drainage systems and collection systems.

In Sweden, for example, several small, well-defined catchment areas exist. In the opinion of the Swedish EPA, it is possilbe to regard such areas as part of the leachate containment system without getting in conflict with the proposed European Union Landfill Directive (EU, 1995).

According to the EU Landfill Directive each landfill shall be classified in one of the following classes: Hazardous waste, nonhazardous waste or inert waste. A landfill must be situated and designed so as to meet the necessary conditions for preventing pollution of the soil, groundwater or surface water. The geological barrier is determined by geological and hydrogeological conditions, see Table 22.7. In this table the corresponding vertical flow time has been calculated (hydraulic gradient = 1).

Table 22.7
Geological Barrier Requirements According to the Proposed EU Landfill Directive and Corresponding Calculated Vertical Flow Times.

| Landfill class | Permeability, K, m/s | Thickness, D, m | Vertical flow time, $t_v$, year |
|---|---|---|---|
| Hazardous waste | $\leq 1.0 \cdot 10^{-9}$ | $\geq 5$ | $\geq 15.9$ |
| Nonhazardous waste | $\leq 1.0 \cdot 10^{-9}$ | $\geq 1$ | $\geq 3.17$ |
| Inert waste | $\leq 1.0 \cdot 10^{-7}$ | $\geq 1$ | $\geq 0.051$ (=18.5 days) |

However, leakage often takes place in the unsaturated zone where the flow rate is much slower than it is in the saturated zone. Also, in the unsaturated zone, conditions are favour sorption and degradation of leachate components. In view of these circumstances it is proposed to assign a weighting factor of 10 to the vertical flow component. The total flow time through the geological barrier is then as follows:

$$t = 10 \cdot t_v + t_h = 10 \cdot (D \cdot n_e)/(K_v \cdot I) + (L \cdot n_e)/(K_h \cdot I) \qquad (22.1)$$

where

t = flow time, s
D = vertical flow length, m
K = hydraulic conductivity, m/s
L = horizontal flow length, m
I = hydraulic gradient, dimensionless
$n_e$ = effective porosity of the groundwater aquifer, dimensionless
(subscripts *v* and *h* denotes *vertical* and *horizontal* directions, respectively.)

Nominal time values for groundwater are proposed: >200 years (hazardous waste), >50 years (nonhazardous waste) and >1 year (inert waste). Corresponding calculations for surface water show that they have little barrier effect.

### 22.4.3 Caps and Top Covers

The capping of a landfill serves the dual purposes of modifying the infiltration of precipitation into the landfill and isolating the waste from the surroundings. The final capping of landfills with low permeability materials, such as clays, can greatly reduce the rate of leachate production. It is general practice to complete landfill sites with a layer of compacted clay or an artificial membrane, followed by soil and/or subsoil to support vegetation. In some cases (e.g. for MSW landfills) a gas drainage layer must be placed underneath the cap. Sloping surfaces, surface drainage systems and capillary barriers may be used to help reduce the infiltration of precipitation into the landfill if this is the part of the disposal strategy. It should be noted that synthetic cap materials, although impermeable in principle, may not be durable over a longer period of time. Furthermore, unlike most clay caps, a synthetic cap may (over a certain period of time) entirely eliminate infiltration, but it does not allow controlled infiltration at a predetermined rate which may be called for by the disposal strategy (e.g. leachate containment and collection in conjunction with waste mineralisation processes within the landfill or controlled contaminant release). A European review of clay cap performance (Knox, 1991) produced the following conclusions:

- Percolation through clay caps may range from 0 to 200 mm/annum depending on its quality and on materials and drainage arrangements above the clay layer. In many parts of Europe this represents a large reduction compared to effective rainfall.

- The performance of the cap is determined as much by what is put on top of it, as by the quality of the barrier layer itself.

- Percolation through the cap is extremely dependent on the efficiency of lateral drainage above it. For a given amount of effective rainfall, percolation is minimised when lateral drainage is maximised. To achieve a low percentage percolation, the ratio of hydraulic conductivity in the cap to that in the soil or drainage layer above it should be no greater than $10^{-4}$. The distance between field drains should be no greater than 20 m.

- Desiccation cracking of caps may lead to a large increase in percolation.

