RICHARDSON SMITH GARDNER
& ASSOCIATES
ENGINEERING & GEOLOGICAL
SERVICES

November 6, 2009

Ms. Jaclynne Drummond
NCDENR DWM Solid Waste Section
1646 Mail Service Center
Raleigh, North Carolina 27699-1646

RE: Soil Analytical Results
Leachate Release Compliance Order
Davidson County Landfill
Thomasville, North Carolina

Dear Ms. Drummond:

As you know, Richardson Smith Gardner and Associates, Inc. (RSG) has been assisting Davidson County Landfill personnel in responding to a Compliance Order from NCDENR dated July 15, 2009. The Compliance Order addressed, in part, a leachate release that occurred on June 8, 2009. Under Condition #4 for Compliance, Davidson County agreed to a plan to collect soil and water samples in the area where the release occurred. The results of this preliminary sampling were transmitted to your office in the response to the Compliance Order dated July 29, 2009.

The analysis of soil sample #3 (location shown on the attached **Figure 1)**, indicated concentrations of total phosphorus above the site background (soil sample #1). At the time of the initial sampling, Davidson County personnel were responding to the release and cleaning up the area. An evaluation of the phosphorus in this area and resampling of this location was requested to confirm the analysis. Since the initial sampling, the area has been regraded, fertilized and seeded.

On September 25th 2009, RSG personnel mobilized to the site to collect a second soil sample from this location. The results indicate a concentration of 539 mg/kg. Analysis of this sample for total phosphorus indicates lower concentrations than the original analysis (793 mg/kg), however, the concentration of phosphorus remains above the site background concentration and soil standards. The most recent laboratory analytical report is included in **Attachment 1**.

To further evaluate potential sources of the phosphorus, RSG personnel reviewed published leachate analytical data of phosphorus concentrations, as well as phosphorus concentrations of sites that we currently work with. Published sources that were reviewed included the following papers:

- Present and Long Term Composition of MSW Landfill Leachate: A Review, Kjelsen and Barlaz, et. al., Critical Reviews in Environmental Science and Technology32(4) pp 297 – 336, 2002.

- Guidance Note of Leachate Management for Municipal Solid Waste Landfills, Johannessen, The World Bank Urban and Local Government Urban Waste Management Thematic Group, 1999.

Copies of these papers are included in **Attachment 2** for your review.

Each of these reports indicate that phosphorus concentrations at landfills in the U.S. and Europe range from 0.1 to 30 mg/l. This is consistent with laboratory data we've analyzed at other landfills in North Carolina. Given the low concentrations of phosphorus generally found in leachate, it is unlikely that the leachate release was entirely responsible for the elevated concentrations of phosphorus in soils in this area.

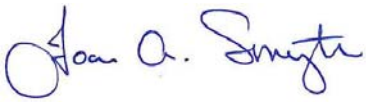
Additionally, prior to the second sample collection event, Davidson County personnel graded, fertilized and seeded the area for a grass cover. It is likely that the fertilizer used (which contains phosphorus) in this area has contributed to the elevated phosphorus concentrations detected in the sample.

A review of published information regarding the removal of phosphorus from soils included the article *Agriculture and Its Relationship to Toxic Dinoflagellates in the Chesapeake Bay*[1] which recommends removal through the use of crops. A conversation with representatives of the NC Agricultural Extension Office in Davidson County, North Carolina[2] indicated that a cover crop of grass would be sufficient to decrease phosphorus concentrations in the soils as long as the cuttings are removed from the area after they are cut.

Based upon the likelihood of multiple sources for the elevated phosphorus concentrations, we recommend phytoremediation of the area by establishing a grass cover crop which will take up the phosphorus from the soils for growth. The crop should be mowed and grass cuttings removed from the area to prevent the decomposing grasses from re-depositing the phosphorus in the soils.

We further recommend that a soil sample be collected in mid-2010 to evaluate phosphorus concentrations and the effectiveness of this strategy. If you have any questions, or require additional information, please call me at 919-828-0577 x 122 at your earliest convenience.

Sincerely,
Richardson Smith Gardner and Associates, Inc.



Joan A. Smyth, P.G.
Senior Hydrogeologist

Attachments

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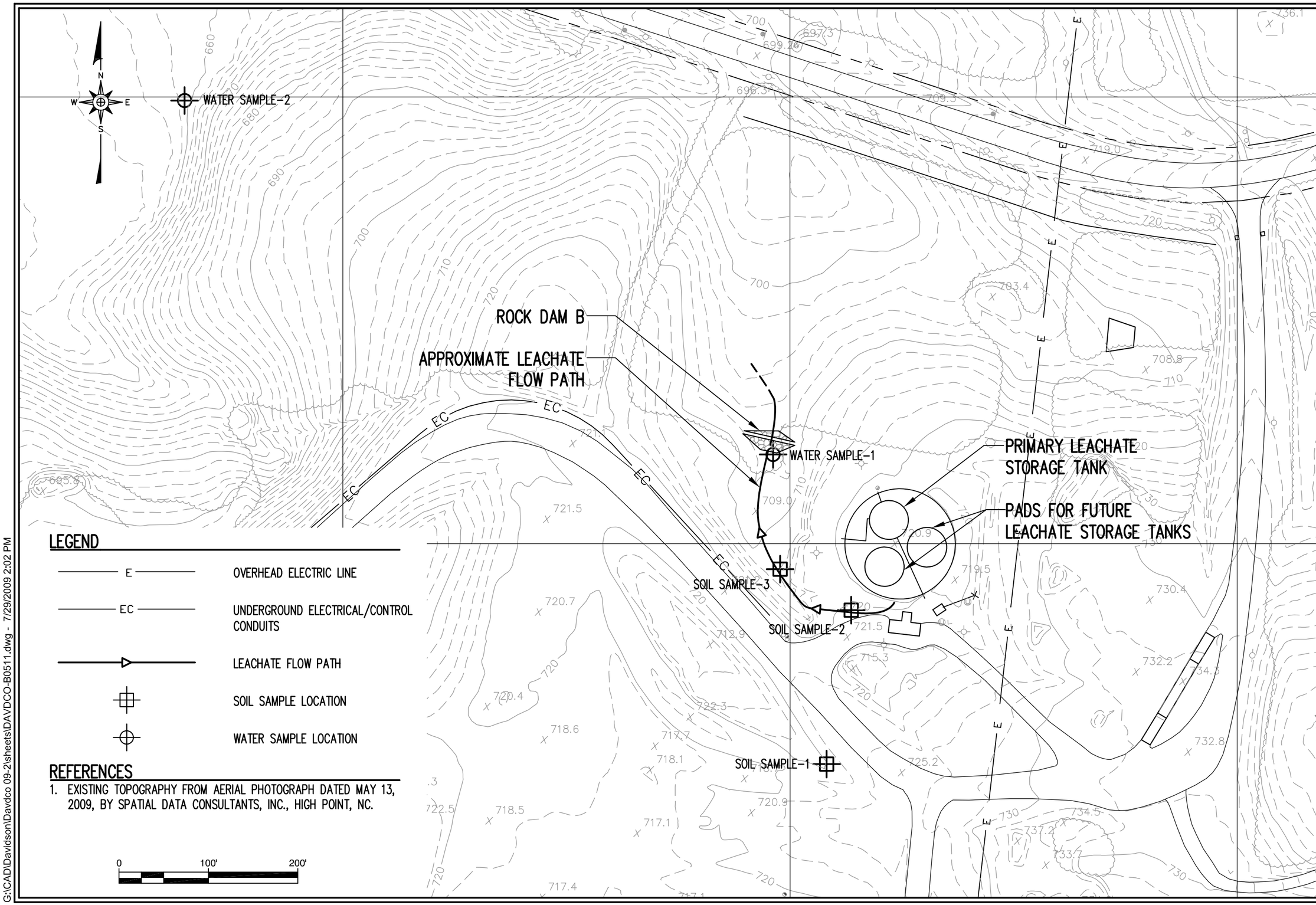
[1]Agriculture and Its Relationship to Toxic Dinoflagellates in the Chesapeake Bay, University of Maryland College of Agriculture and Natural Resources, www.agnr.umd.edu/pfiesteria/agpro2.htm#remiss
[2] Phone conversation between Joan Smyth, P.G. of RSG and Karl McNight of the Davidson Co. NC Agricultural Extension Office, 10/22/09

Figure



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Raleigh, N.C. 27603
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www.rsgengineers.com

FIGURE NO.	1
FILE NAME	DAVDCO-B0511
SCALE:	AS SHOWN
PROJECT NO.	DAVDCO 09-2
CHECKED BY:	
DRAWN BY:	J.A.L.
DATE:	Jul. 2009

TITLE:
DAVIDSON COUNTY LANDFILL - PHASE 2
LEACHATE RELEASE 6/8/09
SOIL AND WATER
SAMPLING LOCATIONS

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Attachment 1
Soil Analytical Report

Environment 1, Incorporated

REC'D OCT 13 2009

Drinking Water ID: 37715
Wastewater ID: 10

P.O. BOX 7085, 114 OAKMONT DRIVE
GREENVILLE, N.C. 27835-7085

PHONE (252) 756-6208
FAX (252) 756-0633

ID#: 1024

DAVIDSON COUNTY PHASE 2 (SOILS)
MS. JOAN SMYTH
RICHARDSON SMITH GARDNER
14 N. BOYLAN AVENUE
RALEIGH ,NC 27603

DATE COLLECTED: 09/25/09
DATE REPORTED : 10/08/09

REVIEWED BY: _____

PARAMETERS	Soil Sample #3	Analysis Date	Analyst	Method Code
Phosphorus, T. (dry wt.), mg/kg	539	10/08/09	TWA	EPA365.4

Environment 1, Inc.
P.O. Box 7085, 114 Oakmont Dr.
Greenville, NC 27858

Phone (252) 756-6208 • Fax (252) 756-0633

CHAIN OF CUSTODY RECORD

CLIENT: 1024

DAVIDSON CO.

DISINFECTION
- ☐ CHLORINE
- ☐ UV
- ☐ NONE
- ☐ _____

CHLORINE NEUTRALIZED AT COLLECTION

pH CHECK (LAB)

CONTAINER TYPE, P/G

CHEMICAL PRESERVATION

PARAMETERS

A - NONE
B - HNO_3
C - H_2SO_4
D - NAOH
E - HCL
F - ZINC ACETATE
G - NA THIOSULFATE

SAMPLE LOCATION	COLLECTION DATE	COLLECTION TIME	TOTAL CHLORINE, mg/l AT COLLECTION	TEMPERATURE, °C AT COLLECTION	# OF CONTAINERS																	
									A													
									T. PHOS													
SOIL SAMPLE #3	9/25/09	1:00			1				G													

CLASSIFICATION:
- ☐ WASTEWATER (NPDES)
- ☐ DRINKING WATER
- ☐ DWQ/GW
- ☐ SOLID WASTE SECTION

CHAIN OF CUSTODY MAINTAINED DURING SHIPMENT/DELIVERY: (Y) N

SAMPLES COLLECTED BY: (Please Print) Don Misenheimer

SAMPLES RECEIVED IN LAB AT 0.3 °C

RELINQUISHED BY (SIG.) (SAMPLER)	DATE/TIME	RECEIVED BY (SIG.)	DATE/TIME
Don Smyth	9/28/09 3:10	B	9/29 10:24
RELINQUISHED BY (SIG.)	DATE/TIME	RECEIVED BY (SIG.)	DATE/TIME
RELINQUISHED BY (SIG.)	DATE/TIME	RECEIVED BY (SIG.)	DATE/TIME

COMMENTS:

PLEASE READ Instructions for completing this form on the reverse side.

Sampler must place a "C" for composite sample or a "G" for Grab sample in the blocks above for each parameter requested.

FORM #5

№ 188903

Attachment 2
Published Information

Critical Reviews in Environmental Science and Technology, 32(4):297–336 (2002)

Present and Long-Term Composition of MSW Landfill Leachate: A Review

Peter Kjeldsen,[1] Morton A. Barlaz,[2] Alix P. Rooker,[2] Anders Baun,[1] Anna Ledin,[1] and Thomas H. Christensen[1]*

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ABSTRACT: The major potential environmental impacts related to landfill leachate are pollution of groundwater and surface waters. Landfill leachate contains pollutants that can be categorized into four groups (dissolved organic matter, inorganic macrocomponents, heavy metals, and xenobiotic organic compounds). Existing data show high leachate concentrations of all components in the early acid phase due to strong decomposition and leaching. In the long methanogenic phase a more stable leachate, with lower concentrations and a low BOD/COD-ratio, is observed. Generally, very low concentrations of heavy metals are observed. In contrast, the concentration of ammonia does not decrease, and often constitutes a major long-term pollutant in leachate. A broad range of xenobiotic organic compounds is observed in landfill leachate. The long-term behavior of landfills with respect to changes in oxidation-reduction status is discussed based on theory and model simulations. It seems that the somewhere postulated enhanced release of accumulated heavy metals would not take place within the time frames of thousands of years. This is supported by a few laboratory investigations. The existing data and model evaluations indicate that the xenobiotic organic compounds in most cases do not constitute a major long-term problem. This may suggest that ammonia will be of most concern in the long run.

KEY WORDS: waste disposal, xenobiotic organic compound, ammonia, dissolved organic matter, heavy metals, toxicological testing.

I. INTRODUCTION

The burial of municipal solid waste in landfills is the most common disposal alternative in most countries. Landfill leachate is generated by excess rainwater percolating through the waste layers in a landfill. A combination of physical, chemical, and microbial processes in the waste transfer pollutants from the waste material to the percolating water (e.g., Christensen and Kjeldsen, 1989). The biodegradability of organic content in the MSW and the compaction of the waste layers make the landfill an anaerobic environment, giving many similarities to the composition of leachates among different landfills.

Focusing on the most common type of landfill that receives a mixture of municipal, commercial, and mixed industrial waste, but excludes significant amounts

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of concentrated specific chemical waste, landfill leachate may be characterized as a water-based solution of four groups of pollutants (dissolved organic matter, inorganic macro components, heavy metals, and xenobiotic organic compounds [Christensen et al., 1994]).

The major potential environmental impacts related to landfill leachate are pollution of groundwater and surface water. The risk of groundwater the pollution is probably the most severe environmental impact from landfills because historically most landfills were built without engineered liners and leachate collection systems. More recently, regulations in many countries have required the installation of liners and leachate collection systems as well as a plan for leachate treatment. Christensen et al. (1994 and 2001) reviewed the characteristics of leachate plumes downgradient of landfills. Surface water pollution caused by leachate has also been observed, although relatively few cases have been described in the literature. The major potential effects of a leachate release to surface water are expected to be oxygen depletion in part of the surface waterbody, changes in the stream bottom fauna and flora and ammonia toxicity.

When refuse is buried in a landfill, a complex series of biological and chemical reactions occur as the refuse decomposes. Generally, it is accepted that landfills undergo at least four phases of decomposition, (1) an initial aerobic phase, (2) an anaerobic acid phase, (3) an initial methanogenic phase, and (4) a stable methanogenic phase (Christensen and Kjeldsen, 1995). Recently, an additional aerobic or humic phase of decomposition has been proposed (Christensen and Kjeldsen, 1995; Bozkurt et al., 2000). Once the refuse is very well decomposed, the rate of oxygen diffusion into the landfill may exceed the rate of microbial oxygen depletion. Thus, over time the anaerobic landfill is hypothesized to become an aerobic ecosystem. As refuse is buried in landfills over many years in a series of cells and lifts, it is quite common for different parts of the landfill to be in different phases of decomposition. As described below, there is a strong relationship between the state of refuse decomposition and its associated leachate characteristics. Thus, leachate composition can vary throughout a landfill. More detail on each phase of refuse decomposition is presented in Sections II and IV.

An understanding of leachate composition is critical for making projections on the long-term impacts of landfills. Even after a landfill stops accepting waste and a final cover is placed over the landfill, the refuse will continue to decompose. While leachate production decreases significantly with placement of the final cover, there is little data on leachate production over long periods of time. Furthermore, in assessing the long-term stability of a landfill, the possibility that the integrity of the landfill cover will decrease must be considered. Should the cover integrity deteriorate, the quantity of leachate could actually increase long after landfill closure.

In the U.S., the regulatory structure for landfills specifies a 30-year post-closure monitoring period during which the landfill is monitored. It is presumed that at the end of the 30-year period, the landfill will be stable and will no longer

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require intensive monitoring. However, there is little information on the long-term behavior of lined landfills and 30 years may be an insufficient long-term monitoring period. In developing assessment methods to determine whether post-closure monitoring can be terminated, it is important to understand the potential future composition of the leachate, should it be released to the environment at some time in the future.

The objective of this article is to give an overview of the composition of leachate from landfills receiving municipal solid waste. This overview is based on published reports that represent landfills that are on the order of 40 years old and younger. A second objective is to evaluate the expected long-term composition of leachate once the refuse is completely degraded. This is done by reviewing models and laboratory experiments addressing stabilization processes in landfills. Our focus is on landfills that are operated under anaerobic conditions and that receive a mixture of municipal and nonhazardous commercial and industrial waste. The composition of leachate from landfills that are maintained under aerobic conditions, as well as landfills that receive incineration residue and other inorganic wastes will be considerably different and is not addressed here.

A description of the first four phases of refuse decomposition is presented in the following section. This is followed by presentation of information on the composition of leachate based on our current knowledge. Leachate composition is considered in four categories: (1) dissolved organic matter, (2) inorganic macro components, (3) heavy metals, and (4) xenobiotic organic compounds. The final section of this article evaluates the potential for changes in leachate composition over very long (geologic) periods of time.

II. STABILIZATION PROCESSES IN LANDFILLS — AN OVERVIEW

Landfills have been controlled and monitored for about 30 years. Throughout this period an increasing understanding of the complex series of chemical and biological reactions that initiates with the burial of refuse in a landfill has been developed. Figure 1 shows the gas and leachate composition as refuse decomposes. The figure is developed from the first description of the landfill phases given by Farquhar and Rovers (1973). The first four phases shown in the figure are referred to as the aerobic phase, the anaerobic acid phase, the initial methanogenic phase, and the stable methanogenic phase. Subsequent phases of decomposition, in which the waste cell begins to turn aerobic are based on theory and are somewhat speculative because no field data are available to document the onset of aerobic conditions (Christensen and Kjeldsen, 1995). This is due to the fact that most well-monitored landfills are less than 30 years old and are still in the stable methanogenic phase.

During the initial aerobic phase, oxygen present in the void spaces of the freshly buried refuse is rapidly consumed, resulting in the production of CO_2 and

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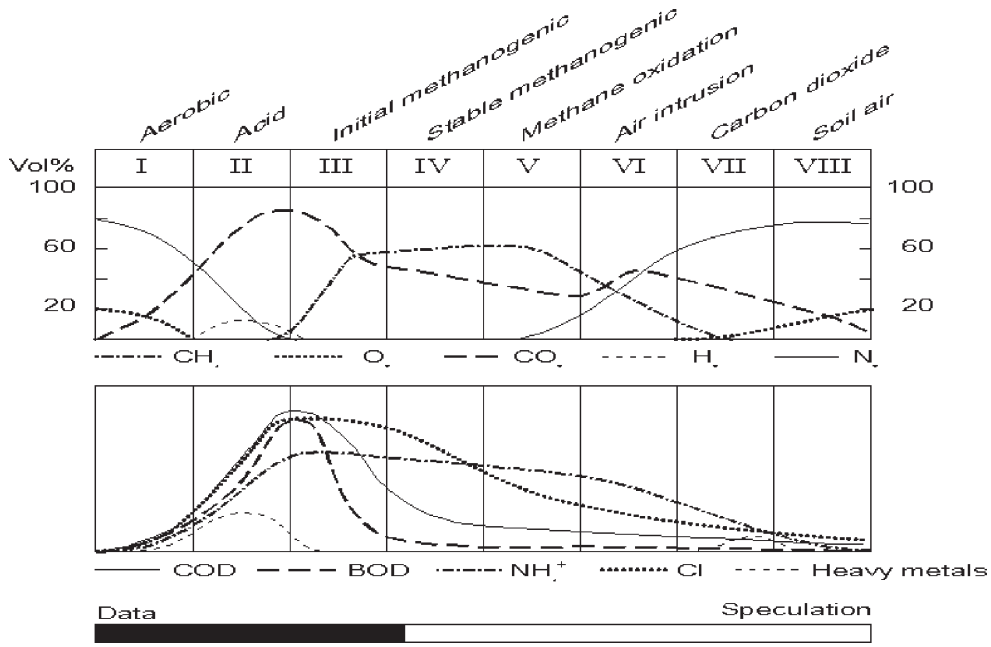

FIGURE 1. The lifetime of a landfill showing general trends in gas and leachate quality development.

maybe an increase in waste temperature. The aerobic phase in a landfill lasts only a few days because oxygen is not replenished once the waste is covered. During the aerobic phase, the waste is not typically at field capacity (Barlaz and Ham, 1993). Most leachate produced during this phase results from the release of moisture during compaction as well as short-circuiting of precipitation through the buried refuse.

