ABSTRACT

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The objective of this research is to identify and evaluate parameters that could be useful for defining the end of the post-closure monitoring period, or that time at which the landfill is stable with respect to environmental emissions. Parameters evaluated in the report include leachate composition, leachate production, and gas production. In addition, the evaluation of leachate composition and quantity is combined to evaluate surface water impacts associated with the release of leachate to surface water.

To determine if a landfill has reached stability, the following data are needed: the concentrations of ammonia and BOD in the leachate, leachate production rates, and gas production rates. In order for the landfill to be declared stable, the leachate should not deplete oxygen resources, exert toxicity, or increase eutrophication if it is released to surface water. In addition, gas production rates should be low enough so that their emission does not represent a significant release of a greenhouse gas or cause an odor nuisance or explosion hazard. Ultimately, the technical criteria for landfill stability evaluated in this research should be applied to a full-scale landfill to test the usefulness of the proposed approach.

A CRITICAL EVALUATION OF FACTORS REQUIRED TO TERMINATE THE POST-CLOSURE MONITORING PERIOD AT SOLID WASTE LANDFILLS

By

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BIOGRAPHY

Alexandria Pettway Rooker was born in April, 1976 in Raleigh, NC. She grew up in Hickory, NC but returned to Raleigh for her education. She received her Bachelor's of Science in Environmental Engineering from North Carolina State University in 1999. Her interests in surface water protection and restoration led her to pursue her Master's of Science in Civil Engineering as well.

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A Critical Evaluation of Factors Required to Terminate the Post-closure Monitoring Period at Solid Waste Landfills

Table of Contents

List of Tables

List of Figures

I. Introduction

In 1997, approximately 217 million tons of municipal solid waste (MSW) were generated in the U.S., with approximately 55% of this waste buried in landfills. While diversion of waste for recycling and composting are increasing, landfills will remain a significant part of MSW management for the foreseeable future (US EPA, 1999a). In 1991, the U.S. EPA promulgated regulations governing the disposal of MSW in landfills (40CFR Part 258). The regulations, commonly referred to as Subtitle D of RCRA, specify a number of aspects of landfill design and operation including requirements for site closure and post-closure monitoring. The regulations specify that after a landfill has received a final cap the owner must monitor the site for 30 years unless this period is extended by the regulatory agency. Once this post-closure period is over, the state will take responsibility for the site unless it is determined that the site is not stable. However, a technical definition of stability is required to make this determination. Once environmental stability is defined, it could be used to shorten or extend the 30-year requirement.

Subtitle D landfills are required to have a liner system that includes a flexible membrane liner (FML) and a compacted clay layer (CCL). Final covers may be no more permeable than the liner, so they often consist of the same materials. After decades of operation and post closure, deterioration of both the landfill cover and liner may occur so that containment of leachate and gas cannot be insured in perpetuity. Before a landfill receives its permit for closure and monitoring is no longer required, the site should be stable so that failure of either the cover or liner does not result in deleterious impacts.

In order for leachate to safely be released to the environment, several criteria should be met: (1) the leachate should not contain high concentrations of organics, ammonia, or heavy metals; (2) the leachate should not exhibit toxicity to the organisms in the ecosystem. In addition, the landfill should not be releasing gas that may cause contamination of ground water supplies, odor nuisances, or explosions.

The objective of this paper is to identify and evaluate parameters that could be useful for defining the end of the post-closure monitoring period, or that time at which the landfill is stable with respect to environmental emissions. Parameters evaluated in the report include leachate composition, leachate production, and gas production. In addition, the evaluation of leachate composition and quantity will be combined to evaluate surface water impacts associated with the release of leachate to surface water.

II. Landfill Leachate Composition

There are numerous reports summarizing leachate composition. In this section, the objective is to synthesize these reports to evaluate the composition of leachate from well-decomposed refuse. Landfill leachate typically contains heavy metals, organic matter, and inorganic ions such as ammonia, phosphate, and sulfate (Christensen et al., accepted for publication). Its composition depends on several factors including water infiltration, the state of waste decomposition, the original waste composition, codisposal practices, and site hydrology (Reinhart and Grosh, 1998; Pohland and Harper, 1985). Though it is impossible to predict the stability of a landfill from any one of these factors, chemical indicators may be used to estimate the stage of waste decomposition. The concentrations of dissolved organics, ammonia, and heavy metals are reviewed in the following sections with a focus on concentrations typically present in well-decomposed or stable refuse.

A. The Concentration of Dissolved Organic Matter in Leachate

There are three ways to measure organic content including the total organic carbon (TOC), the chemical oxygen demand (COD), and the biological oxygen demand (BOD). As waste is degraded, the concentrations of all three measures decrease (Reinhart and Grosh, 1998). However, BOD decreases faster, and its concentration can approach zero. COD and TOC will remain in the leachate and is comprised of humic and fulvic compounds (Christensen et al., 1994). Table 1 presents typical concentrations of BOD and COD found in leachate from well-degraded wastes.

Table 1. Organic and Ammonia Concentrations (mg/L) in Landfill Leachates from Older, Methanogenic Landfills[1, 2]

BOD	COD	BOD:COD	Ammonia as N	Reference
5.7 - 1100 (50, na)	76 - 6997 (50, na)	--- (na, na)	12.4 - 1571 (50, na)	Range of concentrations from 21-30 year old, German landfills (Krumpelbeck and Ehrig, 1999)
290 (50, na)	1225 (50, na)	0.24 (50, na)	445 (50, na)	Average concentrations from 21-30 year old, German landfills (Krumpelbeck and Ehrig, 1999)
44 (35, 120)	320 (85, 550)	0.11 (35, -)	110 (104, 190)	Average concentrations from old, Danish landfills (Kjeldsen and Christophersen, accepted)
39 (1, na)	398 (1, na)	0.10 (1, na)	233 (1, na)	A sample composition at Sandsfarm Landfill (Robinson, 1995)
11 (1, na)	190 (1, na)	0.06 (1, na)	282 (1, na)	A sample composition at Bishop Middleham Landfill (Robinson, 1995)
38 (1, na)	517 (1, na)	0.07 (1, na)	399 (1, na)	A sample composition at Odsal Wood Landfill (Robinson, 1995)
1.0 (1, na)	53 (1, na)	0.02 (1, na)	42.6 (1, na)	A sample composition at East Park Drive Landfill (Robinson, 1995)
2.5 (1, na)	64 (1, na)	0.04 (1, na)	29.8 (1, na)	A sample composition at Marton Mere Landfill (Robinson, 1995)
20 - 550 (na, na)	500 - 4500 (na, na)	--- (na, na)	30 -3000 (na, na)	Range of concentrations reported in the literature for methanogenic leachate (Ehrig, 1988)
180 (na, na)	3000 (na, na)	0.06 (na, na)	750 (na, na)	Average concentrations reported in the literature for methanogenic leachate (Ehrig, 1988)
33 (3, 5.57)	75 (3, 7.23)	0.44 (3, 0.05)	0.5 (3, 0)	Average values from the last three days of a laboratory experiment with leachate recycle (Pohland, 1975).
34 (3, 9.54)	224 (3, 23.3)	0.15 (3, 0.026)	1.07 (3, 0.40)	Average values from the last three days of a laboratory experiment with leachate recycle and pH adjustment (Pohland, 1975).
36 (3, 4.04)	194 (3, 17.6)	0.19 (3, 0.035)	3.5 (3, 0.91)	Average values from the last three days of a laboratory experiment with leachate recycle, pH adjustment, and nutrient addition (Pohland, 1975).

1. The values represent actual composition results, averages, or ranges of concentrations (or ratios) from leachate composition data. The landfills sampled are older, methanogenic landfills. The laboratory experiment data (Pohland, 1975) comes from stabilized waste.

2. Number of samples and standard deviation are given in parentheses (n, std. dev.)

The dissolved organics in landfill leachate include intermediates of refuse decomposition, humic and fulvic acids, and compounds leached from the buried waste. Trends in leachate composition during decomposition have been reviewed previously (Barlaz et al., 1990). In the leachate representative of well-decomposed refuse, the decomposed intermediates will have been consumed, and humic matter will dominate.

1. Organic Indicators of Waste Decomposition

Several parameters have been suggested as indicators of the stage of waste decomposition: pH, oxidation reduction potential, COD, BOD, BOD/COD, alkalinity, heavy metals, ionic strength, and the sulfate to chlorine ratio (Pohland and Harper, 1985; Chian and DeWalle, 1977; Reinhart and Grosh, 1998). The most commonly used indicators are BOD, COD, and the ratio of the two. In any given sample, the COD must be greater than or equal to BOD.

Though no standard values exist for these indicators, Table 2 summarizes some suggested ranges that correlate with the phase of decomposition. Several authors assign a BOD:COD value of less than 0.1 to stable leachate (Reinhart and Grosh, 1998; Pacey, 1999; Pohland and Harper, 1985; Ehrig, 1988). Pacey (1999) has suggested that the BOD should be less than 100 mg/L and the COD should be less than 1000 mg/L. This pattern is consistent with the data summarized in Table 1 where the composition of leachate from older landfills containing well-decomposed refuse generally exhibits a BOD:COD of less than about 0.1. There is no apparent explanation for the higher values in the laboratory work though it should be noted that gas production was inferred from gas concentrations and not measured directly. Thus the refuse in the laboratory study may not have been as well-decomposed as otherwise assumed.