- To prevent desiccation of a cap placing sufficient soil or other material on top of it is necessary. Typically, at least 0.9 m of soil or subsoil may be needed under northern European conditions. In more arid climates greater depths may be needed to counteract desiccation.

### 22.4.4 Geotechnical Stability

The geotechnical stability of the landfilled material (e.g. bearing capacity and slope stability) must in general be sufficiently high to accommodate trucks and various types of waste moving and compacting equipment. In the longer term, it must be ensured that settlements, particularly differential settlements, do not hamper or interfere with the intended performance of liners, caps and drainage systems.

### 22.4.5 Abatement of Noise, Odour and Fugitive Dust Problems

Proper precautions must be taken to minimise noise, odour and fugitive dust problems during the operation of a landfill. Keeping the residues moist or contained are among the measures used to avoid fugitive dust problems at residue landfills. In traditional landfilling, it is common practice to apply a daily cover of topsoil to the landfilled waste in order to minimise problems with odour, dust and vermin. Depending on the quality of the soil used, this may have the undesirable effect of creating a number of hydraulically isolated waste cells within the landfill.

### 22.4.6 Monitoring of Leachate Quantity and Quality

Both the quantity and the quality of the leachate generated at any landfill equipped with leachate collection systems should be monitored during the period of operation and during the postclosure period until the waste in the landfill has reached final storage quality. Climatic data (e.g. precipitation and temperature) should also be monitored.

The main objectives of monitoring the quantity of leachate produced at a landfill are 1): to evaluate through water balance calculations whether the leachate collection system is functioning as intended; 2): to check the efficiency of any systems intended to modify the rate of production of leachate and to obtain information on the actual rate of production of leachate at the landfill in question, thus enabling proper planning of leachate management; and 3): to provide information on the accumulated amount of leachate produced at a given landfill and hence allow an evaluation of the attained degree of leaching of the waste.

The main objectives of monitoring the quality of leachate produced at a landfill are 1): to provide a basis for the selection of indicator parameters for the monitoring of groundwater (and surface water) monitoring; 2): to ensure that the leachate quality complies with the criteria in relation to on-site management and direct discharge, on-site treatment or treatment at a municipal wastewater treatment plant; and 3): to provide information on the progress of the waste stabilisation processes occurring within the landfill and, eventually, to provide background for an assessment of whether or not the waste has reached final storage quality.

The leachate sampling frequency should be higher during the period of landfill operation than during the subsequent postclosure period. The analytical programme will depend on the type of waste present in the landfill and possibly also on the perceived environmental risks. A programme may consist of a routine analytical programme with relatively few parameters carried out on leachate samples collected relatively frequently and a more comprehensive extended analytical programme which is carried out on samples collected at longer time intervals. For instance, for leachates from incinerator residues, the simple analytical programme could comprise the determination of pH, conductivity, total dissolved solids (TDS), nonvolatile organic carbon (NVOC), chloride, sulphate, alkalinity and temperature. In addition to this, the extended programme could include measurement of adsorbable organic halogen (AOX), sulphide, total N, ammonia N, $NO_x$ N, Na, K, Ca, Mg, Fe, Pb, Cu, Cr, Hg, Ni, Zn and possibly As and Mo. The sampling point should be chosen carefully and the sampling procedure should allow for correct measurement of sensitive leachate quality parameters such as pH, redox potential, sulphide content, etc.

### 22.4.7 Monitoring of Groundwater and Surface Water Quality

The main objectives of groundwater quality monitoring at landfills are 1): to provide information on background levels and natural variations of downstream groundwater quality prior to operation of a landfill; 2): to detect at the earliest possible time any unintended leakage of leachate from a lined landfill into the aquifer; and 3): to ensure that the anticipated impact on the aquifer is not exceeded if a controlled contaminant release strategy is applied to a landfill.

The implementation of a groundwater monitoring programme which is highly dependent on local conditions includes the performance of geological and hydrogeological surveys to determine how many monitoring wells are needed, where they should be located and how they should be designed to ensure maximum likelihood of detecting any leachate plume at an early stage. It also includes the installation of such monitoring wells and implementation of proper groundwater sampling procedures. A detailed discussion of these subjects is, however, beyond the scope of this study. Several books and manuals are available on these issues. An overview is provided by Christensen et al. (1992).