As oxygen sources are depleted, the waste becomes anaerobic, which supports fermentation reactions. Cellulose and hemicellulose comprise 45 to 60% of the dry weight of MSW and are its major biodegradable constituents (Barlaz et al., 1989b). The decomposition of these compounds to methane and carbon dioxide in landfills under anaerobic conditions is well documented (Barlaz et al., 1990; Pohland and Harper, 1986; Bookter and Ham, 1982). Cellulose and hemicellulose biodegradation is carried out by three groups of bacteria: (1) the hydrolytic and fermentative bacteria that hydrolyze polymers and ferment the resulting monosaccharides to carboxylic acids and alcohols; (2) the acetogenic bacteria that convert these acids and alcohols to acetate, hydrogen, and carbon dioxide; and (3) the methanogens that convert the endproducts of the acetogenic reactions to methane and carbon dioxide (Zehnder, 1982). This process proceeds efficiently over a relatively narrow pH range around neutral. In the second phase the hydrolytic, fermentative, and acetogenic bacteria dominate, resulting in an accumulation of carboxylic acids, and a pH decrease. The highest BOD and COD concentrations in the leachate will be

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measured during this phase (Barlaz and Ham, 1993; Reinhart and Grosh, 1998). The BOD:COD ratio in the acid phase has been reported to be above 0.4 (Ehrig, 1988) or 0.7 (Robinson, 1995). As the pH is acidic, acid phase leachate is chemically aggressive and will increase the solubility of many compounds.

The onset of the initial methanogenic phase (3) occurs when measurable quantities of methane are produced. The onset of this phase is likely associated with the pH of the refuse becoming sufficiently neutralized for at least limited growth of methanogenic bacteria. During this phase the acids that accumulated in the acid phase are converted to methane and carbon dioxide by methanogenic bacteria, and the methane production rate will increase (Christensen and Kjeldsen, 1989, Barlaz et al., 1989a). Cellulose and hemicellulose decomposition also begins. COD and BOD concentrations begin to decrease and the pH increases as acids are consumed. The BOD to COD ratios will also decrease as carboxylic acids are consumed.

In the stable methanogenic phase, the methane production rate will reach its maximum, and decrease thereafter as the pool of soluble substrate (carboxylic acids) decreases. In this phase, the rate of CH_4 production is dependent on the rate of cellulose and hemicellulose hydrolysis. The pH continues to increase to steady-state pool concentrations that are on the order of a few mg/L. Some COD is present in the leachate, but it is mostly recalcitrant compounds such as humic and fulvic acids (Barlaz and Ham, 1993; Christensen et al., 1994). As discussed in the following section, the BOD:COD ratio generally will fall below 0.1 in this phase because carboxylic acids are consumed as rapidly as they are produced.

The four phases of refuse decomposition described above have been defined on the basis of both field and laboratory-scale data that have been summarized in earlier reviews (see Barlaz et al., 1990). However, environmental conditions in the landfill will have a significant impact on the rate of refuse decomposition, and subsequently the time required for decomposition to proceed to the point where methane production decreases to zero. Studies on the effect of a number of factors on refuse decomposition have been summarized (Barlaz et al., 1990; Christensen et al., 1992). The factor that has most consistently been shown to affect the rate of refuse decomposition is the moisture content, and it is generally accepted that refuse buried in arid climates decomposes more slowly than refuse buried in regions that receive greater than 50 to 100 cm of annual infiltration into the waste. Refuse decomposition can also be accelerated during the operational phase of the landfill. The most common enhancement technique is the use of leachate recycle, whereby leachate is recirculated through the refuse as opposed to it being treated and released to the environment. By recirculating leachate, the refuse moisture content is increased from its initial value, which is typically 15 to 25% (wet weight basis) to 40 to 50%. In addition, leachate recirculation results in better distribtion of nutrients, substrates, and bacteria. Other factors that can be used to accelerate decomposition include shredding and an initial aeration step in which the refuse is aerated for a period of 1 to 2 months after burial to increase the temperature and

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allow for the aerobic biodegradation of the initial accumulation of soluble organic carbon (Komilis et al., 1999a,b). However, field experience with shredding and aeration are considerably more limited.

Wheras the authors are not familiar with any landfills that have progressed beyond the stable methanogenic phase, in theory refuse will continue to decompose until no more degradation occurs and the landfill becomes aerobic. This process can be described as a series of four phases in which the methane production rate continues to decrease to a point at which air begins to infiltrate into the waste cell (Christensen and Kjeldsen, 1995). The four phases are discussed in detail in Section IV on long-term landfill stabilization.

III. PRESENT MSW LANDFILL LEACHATE COMPOSITION

This section presents data on the composition of landfill leachate, and further discuss the importance of the landfill phases presented in the previous section on compositional changes. Pollutants in MSW landfill leachate can be divided into four groups:

- Dissolved organic matter, quantified as Chemical Oxygen Demand (COD) or Total Organic Carbon (TOC), volatile fatty acids (that accumulate during the acid phase of the waste stabilization, Christensen and Kjeldsen, 1989) and more refractory compounds such as fulvic-like and humic-like compounds.

- Inorganic macrocomponents: calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^{+}), potassium (K^{+}), ammonium (NH_4^{+}), iron (Fe^{2+}), manganese (Mn^{2+}), chloride (Cl^{-}), sulfate (SO_4^{2-}) and hydrogen carbonate (HCO^{3-}).

- Heavy metals: cadmium (Cd^{2+}), chromium (Cr^{3+}), copper (Cu^{2+}), lead (Pb^{2+}), nickel (Ni^{2+}) and zinc (Zn^{2+}).

- Xenobiotic organic compounds (XOCs) originating from household or industrial chemicals and present in relatively low concentrations (usually less than 1 mg/l of individual compounds). These compounds include among others a variety of aromatic hydrocarbons, phenols, chlorinated aliphatics, pesticides, and plastizers.

Other compounds may be found in leachate from landfills: for example, borate, sulfide, arsenate, selenate, barium, lithium, mercury, and cobalt. However, in general, these compounds are found in very low concentrations and are only of secondary importance. Leachate composition may also be characterized by different toxicological tests, which provide indirect information on the content of pollutants that may be harmful to a class of organisms.

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A. How Representative Are Leachate Samples?

Many investigations on leachate composition have been restricted to one or a few leachate samples from each landfill. Depending on the study objectives, this may suffice if the leachate collection system averages the leachate from different sections of the landfill and the leachate is pumped out of the landfill for treatment. However, a landfill contains areas of refuse of varying ages and states of decomposition. Thus, where leachate from older methanogenic refuse is mixed with leachate from fresher refuse in the acid phase, it is not possible to relate leachate composition to processes within the waste layers. Furthermore, where leachate from refuse in the acid phase percolates through well-decomposed refuse, the leachate can be expected to reflect the composition of methanogenic leachate. This is because the high COD of the acid phase leachate will be consumed as the leachate passes through the well decomposed, and thus carbon limited refuse. In cases where leachate is released to groundwater, such as in the case of older landfills that are not lined, the spatial distribution of the leachate quality is especially important to evaluate the leaching to the underlying strata. This, however, require a large number of sampling points (Kjeldsen et al., 1998; Assmuth, 1992).

Leachate composition varies significantly among landfills depending on waste composition, waste age, and landfilling technology. Leachate sampling methods and sample handling routines may also influence the measured leachate quality. For example, colloids have a high affinity for heavy metals (Gounaris et al., 1993), thus the concentration of heavy metals measured in a leachate sample may depend strongly on the amount of colloidal matter present in the sample and the handling of the sample (see also the section on heavy metals). No standard protocols for sampling, filtration, and storage of leachate samples exist. The content of colloidal matter in a sample depends to a large extent on the sampling technique used (e.g., McCarthy and Zachara, 1989; Backhus et al., 1993). Where samples are obtained from groundwater monitoring wells, a high pumping rate will increase the colloid content of the sample significantly (Backhus et al., 1993), and the heavy metal concentration may also be increased. Therefore, leachate samples should be filtered in the field before analysis of heavy metals, especially if the sampling is done quickly. Alternatively, samples could be withdrawn under very low pumping rates and after sufficient purging of the well. Sampling and sample handling may explain part of the variation seen among landfills with respect to the heavy metal concentration in leachates. Maintaining leachate samples under anaerobic conditions until they are preserved is also important because metal solubility varies with their oxidation stage.

Leachate analysis for XOCs may be biased by loss of pollutants due to volatilization or sorption to the sampling equipment. However, recent studies have shown that the quality of leachate analysis for XOCs is seriously jeopardized only if very poor sampling protocols are used (see review by Parker, 1994).

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B. General Leachate Composition

Table 1 presents ranges of general leachate parameters from various reports. The table is based mainly on data originating from newer landfills. Data from older uncontrolled landfills may exhibit lower values than the minimum values given in the table (Assmuth and Strandberg, 1993; Kjeldsen and Christophersen, 2001). In general, landfill leachates may contain very high concentrations of dissolved organic matter and inorganic macrocomponents. The concentrations of these components may typically be up to a factor 1000 to 5000 higher than concentrations found in groundwater.

Several parameters change dramatically as the landfill stabilizes. During the acid phase, the leachate may show low pH values and high concentrations of many compounds, in particular high concentrations of easily degradable organic compounds as volatile fatty acids. In the later stable methanogenic phase, the pH increases and the BOD5/COD ratio (biological oxygen demand measured over 5 days divided by chemical oxygen demand), reflecting the degradability of the organic carbon is lowered dramatically (Ehrig, 1988). The increased pH also affects many of the inorganic parameters as illustrated in Table 2, where data are presented for acid phase leachate and methanogenic leachate. For parameters not significantly affected by landfill stabilization, only average values are given.

Besides the time dependency in relation to change from acid to methanogenic phase, short-term variations in leachate quality are also expected. Seasonal variations in leachate composition have been observed in several cases. Åkesson and Nilsson (1997) observed lower leachate concentrations in the wet season in a Swedish landfill test cell. Similar observations were found by Chu et al. (1994) in a Hong Kong landfill.

C. Dissolved Organic Matter

As described previously, several bulk parameters are used to describe the content of dissolved organic matter in leachate; TOC (Total Organic Carbon), COD (Chemical Oxygen Demand), and BOD (Biological Oxygen Demand). Table 3 gives further details on observed BOD and COD values and the ratio BOD/COD in leachates from older landfills that are well into their methane phase. The table shows that the BOD/COD ratio in most cases is below 0.10 for methanogenic leachates.

One source of variability of the COD measurement that may have some effect on the values presented in Table 1 is the presence of inorganic constituents that may contribute to COD. For instance, Kylefors et al. (1999) found that Fe(II), Mn(II), and sulfide contributed up to one-third of the COD in the leachates they tested. Chloride may also contribute to COD if not accomodated for in the procedure. In addition, poor sampling methods that expose anaerobic leachate to oxygen

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TABLE 1
Composition of Landfill Leachate* (Values in mg/l unless otherwise stated.)

Parameter	Range
pH	4.5-9
Spec. Cond. ($\mu S\ cm^{-1}$)	2500-35000
Total Solids	2000-60000
Organic Matter	
Total Organic Carbon (TOC)	30-29000
Biological Oxygen Demand (BOD5)	20-57000
Chemical Oxygen Demand (COD)	140-152000
BOD5/COD (ratio)	0.02-0.80
Organic nitrogen	14-2500
Inorganic macrocomponents	
Total phosphorous	0.1-23
Chloride	150-4500
Sulphate	8-7750
Hydrogenbicarbonate	610-7320
Sodium	70-7700
Potassium	50-3700
Ammonium-N	50-2200
Calcium	10-7200
Magnesium	30-15000
Iron	3-5500
Manganese	0.03-1400
Silica	4-70[a]
Heavy metals	
Arsenic	0.01-1
Cadmium	0.0001-0.4
Chromium	0.02-1.5
Cobolt	0.005-1.5
Copper	0.005-10
Lead	0.001-5
Mercury	0.00005-0.16
Nickel	0.015-13
Zinc	0.03-1000

*The ranges are based on Andreottola and Cannas (1992), Chu et al.(1994), Robinson (1995), Ehrig (1980), Ehrig (1983), Ehrig (1988), Garland and Mosher (1975), Johansen and Carlson (1976), Karstensen (1989), Krug and Ham(1997), Lu et al. (1985), Naturvårdsverket (1989), Owen and Manning (1997), and Robinson and Maris (1979).
[a] Values based on Owen and Manning (1997).

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TABLE 2
Leachate Composition in Terms of Average Values and Ranges for Parameters with Differences between Acid and Methanogenic Phase (Ehrig, 1988) and Average Values for Parameters with No Observed Differences between Acid and Methanogenic Phase (Ehrig, 1983) (All values in mg/l except pH and BOD5/COD)

Parameter	Acid phase		Methanogenic phase		Average
	Average	Range	Average	Range	
pH	6.1	4.5-7.5	8	7.5-9	
Biological Oxygen Demand (BOD_5)	13000	4000-40000	180	20-550	
Chemical Oxygen Demand (COD)	22000	6000-60000	3000	500-4500	
BOD_5/COD (ratio)	0.58		0.06		
Sulfate	500	70-1750	80	10-420	
Calcium	1200	10-2500	60	20-600	
Magnesium	470	50-1150	180	40-350	
Iron	780	20-2100	15	3-280	
Manganese	25	0.3-65	0.7	0.03-45	
Ammonia-N					740
Chloride					2120
Potassium					1085
Sodium					1340
Total phosphorus					6
Cadmium					0.005
Chromium					0.28
Cobalt					0.05
Copper					0.065
Lead					0.09
Nickel					0.17
Zinc	5	0.1-120	0.6	0.03-4	

may cause Fe(II) to oxidize to Fe(III) and precipitate out of the leachate. This would result in a decreased COD relative to a study in which Fe(II) was oxidized as part of the COD analysis. This would occur when the sample was maintained under anaerobic conditions until after filtration, at which point it could be acidified to reduce iron oxidation.

Dissolved organic matter in leachate is a bulk parameter covering a variety of organic degradation products ranging from small volatile acids to refractory fulvic and humic-like compounds (Chian and DeWalle, 1977). Dissolved organic matter

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TABLE 3
Observed Values of BOD, COD and BOD/COD-Ratio for Landfill Leachates Samples from Landfills in the Methanogenic Phase

BOD (mg/l)	COD (mg/l)	BOD/COD	Reference
5.7-1100	76-6997	---	Concentration ranges from 21-30 year old German landfills (Krumpelbeck and Ehrig, 1999)
290	1225	0.24	Average concentrations from 21-30 year old German landfills (Krumpelbeck and Ehrig, 1999)
44	320	0.11	Average concentrations from old, Danish landfills (Kjeldsen and Christophersen, 2001)
39	398	0.10	Composite results at Sandsfarm Landfill, (Robinson, 1995)
11	190	0.06	Composite results at Bishop Middleham Landfill, (Robinson, 1995)
38	517	0.07	Composite results at Odsal Wood Landfill, (Robinson, 1995)
1.0	53	0.02	Composite results at East Park Drive Landfill, (Robinson, 1995)
2.5	64	0.04	Composite results at Marton Mere Landfill, (Robinson, 1995)
180	3000	0.06	Average concentrations in methanogenic leachate (Ehrig, 1988)

can affect leachate composition in relation to other constituents through the complexing properties of the high-molecular-weight component of the dissolved organic matter. Unfortunately, we usually have very little information on the composition of the dissolved organic matter in landfill leachate. At the most general level, a low BOD/COD ratio suggests a leachate with low concentrations of volatile fatty acids and relatively higher amounts of humic and fulvic-like compounds. A few investigations concerning DOC in landfill leachate are available.

Harmsen (1983) analyzed an acid-phase leachate and a methanogenic-phase leachate. In the acid-phase leachate, more than 95% of the DOC content of 20,000 mg/l consisted of volatile fatty acids and only 1.3% of the DOC consisted of high-molecular-weight (MW) compounds (MW>1000). In addition, volatile amines and alcohols were detected. In the methanogenic-phase leachate; however, no volatile acids, amines, or alcohols were detected, and 32% of the DOC (2100 mg/l) consisted of higher-molecular-weight compounds (MW>1000). Also in a methanogenic-phase leachate, Artiola-Fortuny and Fuller (1982) described more than 60% of the DOC content as humic-like material. Investigating anaerobic and aerobic leachates and leachates that have passed a model aquifer, Frimmel and Weis (1991) found that only 6 to 30% of the DOC could be described as fulvic acids.

More detailed characterization of the DOC in leachate and leachate polluted groundwater is based on isolation and purification of the DOC. These procedures may potentially change the properties of the organic matter because it is necessary

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to purify the material to obtain results. Weis et al. (1989) compared fulvic acids from landfill leachates with those from soil and bog lake water. The fulvic acids isolated from landfill leachate had higher carbon, hydrogen, and sulfur content, lower quantities of phenolic groups, lower complexation capacities for copper and lower molecular weight. Christensen et al. (1998a) characterized 82% of the DOC in leachate-polluted groundwater sampled less than 10 m downgradient from the Vejen Landfill (DK) and found 49% fulvic acids, 8% humic acids, and 25% hydrophilic fraction. Based on molecular weight, elemental composition, and acidity, the fulvic acid fraction and the hydrophilic fraction resembled fulvic acids from other origins, whereas the humic acid had rather low molecular weight. This analysis showed that the three fractions of DOC were rather similar with respect to the features of importance to complexation of metals.

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D. Inorganic Macrocomponents

The concentrations of some inorganic macrocomponents in leachate depend, as in the case of the dissolved organic matter, on the stabilization of the landfill. Table 2 shows that the cations calcium, magnesium, iron, and manganese are lower in methanogenic phase leachate due to a higher pH (enhancing sorption and precipitation) and lower dissolved organic matter content, which may form complexes with the cations. Sulfate concentrations are also lower in the methanogenic phase due to microbial reduction of sulfate to sulfide.

Table 2 also presents average concentrations for parameters with no observed difference between acid and methanogenic phase. These are the macrocomponents chloride, sodium, and potassium for which the effects of sorption, complexation, and precipitation are minor. Decreasing trends in concentration with time of these pollutants could be due to wash out by the leaching, although Ehrig (1983, 1988) did not observe any decrease in concentration for these parameters after up to 20 years of leaching.

Table 2 is based on detailed studies (Ehrig 1983, 1988) on a large number of landfills in Germany. Similar findings are presented in a study of 13 sanitary landfills in Wisconsin, USA (Krug and Ham, 1997), where equivalent concentration ranges and time dependency of the selected parameters were found.

Many investigations report concentrations of ammonia-nitrogen in the range of 500 to 2000 mg/l, and no decreasing trend in concentration with time. Ammonia is released from the waste mainly by decomposition of proteins. The only mechanism by which the ammonia concentration can decrease during refuse decomposition is leaching because there is no mechanism for its degradation under methanogenic conditions (Robinson, 1995; Burton and Watson-Craik, 1998). For this reason, several researchers have identified ammonia as the most significant component of leachate for the long term (Robinson, 1995; Krumpelbeck and Ehrig, 1999; Christensen et al., 1994; Christensen et al., 1999).

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In a study of 50 German landfills, ammonia concentrations did not show a significant decrease even 30 years after landfill closure (Krumpelbeck and Ehrig, 1999). Ehrig (1988) reports that there is no significant change in ammonia concentrations from the acidic to methanogenic phase, and that the average value is 740 mg-N/L (Table 2). Ammonia concentration data from several studies is given in Table 4. These data document that ammonia concentrations will remain high even in leachate from older landfills that is otherwise low in organic content.

E. Heavy Metals

There is wide variation in the reported concentrations of heavy metals from different landfills (Table 1). However, average metal concentrations are fairly low. This has been shown by several studies in which researchers have reported metals concentrations from full-scale landfills, test cells, and laboratory studies. The ultimate conclusion of all of these studies is that heavy metals in landfill leachate at present are not at major concern (Christensen et al., 1999; Robinson, 1995; Reinhart and Grosh, 1999; Revans et al., 1999; Kjeldsen and Christophersen, 2001; Christensen et al., 1994). Table 5 gives more details of observed heavy metal concentrations. For comparison, the U.S. Drinking Water Standards are presented as well. The table shows that most heavy metal concentrations in landfill leachate are at or below the US drinking water standards.

TABLE 4
Ammonia Concentrations in Landfill Leachate (All values are from older landfills in the methanogenic phase)

Ammonia-N (mg/l)	Reference
110	Average ammonia concentration from 104 old, Danish landfills (Kjeldsen and Christophersen, 2001)
233	Composite results at Sandsfarm Landfill, (Robinson, 1995)
282	Composite results at Bishop Middleham Landfill, (Robinson, 1995)
399	Composite results at Odsal Wood Landfill, (Robinson, 1995)
43	Composition results at East Park Drive Landfill, (Robinson, 1995)
30	Composition results at Marton Mere Landfill, (Robinson, 1995)
12-1571	Range of concentrations from 21-30 year old, German landfills (Krumpelbeck and Ehrig, 1999)
445	Average concentration from 21-30 year old, German landfills (Krumpelbeck and Ehrig, 1999)
740	Average concentration (Ehrig, 1988)

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TABLE 5
Heavy Metal Concentrations in Leachate[a]

Reference	1	2	3	4	5	6	7	8	9	10	US Stand.[b]
Metal											
Cd	0.006	0.005	0.006	0.0002	0.0004	0.0003	0.0036	0.002 - 0.008	0.0002 - 0.018	< 0.01 - < 0.04	0.005
Ni	0.130	0.17	0.05	0.028	0.084	0.054	0.062	0.01 - 0.08	0.0036 - 0.348	< 0.01 - 0.1	---
Zn	0.67	0.6	2.2	0.2	0.36	0.085	5.31	0.003 - 0.011	0.05 - 9	< 0.01 - 0.47	5.0
Cu	0.07	0.065	0.04	0.002	0.007	0.034	0.002	---	0.004 - 0.27	< 0.02 - 0.17	1.3
Pb	0.07	0.09	0.02	< 0.005	< 0.005	0.056	0.188	0.016 - 0.067	0.005 - 0.019	< 0.04 - 0.13	0.0
Cr	0.08	0.28	0.01	0.003	0.016	0	0.002	0.033 - 0.085	0.005 - 1.62	< 0.01 - 0.05	0.1

1. Average (undiluted) leachate concentration from 106 old, Danish landfills (Kjeldsen and Christophersen, 2001).
2. Average leachate concentration in 20 German landfills in the methanogenic phase (Christensen et al., 1999).
3. Average leachate concentration in a full-scale test cell operated with leachate recirculation (Flyhammar et al., 1998).
4-7. Average leachate concentrations at four Danish landfills. Only site 4 has been closed (Jensen and Christensen, 1999).
8. Ranges of leachate concentrations in the most contaminated groundwater wells at North Bay Landfill, Canada (Christensen et al., 1999).
9. Ranges of typical leachate concentrations in a study of 21-30 year old, German landfills (Krumpelbeck and Ehrig, 1999).
10. Ranges of typical leachate concentrations at six old landfills in the UK (Robinson, 1995).
--- Data not available

a. Data listed are either the average or range for a given study in mg/l.

b. National primary drinking water regulations, USA (http://www.epa.gov/OGWDW/wot/appa.html).