A low BOD:COD ratio should be considered a necessary but insufficient condition to insure leachate stability because (1) refuse in an active state of methane production may have a low ratio and (2) leachate from refuse in the acid phase that percolates through well-decomposed refuse will have a low BOD:COD ratio even though the landfill is not stable.

Table 2. The Use of Organic Indicators to Define Waste Decomposition[1]

COD and BOD are expressed in mg/L.

Phase:	Acid	Methanogenic	Stabilized	Reference
Organic Indicator:				
BOD COD BOD/COD	>10,000 --- > 0.7	20% of COD < 2000 ---	--- --- ---	Robinson, 1995
BOD COD BOD/COD	--- 3×10^{-4} - 3×10^{-7} ---	--- < 2000 ---	--- --- < 0.1	Reinhart and Grosh, 1998
BOD COD BOD/COD	--- --- ---	--- --- ---	--- --- 0.02 - 0.013	Pohland and Harper, 1985
BOD COD BOD/COD	--- --- > 0.4	--- 3000 - 4000 < 0.1	--- --- ---	Ehrig, 1988
BOD COD BOD/COD	--- --- ---	--- --- ---	< 100 < 1000 < 0.1	Pacey, 1999

1. These values are suggested ranges to define stable leachate.

B. Ammonia in Landfill Leachate

Another long-term pollution problem associated with landfill leachate is ammonia nitrogen. Although it is associated with biodegradable substrate, there is no mechanism for its degradation under methanogenic conditions (Robinson, 1995; Burton and Watson-Craik, 1998). Several researchers have attributed the toxicity of landfill leachate to organics and ammonia (Wong, 1989; Assmuth and Pentillae, 1995; Cheung et al., 1993; Lambolez et al., 1994; Clement et al., 1997). Ammonia also exerts an oxygen demand and contributes nitrogen that can lead to eutrophication. Some have suggested that ammonia is the most significant long-term pollution problem at landfill sites (Robinson, 1995; Krumpelbeck and Ehrig, 1999; Christensen et al., 1994; Christensen et al., accepted for publication).

Ammonia-N concentrations for a number of landfills containing decomposed refuse are presented in Table 1. Krumpelbeck and Ehrig (1999) report that ammonia concentrations did not show a significant decrease even 30 years after closure based on a study of 50 German landfills. Ehrig (1988) reports that there is not a significant change in ammonia concentrations from the acidic to methanogenic phase, and that the average value is 750 mg-N/L. According to Krumpelbeck and Ehrig (1999) and Kjeldsen and Christophersen (accepted for publication) average ammonia concentrations for old landfills range from 100 to 450 mg NH_3-N/L. Table 1 reports ammonia concentrations from these and other studies.

C. Heavy Metals in Landfill Leachate

Metal species are of concern when they are biologically available at toxic levels to organisms in the receiving ecosystem (Smith et al., 1999). This category includes not only free metal ions in solution, which are usually present in low concentrations, but also organic complexes, inorganic complexes, and metals bound to colloids (Christensen et al., accepted for publication). The objective of this section is to summarize information on heavy metal concentrations in the leachate produced from landfills that contain well-decomposed refuse.

There have been a large number of studies in which researchers have reported metals concentrations from full-scale landfills, test cells, and laboratory studies. The ultimate conclusion of these studies is that metals concentrations are not of concern in landfill leachate (Christensen et al., accepted for publication; Robinson, 1995; Reinhart and Grosh, 1999; Revans et al., 1999; Kjeldsen and Christophersen, accepted for publication; Christensen et al., 1994).

As presented in Table 3, most heavy metal concentrations in landfill leachate are at or below the US drinking water standard which are an exceedingly strict standard for leachate. Only two of the ten studies reported Cd concentrations greater than one order of magnitude higher than the US Drinking Water Standard; two studies also detected Cr concentrations in excess of the drinking water standards. One study showed Zn concentrations that minimally exceeded the standard, 5.31 mg/L compared to 5.0 mg/L. There were no reports of Cu in excess of its drinking water standard (1.3 mg/L). The US Drinking Water Standard for Pb is 0, and all leachates were reported to contain some amount of Pb, ranging from < 0.005 mg/L to 0.19 mg/L. A federal standard for Ni has not been established, but the concentrations found in the 10 studies ranged from 0.0036 mg/L to 0.348 mg/L. No metals data were available for US landfills in which the refuse was well decomposed. However, in developing effluent guidelines for leachate treated at the landfill site, the US EPA elected not to include guidelines for metals as there was no evidence that metals concentrations were ever a problem (US EPA, 2000). Metals are subject to both adsorption and precipitation, and it has been suggested that these attenuation mechanisms are responsible for the relatively low metal concentrations in leachates (Christensen et al., accepted for publication; Revans et al., 1999; Flyhammar and Hakansson, 1999; Christensen et al., 1994).

1. Processes Affecting Metals Concentrations

Four processes have been reported to control heavy metal concentrations in landfill leachates: complexation, oxidation-reduction reactions, sorption, and precipitation (Christensen et al., accepted for publication; Revans et al., 1999; Flyhammar and Hakansson, 1999; Christensen et al., 1994). A brief description of each process is presented in this section.

Christensen et al. (accepted for publication) define sorption to include adsorption, absorption, surface complexation, and ion exchange. Adsorption and absorption occur when the molecules of a gas, liquid, or dissolved substance and a particle "stick" together (Lu et al., 1985). This process generally decreases metals concentrations in landfill leachate. Divalent metal cations tend to favor sorption onto negatively charged sites such as colloidal particles, calcite, clay minerals, organics, and oxides of Fe, Mn, Al, and Si (Christensen et al., accepted for publication).

Complexation is the combination of metal ions with non-metallic ligands by covalent bonds; it is capable of increasing metal solubilities (Lu et al., 1985). In general, a large fraction of the dissolved metals content is complexed with organics compared to the fraction present as the free metal ion (Christensen et al., accepted for publication; Jensen et al., 1999; Jensen and Christensen, 1999). For example, the humic-like substances formed from waste decomposition can serve as ligands for metal complexes (Reinhart and Grosh, 1998).

Precipitation occurs when a metal species falls out of solution as a solid. Sulfides and carbonates are capable of forming precipitates with Cd, Ni, Zn, Cu, and Pb. Occasionally, phosphates and hydroxides will also precipitate metals (Christensen et al., 1999). Hydroxide precipitates form at pH at or above neutral which is typically the case in methanogenic leachates (Reinhart and Grosh, 1998).

Sulfide is formed from sulfate during waste decomposition in landfills, and sulfide precipitation is often cited as an explanation for low concentrations of heavy metals (Christensen et al., accepted for publication; Christensen et al., 1994). Even small concentrations of sulfides will precipitate heavy metals except for Cr, which does not form an insoluble sulfide compound (Christensen et al., accepted for publication). However, Cr does tend to form insoluble precipitates with hydroxide (Christensen et al., accepted for publication; Revans et al., 1999). While carbonates are abundant in landfill leachate, the solubilities of metal carbonates are generally higher than those of metal sulfides (Christensen et al., accepted for publication).

Table 3. Heavy Metal Concentrations in Leachate[a]

Study	1	2	3	4	5	6	7	8	9	10	11
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, accepted for publication).
2. Average leachate concentration in 20 German landfills in the methanogenic phase (Christensen et al., accepted for publication).
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., accepted for publication).
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).
11. National primary drinking water regulations, USA (US EPA, http://www.epa.gov/OGWDW/wot/appa.html).

--- Data not available

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

2. The Effect of Oxidizing Conditions on Metals Concentrations

Once refuse is completely decomposed under anaerobic conditions and the majority of the degradable carbon is consumed, it is theoretically possible that a landfill will gradually become aerobic as air diffuses into the site and oxygen is not consumed in the oxidation of organic matter. To our knowledge, there are no reports of a site turning aerobic, and it is expected that this may only occur in near geologic time. However, in considering long term landfill stability, this issue must be considered as a number of biological and chemical reactions will occur that may affect the mobilization of metals.

Several authors express concern that future oxidation of a landfill could result in increased heavy metals concentrations in leachate (Reinhart and Grosh, 1998; Flyhammar et al., 1998; Martensson et al., 1999). There are three mechanisms by which metals concentrations could increase in an aerobic landfill including (1) oxidation of metal sulfides to metal sulfates that are generally more soluble, (2) increased complexation capacity of oxidized humic acids relative to reduced humic acids (Martensson et al., 1999), and (3) increased metals dissolution due to a pH decrease, which could occur as sulfides are produced.

There have been a limited number of studies on metals concentrations in decomposed refuse that has become aerobic. To test the effects of changing conditions on leachate composition, Flyhammar et al. (1998) used four experimental test cells that were constructed in 1975 on top of an old landfill. The test cells were originally used to monitor changes in leachate composition in a leachate-recycle reactor. In 1993, Flyhammar et al. began to sample the leachate from the anaerobically degraded waste and to collect data on Fe, Mn, Zn, Cu, Pb, Cr, Ni, and Cd. Table 4 presents metals concentrations in the leachate in 1975 compared to 1993. Each concentration decreased by a factor of 5 or more except for Cu.