The analytical requirements for a groundwater monitoring programme should be based upon information on the composition of the leachate from the landfill in question and the background quality of the groundwater. The monitoring parameters should generally be chosen among components which are present in the leachate at significant concentrations, which are relatively mobile in the aquifer, and/or which are present in the groundwater at low background concentrations. The latter condition facilitates the detection of a leachate plume due to a large relative concentration contrast between contaminated and uncontaminated groundwater.

In order to establish background levels, and the natural or seasonal variations of the groundwater to be monitored it is proposed that the samples from the monitoring wells are analysed 4 times a year for 2 years prior to commencement of landfill operation. The analytical programme carried out during this period should correspond to the extended programme for leachate described in Section 22.4.6. When this has been accomplished, both the sampling frequency and the analytical programme should change.

The sampling frequency should be high enough to ensure that a leachate plume will be detected before it has migrated significantly past a monitoring well. The sampling frequency should also be low enough to ensure that the water that is drawn into the well from the aquifer has been replaced between sampling events by water flowing from upstream. The sampling frequency will thus depend on the rate of flow of the groundwater which must be determined for each individual landfill site. It is recommended, however, that a groundwater sampling frequency of not less than once a year is adopted in any case.

An analytical programme could consist of a basic set of parameters (pH, conductivity, TDS, NVOC, chloride, sulphate, ammonia, Na, K and Ca) which for each individual landfill may be supplemented with further parameters. The basic parameters are designed to provide an early warning for components which are present in most incinerator residue leachates. The supplementary parameters must be selected for each individual landfill based on knowledge of the actual waste accepted, its leaching properties, the composition of the leachate and the mobility and background values of various leachate components in the aquifer. Hjelmar et al. (1988) report that very early signs of a plume of bottom and fly ash leachate in an aquifer were observed as continuously rising concentrations of chloride and calcium (due to ion exchange between Na and K in the leachate and Ca in the soil) and other mobile salts in groundwater from a downstream monitoring well.

The monitoring of the groundwater quality should continue during the postclosure period, possible at a reduced sampling frequency, until the waste in the landfill has reached final storage quality.

The main objectives of surface water quality monitoring at landfills are 1): to provide information on background levels and natural variation of the quality of surface waters to which leachate is directly or indirectly discharged; and 2): to detect any unintended impact of leachate on a surface water system and to ensure that the anticipated impact on the surface water system is not exceeded if a controlled contaminant release strategy is applied to a landfill.

It should be noted that monitoring of surface water systems generally is more difficult and much less likely to produce useful results than monitoring of groundwater. Surface water systems are usually much more diverse and dependent on local conditions than aquifers. There may be none or several surface water bodies downstream of a landfill and within the same catchment area or close enough to be affected by spillages of leachate. With the exception of stagnant waters, most surface waters systems generally have much higher dilution potentials than aquifers do. This means that leachate which may be discharged directly, may be removed rapidly or diluted by the surface water to the extent that direct detection by chemical analysis is difficult or impossible.

In cases where surface water quality monitoring is found desirable, the design of the chemical analytical programme may be based on the same principles and include similar ranges of analytical parameters as for groundwater while taking into account the nature of the surface water body in question. Analytical parameters such as chloride, sulphate, Na, K, Ca and Mg are, of course, irrelevant as contamination indicators in marine surface water bodies. Due to the low concentration levels of contaminants which must be expected for affected surface water systems, the monitoring programme may in certain cases alternatively be based on eutrophication indicators (e.g. chlorophyll-a) or on biological monitoring (in marine waters, e.g. placement and subsequent analysis of mussels).