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1. Heavy Metal Attenuation Processes in Landfills

The reason for the low concentrations of heavy metals in methanogenic leachates is not a lack of heavy metals present in the waste. Heavy metal balances for landfills have shown that less than 0.02% of heavy metals received at landfills are leached from the landfill after 30 years (Flyhammer, 1995; Aulin and Neretnieks, 1995; Belevi and Baccini, 1989). Both sorption and precipitation are believed to be significant mechanisms for metals immobilization and the subsequent low leachate concentrations. Waste contains soils and organic matter, which, especially at the neutral to high pH values prevailing in methanogenic leachate, has a significant sorptive capacity (Bozkurt, 1999). In addition, the solubilities of many metals with both sulfides and carbonates is low, and these anions are typical in landfills. Sulfide is formed from sulfate reduction during waste decomposition in landfills, and sulfide precipitation is often cited as an explanation for low concentrations of heavy metals (Christensen et al., 2000; Christensen et al., 1994). Sulfides and carbonates are capable of forming precipitates with Cd, Ni, Zn, Cu, and Pb. While carbonates are abundant in landfill leachate, the solubilities of metal carbonates are generally higher than that of metal sulfides (Christensen et al., 2000). In general, sulfide precipitation is expected to dominate heavy metal attenuation compared with complexation agents (Reinhart and Grosh, 1999). Cr is an exception to this because it does not form an insoluble sulfide precipitate (Christensen et al., 2000). However, Cr tends to form insoluble precipitates with hydroxide (Christensen et al., 1999; Revans et al., 1999).

Investigations of the sulfur content of landfilled waste have shown that the waste does not contain sufficient sulfur to bind all the heavy metals present in the waste. The landfill sampled by Martensson et al. (1999) contained only enough sulfur to bind 5% of metals present. Occasionally, phosphates and hydroxides will also precipitate metals (Christensen et al., 1999). Hydroxide precipitates form at pHs at or above neutral, which is typically the case in methanogenic leachates (Reinhart and Grosh, 1998).

2. Heavy Metal Mobilizing Processes in Landfills

Several processes, including complexation to inorganic and organic ligands, and sorption to colloids are capable of mobilizing heavy metals by increasing the concentration in the mobile aqueous phase. Jensen and Christensen (1999) separated leachate samples from four Danish landfills into size fractions to obtain information about size distribution of colloids and associated heavy metals (Cd, Ni, Zn, Cu, Pb, Cr). A significant but highly varying fraction of the heavy metals was associated with colloidal fractions. Similarly, Gounaris et al. (1993) reported that in an American landfill a significant fraction of the Zn, Pb, and Cr were in colloidal fractions. Klein and Niessner (1998) found at a German landfill that the main

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fraction of the heavy metals was associated with the colloidal matter, primarily the 0.001 to 0.01 μm fraction, which is dominated by humic material. In all three investigations, comparison of the distribution of organic matter and heavy metals between the size fractions indicated that the heavy metals in the colloidal fractions were not only related to organic matter, even though the colloidal humic substances are suspected to play a major role with respect to the speciation of the heavy metals.

Different physical separation techniques have been used to divide the heavy metals in landfill leachates into different types of species. Lun and Christensen (1989) determined the distribution between different cadmium species in landfill leachate by a resin method and found that free divalent Cd^{2+} only made up a few percent of the total cadmium content. Most of the complexed fraction was characterized as labile complexes that easily could be redistributed to other dissolved species. However, a small fraction (5 to 15%) was characterized as stable soluble complexes, defined as lack of ability to exchange with a cation exchange resin. The stable complexes were considered to be organic. In leachate from three landfills containing industrial waste, Holm et al. (1995b) found by use of dialysis, ion exchange, and thermodynamic calculations, large variation with respect to the speciation of Cd. In two of the investigated leachates about 20% of the total Cd content was determined to be Cd^{2+}, while most of the Cd was identified as chloride complexes. The third leachate had a high dissolved organic carbon content (3200 mg C/l), and most of the Cd in this leachate was complexed with the organic matter. Knox and Jones (1979) showed that both low-molecular-weight compounds (<500) comparable to simple carboxylic acids and high-molecular-weight compounds (>10000) contributed significantly to cadmium complexation. Bolton and Evans (1991) performed some speciation calculations on four leachates. Their calculations showed that 38% of Cd was present in complexes with organic ligands and 32% with inorganic ligands, while Zn and Ni were present in complexes (36% and 68%, respectively) exclusively with organic ligands.

F. Xenobiotic Organic Compounds (XOCs)

Table 6 presents concentration ranges of some xenobiotic organic compounds (XOCs) found in landfill leachate. The table is based mostly on observations from landfills containing municipal solid waste. The amount of hazardous waste that has been allowed into MSW landfills has decreased significantly over the last 20 years. However, many of the landfills for which data are reported in Table 6 may contain waste from a time period when there were fewer restrictions on the disposal of hazardous waste in MSW landfills. Very broad ranges are observed, reflecting differences in waste composition, landfill technologies, and waste age.

The most frequently found XOCs are the monoaromatic hydrocarbons (benzene, toluene, ethylbenzene, and xylenes) and halogenated hydrocarbons such as tetrachloroethylene and trichloroethylene (Table 6). These pollutants are also the

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TABLE 6
Xenobiotic Organic Compounds (XOCs) Observed in Landfill Leachates

Compound	Range (µg/l)	References[5]
Aromatic hydrocarbons		
Benzene	0.2-1630	a,b,d,f,h,i,k,l,m,n,o,p,q, t, x
Toluene	1-12300	a,b,d,f,h,i,k,l,m,n,o,p,q, t, x
Xylenes	0.8-3500	a,b,d,f,h,i,k,l,m,n,o,p,q, t, x
Ethylbenzene	0.22329	a,b,d,f,m,n,o,p,q, t, x, y
Trimethylbenzenes	0.3-250	b,f,o,p, t, x
n-Propylbenzene	0.3-16	T, x
t-Butylbenzene	2.1-21	T
o-Ethyltoluene	0.5-46	t
m-Ethyltoluene	0.3-21	T
p-Ethyltoluene	0.2-10	T
Naphthalene	0.1-260	c,d,f,m,n,o,p, t, x
Halogenated hydrocarbons		
Chlorobenzene	0.1-110	a,d,f,m,o, t
1,2-Dichlorobenzene	0.1-32	a,c,d,f,o, t
1,3-Dichlorobenzene	5.4-19	Y
1,4-Dichlorobenzene	0.1-16	a,c,d,f,m, t
1,2,3-Trichlorobenzene	I	Y
1,2,4-Trichlorobenzene	4.3	Y
Hexachlorobenzene	0.025-10	Z
1,1-Dichloroethane	0.6-46	a,i, t, y
1,2-Dichloroethane	<6	a,b,d
1,1,1-Trichloroethane	0.01-3810	a,b,d,f,m,o,p,q, t, x
1,1,2-Trichloroethane	2.5-16	Y
1,1,2,2,-Tetrachloroethane	I	Y
trans-1,2-Dichloroethylene	1.6-6582	a,b, y
cis-1,2-Dichloroethylene	1.4-470	a,b, t
Trichloroethylene	0.05-750	a,b,d,f,l,m,n,o,p, t, x, y
Tetrachloroethylene	0.01-250	a,b,f,i,l,m,n,o,p,q, t, x
Dichloromethane	1.0-827	a,b,d,k,m, t, y
Trichloromethane	1.0-70	a,b,d,h,i,k,o,p,q,
Carbontetrachloride	4.0-9.0	h,o,p
Phenols		
Phenol	0.6-1200	c,f,g,k,m,n, x
Ethylphenols	<300	k,l
Cresols	1-2100	c,g,i,k,l,m,n, t, x
Bisphenol A	200-240	t
3,5-Dimethylphenol	0.7-27.3	x
2,5-Dimethylphenol	0.4-4.5	X
2,4-Dimethylphenol	0.1-12.5	x
3,4-Dimethylphenol	0.03-10.4	x
2,6-Dimethylphenol	0.3-1.9	X
2-Methoxyphenol	I[4]	x
2/3-Chlorophenol	0.03-1.6	x
4-Chlorophenol	0.2-1.3	x
4-Chloro-m-cresol	1.2-10.2	x
3,5-Di-chlorophenol	0.08-0.63	x
2,3,4,6-Tetrachlorophenol	0.079-3.0	z

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TABLE 6 (continued)

Alkylphenols		
Nonylphenol	6.3-7	x
Nonylphenolmonocarboxylate	0.5-3	x
Pesticides		
Ametryn	0.12	X
AMPA	3.8-4.3	X
Atrazin	0.16	X
Bentazon	0.3-4.0	X
Chloridazon	1.6	X
Chlorpropham	26	X
Dichlobenil	0.1-0.3	X
Fenpropimorf	0.1	X
Glyphosat	1.7-27	X
Hexazinon	1.3	X
Hydroxyatrazin	0.7-1.7	X
Hydroxysimazin	0.6-1.7	X
Isoproturon	1.2	x
Lindane	0.025-0.95	Z
Mecoprop[1]	0.38-150	c,e,l, u, x
MCPA	0.2-9.1	U, x
Propoxuron	2.6	X
Simazine	2.3	X
Tridimefon	2.1	X
4-CPP	15-19	X
2,4-D[2]	1.0-5	e,l
2,4,5-T	l[4]	U
2,4-DP	0.3-5.2	U, x
2,6-DCPP	0.7-1.3	X
Phthalates		
Monomethyl phthalate	1	X
Dimethyl phthalate	0.1-7.7	S
Diethyl phthalate	0.1-660	c,g,j,m, t, x
Methyl-ethyl phthalate	2-340	X
Mono-(2-ethylhexyl) phthalate	4-14	V, x
Di-(2-ethylhexyl) phthalate	0.6-235.9	s, t, u, v, x
Mono-butylphthalate	4-16	V, x
Di-n-butylphthalate	0.1-70 5.0-15	c,g,i,j,m, t
Di-isobutylphthalate	3-6	T
Mono-benzylphthalate	6-16	V, x
Butylbenzyl phthalate	0.2-8	c,g,j, v
Dioctylphthalate	1-6	T
Phthalic acid	2-14000	V, x
Aromatic sulfonates		
Naphtalene-1-sulfonate	506-616	r
Naphtalene-2-sulfonate	1143-1188	r
Naphtalene-1,5-disulfonate	<2.5-51	r
Naphtalene-1,6-disulfonate	366-397	r
Naphtalene-2,7-disulfonate	129-145	r
2-aminonaphtalene-4,8-disulfonate	73-109	r
p-toluenesulfonate	704-1084	r

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TABLE 6 (continued)

Phosphonates		
Tri-n-butylphosphate	1.2-360	c.f.i.l.m
Triethylphosphate	15	f.i.l.
Miscellaneous		
Acetone	6-4400	a.i.k.o
2(3H)-Benzothiazolone	10-50	t
Camphor[3]	20.6-255.2	c.f.i.n. t. x
Cumen	0.3-7.4	
Fenchone	7.3-83	c.f.n. x
Tetrahydrofuran	9-430	a.i.k.o
Indane	0.2-20	t
Methylethylketone	110-6600	a.k. aa
Methyl-iso-butylketone	1.1-176	t. aa
Dimethoxymethane	1.1	t
MTBE	0.8-35	t
Styrene	0.5-1.6	t

[1] 2-(2-methyl-4-chlorophenoxy)propionic acid (MCPP)

[2] Dichlorophenoxyacetic acid

[3] 1,7,7-trimethyl-bicyclo(-bicyclo[2.2.1]-heptane-2-one

[4] I: Identified

[5] The references are:(a) Sable and Clark (1984), (b) Först et al. (1989), (c) Öman and Hynning (1993), (d) Harkov et al. (1985), (e) Gintautas et al. (1992), (f) Reinhard et al. (1984), (g) Albaiges et al. (1986), (h) Khare and Dondero (1977), (i) DeWalle and Chian (1981), (j) Dunlap et al. (1976), (k) Sawhney and Kozloski (1984), (l) Schultz and Kjeldsen (1986), (m) Karstensen (1989), (n) Kjeldsen (1993), (o) Barker et al. (1987a), (p) Barker et al. (1987b), (q) Krug and Ham (1997), (r) Riediker et al., (2000), (s) Bauer et al. (1998), (t) Greenpeace (1999), (u) Öman (1999), (v) Ejlertsson et al (1999), (x) Ledin et al. (2001), (y) Schrab et al., (1993), (z) Assmuth and Penttilä (1995), (aa) Plotkin and Ram (1984), (ab) Paxeus (2000).

ones found in the highest concentrations. The main reason for the relatively large number of investigations focusing on these two groups of pollutants is their well-documented negative effects in the aquatic environment. In addition, these non-polar organic compounds are relatively easy to analyze for, in spite the very complicated matrix of leachates from landfills. Finally, several of these compounds have been designated as priority pollutants by the U.S. Environmental Protection Agency.

Data on polar and ionic organic pollutants is scarcer, although the number of investigations including these more water-soluble pollutants is increasing. This can be exemplified by the group of phenols, where the older studies usually only were analyzing for phenol and the cresols, while some newer investigations also cover, for example, chloro- and nonyl-phenols (Table 6). The observed quantities of these phenols are generally in the µg/l level.

Twenty-one different pesticides were identified from a screening for a total of 101 pesticides in 10 Danish landfills (Ledin et al., 2001). The most common ones were MCPP (or Mecoprop; present in nine landfills), Bentazon (six landfills), and MCPA (three landfills). These phenoxyalkanoic acid herbicides have been identified in leachates from other landfills as well. Especially MCPP has been observed frequently (Schultz and Kjeldsen, 1986; Gintautas et al., 1992; Kjeldsen, 1993;

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Öman and Hynning, 1993; Öman, 1999). However, other types of pesticides were also identified, for example, Ametryn, Atrazine, Chlorpropham, Dichlobenil, and Hexazinon (Table 6). These findings indicate that pesticides could be of importance when evaluating the impact from landfill leachates on groundwater quality.

Measurements for benzene and naphthalene sulfonates have been carried out in leachates from four Swiss landfills (Riediker et al., 2000). The results showed that benzenesulfonates (*p*-toluenesulfonate) and naphthalenesulfonates (Naphtalene-1-sulfonate, Naphtalene-2-sulfonate, Naphtalene-1,5-disulfonate, Naphtalene-1,6-disulfonate, Naphtalene-2,7-disulfonate, and 2-aminonaphtalene-2,7-disulfonate) were present in the leachates, at a concentration range from a few μg/l up to 11 mg/l (Table 6). The sulfonates include some of the surfactants used in laundry detergents and shower soaps. Another type of surfactant that could be expected to be present in landfill leachates are the alkylphenol polyethoxylates. Ledin et al. (2001) analyzed for two of these, nonylphenol mono- and di-ethoxylate, as well as two degradation products, nonylphenol mono- and di-carboxylate. They found nonylphenol monocarboxylate in leachates from two landfills, although no nonylphenol polyethoxylates were observed. The latter could be due to the relatively high detection limits for these compounds in leachates (20 to 25 μg/l).

The phthalates are also pollutants of concern. The most frequently observed phthalates are di-(2-hexylethyl) phthalate, di-ethyl-phthalate, di-*n*-butyl-phthalate, and butyl-benzyl-phthalate (Table 6). The highest concentrations are, however, observed for the degradation product phthalic acid (up to 14 mg/l) (Ejlertsson et al., 1999).

It could also be noted that MTBE (methyl-tert-butyl-ether), which is used as a gasoline additives, has been found in concentrations up to 35 μg/l in the leachates from eight Swedish landfills (Greenpeace, 1999).

More that 200 individual compounds or classes of compounds were identified in a screening for XOCs in three Swedish landfills (Paxeus, 2000). Among the compounds identified were dioxanes and dioxolans, which have not been reported previously in landfill leachates. These are synthetic cyclic ethers and are known to have very low odor threshold and high odor intensity. They originates form waste disposal from alkyd resin production and from disposed products from painting and coating (Paxeus, 2000).

AOX (Adsorbable Organic Halogen) has in some cases been used as an aggregated parameter for the content of XOCs containing halogens. Robinson (1995) reviewed concentration ranges of AOX in landfill leachates reported in the literature and performed additional analysis on 30 landfill leachate samples. The total range given by Robinson (1995) was 30 to 27,000 μg/l, but in most cases AOX results were in the range of 200 to 5000 μg/l. The usefulness of AOX measurements is limited by the lack of information on identities and concentrations on individual compounds, often with quite different health effects, and also by poor correlation between the measured AOX in the leachate, and the concentrations of identified, halogenated pollutants (Robinson, 1995). Grøn et al. (2000) saw the

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same problems using the related parameter TOX (Total Organic Halogens) in leachate from two Danish landfills. With commonly used screening procedures for organic pollutants, the individual halogenated compounds behind the TOX could not be found.

The results for halogenated aliphatics and the aromatics from two studies are compared in Table 7. The data presented by Krug and Ham (1997) originate from 13 municipal sanitary landfills in Wisconsin, USA, while the data from Kjeldsen and Christophersen (2001) is from 104 old Danish MSW landfills. In both studies, BTEX were the most frequently found compounds and concentrations were generally lower at the Danish landfills. Krug and Ham (1997) also reported the frequent presence of halogenated hydrocarbons at relatively high concentrations, reflecting the co-disposal of hazardous waste at some of the sites. At the old Danish sites the concentrations of the halogenated hydrocarbons were very low, probably reflecting that intense co-disposal has not been practiced on the Danish sites. Table 8 shows the results of a recent study comparing detection frequency and concentrations from old and new MSW landfills (Gibbons et al., 1999). The table shows that the concentrations of xenobiotic organic compounds at the old MSW landfills were generally higher than concentrations at the newer landfills, probably reflecting a lower acceptance of organic chemicals at the newer landfills. These data are in good accordance with an investigation of leachate concentrations over a 1-year period from 40 landfills with no co-disposal (Ecobalance, 1999).

H. Toxicity of Landfill Leachate

Risk assessment of landfill leachate is traditionally based on chemical analyses of specific compounds present in the leachate. However, risk assessment is not sufficiently developed to take into account interactions among chemicals or toxic degradation products for constituents in a complex mixture. In contrast to chemical analysis, bioassays can be used to characterize the toxicity of landfill leachate to integrate the biological effect of all its constituents. Thus, factors like bioavailability, synergistic, antagonistic, or additive effects can be assessed directly without the need for assumptions and extrapolations made from chemical analysis.

Until about 1980, there were only a few published studies on the toxicity of landfill leachate (Cameron and Koch, 1980), but since then the toxicity of landfill leachate has been assessed by several authors using a number of different organisms (Table 9). Single species or test batteries consisting of several different species or organisms from different trophic levels have been applied to landfill leachate to detect both specific and general toxicity. Fish (usually fathead minnow or rainbow trout), crustaceans (daphnids), and luminescent bacteria (commercially available as Microtox test kits) are among the most frequently used methods. Considerable differences in the sensitivities of different test organisms have been observed in most studies (for example, Plotkin and Ram, 1984; Clément et al.,

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TABLE 7
Leachate Concentrations for Halogenated Aliphatic Hydrocarbons and Aromatic Hydrocarbons Found in Two Reviews of Regional Landfill Leachate Data

Compound	Krug and Ham (1997)			Kjeldsen and Christophersen (2001)		
	Percentage of sites detected	Average (μg/l)	Range (μg/l)	Percentage of wells detected	Average (μg/l)	Range (μg/l)
Halogenated Aliphatics						
Tetrachlormethane	n.m.	-	-	50	0.23	<0.1-0.28
Chloroform	40	7.1	7.1	50	0.39	<0.1-1.5
Dichlorodifluoromethane	40	219	2-1030	n.m.	-	-
Tetrachloroethane	n.m.	-	-	0	-	<0.1
1,1,2-Trichloroethane	40	1487	0.5-7130	n.m.	-	-
1,2-Dichloroethane	10	612	212-1030	n.m.	-	-
1,1-Dichloroethane	90	210	6-4120	n.m.	-	-
Chloroethane	40	112	9-410	n.m.	-	-
Tetrachloroethylene	40	19	0.8-44	57	3.2	<0.1-5
Trichloroethylene	n.m.	-	-	70	5.6	<0.1-24
1,2-t-Dichloroethylene	10	5.6	5.6	n.m.	-	-
Aromatics						
Benzene	70	123	2.2-1630	55	22	<1-400
Toluene	100	417	2.8-12300	49	38	<1-900
Ethylbenzene	100	110	4.8-1280	60	30	<1-100
Xylenes	70	461	10-3010	73	210	<1-2400
Naphthalene	n.m.	-	-	70	34	<1-280

n.m.: not measured.

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TABLE 8
Leachate Concentrations for Halogenated Aliphatic Hydrocarbons and Aromatic Hydrocarbons Found in a New Review of American Landfill Leachate Data (Gibbons et al.,1999) (The landfills have been categorized in old and new landfills.)