Table 4. Leachate Concentrations Reported by Flyhammar et al., 1998[1]

	Concentrations mg/L	
Metal	1975 (23)	1993 (1)
Zn	10	2.2
Cu	0.07	0.04
Pb	0.2	0.02
Cr	0.25	0.01
Ni	0.5	0.05
Cd	0.015	0.006

1. The concentrations represent the mean leachate concentrations during the year. The number of samples is given in parentheses next to the year.

To quantify the amount of reactive metal species in the twenty-year-old waste, the samples were boiled in concentrated HNO_3 for two hours. Only 5% of Pb, Cr, Ni, and Cd were released, while less than 15% and 50% of Cu and Zn were released, respectively (Flyhammar et al., 1998). Thus, acid-extractable metals remain after 20 years, but their release to leachate does not appear to increase.

Revans et al. (1999) studied the effects of changing redox conditions on heavy metal concentrations. Six test columns, 500 mm in length and 100 mm in diameter, were constructed with previously decomposed, anaerobic, domestic waste to which Cd, Zn, and Cr were added. Once anaerobic conditions were reestablished in the columns, three of the six were converted to aerobic conditions by aeration with moist air. Water was added weekly to all six reactors to produce leachate that was collected and analyzed, but not recirculated. No significant increase in metals concentrations was detected despite the heavy load of metals applied initially. In fact, less than 1% of each metal was leached by any test column, anaerobic or aerobic. Although metal sulfides would have been oxidized by the aerobic conditions that were established, processes such as sorption, carbonate precipitation, and hydroxide precipitation apparently retained the metals (Revans et al., 1999). In a similar experiment, Flyhammar and Hakansson (1999) reported no evidence of Zn or Fe sulfide dissolution upon aeration though changes in solid species of Zn and Fe did occur.

To test the effects of aerobic conditions on landfill leachate, Martensson et al. (1999) conducted an experiment similar to Revans et al. (1999). Samples of degraded waste from a codisposal landfill in Sweden were placed in reactors and treated with either nitrogen or air to establish anaerobic or aerobic conditions, respectively. After 82 days, leachate was produced by adding 100 mL of distilled water to the waste followed by centrifugation. Aeration did result in a decrease in pH from 8.52 to 8.18. An increase in Cd, Ca, Cr, and Zn concentrations did occur in aerobic waste, but Al, Fe, and Mo concentrations decreased (Martensson et al., 1999).

Metals in landfills are generally assumed to be bound by sulfides. However, the landfill sampled by Martensson et al. (1999) was reported to contain only enough sulfur to bind 5% of metals present which could explain why the results differed from those reported by Revans et al. (1999). Thus, other mechanisms such as complexation, carbonate precipitation, and hydroxide precipitation apparently attenuated the metals as aeration did not result in a significant metals release. Another potential explanation for the difference between the results of Martensson et al. and Revans et al. is that the bottom of the columns used in Revans' work may have remained anaerobic.

In conclusion there is little evidence to date to indicate that the concentrations of heavy metals will increase as a landfill becomes aerobic though the results of two studies on this issue are not consistent. Processes such as sorption, precipitation, and increased cation-exchange capacity are believed to inhibit the metal's mobility (Martensson et al., 1999) though additional data on metal transport in refuse would be desirable.

3. Summary

In summary, theory suggests that metals could be released if a landfill becomes aerobic. Evidence of this has only occurred in some laboratory experiments, while others indicate that the problem is insignificant. To our knowledge, there are no reports in the literature of a landfill becoming aerobic and field studies are required to identify any such landfills. Given the conflicting experimental evidence on the effects of aeration on heavy metals, additional research is recommended to conclusively address the potential for higher metals concentrations in leachate after long periods of time.

D. Toxicity Testing of Landfill Leachate

The objective of this section is to identify and compare available toxicity tests and their applicability to monitoring landfill leachate. Such tests may be useful as part of a testing program to determine whether landfill leachate can be considered stable. In the case of landfill leachate, stability should infer that it can be safely released to the environment. Leachate should not deplete oxygen sources, adversely effect the ecosystem of the receiving body, or emit toxic chemicals.

To date, there are no federal requirements that landfill leachate undergo toxicity testing before it is released to the environment. Bioassays have typically been used as a research tool rather than a regulatory device. Research has shown, however, that many leachates do contain toxic and sometimes carcinogenic compounds (Baun et al. 1999, 1998; Brown and Donnelly, 1988; Christensen et al., accepted for publication). Additional studies have shown that leachate from a municipal solid waste landfill can be more toxic than leachate from hazardous waste landfills (Brown and Donnelly, 1988; Clement et al., 1996; Schrab et al., 1993). The higher toxicity may be attributed to the mixture of compounds present in a municipal landfill, as opposed to the uniform waste streams that a particular hazardous waste landfill will normally receive. Even though large scale co-disposal is no longer practiced, small quantity generators such as small businesses and households do continue to dispose of hazardous chemicals in municipal landfills (Brown and Donnelly, 1988).

Releasing toxic leachate into the environment may potentially contaminate groundwater supplies, harm aquatic ecosystems, and transfer toxic compounds into other media such as air and soil. Chemical analyses can identify the presence of specific compounds, but the identification is not enough to assess the potential danger (Baun et al., 1999; Christensen et al., accepted for publication; Clement et al., 1996; Cheung et al., 1993). In addition, chemical analyses only detect a small percentage of the toxic compounds present (Baun et al., 1999; Galassi et al., 1988; Bihari and Batel, 1994). Bioassay toxicity tests that measure effects on organisms can detect toxicity even when the contaminants are not identified by chemical analysis. They reflect the cumulative and synergistic effects of all the compounds (Baun et al., 1999; Christensen et al., accepted for publication; Lambolez et al., 1994).

This section reviews general considerations for bioassay tests and the organisms that are typically used. Studies in which bioassays were used to measure the toxicity of landfill leachate are then summarized. This is followed by a discussion of future research requirements for assessing the toxicity of landfill leachates.

1. General Principles of Bioassays

Bioassays use organisms to measure the potential adverse effects of pollutants on the environment. They are capable of measuring not only organism death, but also physiological, metabolic, and reproductive changes. Fish, crustaceans, bacteria, and algae are among the most common organisms used for testing complex environmental samples such as landfill leachate for toxicity (Christensen et al., accepted for publication). Many authors suggest using a battery of tests with organisms from different trophic levels to increase toxicity detection (Baun et al., 1999; Christensen et al., accepted for publication; Lambolez et al., 1994; Maciorowski and Clarke, 1980; Clement et al., 1996; Rand, 1980; Schrab et al., 1993; Cheung et al., 1993). Bacteria are typically used for genotoxicity testing. Genotoxicity should be analyzed if downstream ground water supplies are used for human consumption (Baun et al., 1999).

Most bioassays report toxicity as LC50 or EC50, the lethal or effective concentrations that affect 50% of the population (Rand, 1980). A lower number for the LC50 or EC50 means a leachate is more toxic relative to another sample because a lower concentration will show lethal or adverse effects (Baun et al., 1999). Results may also be expressed as toxic units, 1 / (LC50 or EC50). The advantage of using the toxic unit is that a higher value signifies a higher toxicity.

Different types of tests are available to assess different environmental impacts. Acute toxicity tests expose organisms to at least four different concentrations of a sample for 24 to 96 hours to determine the concentration that causes death in some fraction of organisms. These tests may be conducted under static or flow through conditions. The static tests are easier to perform but have a characteristic disadvantage: the toxicant may volatilize, degrade, or react with metabolic products. Thus, the concentration of a toxicant may change with time. For flow-through conditions, water is only cycled through once, so the concentrations of toxicant and dissolved oxygen remain constant (Rand, 1980).

There are several limitations to bioassay testing. First, organisms with special ecological needs should not be used if these requirements cannot be met in the laboratory. For example, test organisms may build up a tolerance to a toxicant and could pass this acclimation on to future generations. Therefore, some organisms should be isolated from pollutant exposure for breeding purposes. Second, it is often difficult to distinguish organism death from immobility. Immobile organisms may recover, and other organisms may die after the test is complete (Maciorowski and Clarke, 1980).

A third limitation to acute bioassays is that in the environment, organisms are usually exposed to sublethal concentrations due to dilution of the water body. Because acute tests measure lethal concentrations, they will not reflect chronic or latent effects due to lower concentrations. Chronic tests expose the organisms for their entire life cycle and measure effects on hatching, growth, reproduction, and survival. Chronic tests use the flow through method and test a minimum of four concentrations (Rand, 1980).

Once the toxicity of a complex mixture has been established, it may be difficult to determine which compound or compounds are contributing to the toxicity. Fractionation divides the sample into groups of similar compounds. Separate bioassays can then be performed to identify the responsible fraction. A toxic group can either be analyzed for chemical constituents or refractionated and tested first (Parkhurst, 1982).

An additional limitation of bioassays is that results may reflect harmful interactions between test organisms and test conditions rather than toxic organic compounds (Baun et al., 1999). For example, the high conductivity of landfill leachate may contribute a significant amount of toxicity to test organisms and may mask the toxicity of organic chemicals. One solution is sample fractionation by solid phase extraction. This method extracts organic chemicals onto a resin while other constituents such as inorganic ions remain in the effluent. The organics are removed from the resin with acetone, which is then evaporated from the sample with either nitrogen gas or air.