### 22.4.8 Leachate Treatment

The most common leachate disposal route for landfills with active leachate removal systems is to sewer and subsequently to a biological wastewater treatment plant without any pretreatment. Biological treatment only affects the content of biodegradable organic contaminants (and perhaps ammonia) in the leachate. Leachates with high organic loadings may require aerobic biological pretreatment to remove organics and ammonia prior to discharge to sewer or surface water. Incinerator residue leachates, however, normally have relatively low contents of organics, but may have high contents of inorganic salts (e.g. $Cl^-$, $SO_4^{2-}$, $Na^+$, $K^+$, $Ca^{2+}$) and variable contents of trace elements/heavy metals. Such leachates may be treated/pretreated by pH and/or redox potential adjustment, precipitation (e.g. with TMT)/filtration and/or adsorption on activated carbon to remove dissolved trace metals, suspended solids and non-degradable organics prior to discharge to sewer or surface water bodies. Preconcentration or removal of salts may be accomplished by reverse osmosis or evaporation, although this may not be advisable for this type of leachate from an economic or a life cycle perspective (Hjelmar et al., 1995). In most cases, biological wastewater treatment will have little effect on incinerator residue leachates dominated by inorganic constituents, but discharge to sewer may be convenient. If such leachates have very low contents of trace elements and organics, they are compatible with seawater and may be discharged directly into marine surface water bodies without pretreatment. Leachates from landfilling of poorly combusted residues may have high contents of organic material and may require (and benefit from) biological wastewater treatment.

## 22.5 DISPOSAL PRACTICES

Data have been compiled for a number of countries regarding present and future disposal practices for incinerator residues. In almost all jurisdictions, regulations have been or are presently undergoing modifications. In Europe, for instance, a number of countries will have to modify regulations to come into compliance with an anticipated European Union (EU) waste disposal directive (European Union, 1995) and other EU waste management legislation. In the US, the Supreme Court ruling in May 1994 which lifted the exemption of MSW incinerator ash from being tested as a potentially hazardous waste may give rise to changes in some states (Supreme Court of the United States, 1994). Consequently, the situation is somewhat dynamic, and since the collected information may be outdated fairly rapidly, only a brief, general outline of the current disposal practices in some European and North American countries is presented in this section.

Disposal practices for MSW incinerator residues vary widely across the world. Substantial variations in disposal practices are also found within countries consisting of federations of states or provinces such as US, Canada and Germany. Table 22.8 summarises the disposal strategies corresponding to the landfilling practices and/or policies of some countries.

Table 22.8
MSW Incinerator Residue Disposal Practices in Various Countries

| Disposal strategy | Bottom ash | APC residues | Combined ash |
|---|---|---|---|
| Total containment/dry tomb | Germany | Canada<br>Denmark (d)<br>The Netherlands (d)<br>Germany | US |
| Leachate containment and collection | Denmark<br>France<br>Germany<br>Sweden<br>Switzerland<br>The Netherlands | Denmark (d)<br>France (a)<br>Germany (wet scrubber)<br>Sweden (a)<br>Switzerland (a) | US |
| Controlled contaminant release | Sweden<br>Denmark (b,c) | Sweden (a) | |
| Unrestricted contaminant release | Canada | | |

(a): Residue treatment or stabilisation required
(b) Past practice
(c) Planned future practice (pending legislation)
(d) Temporary practice (awaiting improved treatment and disposal technology)

Monofill disposal represents the most common method of bottom ash disposal in Europe and Canada, although co-disposal of incinerator residues with other wastes, including MSW, does occur. The strategies employed for disposal of bottom ash cover the entire range from total containment over leachate containment and collection and controlled contaminant release to unrestricted contaminant release. At present, however, the prevailing disposal strategy for bottom ash is containment with some type of leachate collection.

In Canada and several European countries, APC system residues are listed as hazardous or special wastes, and disposal at highly engineered monofills with extensive environmental protection systems corresponding to total containment or leachate containment and collection is normally required, often in conjunction with treatment/stabilisation of the residues. In France, for instance, stabilisation of APC residues is required prior to disposal at hazardous waste landfills, and in Germany most of the residues from dry/semi-dry APC processes are placed in underground storage in old salt mines. At one disposal site in Sweden, residue from the dry APC process is stabilised with approximately 30% (w/w) of a special type of cement; the material (37% residue, 16% cement and 47% water) is poured onto the site as a slurry in 2.5 m deep layers (cells) and allowed to cure. The median concentration of chloride in the leachate which consists primarily of surface runoff water is approximately 20,000 mg/l and the median concentration of lead is approximately 0.06 mg/l over a period from 2 to 6 years after the commencement of disposal (Sundberg and Tuutti, 1994).