Compound	Old MSWLF		New MSWLF	
	Percentage of locations detected	Average (μg/l)	Percentage of locations detected	Average (μg/l)
Halogenated Aliphatics				
Chloroform	3	n.g.	4	n.g.
Tetrachloroethane	0	-	0	-
1,1,1-Trichloroethane	2	5	23	178
1,1-Dichloroethane	17	400	41	116
Chloroethane	8	n.g.	8	n.g.
Tetrachloroethylene	5	n.g.	4	n.g.
Trichloroethylene	13	51	71	5.6
1,2-t-Dichloroethylene	33	492	32	104
Vinylchloride	10	107	15	51
Aromatics				
Benzene	49	65	13	7
Toluene	69	583	89	406
Ethylbenzene	57	198	39	60
Xylenes	20	461	n.m.	210

n.m.: not measured.
n.g.: not given
a Old MSWLF are landfills first receiving waste before 1985
b New MSWLF are landfills first receiving waste after 1985

TABLE 9
Biological Species (Number of Species) Used in Toxicity Tests for Assessment of Toxicity of MSW Landfill Leachate

No. of MSW landfills	F	C	D	A	B	G/M	Other	Study
1	1							McBride et al. 1979
7	2	1						Cameron and Koch 1980, Atwater et al. 1983
1	1	1		1	1			Plotkin and Ram 1984
1	1							Wong 1989
8*						1		Omura et al. 1992
19	1	1			1			Kross and Cherryholmes 1993
4					1	3		Schrab et al. 1993
2				4				Cheung et al. 1993
2			1		1		2	Devare and Bahadir 1994
1	2	1						Ernst et al. 1994
1					1			Nohava et al. 1995
9			1					Clément and Merlin 1995
35		1						Assmuth and Penttilae 1995
2						4		Helma et al. 1996
8		3	1	1	1		2	Clément et al. 1996
1						1		Beg and Al-Muzaini, 1998
1						3		Cabrera and Rodrigues, 1999
10				1	1	1		Ledin et al. 2001

F: Fish, C: Crustacean, D: Duckweed, A: Algae, B: Bacteria, G/M: Genotoxicity/mutagenicity, Other: Protozoans, Rotifers, Plants.
*=Leachate collected from mixing tank covering 8 landfills.

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1996). As with the case for other complex environmental matrices, batteries of bioassays with organisms from different trophic levels increase the potential of detecting toxicity in a leachate sample (Kross and Cherryholmes, 1993; Keddy et al., 1995).

The toxicity of leachate from more than 98 different landfills has been reported in the studies listed in Table 9. In general, high toxicity was observed in leachate from landfills receiving MSW (Cameron and Koch, 1980; Atwater et al., 1983; Plotkin and Ram, 1984; Schrab et al., 1993). Clément et al. (1996) found that the most toxic leachates were those originating from landfills with co-disposed hazardous industrial waste, but other studies found that leachates from landfills with no known co-disposal were as toxic as leachates from mixed landfills or hazardous waste landfills (Schrab et al., 1993). In a lysimeter experiment with municipal waste Cameron and Koch (1980) found that 5 years of aging of the waste resulted in an 80-fold decrease in toxicity toward rainbow trout.

Ernst et al. (1994) concluded that ammonia was the primary cause of acute toxicity of municipal landfill leachate, whereas the chronic effects of the range of XOCs identified in the leachate could not be determined. Based on toxicity testing of 27 landfill leachates, Clément et al. (1997) concluded that ammonia and alkalinity were the most probable factors contributing to the observed toxicity. In the bioassay studies by Cheung et al. (1993) and Clément and Merlin (1995), it was also concluded that ammonia was the main cause of the toxicity measured in the biotests. As illustrated in Table 10, other studies have indicated that factors like pH, conductivity, and the concentrations of chloride, copper, or zinc may also be of major importance to aquatic toxicity assessed by aquatic bioassays (Cameron and Koch, 1980; Atwater et al., 1983; Kross and Cherryholmes, 1993; Assmuth and Penttilae, 1995; Clément and Merlin, 1995).

The chronic effects of landfill leachates have not received much attention, but of the different long-term effects mutagenicity/genotoxicity has been studied to some detail. Omura et al. (1992) covered leachates collected from eight MSW landfills. It was found that the leachates were mutagenic after preconcentration, and the authors suggested that organic compounds in the leachate caused the mutagenic activity. Based on multiple genotoxicity tests of leachate from MSW landfills, Schrab et al. (1993) found that three of four tested samples exerted genetic toxicity. This was also found by Helma et al. (1996) in a study of different aqueous samples, including landfill leachates. These experiments showed that landfill leachates had higher genotoxic potency than effluents from pulp production and wastewater. Moreover, risk calculations indicated that MSW leachates may present as great a cancer risk as those from co-disposal and industrial solid waste landfills (Schrab et al., 1993; Brown and Donnelly, 1988).

A number of studies reported that analytical measurements of XOCs did not correlate well with the toxicity observed in bioassays (Plotkin and Ram, 1984; Kross and Cherryholmes, 1993; Lambolez et al., 1994). It is, however, a general observation that landfill leachates may contain a large variety of XOCs that are

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TABLE 10
Relationships between Toxicity of Different Landfill Leachates and Physico-Chemical Parameters (Estimated by stepwise multiple linear regression.)

Organism	Relation	R^2	Study
Fish	$\log LC_{50,\ 96h} =$	0.94	Cameron and Koch, 1980
(Salmo gairdneri)	$1.427-0.386[NH_3]-101400[H^+]-0.000539[Tan]-4.074[Cu]$		
Duckweed	$EC_{50}=-23.3\log[Alk]-9.0\log[NH_4^+]+78.9$	0.96	Clément et al., 1997
(Lemna minor)	$EC_{50}=-21.6\log[Alk]-15.5\log[Cond]$	0.89	Clément and Merlin, 1995
Crustacean	$LC_{50}=27.73[Zn]^{-1.05}$	0.97	Altwater et al., 1983
(Daphnia pulex)	$\log LC_{50}=0.969-0.00884[Zn]-0.00152[Tan] + 3.804\ [NH_3]$	0.83	Clément and Merlin, 1995
Crustacean	$EC_{50}=2489\ [NH_3]^{-0.62}[Alk]^{-0.386}$	0.96	Clément et al., 1997
(Daphnia magna)	$EC_{50,\ 48h}=102-0.00507(LC_{50,24h})^2-0.396(\log[Cl^-])^2$	0.73	Assmuth and Penttilae, 1995

Tan: Tannin; Alk: Alkalinity; Cond.: Conductivity

acutely and chronically toxic (e.g. Brown and Donnelly, 1988; Schrab et al., 1993; Ernst et al., 1994). Despite all of the aforementioned studies, the importance of XOCs as a contributor to leachate toxicity remains largely unknown. This maybe due to the fact that toxicity caused by the sample matrix (e.g., ammonia, alkalinity, and salts) masks the toxic effect of XOCs (Baun et al., 1999). A way to overcome this problem is to use fractionation and preconcentration procedures such as the ones used in mutagenicity testing (e.g., Schrab et al., 1993; Helma et al., 1996). This approach was used by Ledin et al. (2001) in a study of the toxicity of leachate from 10 Danish MSW landfills. In general, the organic fraction of all landfill leachates were toxic toward algae and bacteria, whereas genotoxic responses were found in only a few of the samples in the concentrations used (Ledin et al., 2001).

I. Summary

The review on present leachate concentrations shows relative high concentrations of many parametres, especially COD, ammonia, and in some cases salts and xenobiotic organic compounds. There is still relatively little information regarding XOCs. The concentrations of heavy metals are generally low, even in cases where

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considerable amounts of heavy metals have been disposed of at the landfill. In the next section potential changes in leachate composition over periods of time beyond the currently available data are considered.

IV. LONG-TERM MSW LANDFILL STABILIZATION PROCESSES AND LEACHATE COMPOSITION

The decomposition of refuse and its relationship between the state of decomposition to leachate composition has been described in the previous sections. This description is appropriate for landfills in which decomposition is enhanced such that decomposition may occur over a period of as little as 5 to 10 years, as well as for landfills where there is sufficient moisture infiltration to promote decomposition over periods of 20 to 50 years. At landfills in arid regions, where the waste is dry and there is little infiltration, the landfill may remain in the acid phase or early methane production phase for decades or longer. To evaluate the long-term properties of landfills, it is important to look beyond the time when gas production is significant and to evaluate how the landfill ecosystem can be expected to behave over centuries and even thousands of years, once the bulk of the degradable components of the refuse have been degraded.

The objective of this section is to look into the future at the end of the refuse decomposition cycle, recognizing that the time to reach the end of the decomposition cycle will vary based on environmental conditions, primarily moisture, within the landfill. An understanding of the likely changes in leachate composition over long periods of time is required to assess the safety of landfills for the environment in the future. Interestingly, the time frames in this section go well beyond the 30-year post-closure monitoring period prescribed for landfills in the U.S.

A. Long-Term Landfill Phases

The four phases following the stable methanogenic phase are shown in Figure 1. The phases are based on speculation because we are not familiar with any landfills that have progressed beyond the stable methanogenic phase. Figure 1 is generally valid for a single waste cell, that is, a mass of waste having the same age. However, the last four phases on the figure are valid for a whole landfill because any age differences of the waste layers will be small compared with the total time to decompose beyond the methanogenic phase. In the following description we also focus on the gas composition within the landfill. The evolution of the landfill environment, and by that any potential changes in leachate composition will be driven by the supply of oxygen to the waste layers. The intrusion of oxygen to the landfill is dependent on the remaining degradable organic carbon, the gas genera-

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tion rate, the cover properties, and the topography of the landfill. Phases V and VI may be seen as intermediate phases between the stable methanogenic phase and the carbon dioxide phase, where the landfill becomes aerobic.

1. Phase V. Methane Oxidation

Through the stable methanogenic phase, the gas production rate is continuously decreasing. For landfills with an active gas extraction and utilization system, air might be pulled into the landfill due to overpumping in the later phases when gas generation rates are lower. For landfills without active gas extraction, a small elevated gas pressure drives gas emission, and air intrusion will only occur due to barometric pressure changes, which might push air into the upper waste layers. If the landfill is equipped with a tight cover and open gas vents (assuming that active gas extraction or gas flaring has ceased), then air intrusion may be very local and probably of minor importance. For more permeable covers, the oxygen intruding into a landfill at high barometric pressure or due to diffusion will promote oxidation of the methane still produced in the landfill. The methane oxidation will mostly take place in the cover soil or in the waste mass surrounding the gas vent screens, that is, an insignificant part of the landfilled waste will be intruded by air.

2. Phase VI. Air Intrusion

As methane production continuously decreases, air will intrude through the cover and into the actual waste mass. At some time, almost all of the methane produced should be oxidized in the waste mass and the cover. This means that the average methane content within the landfill will continuously decrease, and carbon dioxide concentrations can be expected to increase through its production in the methane oxidation reaction. In addition, the nitrogen content will increase due to air intrusion, while oxygen will nearly be undetectable due to its rapid consumption. The overpressure produced by the generated methane will approach zero.

3. Phase VII Carbon Dioxide

Methane production will be insignificant in Phase VII, and the oxygen intruding the landfill will be consumed for oxidation of any residual methane, organic materials and reduced inorganic species (sulfur, nitrogen, and iron-containing species) buried in the landfill. Under aerobic conditions, additional refuse oxidation will occur, as some of the lignocellulosic substrate is more degradable under aerobic conditions relative to anaerobic conditions. The gas composition will be comprised largely of carbon dioxide, oxygen, and nitrogen. The resulting CO_2 production has been reported to result in

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a pH decrease in both refuse (Revans et al., 1999) and harbor sediments (Calmano et al., 1993). Another factor that exerts downward pressure on the pH of the refuse mass is the oxidation of reduced sulfur, nitrogen, and iron as each of these oxidation reactions releases protons (Calmano et al., 1993). It has been suggested that as the pH decreases, metal carbonate precipitates will dissolve and the carbonate release will buffer the pH, although this has not been demonstrated experimentally (Calmano et al., 1993). This phase has also been called the humic phase (Bozkurt et al., 1999).

The most important factors governing the exchange of air between the landfill and the atmosphere are diffusion, wind-induced exchange, natural convection (density differences), and barometric pumping (Bozkurt et al., 1999). Concentration differences between the landfill and the atmosphere will drive a *diffusional transport* of atmospheric components in and landfill gas components out of the landfill. One very important parameter is the diffusivity of the cover and the waste layers, which again is mainly governed by the water content. The presence of an engineered top cover aiming at preventing infiltration to the waste layers often includes a clay layer with a very low gas diffusivity at normal water content. In waste layers saturated with water, gas diffusion is not active and the diffusion through water is 10,000 times lower than in air. Diffusion will also be reduced significantly by the presence of plastic flexible membrane liners.

For above grade landfills, winds blowing past the landfill will create a pressure gradient between the two opposite sides of the landfill. This will cause air to flow through the landfill, the magnitude governed by the wind speed and the permeability of the cover/waste layers. However, Bozkurt et al. (1999) suggested that this process is of minor importance in most cases.

Natural convection is governed by differences in gas densities between the atmosphere and the interior of the landfill. This variation may be due to temperature differences, differences in air humidity and differences in concentration (CO_2/O_2), the interior gas being both lighter or heavier than the atmospheric air (depending on time over the day as well as over season) (Bozkurt et al., 1999).

If the landfill has a cover with a lower permeability than the waste layers, and if the cover has imperfections such as cracks, or open gas vents, or maintenance shafts, any *barometric pressure changes* will pump gas in or out of such openings. The flow direction will depend on the direction of the pressure change. Christensen et al. (2000) have successfully used passive venting as a remedial technology for removing volatile pollutants from sandy unsaturated layers covered by clay and showed that barometric changes led to a significant exchange of air through the installed screened wells.

Bozkurt et al. (1999) modeled oxygen intrusion considering the above stated processes (except barometric pumping) assuming that all oxygen is used for the oxidation of organic matter, and that all organic matter is available for oxidation. They used the model for different landfill scenarios (with/without cover, water saturated/unsaturated) and showed that in most cases the diffusional oxygen transport dominates the other transport processes. They estimated the time required to deplete all organic matter and found values ranging from a few centuries (for an

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unsaturated, uncovered landfill) to more than half a million years for a landfill fully saturated with water (i.e., waste placed under the groundwater table).

B. Long-Term Leachate Composition

In this section, changes in leachate composition through phases V–VII are evaluated. This evaluation is based on theory and limited laboratory data as field data from landfills in phase V-VII are not available.

1. Dissolved Organic Carbon and Inorganic Macrocomponents

The concentration of dissolved organic carbon and inorganic macrocomponents in landfills containing well-decomposed refuse was summarized in Section III.C and III.D, and these concentrations were shown to be relatively high, certainly higher than what can be allowed for direct release to the environment. Data from specific landfills generally show a slow decreasing trend with time (Kjeldsen and Christophersen, 2001). BOD/COD ratios below 0.1 can be expected and the absolute concentrations of BOD and COD are likely to decrease slowly through Phase V–VII until a BOD of near zero and a COD that consist of recalcitrant humic matter is all that remains. Ammonia-nitrogen will be converted to nitrate as a landfill becomes aerobic.

Belevi and Baccini (1989, 1992) developed a model to predict the long-term leachate concentration from landfills based on the extrapolation of leachate fluxes from the early phases of decomposition coupled with laboratory leaching experiments. Their objective was to evaluate when a landfill leachate would meet Swiss water quality standards without treatment. Elements modeled include C, N, P, Cl, Fe, Cu, Zn, Pb, and Cd. They present a first-order decay model that requires both a remobilization rate and knowledge of the fraction of a given element in MSW that can be mobilized. This fraction was determined in laboratory leaching experiments conducted with multiple distilled water extractions. The total volume of water used for the extractions was intended to simulate 2000 years of infiltration at a hypothetical landfill. The proposed methodology is largely empirical and does not take into account (1) the effect of biological transformations and stabilization on element release, (2) potential changes in the redox potential of a landfill, (3) the potential for dissolved organic matter to complex certain elements and increase their solubility, or (4) immobilization (sorption, precipitation) reactions.

2. Heavy Metals

As described in Section III.E, the concentration of heavy metals is low under the long, stable methanogenic phase, and the metal release is not considered

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problematic through this phase. However, as the refuse mass turns aerobic, a number of biological and chemical reactions will occur (see Section IV.A), and these reactions may well effect metal mobilization. As previously described, a decrease in pH is expected through the carbon dioxide phase, and it is well known that metal dissolution is enhanced at low pH. During anaerobic refuse decomposition, sulfates in refuse are reduced to sulfides that form insoluble precipitates with most metals (Christensen et al., 2000). However, as the refuse mass turns aerobic, these sulfide precipitates can be expected to oxidize to metal sulfates. The sulfate compounds are considerably more soluble than the corresponding metal sulfides that may increase metal mobilization. For example, pK_{so} for PbS and $PbSO_4$ is 27.6 and 7.73, respectively.

In addition to pH, metal solubility will be influenced by the oxidation-reduction potential (E_h). As the E_h of the refuse mass increases, the solubilities of some metals may increase. For example, Charlatchka and Cambier (2000) showed that the concentrations of Pd, Zn, Mn, and Fe increased with increasing E_h in agricultural soils polluted with metals. While there are certainly analogies between soils and decomposing refuse, the concentrations of organic carbon and the diversity of biological and chemical reactions are likely broader in landfills.

Other factors that will influence metals solubility include the cation exchange capacity of the refuse and how it changes during refuse oxidation and the presence of more oxidized functional groups on the solid humic matter as well as humic matter in leachate (Martensson et al., 1999). For example, carboxylic acids are known to act as chelators, and there could be an increase in carboxylic functional groups on humic compounds because the refuse is oxidized. Finally, iron hydroxides and oxyhydrates are formed during refuse oxidation. These compounds are reactive and may adsorb other metals (Calmano et al., 1993 Bozkurt et al., 2000).

In summary, there are multiple processes that can affect metals mobilization during refuse oxidation. Bozkurt et al. (2000) developed a model to predict long-term emissions of metals from landfills. Their model considered the oxidation of all refuse organic matter, humic substances that bind metals, and metal sulfide precipitates as well as pH buffering associated with calcite dissolution. They considered oxygen diffusion into landfills with a range of landfill geometry's and degrees of saturation and predicted that heavy metals mobilization will not occur for thousands of years. Belevi and Baccini (1989) suggested by measuring the alkalinity of landfill samples and assuming that 50% of the organic matter content of the waste is degradable and produces protons that landfills contain sufficient buffer to maintain alkaline conditions for more than 2000 years. Consequently, they did not expect remobilization of heavy metals due to lower pH for many centuries.

There has been a limited amount of experimental work to measure metals mobilization under an aerobic phase in well-decomposed refuse. Martensson et al. (1999) identified a number of factors that could effect metal's mobility in well-decomposed refuse that is undergoing a transition from anaerobic to aerobic

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conditions. Working with samples of 20-year-old refuse excavated from a landfill, they showed that the concentrations of Zn, Cd, Cr, S roughly doubled when the refuse was decomposed under aerobic conditions relative to anaerobic conditions in reactors. They suggested that aeration of the decomposed refuse resulted in the production of chelating agents that enhance metals mobility. This suggestion was supported by an increased metal binding capacity of aerated leachate. In contrast to the results of Martensson et al. (1999), Revans et al. (1999) conducted work in experimental columns that were filled with well-decomposed refuse to which Cd, Zn, and Cr had been added. Columns were aerated to simulate an aerobic phase after anaerobic decomposition. The concentrations of Cd, Zn, and Cr in leachate did not increase after aeration began and a mass balance showed that less than 1% of the added Cd, Cr, and Zn were leached after passing three bed volumes of leachate through the columns. Interestingly, the pH decreased approximately 0.8 pH units when the columns were turned aerobic. Altmann and Bourg (1997) carried out laboratory experiments with aqueous systems containing solid phases (aquifer solids from a landfill leachate plume). They showed that Cd that was previously trapped as a sulfide precipitate was released to the aqueous phase when conditions changed from anaerobic to aerobic. The released Cd was removed subsequently from solution by adsorption on iron oxyhydroxide phases or by precipitation as a carbonate mineral. Revans et al. (1999) suggested that the same processes were responsible for the low metal release in their columns.

Flyhammer et al. (1997) studied the fractionation pattern of heavy metals present in the fines and paper fractions of MSW based on the assumption that metals present in plastics, metals, rubber, and leather were not leachable under landfill conditions. They showed an increase in the fraction of Cu, Zn, Pb, Cr, Ni, and Cd present as carbonates and sulfides in anoxic refuse relative to aerobic refuse. This observation confirms the widely held expectation that metals are largely immobilized by precipitation under anaerobic conditions.

Finally, it must be recognized that the form of the added metal will likely impact its behavior in a landfill. Flyhammer et al. (1998) suggested that approximately 30% of the metals in MSW were available in reactive solid form based on a sequential extraction procedure. Using a similar set of extractions, Prudent et al. (1996) reported on the concentrations of Cd, Cr, Cu, Ni, Pb, and Zn in 14 components of MSW. Prudent's work showed that much of the total metal content was present in forms that were not likely to be reactive in landfills. For example, plastics were the major contributors of Cd and scrap metal and rubber were major contributors of Zn. Interestingly, fines were high for each metal analyzed although the components of MSW that contributed to the fines fraction are difficult to define.