Toxicity tests can then be conducted on the organic fraction separately (Baun et al., 1999). If ions are the only contributor to toxicity, then dilution may suffice.

2. Frequently Used Bioassay Toxicity Tests

Bioassay toxicity tests generally use aquatic invertebrates, bacteria, microalgae, fish, and higher plants to estimate the effects of a contaminated waste stream on ecological habitats and humans. They have proven useful for testing wastewater treatment plant effluents, leachate and hazardous wastes, groundwater, and oil drilling waste fluids (Bulich, 1984). The California State Pollution Control Laboratory uses the acute 96 hr, fathead minnow test to measure the toxicity of an extract of solid hazardous waste. If the waste exhibits a 96 hr LC50 of less than or equal to 750 mg/L, then the substance is considered toxic. Another laboratory reportedly has designated a 15 minute Microtox (bacteria) EC50 or a 96 hr fathead minnow LC50 of 750 mg/L or less as toxic (Bulich, 1984). Some municipalities use bioassays to monitor the toxicity of industrial waste streams prior to treatment at wastewater plants (Anderson et al., 1984).

Bailey et al. (2000) used *Ceriodaphnia dubia* (aquatic invertebrate) and *Selenastrum capricornutum* (algae) for whole effluent toxicity tests of sewage treatment plants in Australia. The toxicity to these organisms was attributed to pesticides and ammonia. Fifteen of the 18 treatment plant effluents exhibited acute or chronic toxicity to *Ceriodaphnia dubia*, and 2 exhibited decreased growth rates of *Selenastrum capricornutum*. Vlaming et al. (2000) used *Ceriodaphnia dubia*, *Selenastrum capricornutum*, and fathead minnows (*Pimephales promelas*) for whole effluent toxicity tests in California. They reported that toxicity in the waste water was due to organophosporous insecticides and ammonia. Goodfellow et al. (2000) used *Ceriodaphnia dubia*, *Daphnia magna* (aquatic invertebrate), and fathead minnows to test the ion toxicity of 2900 test solutions. They found that *Ceriodaphnia dubia* was more sensitive than *Daphnia magna* and fathead minnows to all the salt solutions. Rutherford et al. (2000) used rainbow trout (*Oncorhynchus mykiss*), *Daphnia magna, Vibrio fischeri* (bacteria), *Ceriodaphnia dubia,* and *Selenastrum capricornutum* to test the toxicity of landfill leachate after pretreatment in a wetland treatment system. Their results show that the tertiary treatment provided by the wetlands reduced effluent toxicity.

Table 5 ranks the Microtox, *Daphnia magna*, fish, duckweed, and algal bioassays according to cost, simplicity, and sensitivity. While two reports have shown duckweed to be among the most sensitive test species, tests with luminescent bacteria and *Daphnia* are more commonly used. This is justified by their relative cost, their simplicity, and reasonably good correlations between these assays and assays with higher organisms. While the use of algae in bioassays is less common, it does allow an assessment of photosynthetic activity, which is otherwise neglected.

Table 5. Bioassay Comparisons of Cost, Simplicity, and Sensitivity[1]

Cost	Simplicity	Sensitivity
Microtox	Microtox	Duckweed
Daphnia magna	*Daphnia magna*	Algae
Algae	Algae	Microtox
Duckweed	Duckweed	*Daphnia magna*
Fish	Fish	Fish

1. Cost is ranked least expensive to most expensive, simplicity is ranked easiest to most difficult, and sensitivity is ranked from most sensitive to least sensitive.

3. Genotoxicity

Genotoxic compounds, or mutagens, cause DNA mutations to either the chromosome or the gene. If these mutations cause cancer, then the compounds are classified as carcinogens. Usually the initial compound is not genotoxic: it becomes genotoxic only after it has been metabolized by some organism (Zahn et al., 1994). After conversion, the compound will either be released to the environment or bioaccumulated (Bihari and Batel, 1994). If the test species are not capable of metabolizing the compound into a toxic form, then the toxicity will not be detected. If other organisms in the ecosystem are able to biologically transform the compound, then toxicity will be exerted. Therefore, if a bioassay species is not capable of metabolizing a chemical into the mutagenic or carcinogenic compounds, an enzyme such as liver extract must be added to the bioassay (WHO, 1985).

Bacterial genotoxicity assays are the most widely used because the organisms give consistent results and are easy to alter genetically. In various experiments, the Ames test, which is a three-day bioassay that uses four strains of the bacteria *Salmonella typhimurium*, has shown a 90% agreement with known carcinogens and noncarcinogens. The 10% uncertainty can be alleviated by using a battery of tests that will detect damage to both chromosomes and genes (WHO, 1985).

4. The Applicability of Toxicity Bioassays to Landfill Leachate

Several authors have demonstrated the toxicity of landfill leachates to bioassay organisms (Baun et al., 1999; 1998; Brown and Donnelly, 1988; Christensen et al., accepted for publication; Clement et al., 1996; Schrab et al., 1993; Wong, 1989; Assmuth and Penttilae, 1995; Atwater et al., 1983; Cheung et al., 1993; Rutherford et al., 2000).

Lambolez et al. (1994) reported that leachate toxicity was dependent on ammonia, alkalinity, and COD. Clement et al. (1997) suggested that ammonia especially affects duckweed and daphnids, and that alkalinity enhances ammonia toxicity and may even exert direct toxicity to daphnids. Wong (1989) attributed tilapia toxicity to ammonia and organics. Assmuth and Penttilae (1995) report that in general, toxicity correlated with chloride content, ammonia, and hardness. A study by Atwater et al. (1983) asserts that zinc is more toxic to *Daphnia* than fish, and that *Daphnia* are less sensitive to pH toxicity. Cheung et al. (1993) attribute the algal toxicity of the leachate to organic compounds and ammonia. Ammonia, organic matter, and heavy metals found in leachate may be toxic to fish, *Daphnia*, and algae (Wong, 1989).

5. Summary

Much research is still needed in the area of leachate toxicity testing. Standard methods need to be developed that will incorporate the general qualities of leachate into variables such as ion concentrations, alkalinity, and pH. For bioassays where feeding is acceptable, standard methods need to be developed for the concentration and frequency of food and nutrients.

There is a lack of information on the aquatic invertebrates *Thamnocephalus platyurus* and *Ceriodaphnia dubia.* More testing is needed to compare their ion and alkalinity tolerances and organic compound sensitivities to *Daphnia*. Similar tolerance tests are also needed for luminescent bacteria and algal species since these organisms are widely used in bioassays. Marine organisms may be the most tolerant to ion and alkalinity toxicity, and their sensitivity to landfill leachate should also be investigated.

Sample fractionation can facilitate the determination of which constituents in leachate are responsible for toxicity and point to the most effective treatment options. This technique may be the only way to mechanistically evaluate a complex mixture such as leachate.

Finally, the use of bioassay data in evaluating environmental releases must be established. For instance, which assays should be used to test landfill leachate? What is the maximum EC50 concentration for each assay that can be safely released? If a leachate sample exceeds the EC50, what procedures will be followed? How will dilution into the receiving water body be taken into consideration? Once a leachate passes the toxicity bioassays, how often should it be monitored? These questions must be answered before bioassay testing can be used in landfill management.

Despite the additional work that is needed, several conclusions may be drawn:

- The Microtox assay has proven to be relatively simple and cost effective. It has shown strong correlations with other organisms such as fish and has been found more sensitive than tests with *Daphnia magna*.
- The algal assays are more complicated and expensive, but could be a strong indicator of leachate effects on photosynthetic organisms and are generally more sensitive than bacteria (Microtox) or daphnids.
- The relative sensitivity of fish bioassays does not justify the associated increase in cost.
- If a landfill leachate is discharged to a potential source of drinking water, then a genotoxicity test such as the Ames test should also be conducted.
- Because leachate generally contains high concentrations of ions, alkalinity, and ammonia, the sensitivity of the bioassays to these factors must be determined.

- Research is required to understand how to relate bioassay test data to the release of leachate with low concentrations of organic compounds and ammonia.

III. Methods to Estimate Leachate Production Rates From Landfills After Installation of the Final Cover

Both the quantity and composition of leachate must be defined to predict the impact of a leachate release on groundwater or surface water systems. The objective of this section is to estimate leachate production from a traditional landfill after placement of the final cover. It should be noted that the data presented in this section are based on traditional landfills. These data could only be applied to a bioreactor landfill after all free water remaining in the waste has been drained. Once estimates of leachate production rates have been developed, they may be used in concert with leachate quality data for contaminants such as BOD, COD, and ammonia to analyze the environmental impact of a leachate release during and after the post-closure monitoring period.

Several methods to estimate leachate quantity are presented and compared in this chapter. The first method uses published data on the quantity of leachate present in leachate collection systems and in leachate detection systems underlying a liner system. These data were used to estimate leachate collection efficiencies for a number of liner configurations. Additional studies that have provided estimates of the number of defects per area of liner were also reviewed. These data were then used to calculate flowrates through liner systems based on an assumed head on liner.