In the US, most of the incinerator residues generated are managed as combined ash. The combined ash is frequently disposed in landfills with relatively stringent design standards for leachate containment and collection, or total containment strategies. Most regulations evolve from the State level, provided Federal criteria are met. As of 1991, 25 states required monofill disposal, 5 states allowed for monocells inside MSW landfills, and 16 states allowed codisposal of incinerator ash and raw MSW. According to the above mentioned Supreme Court decision, all MSW incinerator ash in the US must now be tested using the TCLP test (see Chapters 14 and 16). If the ash passes the requirements, it may be disposed in a Subtitle D landfill for nonhazardous waste. Conversely, if it fails, it must be placed in Subtitle C landfills for hazardous waste (as specified in the Resource and Recovery Act). The ash may be treated and combined prior to testing.

## 22.6 DISPOSAL RECOMMENDATIONS FOR INCINERATOR RESIDUES

Based on the data presented on incinerator residue leaching characteristics and the discussion of disposal strategy, a number of conclusions may be drawn concerning the feasibility of various disposal and leachate management options.

The disposal strategies applicable to mineral wastes such as incinerator residues and organic waste types are, as previously discussed, very different and generally incompatible, both in the short and long-term. Co-disposal of incinerator residues and raw MSW is therefore generally not advisable. Different types of incinerator residues may also exhibit significant differences in behaviour when landfilled, and separate management and disposal of, e.g. bottom ash and APC residues, is therefore recommended, since the opportunity of applying different disposal strategies to different types of incinerator residues when appropriate is lost if bottom ash and APC system residues are combined. This recommendation differs from the current practice in the US, where disposal of combined ash is frequently managed using very stringent landfill design standards (e.g. in double lined monofills). While this practice generally is environmentally protective, at least in a short-term perspective, it does require perpetual maintenance and can be costly, especially since the bulk of the combined ash (bottom ash) may not require the same level of perpetual care as a combined ash or APC residue fill.

Overall, the optimal disposal strategy for the various incinerator residues may be considerably different from the traditional disposal strategies applied to raw MSW. The most feasible disposal options for bottom ash, APC residues and combined ash are briefly discussed below and summarised in Table 22.9.

Table 22.9
Summary of Incinerator Residue Options and Recommendations

| Disposal strategy option | Bottom ash | APC residue | Combined ash |
|---|---|---|---|
| Total containment/dry tomb | No | Possibly<br>(e.g. salt mines) | No |
| Leachate containment and collection | Yes (a) | Yes (a) | Yes (a) |
| Controlled contaminant release | Yes (b) | Maybe (b) | Maybe (c) |
| Unrestricted contaminant release | No (c) | No (c) | No (c) |

(a) : If requirements for controlled contaminant release are not met (e.g. as a first stage of disposal).
(b) : If requirements are met. May require prior or in-situ treatment of the residues or may be second stage of disposal.
(c) : Only after final storage quality criteria are met.

### 22.6.1 Bottom Ash

With proper siting (e.g. close to the sea or in an area without vulnerable aquifers), a disposal strategy based on controlled contaminant release seems appropriate and should be pursued for landfilling of bottom ash. Pretreatment (e.g. washing or stabilisation) or an initial disposal stage entailing containment, collection and treatment of leachate may in several cases be required (Belevi et al., 1992). The possibilities for controlling the geochemical and biogeochemical conditions within a bottom ash landfill through ash quality requirements, ash treatment and landfill design should be investigated further. The rate of leachate production may be controlled partly through the design of the landfill. The construction of any bottom ash disposal site based on a controlled contaminant release strategy must be preceded by a thorough environmental impact assessment which ensures that the rate of release of contaminants into the surrounding environment will not exceed an acceptable limit, neither in the short nor long-term.

If, for some reason, a solution requiring containment and collection of leachate is chosen, either temporarily or indefinitely, it becomes necessary to manage and dispose of the leachate. Leachate from bottom ash is generally accepted at wastewater treatment plants as long as it does not constitute a major proportion of the total input to the facility. It has in some cases been necessary to reduce the pH of the leachate (by addition of sulphuric acid) and/or to elevate the redox potential from a reducing level to an oxidised level (e.g. by addition of hydrogen peroxide) prior to treatment at a wastewater treatment plant. In most cases no pretreatment has been necessary. Dilution is practically the only beneficial effect of biological wastewater treatment on bottom ash leachate containing mostly inorganic salts and little or no organic degradable matter.