In summary, there are a number of factors that influence metals mobility because refuse goes from an anoxic to an aerobic state, including pH, E_h, functional groups on humic matter, and the sorptive capacity of the refuse mass. Given the complexity of the refuse system, it is not surprising that the limited amount of

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experimental work that has been conducted has resulted in both increased metals mobilization (Martensson et al., 1999) and no observed metals mobilization (Revens et al., 1999; Flyhammer et al., 1997). The experimental work to data is limited and the number of parameters that affect metals solubility makes models complex and uncertain. It is also unrealistic to expect that there will be field data forthcoming given the long period of time that is theoretically required for a landfill to turn aerobic. Thus, laboratory simulations that include the aerobic oxidation of organic matter that remains after anaerobic decomposition are required. These laboratory simulations should probably be performed over a long time frame because the aerobic oxidation may be a slow process.

3. Xenobiotic Organic Compounds (XOCs)

To evaluate long-term leachate concentrations and the time required to reach final storage quality, a multiphase approach is needed, including several processes: volatilization to gas escaping the landfill, diffusional loss through top covers, leaching and degradation. Kjeldsen and Christensen (2001) developed a simple box model, MOCLA, to evaluate the long-term fate of XOCs. The model is based on equilibrium assumptions concerning phase distribution mechanisms, and a first-order degradation process. Model runs using MOCLA showed that most XOCs will be released (for most volatile organic compounds through the gas produced) within a few decades. For more strongly sorbing compounds, such as naphthalenes the release will take several decades. Sensitivity analysis showed that degradation might play a very significant role in the long-term fate of XOCs. However, data in the literature on degradation of organic chemicals in landfills are still very rudimentary and, unless specific degradation rates are retrieved for a specific case, default values on degradation cannot be suggested (confer detailed literature review in Kjeldsen and Christensen, 2001).

Several factors may lead to significant extended life times of XOCs in landfills. The presence of special waste types having very high sorptive capacities such as activated carbon (Kjeldsen and Grundtvig, 1995) or insulation foams releasing CFCs (Kjeldsen and Jensen, 2001) may extend the life time considerably. Also, slow desorption from the organic waste may govern the release of XOCs, a fact observed in several studies with soils (Luthy et al., 1997). Recently, Sanin et al. (2000) reported on the behavior of toluene, acetone, and 1,2-dichoroethane in decomposing refuse. All three compounds were shown to biodegrade. Moisture stimulated toluene but not acetone biodegradation, and it was speculated that higher moisture increased the solubility and subsequent bioavailability of the hydrophobic toluene. In addition, some toluene was shown to be associated with the humic fraction (humic acid, fulvic acid, and humin) of the refuse after decomposition. This association may influence bioavailability and may also provide an alternative stable endpoint for some aromatic compounds.

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In many older landfills and in some sanitary landfills, XOCs have been disposed of in intact iron containers such as drums. There is a considerable chance that such containers in some cases might have kept their integrity through the operational phase of the landfill. Thus, any release from such containers is relying on progressive corrosion eventually combined with the physical load of overlying waste layers. It is well known that iron can corrode under anaerobic conditions, especially during the acid phase when low pH will lead to substantial corrosion (Scully, 1990). However, in many cases only slightly acidic pH is reached due to the presence of adequate buffer in the waste (Christensen et al., 1992). Anaerobic corrosion of iron follows the reaction (Reardon, 1995):

$$Fe(s) + 2H_2O(l) \rightarrow Fe^{2+} + 2OH^- + H_2(g) \rightarrow Fe(OH)_2(s) + H_2(g)$$

As shown, the reactions produces hydroxide-ions leading locally to increased pH, precipitating calcium, magnesium, and iron present in the leachate on the iron surfaces. This phenomenon is known as passivation (Scully, 1990), because it prevents further anaerobic iron corrosion. Iron corrosion in the stable methanogenic phase therefore is expected to be relatively slow.

As previously discussed, acidity may be produced in the landfill phases following the methanogenic phase due to oxidation of organic matter and sulfides. This may locally dissolve the passivating layer on the iron surface and expose it to aerobic corrosion. Aerobic corrosion is generally faster, especially if pitting corrosion takes place literally forming open holes in the iron surface (Scully, 1990).

Sudden releases of organic chemicals in the carbon dioxide phase of a landfill due to corrosion of containers may lead to enhanced sorption and degradation of the pollutants. However, studies have shown that the dehalogenation of chlorinated compounds, which in many cases has been observed in landfills, is especially active at low redox potentials (Leahy and Shreve, 2000; Kromann et al., 1998), but inactive under oxidizing conditions, suggesting that a delayed release of chlorinated compounds in the carbon dioxide phase may leave the landfill unattenuated. Of course, this is a worst case scenario based on a container with chlorinated solvents that does not release the solvents until a landfill is largely aerobic. Such containers may also collapse shortly after refuse burial due to compaction of freshly buried refuse.

V. CONCLUSIONS

The release of leachate to the environment is one of the major environmental impacts related to disposal of waste. Disposed waste in landfills undergoes a series of phases where the waste is decomposed. During the decomposition leachate is generated by excess rainwater infiltrating the waste. The leachate contains four

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groups of pollutants: dissolved organic matter, inorganic macrocomponents, heavy metals, and xenobiotic organic compounds.

Existing data show that the composition of leachate is highly dependent on the degradation stage of the waste. In the acid phase, concentrations are generally higher due to enhanced formation of dissolved organic matter and release of ammonia. In the methanogenic phase, the content of dissolved organic matter significantly decreases and the composition of the organic matter changes indicated by BOD:COD ratios below 0.10. The ammonia concentration seems not to follow the same decreasing trend and may constitute one of the major long-term pollutants in landfill leachate. The content of heavy metals in the leachates is generally very low as a result of attenuating processes (sorption and precipitation) that take place within the disposed waste.

Leachate contains a broad variety of xenobiotic organic compounds. A very broad concentration range for each pollutant is observed in most cases. The most frequently observed compounds are aromatic hydrocarbons and chlorinated aliphatic compounds. Some later studies have also reported more polar compounds, which have not been observed previously with older analytical methods used.

A landfill will in a long time frame undergo several phases in which oxygen from the atmosphere will penetrate deeper and deeper into the waste. The oxidation of residual organic matter not decomposed anaerobically, and oxidation of sulfur, nitrogen, and iron-containing compounds may lead to lower pH and higher redox potentials, which may result in enhanced long-term releases of heavy metals. However, model calculations and a few laboratory experiments suggest that the enhanced release might not occur within a timeframe of several thousands of years. The xenobiotic organic compounds are affected by volatilization and degradation as well as leaching and are in most cases not believed to be a long-term problem.

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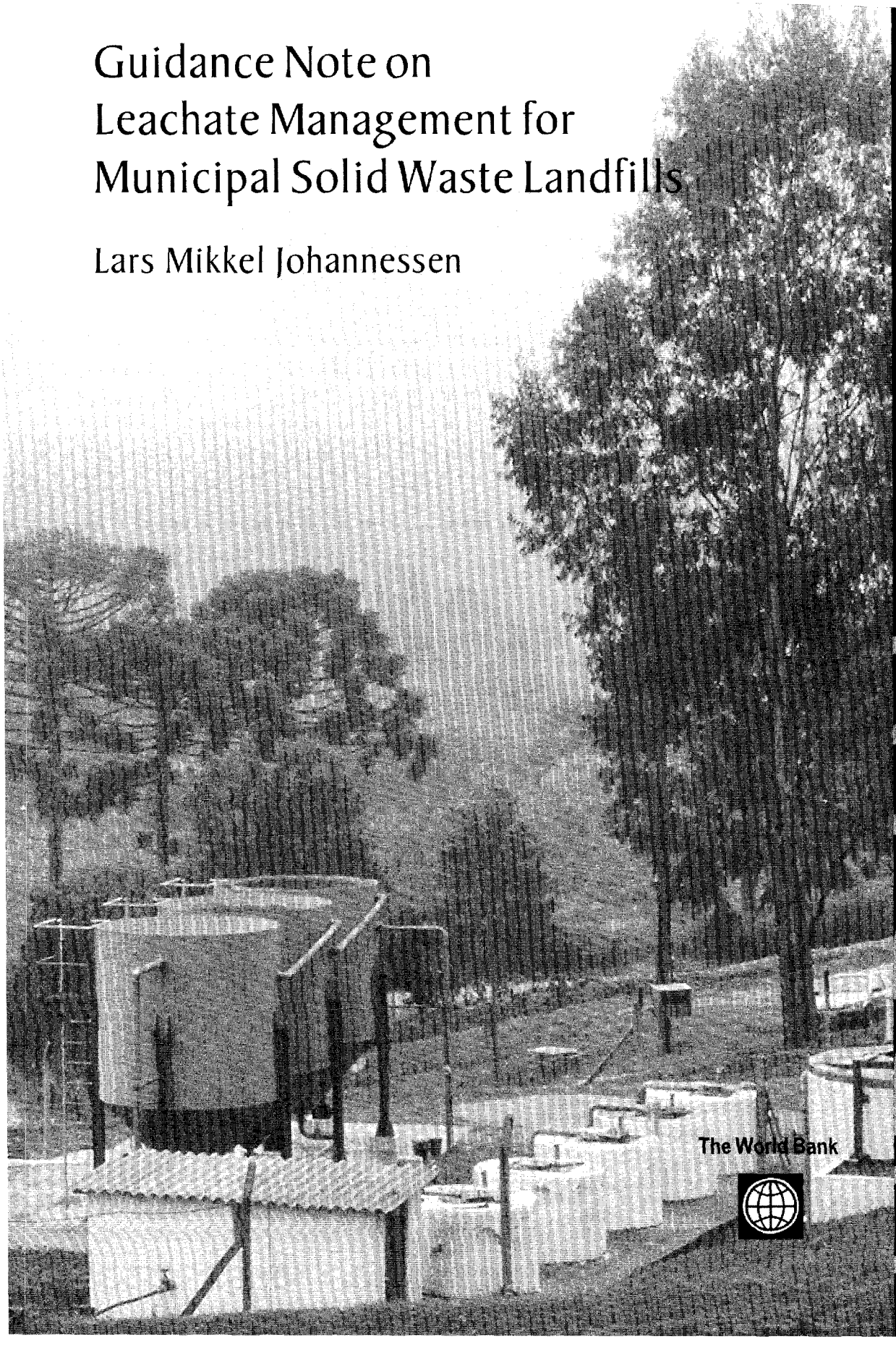
Lars Mikkel Johannessen



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Guidance Note on Leachate Management for Municipal Solid Waste Landfills

Lars Mikkel Johannessen

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1. INTRODUCTION

1.1 Background

One of the most important issues related to siting, planning, design, operation, and long-term management of a municipal solid waste (MSW) landfill is the management of leachate. Leachate is the fluid percolating from landfills that is generated from liquids present in the waste and water from outside percolating though the waste. Leachate will be generated wherever water can enter waste in a landfill or dumped. The quantity and strength of the generated leachate depends on several factors, the most important of which are: a) the amount and characteristics of the discharged waste; b) climatic conditions; c) cell size and phasing of the disposal area; d) operational techniques applied at the landfill; and e) the final top cover applied.

Leachate from MSW landfills contains various contaminants at concentration levels that may have an environmental impact on groundwater and surface water and may therefore be a threat to human health. Leachate may be highly toxic for several decades or even centuries before reaching a level where it is no longer a threat to the environment. "Eternal" leachate collection and treatment at MSW landfills, however, is not a realistic long-term leachate management option because this approach requires external inputs of energy and maintenance. It is therefore necessary to accept that landfills eventually will be left unattended and thus acknowledge the eventual release of some leachate into the environment. Hence, leachate management becomes an important issue in deciding which strategy to apply in any planning process involving closure of dumps and/or siting and development of landfills. The options range from prevention of leachate generation, to sophisticated leachate treatment options, to controlled release of leachate into the environment.

1.2 Objective

The objective of this technical guidance note is to provide practical guidance for better project preparation involving leachate management for municipal solid waste landfills.

1.3. Target groups and limitations of the guidance notes

The document is aimed at local and donor agency task team leaders and their task teams for urban, environmental, and solid waste projects. The guidance note also provides essential information for solid waste and environment decision makers and professionals involved in project preparation in client countries.

The guidance note may not apply to arid climates, as the assumption is that disposal of waste in arid climates rarely generates excess leachate that can affect the environment. However, monitoring may still be essential in arid areas.

The guidance note does not provide detailed technical advice on specific leachate containment (i.e., lining and drainage) or treatment, nor does it set specific discharge standards.

1.4. Cross references

Technical guidelines for designing and construction MSW landfills are provided in the newly developed "Solid Waste Landfills in Middle- and Lower-Income Countries: A Technical Guide to Planning, Design, and Operation," by P. E. Rushbrook and M. P. Pugh. (Published by the World Bank, World Health Organization (WHO), the Swiss Agency for Development and Cooperation (SDC), and the Swiss Center for Development Cooperation in Technology and Management (SKAT), 1998.)

Other World Bank technical guidance notes in this series include:

- Haukohl, J., Marxen, U., and Rand, T., "Decisionmakers' Guide to Municipal Solid Waste Incineration," The World Bank, (under preparation, expected 1999).
- Haukohl, J., Marxen, U., and Rand, T., "Municipal Solid Waste Incineration," The World Bank, (under preparation, expected 1999).
- Johannessen, L. M., Dijkman, M. "Health Care Waste Management," (under preparation, expected 1999).
- Johannessen, L. M., "Guidance Note on Recuperation of Landfill Gas from Municipal Solid Waste Landfills," The World Bank, Urban and Local Government Working Paper Series, September 1999).
- Johannessen, L.M., Bartone, C., "Guidance Note on Approach to Siting of New Landfills," (under preparation, expected early 2000).

In addition, further details on leachate management may be found in:

- Christensen, T.H., Cossu, R., Stegmann, R. (ed.), "Landfilling of Waste: Leachate." Elsevier Applied Science, Elsevier Science Publisher, England, 1992.

2. LEACHATE MANAGEMENT STRATEGIES FOR MSW LANDFILLS

The main objective of leachate management is to ensure that landfilled waste does not impose any unacceptable short- or long-term risks to the environment or to public health. Throughout the world, one of four distinct leachate management philosophies is applied to a lesser or greater extent to meet this objective, either as part of the regulatory framework or as a general practice. The four philosophies and their consequences are as follows:

- **Encapsulation and total containment,** also known as the "dry tomb" concept, is meant to prevent leachate generation by halting the percolation of water through the waste, apart from what may be generated from the moisture content in the waste at the time of disposal, and any moisture generated during the period before final capping. This philosophy makes sense only in wet climates.

The result of this strategy is that the waste is virtually preserved and therefore the environmental risk from the waste may remain virtually unchanged for a very long time. As the encapsulation eventually will fail (due to settling and unforeseen events or activities), perhaps centuries after installation, uncontrolled release of leachate may occur. This strategy therefore passes on the responsibility for leachate management of today's waste to future generations. Hence, this approach should be seen as a preliminary storage strategy and not as a long-term sustainable leachate management option for MSW landfills.

- **Containment, collection, and disposal of leachate,** in principle also known as a "sanitary" landfill, is a strategy in which generated leachate is collected from a liner system of low impermeability, sent through a drainage system, and normally led or pumped for treatment before final discharge to a surface water body. The rate of leachate formation depends on climatic conditions and, after final infilling, is also controlled by the top cover. This strategy enhances stabilization and mineralization of the disposed waste (e.g., through an enhanced bioreactor landfill with options for recirculation of leachate and/or enhanced infiltration of water). The containment, collection, and disposal of leachate strategy is often built on the "eternal" system philosophy. This type of leachate treatment system often requires inputs of energy and may therefore have an active lifetime of 30–50 years after closure, after which it eventually fails (through lack of maintenance, cessation of power, etc.). Consequently, uncontrolled release of leachate may eventually occur and therefore impose unexpected environmental effects from today's waste onto future generations. In combination with other leachate management measures, however, this leachate management strategy can be a first step toward acceptable leachate release.

- **Controlled contaminant release** is a strategy in which the release of the quality and/or quantity of leachate is maintained at all times at an environmentally acceptable level without inputs of energy or other required maintenance. Prior to any release of leachate, the environmental impacts must be assessed and the acceptable level of leachate release determined. MSW leachate may, in the short term (i.e., < 30-50 years), exceed requirements for controlled release. For most MSW landfills under wet climatic conditions, the containment, collection, and disposal strategy may be required as the initial leachate management step. During this time settling will occur and landfill gas will be generated. Consequently, stable sloped top covers, surface drainage systems, and vegetation may require significant maintenance. Hence, the long-term strategy of controlled contaminant release from a MSW landfill may apply only when the waste is stabilized and limited settling will occur. A long-term control measure may be the installation of geologically stable and sloped top covers with a surface drainage system and surface vegetation with high potential evaportranspiration.

- **Unrestricted contaminant release** occurs where no precautions are taken to prevent or reduce the release of leachate. As a strategy, this may be applied to inert waste landfills (i.e., for construction waste or soil) and to MSW landfills in arid climates. However, dumps for MSW apply this leachate management method in wet climates as a lack of strategy. Consequently, the uncontrolled release of leachate will eventually occur and have unpredicted (and often

unacceptable) environmental effects on groundwater and/or surface water. This approach is therefore not a valid leachate management strategy for MSW landfills.

3. REGULATORY AND INSTITUTIONAL ARRANGEMENTS

3.1 Regulatory requirements

Leachate management should be seen as an integral part of landfill siting, planning, design, construction, operation, and maintenance. Leachate management should therefore be integrated into the regulatory framework for landfills.

Where legislation on landfills exists, short-term impacts are often covered. Long-term impacts from leachate, however, should also be reflected in regulations concerned with the landfilling of waste. Leachate management regulations should consider the following factors:

- ***Siting*** is often controlled by several factors. Leachate management strategy and siting should be an iterative process, in which the optimal site for a landfill also provides environmentally acceptable properties for long-term leachate management. Knowledge of hydrogeology and hydrological connection to surface waters becomes important in long-term leachate management strategies, as leachate eventually will be released to groundwater. Groundwater downstream from a landfill may therefore have to be written off and alternative water supplies may be necessary.

- ***Design/construction*** requirements for leachate management should be flexible and based on functional demands rather than specific criteria, as no two landfill sites are identical.

- ***Discharge standards*** for leachate treatment plants under normal conditions may be set as standards for industrial sewage treatment or for municipal sewage treatment plants. For a discussion of direct release, see below.

- ***Monitoring*** requirements should be the single most important regulatory factor in leachate management. From a regulatory point of view, monitoring should: a) track the development and fate of leachate composition; b) either detect if any leachate has leaked to groundwater, or follow development of the intentionally released leachate; c) include indicators for leachate treatment effluent; and d) determine the effects on the receiving surface water. Bio-monitoring of the cumulative effects on the surface water may be required because traditional monitoring of chemical parameters is often inefficient due to dilution and masking effects blur from other sources.

- ***Release of leachate*** to groundwater should only be permitted based on predetermined requirements. These could include a requirement that leachate composition must constantly remain below set site-specific standards (based on dispersion and dilution potential in

groundwater and the vulnerability of surface water) for a period of at least two years. These standards may be based on criteria dictating that: a) the dilution potential in the groundwater be sufficient to ensure that the leachate plume entering the receiving surface water does not exceed discharge standards set for the treatment plant otherwise treating the leachate and discharging to the same receiving water; and b) ensuring that the total flux of contaminants out of the landfill is environmentally acceptable to the receiving surface water. Other requirements might include assumptions for groundwater and surface water that should be fulfilled at least five years prior to release of the leachate, and that monitoring of groundwater should be carried out for five years after the onset of leachate release from the landfill. After that point, monitoring should no longer be necessary, as any leachate that is released is unlikely to pose any environmental risk.

3.2. Institutional arrangements

In addition to including the relevant authority on landfilling of waste, institutional arrangements should involve competent authorities on discharge standards for treatment plants and on the protection of groundwater and surface water.

Leachate will pose an environmental risk longer than any other potential environmental hazards from a landfill. If the decision is made for eventual release of leachate from the landfill, the capacity to do so at a competent authority level is essential. For many countries, this capacity may (and should) exist only at the national level. To ensure a basis for decisionmaking, a mechanism to store and convey leachate monitoring data to the competent authority at the national level should be established.

4. LEACHATE GENERATION AND WATER BALANCE

Leachate is generated primarily from precipitation and thus is principally influenced by climatic conditions such as rainfall and evaporation.

In arid climates, virtually no excess leachate occurs; in semi-arid areas, leachate may be generated irregularly or only at certain times of the year. In wet climates, landfills may produce significant quantities of leachate year-round.

It is thus essential to predict the quantity of leachate that will be generated at the location selected for a landfill, in order to accomplish the following steps:

a) ***Decide which leachate management strategy to employ.*** For arid and semi-arid areas, the quantity of leachate generated may be insignificant and may have no unacceptable impact on the environment; therefore, leachate management may be limited. For wet areas, where significant quantities of leachate may be generated, containment and leachate treatment may be required as a short-term leachate management strategy [*see d) below*].

b) ***Enable properly designed leachate treatment facilities.*** The quantity of leachate is often one of the essential design criteria for leachate collection systems and treatment facilities plant. Where a significant quantity of leachate will be generated, the design should aim to handle large quantity fluctuations (wet/dry season fluctuations) and still meet effluent standards. An alternative is providing for adequate storage of leachate. This may also add flexibility into the design of a leachate treatment facility.

c) ***Enable proper design of disposal areas and leachate collection system.*** Where significant quantities of leachate are generated, it may be necessary to reduce the areas exposed to direct precipitation, as these areas may produce large fluctuations in leachate generation. Construction of small cells to reduce leachate generation should be considered. The leachate collection system should be designed to accommodate average peak leachate quantities.

d) ***Assess the potential for recirculation of leachate.*** In some climatic zones with heavy precipitation during the wet season, recirculation of leachate may be restricted to reduce the risk of landfill slides and "spilling over" of leachate to the surrounding environment.

e) ***Estimate the hydraulic pressure on the designed liner system*** to estimate the magnitude of leachate released to the environment.

f) ***Assess the pollution potential*** as a function of flux (i.e., leachate quality times leachate quantity per frequency).