A. Flowrates through Landfill Liners

1. Typical Flowrates and Liner Efficiencies

Liners or liner systems may be composed of one or more of the following: a flexible membrane liner (FML), a geosynthetic clay liner (GCL), or a compacted clay layer (CCL). A Subtitle D liner, which is the basis for the estimates developed here, is a composite liner consisting of compacted clay and a FML. Under Subtitle D, the final cover may be no more permeable than the liner system. Thus, a final cover typically includes a vegetative layer, a drainage layer, and a composite liner. Water that is not removed by evapotranspiration or the drainage layer passes through the cover, percolates through the waste, and becomes leachate. A leachate collection and removal system (LCRS) is used to remove most of the leachate. The liner system below the LCRS protects the environment. However, some leachate may permeate this liner and enter either the leachate detection system (LDS), if present, or the environment. Landfills that use a LDS are referred to as double lined landfills because there is typically a secondary liner below the LDS.

Gross et al. (1997) studied a database that included 194 cells at 54 double lined landfills. The primary components of the double liner systems that were studied included a FML, FML/GCL, or FML/CCL. The authors defined the apparent efficiency E_a of the liners as

$$E_a\% = (1 - (\text{LDS flowrate} / \text{LCRS flowrate})) * 100\% \qquad \text{Eq. 1}$$

and listed representative values for various liners as presented in Table 6.

Table 6. Liner Efficiencies (Gross et al., 1997)

Liner Type	Efficiency %	Representative Value
FML	97.2 - 99.9	99, n = 15
FML/GCL	99.3 - 100	99.9, n = 24
FML/CCL	na[1]	99.9, estimated value

1. Gross et al. (1997) suggest that consolidation water may interfere with liner efficiency calculations because it contributes excess water to the LDS, so the efficiency calculation is not applicable.

Bonaparte et al. (accepted for publication) studied 31 double-lined cells at 14 landfills; all but four contain MSW. All 31 have FML primary liners, and the secondary liner is either FML or FML/CCL. Only 6 cells received final cover, each with a FML cap. They state that the efficiencies can be as high as 99%, but typically range from 90% to 99%. FML's should not be used alone if an efficiency of 90% is not acceptable. Rather, they should be supplemented with suitable clay as typical for Subtitle D landfills (Bonaparte et al., accepted for publication).

Table 7 presents the average monthly LCRS and LDS flowrates for the 31 landfill cells during three periods: i) initial cell filling, ii) active cell filling, and iii) post closure (Bonaparte et al., accepted for publication). These data are for cells with FML primary liners and from sites with no groundwater intrusion. Sites with and without QA/QC are included. Periods are defined as i) initial - the first few months of cell filling when there is not enough waste to impede precipitation flow directly into the LCRS, ii) active - cell filled with waste and daily and intermediate covers, iii) post closure - after the cell has received its final cover. The sites with 99.99% efficiency in period i, the maximum efficiency, 100% efficiency in period ii, and 99.71% efficiency in period three are three different sites that have a FML primary liner and a FML/CCL secondary liner. All three had a QA/QC program. The site with the lowest efficiency in period i, 83.08%, was a MSW site with a CSPE (chlorosulfonated polyethylene) primary liner, a PVC/CCL secondary liner, and no QA/QC. The site with the lowest efficiency in period ii, 61.3%, was a MSW site with QA/QC that had a FML primary liner and a FML/CCL secondary liner. The site with the lowest efficiency in period iii, 2.8%, had a CSPE primary liner, a PVC secondary liner, and no QA/QC.

Table 7. Average Monthly LCRS and LDS FlowRates From 31 Cells with FML Primary Liners (Bonaparte et al., accepted for publication)

Period	LCRS flowrate lphd (gal/ac/d)	LDS flowrate lphd (gal/ac/d)	Efficiency %
i	1475 - 111,031 (158 - 11,869)	4 - 2144 (0.43 - 229)	83.08 - 99.99
ii	103 - 16,224 (11 - 1734)	0 - 1603 (0 - 171)	61.3 - 100
iii	317 - 1306 (34 - 140)	1 - 328 (0.11 - 35)	2.80 - 99.71

Bonaparte (1995) reports that average monthly leachate flows in the LCRS were up to 3400 lphd (363.5 gal/ac/d) for double lined landfills during the active period. Three years after closure and placement of a FML cover, the average was 70 lphd (7.5 gal/ac/d). The flowrate data for FML/GCL liners is based on a study of 28 cells at 7 landfills with up to 83 months of monitoring data. All of these cells had QA/QC, and the data are presented in Table 8. They do not include mean apparent efficiencies for these cells, only LDS flowrates. However, they suggest that FML/GCL liners can achieve 99.9%

efficiency but will typically range from 99% to 99.9% (Bonaparte et al., accepted for publication).

Table 8. LDS Flowrates for Double Lined Landfills with a FML/GCL Liner (Bonaparte et al., accepted for publication)

Period	LDS Flowrate lphd (gal/ac/d)
i	0 - 290 (0 - 31)
ii	0 - 11 (0 - 1.2)
iii	0 - 2 (0 - 0.2)

The FML/CCL and FML/GCL/CCL liners are represented by 13 cells at 9 landfills with up to 121 months of monitoring data; all have QA/QC. LDS flowrates are given in Table 9.

Table 9. LDS Flowrates for Double Lined Landfills with a FML/CCL or FML/GCL/CCL Liner (Bonaparte et al., accepted for publication)

Period	LDS Flowrate lphd (gal/ac/d)
i	10 - 1400 (1.1 - 150)
ii	0 - 370 (0 - 39.6)
iii	5 - 210 (0.5 - 22.4)

This report also presents leachate generation data, which is expressed as LCRS flowrate (lphd). Table 10 presents the LCRS flowrates along with leachate as a percent of rainfall for various geographic locations in the US (Bonaparte et al., accepted for publication).

LCRS flowrates decreased by a factor of 4 one year after closure, and by one order of magnitude 2 to 4 years after closure. Six years after closure, LCRS flowrates ranged from 5 - 1200 lphd (0.5 - 128.3 gal/ac/d) with a mean of 180 lphd (19.2 gal/ac/d). Nine years after closure these flowrates were negligible. These values are for the closed landfills in the study and are not specific to a certain liner type (Bonaparte et al., accepted for publication).

Table 10. LCRS Flowrates and Percent Precipitation for MSW Landfills, by Geographic Location (Bonaparte et al., accepted for publication)

Period	Geographic Location	LCRS Flow[1] lphd; [gal/ac/d]	Percent Precipitation[1]
i	NE SE W	1050 - 39,900 (10,200); [112 - 4260 (1090)] 1480 - 43, 700 (10,400); [158 - 4670 (1110)] no data	4 -160 (39) 5 - 157 (33) no data
ii	NE SE W	41 - 17,700 (3,530); [4.4 - 1890 (377)] 300 - 10,900 (2,930); [32 - 1160 (310)] 55 - 110 (83)[2];[5.9 - 11.8 (8.9)]	0.1 - 54 (13) 1 - 23 (8) 0.5 - 1 (0.7)[2]
iii	NE SE W	55 - 680 (400); [5.8 - 73 (43)] No data No data	0.2 - 3 (1)

1. Values are presented as range (mean).
2. Values only represent two cells from one landfill.

Tables 6 through 10 present field data from several studies; many do not specify whether or not a Subtitle D liner was used. The study that does focus on sites where a FML/CCL was present had LDS flowrates that ranged from 5 - 210 lphd (0.5 - 22.4 gal/ac/d) in the post closure period (Table 9). Liner efficiencies are not given with this data (Bonaparte et al., accepted for publication). By comparison, in New York State, where MSW landfills must have a primary composite liner and a secondary composite liner, the state requires notification if the LDS flowrate exceeds 20 gal/ac/d (Phaneuf and Vana, 2000). Studies with FML primary liners suggest that achieving liner efficiencies of 99% is possible although efficiencies of 90 - 99% are more reasonable. A sample calculation assuming 99% efficiency in the cover and liner follows.

Liner Efficiency Calculation

The amount of leachate reaching a LDS, or the environment in the absence of a LDS, can be calculated based on assumed liner efficiencies. A sample calculation follows, and the results are presented in Table 11.

Table 11. Landfill Water Balance

Annual Precipitation (in/yr)	40
Efficiency of Landfill Cover %	99
Efficiency of Bottom Liner %	99
Loss due to ET and RO %	60
Conversion factor (inches of rain to gal/ac/d)	74.395
Calculated Values:	
Annual Precipitation gal/ac/d	2975.8
Infiltration through vegetative layer gal/ac/d	1190.32
Infiltration through cap gal/ac/d	11.903
Flow into LCRS gal/ac/d	11.784
Flow into LDS (or the environment) gal/ac/d	0.119

The flow into the LDS, or the environment, is 0.12 gal/ac/d under the specified conditions. If the concentration of BOD is 100 mg/L, COD is 1000 mg/L, and NH_3-N is 250 mg/L, then the mass released will be 45.4 mg/ac/d of BOD, 454 mg/ac/d of COD, and 113 mg/ac/d of NH_3-N. If NH_3-N concentrations remain elevated at 750 mg/L, then the mass released will be 341 mg/ac/d. Concentrations such as these will be analyzed in the section on surface water quality to evaluate the effects of leachate on surface water systems.