### 22.6.2 APC Residues (Fly Ash and Acid Gas Scrubbing Residues)

A sustainable disposal solution for the APC residues, particularly fly ash and residues from the dry/ semi-dry acid gas scrubbing processes, must eventually be based on a controlled contaminant release strategy and will almost certainly require extensive pretreatment of the residues. A two-stage treatment process involving removal and possibly recovery of the soluble salts (washing/extraction) followed by stabilisation, vitrification or fixation of the remnant may be appropriate for this purpose (Hjelmar, 1992). Considerable efforts are currently being spent on the development of such processes. In the meantime, disposal of APC residues must generally be based on less sustainable strategies involving total containment/entombment or containment and collection of leachate. The same requirements concerning proper siting and design of a landfill and performance of an environmental impact assessment as mentioned above for bottom ash apply to a controlled contaminant release disposal strategy for APC residues.

The leachate produced at APC residue disposal sites based on containment and collection of leachate generally has a high concentration of inorganic salts and in some cases also relatively high concentrations of trace elements, particularly Pb and Cd. Such leachate is often accepted at municipal wastewater treatment plants without prior treatment, provided it does not constitute a major proportion of the total input to the plant. As mentioned for bottom ash leachate, the only beneficial effect of such a treatment is dilution. Leachates with a high content of heavy metals may have an adverse effect on the sludge from a biological treatment plant. If necessary, the concentration of several trace elements in the leachate (e.g. Cd and Pb) may be reduced substantially by subjecting the leachate to pretreatment including adjustment of pH and sedimentation/flocculation with TMT. This treatment may be relatively expensive if large amounts of leachate are produced. Removal of the inorganic salts from the leachate (e.g. by evaporation) is not economically feasible or environmentally desirable under most circumstances.

### 22.6.3 Combined Ash

Although separate management and disposal of the different residue streams are believed to be technically and economically advantageous, both in the short and long-term, combined ash is still generated in the US. In principle, the disposal requirements for combined ash are similar to those described for APC residues. The proportion of residue requiring relatively stringent environmental protection measures when landfilled is increased substantially by the mixing of APC residue and bottom ash which also precludes utilisation and renders pretreatment or in-situ treatment of the residue more difficult and less efficient than it would be for separate ash streams.

## REFERENCES

Andersen, L. & J. Boll. "Leaching of APC System Residues from MSW Incinerators, Pilot Scale Experiments", In: Udvaskning fra slagger, jord og affald, ATV -Komiteen vedrørende grundvandsforurening, Lyngby, Danmark, pp. 85-106 (in Danish), 1994.

Belevi, H. and P. Baccini. "Long-Term Behaviour of Municipal Solid Waste Landfills", Waste Management & Research, 7:43-56, 1989.

Belevi, H., D.M. Stämpfli and P. Baccini. "Chemical Behaviour of Municipal Solid Waste Incinerator Bottom Ash in Monofills", Waste Management & Research, 10:153-167, 1992.

Cambotti, R.K. & H.K. Roffman, "Municipal Waste Combustion Ash and Leachate Characterization. Monofill - Fifth Year Study, Woodburn Monofill, Woodburn, Oregon", Report prepared by AWD Technologies, Pittsburgh, Pennsylvania, 1993.

Christensen T.H., P. Kjeldsen and J.L.C. Jansen. "Groundwater Control Monitoring at Sanitary Landfills." In (Christensen, Cossu and Stegemann, eds.): Landfilling of Waste: Leachate. Elsevier, London, 1992.

Comans, R.N.J., H.A. van der Sloot and P.A. Bonouvrie. "Geochemical Reactions Controlling the Solubility of Major and Trace Elements During Leaching of Municipal Solid Waste Incinerator Residues", in Proceedings of the 1993 International Conference on Municipal Waste Combustion, March 30 - April 2, 1993, Williamsburg, Virginia, USA, 1993.

European Union. "Directive on the Landfilling of Waste", Brussels, Belgium, 1995.

Hjelmar, O. "Leachate from Incinerator Ash Disposal Sites", in Proceedings of the International Workshop on Municipal Waste Incineration, Montreal, Canada, October 1-2, 1987.

Hjelmar, O. "Characterization of Leachate from Landfilled MSWI Ash", in Proceedings of the International Conference on Municipal Waste Combustion, Hollywood, Florida, April 11-14, 1989.