Calculating leachate generation can be done through water balance models. Several models exist, from the very advanced and sophisticated, like the U.S. Environmental Protection Agency's (EPA) HELP model (version 3.07)[1] with high demands for data input, to the simplest version:

$$L = R - E_a$$

Where:
L represents the leachate volume
R represents the volume of precipitation
E_a represents the volume of actual evaportranspiration (or simpler evaporation from the ground surface)

Both rainfall and evaporation data are collected routinely by meteorological stations and are usually available.

Any model used to estimate leachate quantity should consider calculations of maximum daily quantity, average quantity for each month of the year, and average annual quantity. As the quantity of generated leachate varies significantly from open cells to closed cells, the evolution in leachate quantity throughout the lifetime (i.e., from the first cell to final cover of the entire disposal area) of the landfill

[1] Prepared by Paul Schroeder for the U.S. EPA, this model can be downloaded from the Internet at the following address: http://www.wes.army.mil/el/elmodels/helpinfo.html.

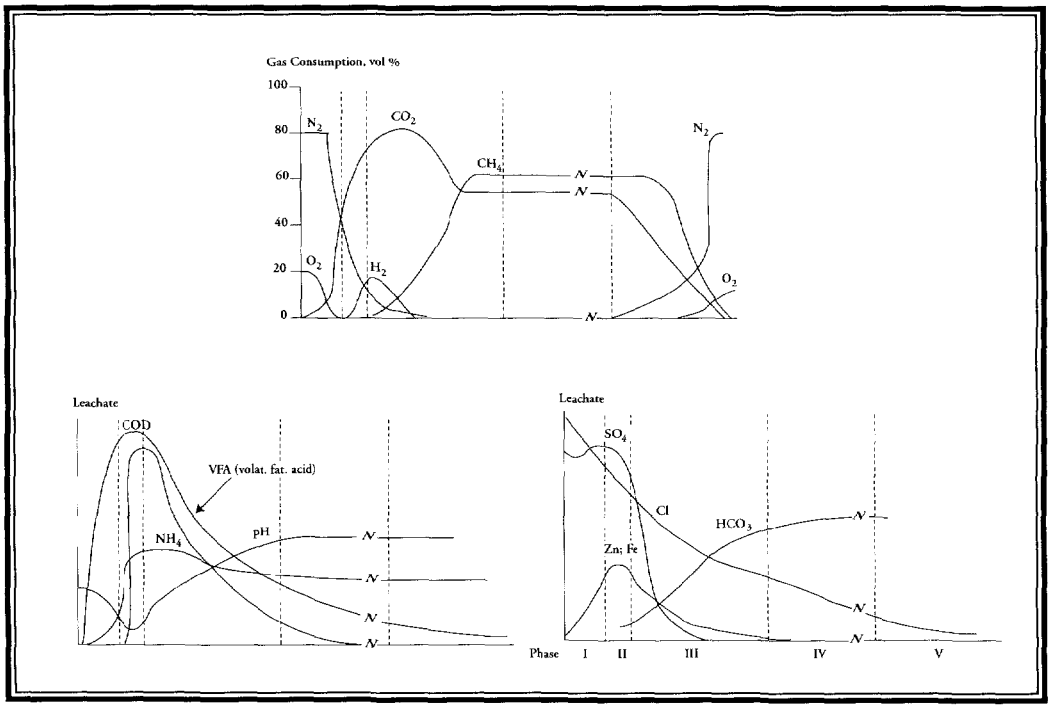
should be reflected in leachate quantity estimates. Water balance models are often subject to considerable uncertainties, given the need to predict or estimate some of the parameters in the equation. Any water balance model should only be used to indicate the magnitude of leachate generation. Thus, for this purpose, a relatively simple model may prove just as useful as the more sophisticated models.

5. LEACHATE COMPOSITION AND POLLUTION POTENTIAL

The chemical composition of leachate is affected by several factors, such as the characteristics of the disposed waste, its moisture content, depth of the disposed waste, the availability of oxygen (redox potential), the temperature and micro flora, compaction rate, and the dissolving of organic and inorganic components in the waste.

The biodegradation of organic waste follows a pattern of five phases. These five phases are fundamental and affect both leachate composition and landfill gas generation and composition. The five phases are illustrated in Figure 1.

Figure 1: Optimal Development of Leachate Composition within a Landfill Cell[2]



[2] Amended from Christensen, T.H., Kjeldsen, P., "Basic Biochemical Processes in Landfills." Christensen, Cossu, Stegmann (ed.), "Sanitary Landfilling: Process, Technology and Environmental Impact." London, Academic Press, 1989.

Each layer of disposed waste in a landfill will undergo the biodegradation phases illustrated in Figure 1. The overall factors influencing the time that elapses for each phase are climatic conditions and operational factors.

Phases I and II last from some weeks to two years (and sometimes longer). A higher ambient air temperature will enhance the biodegradation processes, as will high compaction rates, placing waste in thin layers, and the use of small landfill cells.

Phases III and IV may last at high peak for some 15 years and fade thereafter, depending on the operation of the landfill and, in particular, on the moisture content of the waste. As high moisture content is essential to stimulate bioreactions, high precipitation will reduce the times for Phases III and IV and thus reduce the organic load in the leachate. Operational measures to enhance the biodegradation includes recirculation of leachate and extraction of generated landfill gas.

Phase V of the landfill lifecycle is very dependent on the operational steps taken earlier in the landfill's life. However, it may take several decades before the disposed waste is finally stabilized and no longer constitutes a pollution potential. Ammonia is the limiting factor and will constitute potential pollution for 100 years or longer.

5.1. Leachate composition

The typical composition of leachate from a MSW landfill is given in Table 1. For simplicity, the composition is displayed for the so-called "acidic phase" (Phases I and II of Annex A) and the methanogenic phase (Phases III and IV of Annex A). As for the landfill gas, there are wide variations influenced by climatic and operational factors. For a detailed description of leachate composition, please refer to Annex A.

Parameter	Unit	Acidic Phase (6 months – 2 years)	Methanogenic Phase (2 – 100+ years)
pH		5-6.5	7.5-9
COD[a)]	mg/l	20,000-30,000	1,500-2,000
BOD_5[b)]	mg/l	10,000-25,000	500-1,000
Iron	mg/l	5-20	< 5
Zinc	mg/l	1-5	0.03-1
Cadmium	µg/l	< 30	6
Ammonia	mg/l	900-1,500	900-1,500
Chloride	mg/l	1,200-3,000	1,000-3,000

[a)] Chemical Oxygen Demand
[b)] Biological Oxygen Demand

Table 1: Typical Leachate Composition from Landfill with MSW[3]

[3] Please note that leachate composition is based on data from wet climate conditions in the Northern Hemisphere. Adequate data are not available from developing countries.

5.2. Leachate pollution potential

Leachate from MSW landfills frequently exceeds standards for drinking water and surface water, often for several decades. The leachate therefore often has the frequently significant potential to pollute groundwater and surface water.

The most common pathway for leachate to the environment is from the bottom of the landfill through the unsaturated soil layers to the groundwater, then by groundwater through hydraulic connections to surface water. However, pollution may also result from the discharge of leachate through treatment plants or by direct discharge of untreated leachate. The main factors influencing the pollution potential from leachate are:

- the concentration and flux of the leachate;
- the landfill siting, i.e., the hydrogeological setting and the degree of protection provided; and
- the basic quality, volume, and sensitivity of the receiving groundwater and surface water.

The primary components in leachate from MSW landfills that constitute a significant pollution potential are dissolved organic matter and inorganic salts. Trace elements in leachate are limited and generally do not constitute a groundwater pollution problem due to strong attenuation.

Where groundwater is used (as drinking water or for irrigation) downstream from the landfill, leachate has great potential to pollute. Where groundwater is not used or is not usable downstream, the leachate's pollution potential (if not diluted to ambient concentrations) is transferred to where the groundwater is hydraulically connected to the receiving surface water. The dilution of leachate is faster in surface water than in groundwater, but the contaminants may also spread over larger areas much faster. As well as becoming diluted, biodegradable matter in surface water decomposes, leading to oxygen depletion. Some organic substances in leachate may be toxic to aquatic organisms. The major concern about organic matter from leachate in surface water may therefore be the ecotoxicological effects. Some components (inorganic trace elements) may also have cumulative effects on aquatic organisms.

The inorganic component of concern in leachate is ammonia. Ammonia is toxic to fish and other aquatic organisms and may generate eutrofication.[4] During nitrification of ammonia in surface water, oxygen depletion will occur and may affect the aquatic ecosystem. For freshwater courses, discharge of leachate with high salt concentrations may alter the salinity and thereby affect the aquatic ecosystem. If a body of surface water (e.g., brackish waters) is not sensitive to these effects, however, the byproducts of leachate discharge may not have any adverse environmental impact.

[4] Significant growth of algae in water where nutrients are in surplus, resulting in a lack of oxygen in the water.

6. LEACHATE CONTAINMENT AND COLLECTION

To prevent leachate from emanating in an uncontrolled manner into the surrounding environment, the leachate in a landfill must be contained and collected.

6.1. Leachate containment

A seal/liner system is used to contain leachate. Prices for the different liner types are given in Chapter 8. There are a number of containment option with different installation requirements and varying degrees of leakage (*see Table 2*):

- **Natural liners** of either in-situ clay or clay brought to the site offer a low permeability. The requirements for the thickness of the clay layer and permeability are often set in national regulations and may vary from 0.5 meter–2 meters, with a theoretical permeability of less than 10^{-10}–10^{-9} meters per second (m/sec). The theoretical leakage from this type of liner may be some 5–50 mm of leachate per year.

- **Artificial mineral liners** such as bentonite-enhanced sand spread in thin layers or geosynthetic clay liners (a sandwich with bentonite between two geotextiles or glued to a geomembrane) may achieve theoretical (saturated) permeability of less than 10^{-11} m/sec. The unsaturated permeability may be as high as 10^{-8} m/sec. The theoretical (saturated) leakage may be in the range of 6–10 mm per year. Sodium bentonite is the predominant type of bentonite used as it can withstand chemically aggressive components in leachate better than calcium bentonite.

- **Synthetic liners** of various materials are used, most commonly HDPE[5], LDPE[6], and latex. The thickness of the liners varies from 0.5 mm–2.5 mm. The thickness may be less important for performance than the material's physical, chemical, and biological resistance. The theoretical leakage of leachate through a synthetic liner may be as much as 40 mm per year.

- **Liner combinations** comes in many versions. **Double liner systems** with a leak-detection system may only indicate when leachate leaks through the primary liner. It is now widely accepted that a leak-detection layer between two independent liners simply results in an increased need for leakage control, without increasing the level of security. In contrast, **composite liners**, where two liners of different materials (e.g., a natural clay liner and a synthetic liner) are in direct internal contact, may achieve a significant increase in security. The theoretical leak through a composite liner may be less than 0.2 mm per year.

[5] High Density Polyethylene
[6] Low Density Polyethylene

Liner Type	Theoretical Leakage in mm/year	Sophistication of Installation
Natural clay/ in-situ clay	5 – 50	High demand for local (e.g., road) contractor in-situ clay to install the liner. Price greatly dependent on local availability of clay.
Bentonite mats	6 – 10	Shipment often required; installation may be carried out by local contractor with local labor.
Synthetic liner	~ 40	Shipment often required and highly specialized welding labor and equipment required.
Composite liner (e.g., synthetic/ bentonite matemats)	< 0.2	Shipment often required and highly specialized welding labor and equipment required for the synthetic liner—local labor may install bentonite mats.

Table 2: Overview of Different Liner Types and Their Theoretical Annual Leakage

6.2. Leachate collection

If a landfill is equipped with a containment system, the hydraulic head of leachate on the liner must be controlled by a leachate collection system at the base of the disposal area. A leachate collection system generally consists of the following:

- **Drainage blanket**, consisting of inert (non-calcareous) coarse gravel, typically designed to keep the hydraulic head over the liner below 0.3 meters. With a sufficient thickness (> 0.3 meters), the drainage blanket also serves as a protection layer for the liner system. Artificial drainage grids have been introduced, but have not yet proven adequate, as they are prone to rapid clogging.

- **Drainage elements** (i.e., a means of transporting leachate to the leachate collection point) range from the use of coarse gravel alone to the installation of 300 mm HDPE drainage pipes. Drainage elements surrounded by geotextiles to prevent fine materials from entering the drainage element can develop biological growths and a build-up of chemical precipitates, leading to a reduction in permeability. For this reason, geotextiles should not be used around drains.

- **Leachate collection point**, the lowest point of each cell at the base of the disposal area to which leachate is led by gravity through the drainage system. The collection point may be a shaft

constructed of coated concrete or an artificial material such as fiberglass or HDPE. The collection point is the focus for monitoring leachate and removing leachate from the disposal area for treatment and/or discharge. Removal should be by gravity wherever possible. However, if it is not possible and pumping is required, the pumps must be suitable for pumping agressive liquid. These pumps must be changed at least every five years, and possibly more often.

7. LEACHATE TREATMENT OPTIONS

Wherever leachate is collected, a discharge option must be provided. Most often, leachate requires treatment before final discharge to the environment. Where a municipal sewage treatment plant exists, the preferred discharge of leachate may occur there, providing the plant has the capacity to receive the leachate (i.e., it can meet effluent standards after receiving the leachate). With the expected composition of leachate (*see Table 1 and Annex A*), many municipal sewage treatment plants in developing countries may not be suited to receive the leachate, in particular due to high loading rates of ammonia and chloride and, to a certain extent, organic matter. Large fluctuations in the quantity of leachate may also render certain plants unable to meet basic standards. Where leachate is discharged to a mechanical sewage treatment plant, the treatment will have virtually no effect other than dilution.

In many developing countries, therefore, on-site leachate treatment may be necessary. The main components to be treated in MSW leachate are organic matter, ammonia, and (where the leachate will be discharged to fresh water) chlorides. The degree and type of treatment varies greatly, depending on standards for discharge or, where such standards do not exist, the vulnerability of the receiving water, climatic conditions, and the quality and quantity of leachate generated. A combination of different treatment methods may therefore be required. The three most common treatment methods are:

- ***Physical/chemical treatment***, which primarily includes the addition of simple chemicals followed by mixing, flocculation, coagulation, and settlement before or after other treatment. The physical/chemical treatment primarily reduces suspended solids; precipitates iron, manganese, calcium carbonate, and heavy metals; removes turbidity and color; and removes some of the organic matter. **Air stripping** can be used to remove ammonia by increasing the alkalinity (pH > 10) and aerating the leachate.

- ***Biological treatment***, which is often the most important and most commonly used method to treat leachate for ammonia and organic matter. However, it is not effective in removing inorganic salts. The treatment is predominantly performed by aerobic bacterial degradation of organic matter and nitrification of ammonical nitrogen to nitrite and nitrate. The cheapest and most robust biological process is aeration in lagoons, although lagoons often demand large areas. Activated sludge and rotating biological contractors may also be applied, although these are more sophisticated and entail much higher costs.

- ***Tertiary treatment***, which is often accomplished by highly sophisticated and expensive measures such as **reverse osmosis** and **activated carbon adsorption.** These methods are usually not appropriate in low-income countries. However, less sophisticated methods for the final stages may be achieved by creating **artificial wetlands** (which can also be used for pre-treatment) for nitrification and removal of residual organic matter; or **land treatment** by spraying leachate across sloped grasslands or woods to achieve evaporation, precipitation, oxidation, nitrification, and plant uptake. Fast-growing plants such as bamboo, poplar, or other non-sensitive plants may be used to boost evaporation and removal of nutrients. One disadvantage of the latter two treatment methods is the risk of damaging soil and vegetation, and the possibility of leakage to groundwater. In semi-arid areas and in wet areas with a long dry season, simple **evaporation schemes** may be used to eliminate leachate discharge. However, if there is no pre-treatment, the final residue after evaporation will contain extremely high concentrations of dry organic matter and salts, which may be considered hazardous and therefore difficult and/or expensive to dispose of.

An overview of different treatment options for various components in leachate is given in Annex B.

8. ECONOMIC CONSEQUENCES

The costs of leachate management can be divided among the areas of: a) containment; b) collection; c) treatment; and d) final top cover. All of these depend on several factors, the most important of which are:

- the annual precipitation in the area where the landfill is situated;
- the strategy chosen for leachate management;
- the level of environmental protection (liner system applied and level of leachate treatment); and
- the size of the landfill.

Most landfills are constructed to expand continuously, with expected investments in leachate management systems (primarily liners) for most years during operation. As investment schemes depend heavily on these four factors, the following section provides only unit costs and price ranges.

8.1. Containment

Liner systems for containment are described in Section 6. The minimum size landfill cell feasible to line will be some 2.5–5 ha. The factors influencing the price for liners include: a) shipping distance; b) size of the job; c) market demand in the region; and d) time of year when the liner is to be installed. The price ranges for the various liner types are given in Table 3.

Liner Type	Theoretical Leakage in mm/year	Price Range for Liner Installed [a] in US$/m^2
Natural clay/ in-situ clay	5 – 50	3 – 6
Bentonite mats	6 – 10	4 – 10 [b]
Synthetic liner	~ 40	5 – 8 [b]
Composite liner (e.g., synthetic/ bentonite matemats)	< 0.2	9 – 18 [b]

[a] The price is based on a minimum area of 2.5 ha liner installed.
[b] For certain parts of the world, this price may be significantly higher due to shipping and/or labor costs.

Table 3: Theoretical Leakage and Aproximate Price Range for Installed Liners (1996 price levels)

8.2. Collection systems

Most leachate collection systems can be made from local materials and can be installed by local contractors. The price for leachate collection systems may also depend on the sophistication of the system and specific needs, ranging from less than US$1/m^2 to more than US$4/m^2.

8.3. Leachate treatment

Data available on leachate treatment costs are very scarce. No cost data have been available from developing countries, partly because leachate treatment plants rarely exist and partly because there is little incentive for landfill operators to reveal their leachate treatment costs. The following costs range widely and can therefore only be indicative, reflecting the fact that the available data are very site-specific. Factors likely to affect leachate treatment costs include: a) treatment capacity volume (size of landfill and precipitation in area); b) loading rates of leachate (stage of decomposition of the waste); and c) required effluent standards for the final receiving water body.

The initial investment in a plant that treats 100 m^3/day of leachate (equivalent to 400 mm/year at a 10 ha disposal area) may range from less than US$1.8 million to more than US$5.4 million, depending on the level of sophistication of the plant.

Leachate treatment has relatively high operation and maintenance costs, which constitute 40–60% of the total investment, operation, and maintenance costs per cubic meter. The cost of biological treatment with air stripping, meeting standards for discharge to a robust receiving water, falls in the range of US$6–US$20/m^3.

8.4. Leachate management costs as part of total landfill costs–comparative examples

The magnitude of leachate management investment and operational costs, as compared with all other landfill investment and operational costs, are illustrated with the following examples. The examples assume a 10 ha landfill, which receives 300–350 tonnes per day over a 10-year period. The total capacity of the landfill is 1 million tonnes. Investment and recurrent costs for construction and operation of the landfill are based on 1996 prices from the Philippines. The liner is assumed installed as natural clay. Leachate treatment is assumed as aerobic biological treatment with ammonia stripping. It is assumed that the initial treatment costs (amortized investments and recurrent costs) are US$10/m3 (100% of costs) reduced linearly to zero following the end of aftercare.

The normal aftercare, monitoring, and liability period for a landfill is 30 years. If it is assumed that the landfill can be left unattended 30 years after closure, the total landfill costs will, as indicated in Table 4, range from US$10–US$15/tonne of waste disposed. Leachate treatment costs (investment and operational) constitute 50–67% of the total landfilling costs, as indicated in Figures 2 and 3.

Annual Leachate Generation	Treatment Costs (investment and operational costs)	Liner/Leachate Collection	All Other Landfill Investments	Operation and Maintenance (except leachate treatment)	Total Landfill Costs
200 mm/year	5.1	1.2	2.4	1.4	10.1
400 mm/year	9.9	1.2	2.4	1.4	14.9

Table 4: Leachate Treatment Costs Compared with Total Investment and Operation Costs with a 30-year Aftercare Period (US$/tonne)—An Example

Figure 2

Landfill costs with 30 year aftercare (US/tonne; % of total) 200 mm leachate generated per year

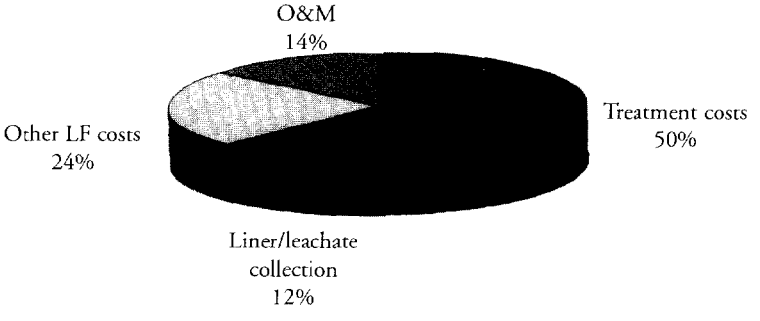
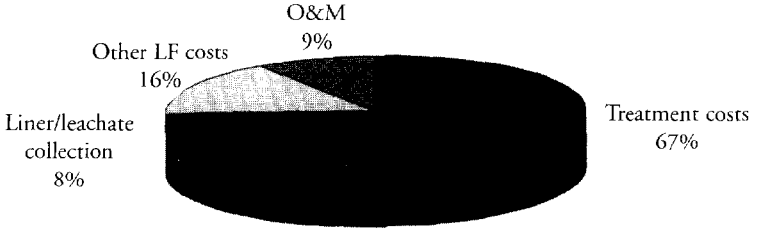


Figure 3

Landfill costs with 30 year aftercare (US/tonne; % of total costs) 400 mm leachate generated per year



It is assumed that leachate will be treated until the pollution potential is at a level where the leachate can be released into the environment with no adverse impacts; under such a scenario, treatment may be required for 150 years or more. Table 5 indicates that the total landfilling costs should then range from US$21 – US$37 per tonne of waste. Leachate treatment may then constitute some 77–87% of the total landfill costs, as illustrated in Figures 4 and 5.