B. Liner Defects

Another approach to estimating leachate release is to estimate the number of liner defects and then calculate the resulting flowrate. Murray et al. (1995) estimate liner leakage rates through low permeability soil and composite liners. They also report leak estimates that have been found by other researchers and use them as input data to the HELP model. Based on modeling results, they estimate that FML liners reduce leakage through CCL liners by 62% - 73% with poor installation or 99% with good installation. Their data are presented in Table 12 as hole frequency per area.

Table 12. Measured Frequency of Holes in FML Liners (Murray et al., 1995)

Leak Estimate (holes / ha)	Reference
3 - 5; liner with good installation and QA/QC program	Giroud and Bonaparte, 1989
2.5 - 55 (mean = 35, n = 61)	Laine and Miklas, 1990
16 in bottom liners, 12 on side slope liners	Saleh, 1992
Mean = 26, range of 1 std dev = 5 - 127	Laine and Darilek, 1993
Mean = 3, range of 1 std dev = 0.61 - 14.8; with good QA/QC Mean = 125, range of 1 std dev = 25 - 612; with poor QA/QC These values are the result of the "Latin-Hypercube" statistical analysis on the values reported above.	Murray et al., 1995

Giroud et al. (1992) describe equations that estimate leakage rates through composite liners as a function of the number of defects, the head on the liner, and coefficients of permeability. A composite liner consists of a geomembrane liner above a low permeability soil, which will be referred to as clay. The assumptions and final equations used to estimate leakage rates are presented in Table 13 and described below.

When water enters a defect in a geomembrane, it flows radially around the hole for some distance and then begins to flow through the clay. The radial area covered by the spread of water is called the wetted area. The radius of the wetted area is largely dependent on the quality of contact between the geomembrane and the clay. Criteria that describe good contact conditions include few wrinkles in the geomembrane and well-compacted, smooth-surfaced clay. Poor contact conditions include the presence of a critical number of wrinkles in the geomembrane, clay that does not have a smooth surface, and clay that was not well compacted (Giroud et al., 1992). The factors that most significantly affect flow through a composite liner are the size and number of defects, the hydraulic conductivity of the clay, and the head of liquid on top of the geomembrane (Giroud et al., 1992).

The equations in Table 13 assume that (1) the liquid head above the defect remains constant and operates under hydrostatic conditions, (2) the wetted area remains saturated, and (3) the liquid is water. Flow through the geomembrane due to permeation is neglected (Giroud et al., 1992). The equations for small holes are based on both theoretical analyses and large-scale model tests. The equations for long defects are based on equations derived for small defects and have not been verified with actual data (Giroud et al., 1992).

The equations for small liquid head assume that the hydraulic gradient in the soil is one and that leakage rates do not depend on the thickness of the soil layer. This assumption is valid for cases where the head on top of the soil approaches zero. In cases where the head on the liner does not approach zero, the average hydraulic gradient is used to calculate flow. Flow is assumed to be vertical (Giroud et al., 1992).

Figures 1 through 4 show flowrates as a function of the number of 3 mm and 10 mm diameter holes with good and poor liner contact. These hole sizes were obtained from the mean values under good and poor conditions, respectively, as reported by Murray et al. (1995). All four figures assume a clay thickness of 2 ft (0.6 m) and a saturated hydraulic conductivity of 1×10^{-9} m/s. According to Giroud and Bonaparte (1989), landfills that do not accept liquid waste will have liquid heads less than 1 ft if a granular leachate collection system is used, or a few millimeters if a geosynthetic is used. Figures 1 and 3 show flowrates that result from 1 ft of head. Figures 2 and 4 show flowrates that result from 2 ft of head. As shown in the graphs, the flowrate is linear with respect to the number of defects due to the method of calculation, and it increases drastically from good to poor contact conditions. These data also illustrate the sensitivity of the equations to the head on the liner. As illustrated in Figure 1, the leakage rate ranges from 20 to 50 lphd (or 2.1 to 5.3 gal/ac/d) for the conditions analyzed assuming 50 holes/ha (20 holes/ac).

Table 13. Equations for Calculation of Leakage Through Composite Liners

Defect and liquid head	Quality of Contact between Geomembrane and Soil Liner	Equations	
Small, circular defect; Small liquid head	Good	$Q = 0.21 a^{0.1} h_w^{0.9} k_s^{0.74}$ $R = 0.26 a^{0.05} h_w^{0.45} k_s^{-0.13}$	Eq. 2 Eq. 3
Small, circular defect; Small liquid head	Poor	$Q = 1.15 a^{0.1} h_w^{0.9} k_s^{0.74}$ $R = 0.61 a^{0.05} h_w^{0.45} k_s^{-0.13}$	Eq. 4 Eq. 5
Small, square defect; Small liquid head	Good	$Q = 0.21 b^{0.2} h_w^{0.9} k_s^{0.74}$ $R = 0.26 b^{0.1} h_w^{0.45} k_s^{-0.13}$	Eq. 6 Eq. 7
Small, square defect; Small liquid head	Poor	$Q = 1.15 b^{0.2} h_w^{0.9} k_s^{0.74}$ $R = 0.61 b^{0.1} h_w^{0.45} k_s^{-0.13}$	Eq. 8 Eq. 9
Small, circular defect; Large liquid head	All Qualities	$i_{avg} = (1 + h_w/(2H_s \ln(R/R_o)))$	Eq. 10
Small, square defect; Large liquid head	All Qualities	$i_{avg} = (1 + h_w/(2H_s \ln(2R/b)))$	Eq. 11
Small, circular defect; Large liquid head	Good	$Q = 0.21 i_{avg} a^{0.1} h_w^{0.9} k_s^{0.74}$	Eq. 12
Small, circular defect; Large liquid head	Poor	$Q = 1.15 i_{avg} a^{0.1} h_w^{0.9} k_s^{0.74}$	Eq. 13
Small, square defect; Large liquid head	Good	$Q = 0.21 i_{avg} b^{0.2} h_w^{0.9} k_s^{0.74}$	Eq. 14
Small, square defect; Large liquid head	Poor	$Q = 1.15 i_{avg} b^{0.2} h_w^{0.9} k_s^{0.74}$	Eq. 15
Long defect	All Qualities	$i^*_{avg} = (1 + h_w/(H_s \ln(2R/b)))$	Eq. 16
Long defect	Good	$Q = 0.52 i^*_{avg}(B-b) b^{0.1} h_w^{0.45} k_s^{0.87} +$ $0.21 i_{avg} b^{0.2} h_w^{0.9} k_s^{0.74}$	Eq. 17
Long defect	Poor	$Q = 1.22 i^*_{avg}(B-b) b^{0.1} h_w^{0.45} k_s^{0.87} +$ $1.15 i_{avg} b^{0.2} h_w^{0.9} k_s^{0.74}$	Eq. 18

Variables:

Q = Rate of Leakage (m^3/s) a = Geomembrane hole area (m^2)

h_w = Head of liquid (m) k_s = Hydraulic conductivity of the clay (m/s)

R = Radius of wetted area (m) R_o = Radius of circular defect (m)

B = Length of long crack (m) b = Length of square defect or width of long crack (m)

H_s = Thickness of clay layer (m)

i_{avg} = Average hydraulic gradient in the soil beneath the circular portion of the wetted area. For a long defect, i_{avg} is the gradient under the circular area ends of the defect. For a small defect, i_{avg} is the gradient under the wetted area.

i^*_{avg} = Average hydraulic gradient in the soil beneath the rectangular portion of the wetted area for a long defect.

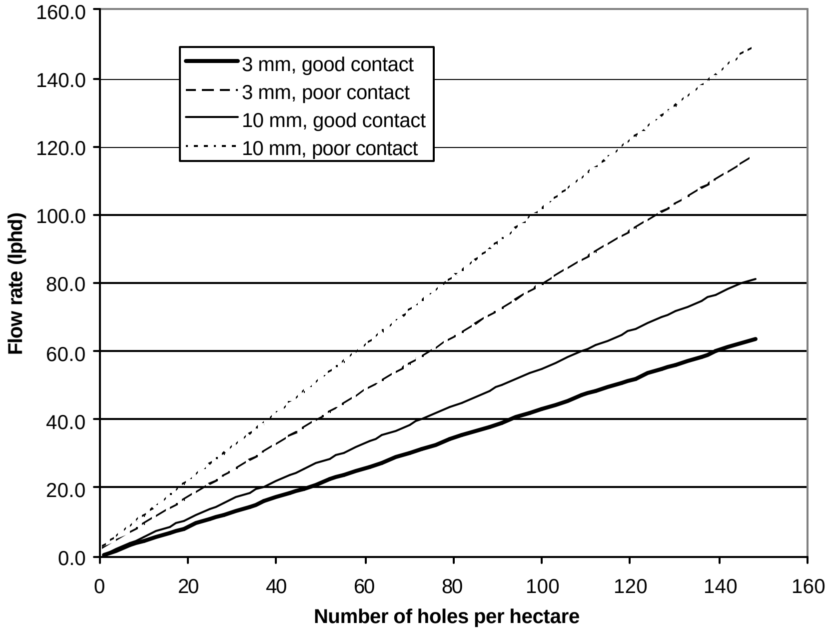

Figure 1. Flowrate Versus Small Hole Density (Head on Liner = 0.3 m)