Hjelmar, O. "Field Studies of Leachates from Landfilled Combustion Residues", Presented at WASCON '91, Environmental Implications of Construction with Waste Materials, Maastricht, The Netherlands, November 10-14, 1991.

Hjelmar, O. "Municipal Solid Waste Incinerator Flue Gas Cleaning in Denmark: Residue Properties and Residue Management Options", in Proceedings of ISWA Specialized Conference on Incineration and Biological Waste Treatment, Amsterdam, The Netherlands, September 1-3, 1992.

Hjelmar, O. "Leaching Properties of Fly Ash from MSW Incinerators", Report for the National Agency for Environmental Protection, VKI Water Quality Institute, Hørsholm, Denmark, 1993

Hjelmar, O. "Disposal Strategies for Municipal Solid Waste Incineration Residues", J. Haz. Mats., 47, pp. 345-368, 1996.

Hjelmar, O., K.J. Andersen, J.B. Andersen, E.A. Hansen, A. Damborg, E. Bjørnestad, A.H. Knap, C.B. Cook, S.B. Cook, J.A.K. Simmons, R.J. Jones, A.E. Murray, M.J. Lintrup, H. Schrøder, F.J. Roethel. "Assessment of the Environmental Impact of Incinerator Ash Disposal in Bermuda", Final Report, Prepared for Ministry of Works & Engineering, Hamilton, Bermuda, by the Water Quality Institute, Hørsholm, Denmark, 1993.

Hjelmar, O., E. Aa. Hansen and A. Rokkjær. "Groundwater Contamination from an Incinerator Ash and Household Waste Codisposal Site", In: UNESCO Workshop on Impact of Waste Disposal on Groundwater and Surface Water, Copenhagen, Denmark, 1988.

Hjelmar O., L.M., Johannessen, K. Knox, H.-J. Ehrig, J. Flyvbjerg, P. Winther and T.H. Christensen "Management and Composition of Leachate from Landfills", Final Report to the Commission of the European Communities, DGXI A.4, Waste '92, Contract No.: B4-3040/013665/92, 1995.

Johannessen, L.M., O. Hjelmar and J. Riemer. "A New Approach to Landfilling of Waste in Denmark", in Proceedings of Sardinia '93, IV International Landfill Symposium, S. Margherita di Pula, Italy, 11-15 October 1993, 1993.

Knox, K. "A Review of Water Balance Methods and Their Application to Landfill in the UK". Report prepared for the UK Department of the Environment, DOE report No. CWM 03/91, 1992

Knox, K. "Control of Landfill Leachate", in Proceedings of the 8th International Conference: Water: Supply and Quality, Cork, Ireland, 1992.

Kosson, D.S. Personal communication, 1995.

Kosson, D.S., H.A. van der Sloot and T.T. Eighmy. "An Approach for Estimation of Contaminant Release During Utilization and Disposal of Municipal Waste Combustion Residues", J. Haz. Mats. 47, 1996.

Lyons, M.R. "The WES-PHix Ash Treatment Process", Wheelabrator Environmental Systems Inc., Hamilton, NH, USA, 1995.

Sundberg, J. and K. Tuutti. "Solidification of APC System Residue from Högdalenverket", Final Report 1994, Terratema ab, Linköping, Sweden (in Swedish), 1994

Supreme Court of the United States. "Syllabus: City of Chicago et al. v. Environmental Defense Fund et al.", Certiorari to the United States Court of Appeals for the Seventh Circuit. No. 92-1639. Argued January 19, 1994 - Decided May 2, 1994.

Reimann, D.O. "Abwasserbehandlung aus Müllverbrennungsanlagen", Müll und Abfall, 19 (1):1-7, 1987.

Thygesen, N., F. Larsen and O. Hjelmar. Environmental Risk Screening of Utilization and Disposal of MSWI Bottom Ash, Miljøprojekter 203, National Agency for Environmental Protection, Copenhagen, Denmark, (in Danish), 1992.

van der Sloot, H.A., R.N.J. Comans, T.T. Eighmy and D.S. Kosson. "Interpretation of MSWI Residue Leaching Data in Relation to Utilization & Disposal", in Proceedings of the International Recycling Conference, Berlin, Germany, 1992.