Annual Leachate Generation	Treatment Costs (investment and operational costs)	Liner/Leachate Collection	All Other Landfill Investments	Operation and Maintenance (except leachate treatment)	Total Landfill Costs
200 mm/year	16.6	1.2	2.4	1.4	21.6
400 mm/year	32.4	1.2	2.4	1.4	37.4

Table 5: Leachate Treatment Costs Compared with Total Investment and Operation Costs, Assuming a 150-year Aftercare Period (US$/tonne)—An Example

Figure 4

Landfill costs with 150 year aftercare (US/tonne; % of total costs)
200 mm leachate generated per year

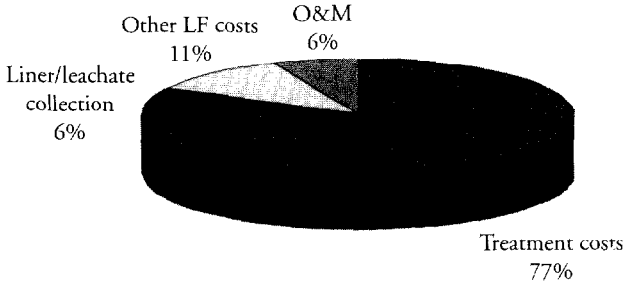
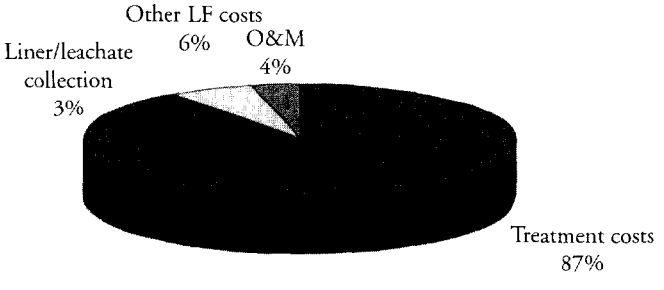


Figure 5

Landfill costs with 150 year aftercare (US/tonne; % of total costs) 400 mm leachate generated per year



It should be emphasized that local conditions will create many uncertainties and variations for these estimates of leachate management costs. It is, however, clear that leachate management constitutes by far the largest investment and operational costs for a landfill and therefore should have a significant influence on the actual tipping fees to be charged for disposal of each tonne of waste. This is true whether the leachate is managed for 30 years or until the leachate reaches direct discharge standards. The choice of leachate management system therefore becomes extremely important during the initial stages of landfill planning and development.

9. CHOICE OF LEACHATE MANAGEMENT SYSTEMS

The choice of leachate management system may be tied to regulations at the national or regional level. In some countries, regulations may allow flexibility as long as the final choice is justified with an in-depth environmental assessment, including serious evaluation of alternatives. Other countries have very specific standards, leaving little flexibility in the choice of which leachate management system to apply.

9.1. Strategy

Leachate management is usually not an issue for landfills in **arid climates** where no excess leachate is generated. For landfills in **semi-arid climates**, where excess leachate is generated occasionally, there may be a need for leachate containment, collection, and treatment/handling. Leachate treatment/handling may simply include storage and evaporation, or the adoption of a more sophisticated approach with recirculation of leachate into already disposed waste, with the intention of stabilizing the waste and evaporating excess leachate during recirculation. However, under some semi-arid climatic conditions, the leachate levels may be low enough to allow for controlled releases into the environment. The assessment in this circumstance should be based on the same considerations as those for long-term leachate management in **wet climatic conditions**, as outlined below.

All landfills subjected to **wet climatic conditions**, where excess leachate is generated, will have an impact on groundwater and/or surface water. These landfills will require some type of leachate containment. If containment is applied, it is imperative that leachate collection and treatment also be part of the short-term leachate management strategy.

Long-term leachate management is necessary when the landfill has reached stable conditions and there is no longer a potential for unacceptable environmental impacts from release of leachate into the environment. The release of leachate into the environment requires that the following conditions be met:

- There is no potable use of groundwater that can be adversely affected downstream from the land fill;
- The dilution potential in the groundwater is sufficient to ensure that the leachate plume entering the receiving surface water does not exceed discharge standards set for the treatment plant that would otherwise treat the leachate and discharge to the same receiving surface water; and
- The total flux of contaminants out of the landfill is environmentally acceptable to the receiving surface water.

Long-term leachate management thus requires extensive knowledge of a) the quantity and quality of leachate generated; b) development and composition of the potential contaminants in the leachate; c) the quality and flow of groundwater downstream from the landfill; d) the potential for dilution and dispersion in the groundwater; and e) the vulnerability of the receiving surface water body.

9.2. Containment

Containment for leachate should always be based on site-specific conditions, primarily the vulnerability of groundwater and, eventually, the receiving surface water body.

In comparison with a single liner (natural or synthetic), a composite liner (two liners internally in direct contact) may reduce the leakage of leachate to groundwater by a factor of 25–50. However, the increased security may double the cost for the liner system. There is often less reason to justify the additional cost if long-term leachate management is taken into consideration during siting of MSW landfills.

For most developing countries, a liner of natural clay (including geo-synthetic clay liners) may be preferable because the installation and construction of these liners often can be provided by local contractors. Synthetic liners, if correctly installed, may provide the same level of security as clay liners. However, installation requires highly specialized equipment and often requires international contractors to carry out the installation. Phased installation, which is required for each cell expansion, may therefore become difficult and more expensive.

9.3. Collection

Where containment of leachate is part of the leachate management system, leachate collection should be provided. To the extent possible, natural materials such as coarse gravel should be used for the drainage blanket and drainage elements of disposal areas. Natural flow (by gravity) of leachate will add to the long-term sustainability of leachate management. Siting of landfills may help to provide natural flow of leachate and should be preferable to pumping.

9.4. Treatment

Treatment of leachate to almost any desired discharge standard is possible. The discharge standards are often receiving water-specific; the volume of leachate varies and the composition of leachate may vary. The required extent of leachate treatment therefore varies. The choice of leachate treatment may be made based on the following key issues:

- The availability of appropriate capacity at the local sewage treatment plant;
- local discharge standards and the vulnerability of the receiving water body;
- the operational capacity for on-site treatment;
- provisions for recirculation, through appropriate operational procedures; e.g., limited if any daily soil cover and avoiding the use of soil with clay content;
- power prices/affordability/willingness to pay for treatment; and
- the area available for construction of an on-site treatment facility.

In general, highly sophisticated treatment methods should not be applied on-site. Regardless of the level of sophistication, leachate treatment is the costliest item in a landfill's budget, even if leachate is

only treated for a 30-year period. As "eternal" leachate treatment is both unrealistic and prohibitively expensive, landfill planners must accept that leachate will eventually be released into the environment. It is therefore advisable to use a treatment system that is simple to operate and that provides some level of treatment even if it fails. In warmer climates, this may be achieved through aerobic biological treatment (e.g., aerated lagoons) followed by finishing and post-treatment in a natural or artificial wetland system before final discharge to the receiving water or groundwater.

Good siting of a landfill therefore becomes a crucial leachate management step.[7] A landfill should be sited where it eventually can be left unattended without any unacceptable environmental impacts. To make the system sustainable over the long term (where energy can no longer be provided), sites with a high degree of natural flow for the leachate are preferable.

9.5. Other leachate management approaches

Other options for leachate management to be considered include:

- ***A final top cover*** may be used to control the volume of leachate generated. A simple top cover with vegetation will reduce infiltration through evaporation. More significant reductions in infiltration may be achieved by applying a low-permeable top cover, such as clay. Reduced infiltration will lessen the annual release of contaminants from the landfill, while at the same time extending the time during which leachate management is required.

- ***Recirculation of leachate*** may have multiple purposes: a) **treatment**, by reducing the load of organic matter in the leachate[8] with virtually no other leachate components affected; and b) leachate **storage**, by building up leachate inside the landfill until treatment capacity is available. In a worst-case scenario, this can lead to disastrous landfill slides. A further goal may be c) leachate **reduction**, by sprinkling and consequent evaporation, which, over longer time periods, may lead to increased concentration and precipitation of salts and may introduce occupational health hazards.

- ***"Flushing"*** by water is one method to increase infiltration and thus boost leachate generation for a period. Flushing washes out salts in the disposed waste, allowing it to more rapidly reach a level where leachate can be released into the environment. This method has not yet been widely applied.

[7] Please refer also to Johannessen, L. M., Bartone, C., "Guidance Note on Approach to Siting of New Landfills," (under preparation, expected early 2000).

[8] See also Johannessen, L. M., "Guidance Note on Recuperation of Landfill Gas for Municipal Solid Waste Landfills," World Bank, Washington, DC, September 1999.

ANNEX A: LEACHATE COMPOSITION FROM GERMAN MSW LANDFILLS

	Average	Range
Acidic phase		
pH	6.1	4.5 – 7.5
BOD_5 (mg/l)	13,000	4,000 – 40,000
COD (mg/l)	22,000	6,000 – 60,000
BOD_5/COD	0.58	
SO_4 (mg/l)	500	70 – 1,750
Ca (mg/l)	1,200	10 – 2,500
Mg (mg/l)	470	50 – 1,150
Fe (mg/l)	780	20 – 2,100
Mn (mg/l)	25	0.3 – 65
Zn (mg/l)	5	0.1 – 120
Methanogenic phase		
pH	8	7.5 – 9
BOD_5 (mg/l)	180	20 – 550
COD (mg/l)	3,000	500 – 4,500
BOD_5/COD	0.06	
SO_4 (mg/l)	80	10 – 420
Ca (mg/l)	60	20 – 600
Mg (mg/l)	180	40 – 350
Fe (mg/l)	15	3 – 280
Mn (mg/l)	0.7	0.03 – 45
Zn (mg/l)	0.6	0.03 – 4
No differences between phases	**Average**	**Range**
Cl (mg/l)	2,100	100 – 5,000
Na (mg/l)	1,350	50 – 4,000
K (mg/l)	1,100	10 – 2,500
Alkalinity (mg $CaCO_3$/l)	6,700	300 – 11,500
NH_4 (mg N/l)	750	30 – 3,000
orgN (mg N/l)	600	10 – 4,250
total N (mg N/l)	1,250	50 – 5,000
NO_3 (mg N/l)	3	0.1– 50
NO_2 (mg N/l)	0.5	0 – 25
total P (mg P/l)	6	0.1 – 30
AOX (µg Cl/l)*	2,000	320 – 3,500
As (µg/l)	160	5 – 1,600
Cd (µg/l)	6	0.5 – 140
Co (µg/l)	55	4 – 950
Ni (µg/l)	200	20 – 2,050
Pb (µg/l)	90	8 – 1,020
Cr (µg/l)	300	30 – 1,600
Cu (µg/l)	80	4 – 1,400
Hg (µg/l)	10	0.2 – 50

*absorbable organic halogen

Source: Ehrig, H-J., "Quantity and Quality of MSW Landfill Leachate" Sardinia, Second International Landfill Symposium, 1989.

ANNEX B: LEACHATE TREATMENT OPTIONS

Treatment Objectives	Main Treatment Options
Removal of degradable organic (BOD)	**Aerobic biological**: Aerated lagoon/extended aeration Activated sludge Sequencing batch reactor (SBR) **Anaerobic biological:** Upflow sludge blanket
Removal of ammonia	**Aerobic nitrification:** Activated sludge Aerated lagoon/extended aeration Rotating biological contractor Sequencing batch reactor Vegetated ditch (Artificial wetlands) **Air stripping**
Denitrification	**Anoxic biological** Sequencing batch reactor Vegetated ditch
Removal of non-degradable organic and color	Lime/coagulant addition Activated carbon Reverse osmosis Chemical oxidation
Removal of hazardous trace organic	Activated carbon Reverse osmosis Chemical oxidation
Odor removal	Hydrogen peroxide
Removal of dissolved iron and heavy metals and suspended solids	Lime/coagulant addition, aeration and setting
Final polishing	**Artificial wetlands** (e.g., reed beds, ponds)
Disinfection	Hypochlorite
Volume reduction/pre-concentration	Reverse osmosis **Evaporation**

Main treatment options in bold may be preferable in many low-income countries.

Source: Adapted from Hjelmar, O., Johannessen, L. M., Knox, K., Ehrig, H.-J., Flyvbjerg, J., Whinter, P., Christensen, T.H., " Management and Composition of Leachate from Landfills" Final report prepared for DGXI, A.4. Waste 92 Contract no.: B4-3040/013665/92. Denmark, 1994.

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College of Agriculture and Natural Resources

Agriculture and Its Relationship to Toxic Dinoflagellates in the Chesapeake Bay

November 27, 1997

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BACKGROUND

Although the evidence is circumstantial and inconclusive at this point, it has been suggested that nutrients lost from agricultural operations through runoff and leaching may be partially responsible for the recent outbreaks of *Pfiesteria*-like organisms in the lower Pocomoke River and several other rivers on the lower Eastern Shore. Nutrients enter water from many sources. Nutrients added to land, however, may represent a significant source of aquatic nutrients. Sewage sludge, septic tank effluent, organic manufacturing waste, and animal manures contain high concentrations of nitrogen and phosphorus. Most of this material is recycled onto land for disposal.

For many years the agricultural community has recognized the need to reduce the loss of nutrients from farmland and has implemented several programs to control nutrient loss. Past efforts, while successful, have focused on reducing nitrogen losses. Recent evidence collected by Dr. Frank Coale, of the University of Maryland College of Agriculture and Natural Resources, demonstrates that some soils of the lower Eastern Shore are highly enriched with phosphorus-to the point where soluble phosphorus may now be lost through runoff. This finding is counter-intuitive to previous scientific thought. A change in control practices to reduce losses of soluble phosphorus represents a major shift in paradigm.

In response to a request by Commission Chair and former Governor Harry Hughes, Thomas A. Fretz, Dean of the College of Agriculture and Natural Resources, brought together a panel of regional experts in nutrient and animal management not only to examine the most current information related to nutrient losses, but to develop a strategy for reducing those losses. While the relationship between the outbreak of *Pfiesteria* and nutrient loading into aquatic systems remains unclear, the agricultural community recognizes the need to take action. Thus, a primary goal of this document is to review current practices and recommend methods for controlling losses of nutrients, especially phosphorus, from agricultural land.

This document contains scientific background information for the comments presented to former Governor Harry Hughes and the Blue Ribbon Commission. To the extent possible, we have attempted to discuss the level of uncertainty and the potential for recommended practices to contribute to reducing nutrient losses--especially soluble phosphorus--from land. In addition, we also discuss the length of time (immediate, short term, long term) required for implementing practices.

OBJECTIVES

The objectives of the Agricultural Scientific Advisory Committee were to:

1. Review current State efforts for controlling losses of nutrients from agricultural land.

2. Review and critically evaluate relevant research on nutrient losses from agricultural land.

3. Recommend practices for reducing losses of nutrients, especially soluble phosphorus, from agricultural land.

4. Identify important scientific questions that limit our ability to recommend effective practices.

APPROACH

A group of 10 scientists, chaired by Dean Fretz, met on October 7, 1997, to review relevant issues and to develop a format for completing the assignment. Review and writing teams were established and charged with drafting this document by October 13, 1997. The Scientific Advisory Committee met twice more to review progress and discuss possible recommendations. This document forms the basis of much of Dean Fretz's verbal testimony.

ISSUES AND SOLUTIONS

Land Application Issues

Nutrients, essential for crop production, come from various sources. Most commonly, farmers obtain nutrients from inorganic commercial fertilizers and from organic sources such as animal manures and biosolids from wastewater treatment plants. Generally, inorganic nitrogen and phosphorus compounds are water soluble and readily available to plants. Most organic nutrient sources must first be mineralized or decomposed to become available to plants, although some may contain inorganic forms of nutrients. Mineralization occurs when moisture and soil temperature are optimal for the growth of microorganisms, which must actively convert these organic forms to inorganic forms.

The movement of nitrogen and phosphorus through soil can differ dramatically. We have long understood that if nitrogen is converted to the highly water soluble nitrate-nitrogen form, and is not used during plant growth, it can move through the soil-water system and be vulnerable to leaching into the groundwater. In contrast, however, the general understanding has always been that phosphorus does not move through the soil-water system. Most phosphorus applied to soils was thought to be readily converted into insoluble forms associated with the iron, aluminum, and calcium ions in the soil system; it was only transported from the field through soil erosion. Recent research, however, demonstrates that soils amended with large quantities of organic or inorganic phosphorus may generate significant amounts of soluble phosphorus that can be readily transported by surface runoff water, even when erosion losses are minimal.

When we use organic sources of nutrients, the ratios of nitrogen and phosphorus do not usually correlate with the crop's actual nutrient needs. When we apply commercial fertilizers, we can tailor the ratios of nitrogen and phosphorus to meet the crop's nutrient needs. If we apply an organic nutrient to meet the nitrogen needs of the crop, what generally happens is we overapply phosphorus. If we apply an organic source to meet the crop's recommended phosphorus needs, organic material application rates are necessarily lower and underapplication of nitrogen occurs. We must then apply additional nitrogen from other sources--generally, commercial nitrogen fertilizers. Thus the challenge is to develop systems that allow the optimum use of organic sources of nutrients while maintaining environmental integrity.

Nutrient losses are associated with water movement. Nutrient loss from the field occurs either through surface water runoff or when nutrients leach down through the soil column. The local hydrology is very important when assessing and controlling nonpoint source nutrient pollution. Local variations in hydrology often preclude a singular approach to nutrient management and control. Rather, we need to develop a range of best management practices (BMPs).

Issues Specific to Nitrogen

Except for legumes, most crops require nitrogen fertilization. We recognize that crop systems are relatively inefficient in using applied nitrogen. When we apply nitrogen as a nutrient, losses do and will occur. Even under natural systems, nitrogen is lost from the soil.

Nitrogen losses, other than by crop removal, occur in the following ways:

a) inorganic and organic forms of nitrogen compounds are lost through surface water runoff and the related movement of particulate organic materials and through soil erosion;

b) soluble compounds (primarily nitrate-nitrogen) leach through the soil-water system, if the nitrogen is not used by the crop or if the nitrogen migrates below the root zone of the crop; and

c) inorganic nitrogen compounds vaporize into the atmosphere.

The purpose of current management systems for limiting nitrogen losses is to increase the efficiency of nitrogen use and to optimize nutrient input. Best management practices for obtaining these goals are: setting realistic yield goals; following recommendations for nutrient needs; increasing the adoption of winter cereal cover crops for water quality enhancement; testing manure and biosolids; applying nutrients at the correct time and using the correct methods; in-season soil and plant tissue testing, such as the pre-sidedress nitrate test for corn; and using efficient crop rotations. While many farmers are following these BMPs, their wider adoption should further limit nitrogen losses from the agroecosystem. Adopting nitrogen-management BMPs, however, may increase operation and crop production costs.

Issues Specific to Phosphorus

Growing crops generally use small amounts of phosphorus relative to nitrogen. However, either through the continued application of commercial phosphorus fertilizers, the disposal of animal waste, or the calculated application of animal manures and biosolids through nitrogen-based nutrient management plans, the phosphorus content of many Maryland soils may become very high. While most of this phosphorus is generally in rather insoluble forms, losses may occur.

Phosphorus losses from agricultural systems, other than by crop removal, generally occur through the following pathways associated with surface water runoff:

a) particulate losses either as phosphorus absorbed into soil particles or organic materials, and/or

b) soluble inorganic and organic phosphorus compounds.

Little phosphorus is lost through leaching into groundwater. However, recent research indicates that the potential for leaching losses may exist on sandy soils with high phosphorus contents.

To control phosphorus losses, we generally recommend the BMPs currently in use to limit soil erosion. These are, among others: no-till farming, contour/strip cropping, grass waterways, buffered streams, and related structural controls.

Soil testbased recommendations for applying phosphorus take into account the phosphorus status of soils and, in turn, only recommend phosphorus applications on soils with low or moderate phosphorus contents. Farms using inorganic fertilizers as a nutrient source generally follow these recommendations. Farms using organic materials (e.g., manures) as a fertilizer source often overapply phosphorus.

Before we understand the true ecological impact of phosphorus overapplication, we must establish a better understanding of the potential for phosphorus transport from the field to surface water and the ultimate bio-availability of the transported phosphorus. Without a broad agroecosystem approach to minimizing phosphorus enrichment of surface waters, singularly focused management practices may be counterproductive. For example, no-till farming is extremely useful in reducing soil erosion and particulate phosphorus movement. However, it may actually increase the amount of soluble phosphorus lost in surface water runoff. Other options for managing phosphorus include BMPs that reduce soluble phosphorus losses through plowing previously no-till or minimum-tilled soils and incorporating manure upon application. We must, however, carefully weigh these suggestions against the increased risk of soil erosion and particulate losses and the increased cost of labor, fuel, and machinery.

Nitrogen- and Phosphorus-based Nutrient Management Planning

Realizing that both nitrogen and phosphorus are nutrients of concern for the Chesapeake Bay ecosystem, the current University of Maryland/Maryland Department of Agriculture Nutrient Management Program concentrated on managing nitrogen, which at the time the program was initiated in 1989, appeared to be the more problematic of the two nutrients. Also, the established thinking maintained that controlling soil erosion losses through soil conservation practices would also control phosphorus losses.