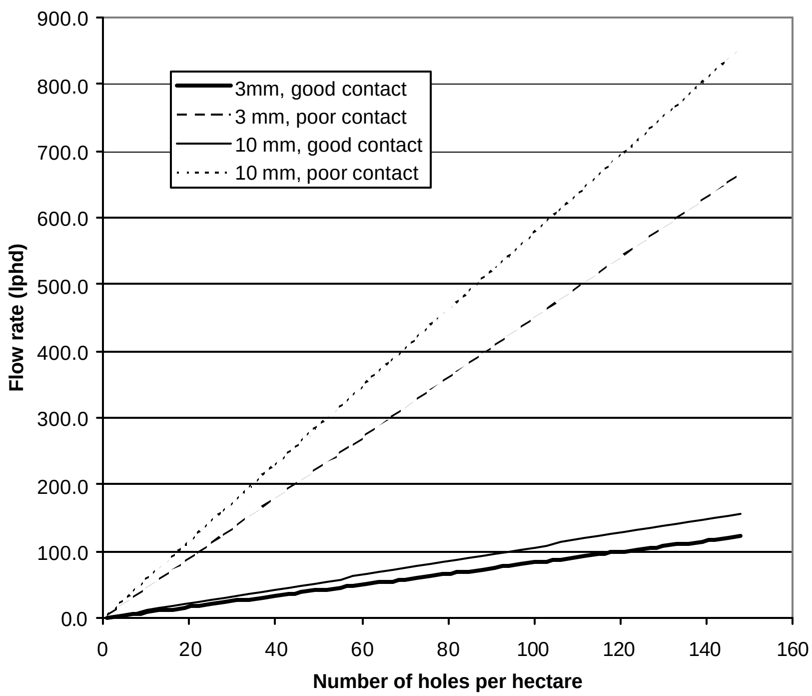

Figure 2. Flowrate Versus Small Hole Density (Head on Liner = 0.61 m)

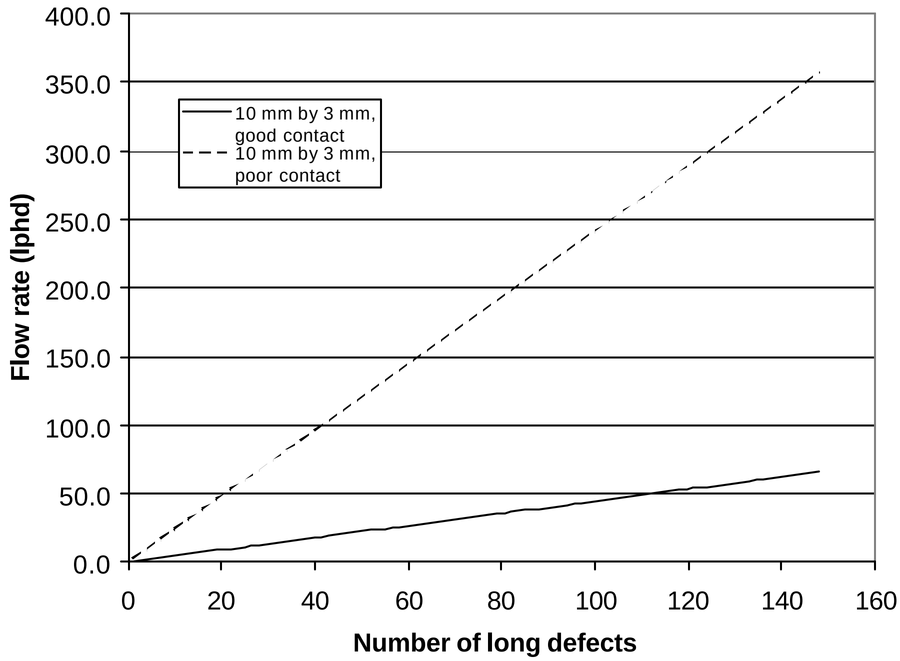

Figure 3. Flowrate Versus Long Defect Density (Head on Liner = 0.3 m)

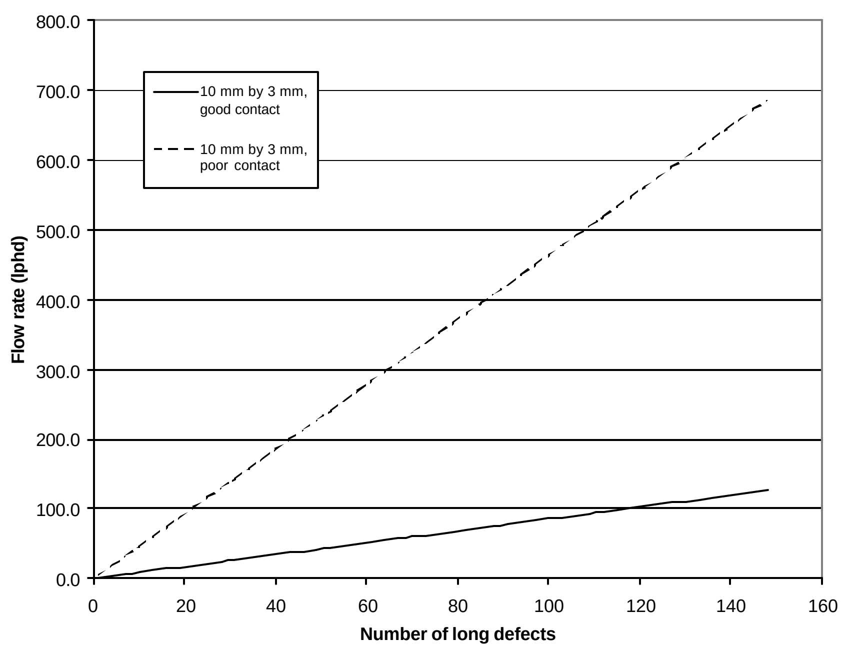

Figure 4. Flowrate Versus Long Defect (Head on Liner = 0.61 m)

C. Head on Liner

Few studies have measured liquid head on liners in landfills. One equation used to calculate the maximum leachate head is

$$h = \frac{L}{2} * \left(\frac{e}{k}\right)^{\frac{1}{2}} \qquad \text{Eq. 19}$$

where h is the maximum leachate head (ft), L is the drainage length (ft), e is the average percolation rate into the landfill (in/yr), and k is the hydraulic conductivity of the refuse (in/yr) (Oweis et al., 1990). For example, if the drainage length is 100 ft, the hydraulic conductivity of the refuse is 12,415.7 in/yr (10^{-3} cm/s), and the average percolation rate is 1.6 in/yr, then the resulting head h = 0.56 ft. The average percolation rate is based on an annual precipitation of 40 in/yr, a 60% loss due to evapotranspiration and runoff, and 90% removal efficiency for the final cover. If the cap has not yet been installed, then average percolation is 16 in/yr, and the calculated head is 1.8 ft. These values of leachate head roughly correspond to those used to develop Figures 1 through 4. However, they are considerably higher than is generally assumed for head on the liner.

Oweis et al. (1990) estimate the hydraulic conductivity of fresh refuse to be 10^{-3} cm/s (12,415.7 in/yr) based on laboratory data. However, this value is expected to change with decomposition and settlement.

D. Landfill Performance in New York State

New York is one of a few states in which double liners are required for MSW landfills. Electronic copies of 1999 annual reports for operating landfills were provided by the NY State Department of Environmental Conservation. These reports contained data on flowrates both in leachate collection and detection systems as well as the applicable acreage. These data are summarized in Table 14.

Table 14. Landfill Performance in NY, 1999

Landfill Type	Landfill ID	LCRS flowrate (gal/ac/d)	LDS flowrate (gal/ac/d)	Liner Efficiency %[a]
Ash	1	1571	2.6	99.8
Ash	2	2142	18.8	99.1
C&D	3	1103	22.5	98.0
C&D	4	1059	16.8	98.4
Ind/Comm	5	1822	0.8	99.9
Ind/Comm	6	2718	1.4	99.9
MSW	7	256	1.8	99.3
MSW	8	1722	1.3	99.9
MSW	9	1083	7.4	99.3
MSW	10	421	2.5	99.4
MSW	11	1304	19.6	98.5
MSW	12	851	24.7	97.1
MSW	13	222	4.2	98.1
MSW	14	359	1.0	99.7
MSW	15	152	0.6	99.6
MSW	16	666	6.2	99.1
MSW	17	450	8.2	98.2
MSW	18	284	2.8	99.0
MSW	19	2651	4.4	99.8
Max		2718	24.7	99.9
Min		152	0.6	97.1
Mean		1097	7.8	99.1

a. $E_a\% = (1 - (\text{LDS flowrate} / \text{LCRS flowrate})) * 100\%$

Liner efficiencies for the LCRS and LDS flowrates are also presented in Table 14. The mean efficiency is 99.1%. This leachate collection efficiency was then used to calculate a hypothetical leachate flowrate in both the LCRS and the LDS based on the following assumptions and the method presented in Table 11:

- annual percipitation of 36.7 in (average of the available 1999 precipitation data for counties represented by a landfill in Table 14)
- evapotranspiration and runoff losses of 60%

The results of the liner efficiency calculation are reasonably consistent with the mean values reported in Table 14. The actual mean LCRS flowrate is 1097 gal/ac/d, and the calculated value is 1092 gal/ac/d. The actual mean LDS flowrate is 7.8 gal/ac/d, and the calculated value is 9.8 gal/ac/d.