We have made progress in reducing both nitrogen and phosphorus inputs. These reductions are the result of reductions in both nitrogen and phosphorus application recommendations and farmers' realization that manures should be used as a nutrient source rather than treated as a waste disposal problem. How these reductions have translated into water nutrient-loading reductions remains speculative, since monitoring of nonpoint source inputs has been minimal.

The emphasis on nutrient management planning has been based, in most part, on optimizing nitrogen inputs. On farms where animal manure is the major or only fertilizer source, this system has allowed phosphorus to be overapplied. Nitrogen-based plans have made use of animal wastes, been cost effective, and reduced nitrogen application to land. However, this has resulted in overapplication of phosphorus. These applications frequently occur on soils with an already extremely high phosphorus content caused by past applications of manure of a much higher phosphorus content. This overapplication increases the potential for phosphorus to move from farmland to nearby water, especially from land maintained under reduced-till cultivation.

It is possible to develop nutrient management plans for farms by using either nitrogen or phosphorus as the limiting nutrient. Nitrogen-based nutrient management planning may continue to be the most effective one for many farms. Farms where nitrogen-based plans are recommended need to implement improved monitoring of pre-, post-, and in-season soil nitrate-nitrogen levels. Farmers need to calibrate and use existing and evolving nitrogen management tools--corn stalk nitrate-nitrogen testing, corn leaf chlorophyll meter analysis, preplant soil nitrate-nitrogen testing, and in-season soil nitrate-nitrogen testing for wheat. Overall, crop nitrogen management will need to be intensified. Winter cereal grain cover crops should be effectively managed on the maximum possible acreage following summer annual crop production where land would otherwise remain fallow.

Many instances will require using nutrient management plans based on phosphorus. These plans would be environmentally sound and should begin to limit both phosphorus and nitrogen enrichment of the associated water bodies. However, this type of planning would have a very serious and potentially expensive impact on farms that generate or use animal manures. Currently, most of the soil in certain regions of the State has a high phosphorus content. We should limit additional phosphorus applications, therefore, to the rather small land base with soil of a medium to low phosphorus content. Thus, phosphorus-based planning would require a land base that is unavailable locally, requiring increased on-farm storage capacity, transporting and marketing manures far from the point of generation, and developing alternative manure uses. Phosphorus-based plans would also require importing additional synthetic fertilizer nitrogen sources, since the nitrogen available from manure under a phosphorus-based plan would likely be inadequate for most crops.

To illustrate the magnitude of the impact of phosphorus-based nutrient management plans, we examined the effect of the impact on a 1,000-acre farm (500 acres of corn, 500 acres of soybeans) with four poultry houses producing a total of 660,000 broilers per year. We compared the economic consequences of the current situation to the case where the farmer is unable to apply poultry litter to the corn. The farmer must bear the cost of disposing of the litter and purchasing and applying inorganic nitrogen and potassium for crop production to replace the nitrogen and potassium from the poultry litter. Based on 1996 corn budgets and prices, the net income per acre of corn is $161.75 if poultry litter application is allowed. This decreases to $116.77 if the poultry litter application must be replaced by inorganic fertilizer. Thus, the net income from corn production to our hypothetical farm is reduced by $22,500.

The cost of transporting the litter out of the region is based on a trucking cost of $0.16 per ton per mile (these costs may be lower for long hauls, but we do not consider handling costs to load, unload, and store the litter). For illustrative purposes, we assume the litter is transported 200 miles. Our hypothetical farm produces 792 tons of litter. Thus, the transportation costs are $25,340. If the litter is sent to a region

where inorganic phosphorus is currently being applied, the litter can be used in place of the inorganic phosphorus, saving that grower $45 per acre for phosphorus, nitrogen, and potassium. This translates into a value of $28.40 per ton of litter. The market value of the litter, therefore, lowers the net cost of disposal by 89 percent to $2,850. Combined with the increase in corn production costs, the net income loss to the poultry/corn/soybean operation is $25,350 per year.

Adopting phosphorus-based nutrient management plans would affect the farm enterprise through increased operation and crop production costs. Major changes to current farming practices such as adopting phosphorus-based nutrient planning and the resulting import of fertilizer nitrogen and export of manures, can have a significant impact on the profitability of any farming system. Thus, it is imperative that if phosphorus-based planning is to become a reality, it be carefully targeted to the locations at greatest risk for phosphorus losses in farmland runoff and be implemented over a period of time sufficient to enable the farming community to adjust to this very different set of management circumstances. This will also require finding alternative uses and remediation strategies for existing animal waste. Any recommendations given consideration must be done concurrently with full development and implementation of cost-effective alternatives.

Land Application Solutions

1. Establish a State-supported program to facilitate implementation of the use of winter cereal cover crops for water quality and erosion control.

2. Establish a scientific advisory group to advise Maryland's Department of Agriculture on the judicious and incremental transition to phosphorus-based nutrient management planning.

3. Target phosphorus-based nutrient management plans to fields with a very high potential for phosphorus loss, as defined by fields with "excessive" soil-test phosphorus levels that also have a "very high" Phosphorus Index rating.

4. Provide the cost share for the cost difference between following a phosphorus-based plan and a nitrogen-based plan.

5. Reliably evaluate the effectiveness of acceptable agronomic practices to control or minimize surface and subsurface losses of nutrients from agricultural systems. This would include impacts of tillage. Incorporation through tillage needs to be used more intensively in land application of poultry litter to distribute the phosphorus levels in greater soil mass. With conventional no-till and minimum-till systems to reduce erosion, we have concentrated the phosphorus levels in the upper soil horizons. Plowing in litter at the time of application would distribute the phosphorus more uniformly throughout the root zone, and could to some degree reduce runoff of soluble P.

Animal Feeding and Management Issues

Feeding and managing animals affects manure quality and, potentially, the extent of nutrient losses from the farm. Two issues are important: improved management decreases the nitrogen and phosphorus content of manure; and improved management decreases feed nutrient requirements and cropland acres

needed to support a given level of animal and plant production.

A significant portion of manure nitrogen and a highly variable portion of manure phosphorus are lost from the farm once they are applied to crops. Therefore, reducing the amount of manure nutrients directly decreases the subsequent nutrient losses. Ultimately, it is much more logical to reduce the production of a waste product than to find alternative solutions once the waste has been generated.

Poultry Industry

Approximately 625 million birds (three billion pounds of meat) are raised each year on the Delmarva Peninsula. Assuming these flocks are fed according to the National Research Council recommendations, 53 million pounds of manure N and 22 million pounds of manure P will be excreted per year. Poultry farming therefore represents one of the significant sources of nutrients that have a potential impact on water resources.

The phosphorus content of plant-source feed ingredients is considered to be only 30 to 40 percent available for use by nonruminants such as poultry and swine. The remaining 60 to 70 percent is bound in the form of phytate phosphorus, which requires the enzyme phytase to break down to forms of phosphorus available to animals. Nonruminants do not secrete the phytase enzyme and therefore any dietary phosphorus in the phytate form will pass through the intestines and into the animal's manure. Typically, because of this low availability of feed grain phosphorus, supplemental inorganic phosphorus, such as dicalcium phosphate, is added to nonruminant diets to meet the animal's available phosphorus requirement for maximal production.

Poultry manure accumulates in the bedding of the poultry house during a period of several years. As manure accumulates, so do the nutrients. Nitrogen levels stabilize after the third flock, but phosphorus and potassium levels continue to increase with each flock produced. The clean-out and storage of bedding from poultry houses represent a significant source of the nutrients applied to land. Complete housecleaning occurs every two to three years, but manure sheds were designed for 180- to 210-day storage. Manure disposal, whether on the land, or via an alternative technique, must be available for much of the year.

Poultry Solutions

1. Phytase is the enzyme that converts unavailable organic phosphorus into a form that poultry can use. Commercial phytase is presently available. Phytase can be sprayed onto poultry feed, reducing the amount of inorganic phosphorus added to feed. The use of phytase should reduce the amount of phosphorus in manures. A program is needed to increase the use of phytase in feeds and therefore reduce supplemental phosphorus feeding and manure P output by 20 to 25 percent. Further research is needed to examine phytase stability and feed processing to improve its use in the field. Technology problems involve uniformity of application to pelletized feed but should be worked out quickly. When implemented, the technology will have immediate impacts.

2. Commercially available low phytic acid corn will be available in the year 2000. Use of this corn in feed mixtures should reduce the amount of phosphorus added to feed. This results in a concurrent

reduction in the phosphorus content of manure. Programs could be established to encourage the use of low phytic corn as soon as it is available to farmers. This technology will likely result in some increased grower cost at the time of seed purchase. Any seed with intrinsic properties that must be identified and maintained in the marketplace will cost more. Also, there will be increased costs to the feed mill operators, as the grain will require identity preservation in the marketplace before being used to make feed. Impacts are likely to be significant.

3. Long-term research is needed to develop ways to improve the efficiency of nutrient use by broilers and layers.

4. Programs could be established to encourage whole house year round use of litter treatments--for example, iron-based products to stabilize manure phosphorus into environmentally inactive forms.

Alternative Uses of Manure

Issues

Many of the lower Eastern Shore counties have inadequate cropland available for efficient utilization of manure phosphorus, therefore, alternative disposal options are needed. It is critical to develop alternatives to the common practice of applying manure to land. Ideally, it is important to transport nutrients to areas of the state or region where soils do not contain excessive concentrations of phosphorus and phosphorus inputs are necessary for optimum crop production. Distribution is not so much limited by lack of available technology but rather by the economics of transporting manure long distances.

Composting can be defined as a controlled decomposition of organic materials. By providing a proper environment through a proportional allocation of ingredients, microorganisms will, over time, change the form of the organic material into a stable humiclike material. During composting, temperatures will reach 145 to 160 °F if the proper environment is provided. Common pathogenic organisms will be destroyed at the elevated temperatures. In addition to pathogen control, composting organic material such as broiler litter will minimize the odor from the product. Deodorizing broiler litter through composting will enhance its market potential.

After the compost is cured, it can also be applied to the land as a soil amendment, or it can be enhanced with either inorganic or organic nutrient sources to develop a value-added compost of known and marketable fertilizer value. Specialty products could be developed using broiler litter compost as a base. Pelletizing the compost will increase its density and assist with proper application.

Once pelleted, compost could be readily used in a variety of markets including golf courses, nursery crops, container-grown nursery crops, sod production, lawns and gardens, landscaping, silviculture, and median strips and ramps on freeways. This will, however, require considerable market analysis and development.

Burning manures is another disposal option. In the early 1980's, Delmarva Power burned broiler litter in their Indian River Generation Facility at Millsboro, Delaware. Nitrous oxide (NOx) emissions and a constant litter supply were problems. However, the facility has been upgraded since then and NOx emissions may not be a problem, today. Even today, however, litter supply remains a problem. The BTU

value of broiler litter is about 6,800 BTU's per pound at 30 percent moisture in a large fluid bed burner. The ash content for broiler litter is approximately 11.3 percent. This shows a large volume and weight reduction. Burning raw litter in small on-farm furnaces has presented some problems such as slag formation because of incomplete combustion, odors, particulates, and loading difficulties. Poultry litter as a fuel is used in the United Kingdom. This practice is an environmentally acceptable alternative to land filling. Therefore, broiler litter as a feedstock in the production of heat or electrical energy is feasible and technically possible.

Adequate litter supply, proper combustion equipment and a use for the energy generated have to be considered in designing a proper system. One fluidized bed furnace is under construction on the Lower Shore for use in a poultry production operation. A third boiler unit is being designed for the Eastern Correctional Facility by Maryland Environmental Services, who are considering broiler litter as a fuel feedstock. Ash from burning broiler litter may be used to make artificial soils, as a component of a fertilizer mix, or added to feed as an ingredient for poultry rations.

Solutions

1. Expand efforts to distribute manure to areas of low manure availability and low soil phosphorus concentrations. This remains a possibility, however, we must raise a concern with regard to biosecurity. Transporting raw litter will be viewed as a possible means of spreading avian pathogens and for that reason composting or heat-treating is preferred. Transport to new markets out of the production region may offer up the greatest opportunity for reducing the application of poultry litter on land already high in phosphorus.

2. Conduct research to examine the potential for the burning and energy recovery of manures. Burning efficiency must be investigated, as well as the availibility of the residual phosphorus and metals in the ash.

3. Research on composting, post-compost processing, and compost market potential is also needed.

4. Develop pelletized fertilizer products. A number of products are currently in the marketplace that are manufactured from poultry waste (e.g., Nutri-Mate from Alabama which is shipped to Australia).

Remediation Issues

Regardless of any other measures either to reduce the phosphorus content of manure or to find alternative uses for manure, these actions will have no effect on soils that already have high phosphorus levels and are at risk for phosphorus losses to the surrounding environment. However, remediation of high-phosphorus soils has never been implemented on a large scale. Hence, no standard practices exist; the concepts presented are theoretically viable. As previously noted, many soils within Maryland, and especially the Pocomoke drainage basin, contain very high phosphorus levels, which may restrict any further land application of manure. Remediating phosphorus-enriched soils includes: removing phosphorus from the soil; or phosphorus immobilization by making the soil phosphorus unavailable for transport to surface waters without removing it from the soil. Remediation assumes that subsequent farming practices will add phosphorus in levels that plants can utilize, precluding phosphorus buildup of

the scale seen on the Lower Shore in the past 60 years.

The only practiced removal technique is through the use of crops. Growing crops without adding phosphorus provides an income source (or at least a net cost decrease) and leaves the soil undisturbed. It will require the application of nitrogen, which is an expense to the farmer, as well as having a pollution potential of its own. Plants accumulate phosphorus at differing rates and the current wisdom indicates that the dominant factor in phosphorus removal is the amount of plant biomass removed and exported. Hence, independent of phosphorus accumulation rates, an alfalfa crop removes more phosphorus than a soybean crop because the entire plant is removed when alfalfa is harvested. An important advantage of mining phosphorus through crop removal is that it can be accomplished entirely by the individual farmer and requires no additional equipment. Estimates vary, but returning a field with excessive soil phosphorus levels to agronomic levels of phosphorus could require from 20 to as much as 100 years, depending on the initial phosphorus level and the crop used for mining the phosphorus. This time frame is of concern and the economic implications to the individual farmer have not been explored.

A novel variation of mining phosphorus through crop removal, called phytoremediation, is being explored for removing inorganic contaminants from soil. Phytoremediation, or "Green Remediation" uses unusual plants that have developed the ability to concentrate high levels of elements, usually heavy metals, in plant tissue. The primary limitation of phosphorus phytomining is no one has identified reliable phosphorus hyperaccumulators. Although it is likely such hyperaccumulators exist in nature, a concerted effort is first required to identify and collect these plants. These plants could be grown on soils with a high phosphorus concentration. After several years, the soil phosphorus would be low and nutrient losses would not be of concern.

Conceptually, a single chemical treatment of phosphorus-enriched soils would be sufficient to change the relatively mobile forms of phosphorus to stable and far less mobile mineral form. The impact of this treatment on soil pH, nutrient availability, and the ecosystem in general is unknown. There are technological questions concerning application uniformity and treatment efficiency. The economics are not documented.

Tillage, a second immobilization technique, would place the phosphorus-rich surface soil well below the surface and out of reach of surface runoff water, hence effectively stopping surface transport of phosphorus. The best scientific judgment is that leaching through the soil column would not be significant. The phosphorus would be below the surface but still within the root zone, enabling it to be taken up by plants over time. One major concern is that the soil that would be brought to the surface must be equally good for crop production as the original surface soil or it would create a permanent liability for the farmer. Furthermore, the subsurface soils that will be brought to the surface must also be low in phosphorus or tillage will have no impact. The ramification of this is that extensive soil testing of the deeper soil would be necessary prior to performing tillage, and it would not be suitable on some farms. The efficiency of tillage with respect to phosphorus in runoff is unknown, but this reclamation technique holds promise.

Off-field treatment, it can be argued, is not true remediation, but it does have the potential to result in an immediate change in the quality of surface-drained waters. A portion of the lower Eastern Shore is drained by field drainage ditches. Collector drainage ditches then flow into the closest tributary. In soils where soluble phosphate is reaching drainage ditches, it is feasible to chemically precipitate the phosphorus out of the drainage water prior to allowing water to enter a major surface tributary. The ditch treatment concept is to partially fill a length of collector ditch with crushed limestone and create a flow-through treatment system.

Calcium carbonate dissociates and eventually combines with phosphorus to form a very stable form of

phosphorus. This approach requires activity only in the collector ditches and permits the continued water table depression that the ditches were designed for in the first place. The cost of crushed limestone is minimal compared to many reclamation options. However, we do not understand the chemical dynamics well enough to predict where the phosphorus would precipitate out. This is a realistic tool to explore for some site-specific applications.

Solutions

1. No remediation technique is understood well enough to endorse without qualification. Therefore, we need to carry out research and demonstration projects before wide scale implementation of remediation practices.

2. Tillage holds potential for acceptance because suitable equipment is widely available and the concept is easily understood. Tillage is appropriate for flat land that has been in no-till cultivation for many years and where manure has been used long enough that soil phosphorus concentrations are high. Intensive soil testing must be done to the depth which the soils will be plowed.

3. Where onsite remediation is not viable, we should consider offsite effluent treatment if water quality sampling indicates significant levels of phosphorus are entering the water system from drained land. However, prior to implementation, additional work is required to assess many of the physical parameters related to this process.

4. There are too many gaps in our knowledge of chemical immobilization and phytoremediation for us to recommend them today. Further, many uncertainties related to the effects on soil chemistry require resolution before we can recommend this practice.

ECONOMIC CONSIDERATIONS

Each of the strategies for reducing nutrient losses from agricultural land will have an economic consequence that must be addressed. The technical solutions presented here each have costs that may be borne by the grower, the integrator, or the consumer. There may also be administrative and enforcement costs associated with a particular action. Additionally, industry and consumers will adjust their behavior when faced with higher costs and new regulations. When poultry prices rise, consumers adjust by purchasing other products. When farming costs rise, there are a number of possible responses. If farming costs rise in one particular region relative to other growing areas, the competitiveness of that type of farming in the region is eroded. Farmers may make less profit, switch to other crops, or the convert the land from farming into some other use. This scenario is clearly contrary to protecting environmental quality and the Governor's Smart Growth Initiative in particular.

The higher cost of poultry production that would result from many scenarios may result in less poultry production in Maryland. A study of the economic impact of the Delmarva poultry industry conducted by the Department of Agricultural and Resource Economics estimated that just a 4 percent decline in Maryland's poultry production would result in an annual loss of $74 million in economic output in the state, a $29 million loss in personal income and business profits, and a loss of 880 jobs. Some of this loss would be ameliorated as the economy adjusted over time.

APPENDIX

Committee Members

Dr. Thomas A. Fretz, Dean of the College of Agriculture and Natural Resources, is also Director of the Maryland Agricultural Experiment Station and the Maryland Cooperative Extension Service. Dr. Fretz chaired the committee and will make the oral presentation to the Blue Ribbon Panel.

Dr. J. Scott Angle is Associate Director of the Maryland Agricultural Experiment Station and Professor of Natural Resource Sciences. During the 1980's he conducted extensive research on leaching and runoff losses from agricultural systems, including turfgrass, no/conventional-till corn, tobacco, and forested areas. More recently his research has focused on the remediation of contaminated soils.

Dr. James C. Wade, Associate Director of the Maryland Cooperative Extension Service and Professor of Agricultural Resource Economics, provides valuable input on the economics of recommendations affecting the poultry industry.

Dr. Lewis E. Carr, Department of Biological Resources Engineering, is internationally recognized for his work on the beneficial use of poultry wastes. He was responsible for developing and promoting the composting of dead birds-a practice many farmers on the Delmarva Peninsula follow.

Dr. Gary K. Felton, of the Department of Biological Resources Engineering, has been involved in agricultural nonpoint pollution research and extension since 1977. He has particular expertise on the impact of agriculture on groundwater resources.

Dr. Frank Coale is Professor of Soil Fertility in the Department of Natural Resource Sciences. Recognized as an expert on phosphorus in soils, he was one of the first investigators in the United States to note that soils fertilized excessively with phosphorus could lose soluble phosphorus through runoff.

Dr. Tom Simpson, Professor of Soil Science and Coordinator of Chesapeake Bay Agricultural Programs, is an expert on the impact of agricultural practices on water quality. He is also a leader in policy issues relating to agriculture and water quality.

Dr. Richard Weismiller is Chair of the Department of Natural Resource Sciences and Landscape Architecture. He is also a Professor of Soil and Water Resource Management with expertise in nonpoint pollution from agricultural lands and nutrient management.

Dr. Russell Brinsfield is Center Director for the Wye Research and Education Center. He is recognized for his work in nonpoint pollution, especially on the use of cover crops to reduce nutrient losses from agricultural land.

Dr. Jeannine Dennis-Harter, Professor of Poultry Nutrition at the Eastern Shore Campus, is an expert in the nutrition of poultry, with special emphasis on reducing nutrient inputs into feed.

Dr. Richard Kohn is Professor of Animal and Avian Sciences. His area of specialization is animal nutrition, and he has developed an internationally recognized model for demonstrating the flow of nutrients on dairy farms.

Dr. Doug Lipton, Professor of Agricultural and Resource Economics and Director of the Maryland Sea Grant Program, is a state leader on issues related to aquatic resources and an expert in assessing the impact of agriculture on Chesapeake Bay fisheries.

Dr. Jim Hanson, Professor of Agricultural and Resource Economics and Assistant Director for the Cooperative Extension Service, is recognized as an expert in farm economics and the impact of farming practices on revenue and profit.

Dr. Doug Parker, Professor of Agricultural and Resource Economics, is involved in agricultural production and water quality issues.

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