Leachate generation rates were also available from 1998 annual reports for 17 of the 19 landfills. The results are presented in Table 15. The mean LCRS flowrate is 1487 gal/ac/d, and the mean LDS flowrate is 9.8 gal/ac/d. The mean liner efficiency for these sites is 99.2%. Precipitation data for 1998 was not available.

Table 15. Landfill Performance in NY, 1998

Landfill Type	Landfill ID	LCRS flowrate (gal/ac/d)	LDS flowrate (gal/ac/d)	Liner Efficiency %[a]
Ash	1	2631	10.3	99.6
Ash	2	2901	37.7	98.7
C&D	3	1997	6.4	99.7
C&D	4	920	18.3	98.0
Ind/Comm	5	2454	1.1	99.9
Ind/Comm	6	2736	8.2	99.7
MSW	7	256	1.8	99.3
MSW	8	3481	26.3	99.2
MSW	9	1025	6.9	99.3
MSW	10	575	4.7	99.2
MSW	11	1464	15.8	98.9
MSW	12	917	5.0	99.5
MSW	13	320	1.3	99.6
MSW	15	281	0.8	99.7
MSW	16	754	9.2	98.8
MSW	18	309	5.7	98.2
MSW	19	2256	6.4	99.7
Max		3481	37.7	99.9
Min		256	0.8	98.0
Mean		1487	9.8	99.2

a. $E_a\% = (1 - (\text{LDS flowrate} / \text{LCRS flowrate})) * 100\%$

E. Conclusions

Three methods have been presented to estimate leachate quantities released to the environment: field data, liner efficiency, and hole density.

As presented in Section III. A. 1, the actual measurements from landfills with Subtitle D liners, show that typical LDS flowrates in the post closure period range from 5 to 210 lphd (0.5 to 22 gal/ac/d). To compare the three methods listed above, a target flowrate of 100 lphd (10.7 gal/ac/d) was chosen as the midrange of typical LDS flowrates observed in the field. However, this number may overestimate the actual flowrates that will result after a final cap has been placed.

Based on the water balance calculation in Section III. A. 1, if the cover and liner are both 99% efficient, then the resulting LDS flowrate is only 1 lphd (0.12 gal/ac/d), well below the mean field observation of 100 lphd. For the efficiency method to result in a flow of 100 lphd (10.7 gal/ac/d), both the cover and liner must be less than 91% efficient. Some of the field data represent situations where a cover is not in place. If the efficiency of the cover is 0%, and the efficiency of the liner is 99%, then the resulting flowrate is 111 lphd (11.9 gal/ac/d) which is close to the midrange of the field data.

A comparison of the liner efficiency calculation to the New York State landfill data shows that the efficiency method is fairly accurate when a reasonable estimate of annual precipitation and liner efficiencies can be made. For 1999, the average efficiency of the landfills for which data were available was 99.1%. When this efficiency is used to calculate a LDS flowrate, it results in 9.8 gal/ac/d, which is close to the measured value of 7.8 gal/ac/d.

Section III. B presents typical hole densities estimated by Murray et al. (1995) which are 3 holes/ha (1.2 holes/ac) for FML's with good QA/QC and 125 holes/ha (50.6 holes/ac) for FML's with poor QA/QC. If the holes are 3 mm in diameter, and the head above the liner is 1 ft, then the predicted flowrates are 1.3 lphd (0.14 gal/ac/d) for 3 holes/ha (good contact conditions) and 100 lphd (10.7 gal/ac/d) for 125 holes/ha (poor contact conditions). The estimates based on 3 holes/ha are close to the value calculated based on cover and liner efficiencies of 99%. The hole density method approaches 100 lphd only when the hole density exceeds 100 holes/ha (40.5 holes/ac). However, this hole density should rarely be seen in the field as quality control is a standard part of liner construction.

The HELP Model was not used for this analysis it assumes saturated flow through the clay liner and requires an estimation of holes/ac as input.

While it is not clear which of the three methods will give the most realistic approximation of leachate flowrates from closed landfills, it is clear encouraging that the LDS flowrates in the NY landfill were quite low and consistent with LCRS efficiencies of about 99%. Ultimately, it will be necessary to obtain field data from closed landfills to verify that there is little to no flow in the LCRS several years after placement of the final cap.

IV. Surface Water Quality Impacts of Landfill Leachate

A. Introduction

In Section II, the composition of leachate associated with well-decomposed refuse was described. Two components of leachate, ammonia and some amount of dissolved organic carbon, will persist even after the refuse mass has reached a state of near complete decomposition. The objective of this section is to discuss a methodology that can be used to assess the potential impacts of a leachate release to surface water. Clearly, if a release of leachate to surface water at a known rate and composition does not threaten a surface water body (or an aquifer), then the landfill can be considered stable with respect to the threat of a leachate release.

Studies have indicated that leachate from a stable landfill will have a BOD/COD ratio less than 0.1 (Reinhart and Grosh, 1998, Pacey, 1999, Pohland and Harper, 1985, Ehrig, 1988), and Pacey (1999) has suggested that BOD should be less than 100 mg/L, and COD should be less than 1000 mg/L. Both the organics and ammonia in landfill leachate have the potential to decrease oxygen levels in receiving streams. In addition, the ammonia can be highly toxic to aquatic organisms. Average leachate ammonia concentrations range from 100 - 450 mg-N/L in leachate from old landfills (Table 1).

Even after dilution with the receiving water body, ammonia concentrations may exceed safe levels. Ammonia also contributes nitrogen that may contribute to an oversupply of nutrients.

To evaluate the impacts of leachate on a surface water system, both flowrates and pollutant concentrations must be known for both the leachate and the receiving stream. Direct measurement of these parameters is ideal but expensive and time consuming. The method presented in this section combines the Streeter Phelps equation, which predicts the depletion of dissolved oxygen by organic matter, with nitrification terms, which estimate the effect of ammonia on dissolved oxygen concentrations. This method may be used to develop an estimate of the effect of a leachate release on surface water. The results may then warrant the collection of additional data to reduce uncertainty in projected dissolved oxygen concentrations.

To begin, data on the flowrate and quality of the potential receiving stream is required. The stream parameters should remain relatively constant over time unless significant changes in the watershed occur. Stream data obtained during landfill operation or from USGS gage stations, along with statistical analysis, can be used to estimate flowrates and concentrations. Most permitting procedures require that the 7Q10 be used as the receiving stream's low flow. The USGS defines the 7Q10 as " the annual minimum 7-day consecutive low flow, which on average, will be exceeded in 9 out of 10 years" (USGS, 1993). Leachate flowrates and concentrations are assumed to be available for landfills during the post-closure monitoring period.

The first step in the process of protecting aquatic environments is to designate a use for the water body. Potential designations include recreation, water supply, fishery, and ecological balance (Thomann and Mueller, 1987). Once a water use has been selected, it is necessary to assign criteria, or concentrations, for the water body that will protect that use. Water quality standards are the regulations on mass discharge or effluent concentration that enforce the criteria.

This section of the report will discuss surface water quality issues associated with organics and ammonia and present equations to estimate dissolved oxygen and ammonia concentrations downstream of a hypothetical leachate discharge. A few case studies that illustrate the impacts of landfill leachate on surface water systems are then presented.

B. Dissolved Oxygen in Streams

Low dissolved oxygen (DO) concentrations are a common problem in surface water systems that receive effluent discharges with high organic loads. Acceptable minimum concentrations depend on the designated water use, temperature of the water, and presence of early life stages of fish. As a general rule, concentrations below 5 mg/L are stressful to aquatic organisms (Thomann and Mueller, 1987). Many states do allow mixing zones where specified standards are not required. Both standards for DO and limitations on mixing zones vary from state to state.

Sources of DO include photosynthesis, atmospheric reaeration, and inputs from tributaries or effluents. Sinks of DO include oxidation of organic carbon, ammonia, and nitrite; sediment oxygen demand; and the respiration of aquatic organisms. The oxidation of organic carbon requires approximately 2.7 mg-O_2/mg-C, and the oxidation of ammonia-nitrogen and nitrite-nitrogen combined requires about 4.57 mg-O_2/mg-N (Thomann and Mueller, 1987).

Atmospheric reaeration depends on the saturation value of dissolved oxygen in the stream, wind speed, stream turbulence, temperature, surface films, and hydraulic structures. Sediment oxygen demand results from the settlement of organic material (Thomann and Mueller, 1987).

The Streeter Phelps equation is often used to predict oxygen concentrations downstream of an organic carbon discharge (Thomann and Mueller, 1987). This equation is presented below. Note that the term (x/U) may be replaced by time t.

$$c = c_s - \left\{ \frac{K_d}{(K_a - K_r)} \left[\exp\left(-K_r \frac{x}{U} \right) - \exp\left(-K_a \frac{x}{U} \right) \right] \right\} L_o - (c_s - c_o) \exp\left(-K_a \frac{x}{U} \right) \qquad \text{Eq. 20}$$

The following equation may be used to account for oxygen depletion during nitrification. Nitrification only occurs at DO concentrations greater than or equal to 1 - 2 mg-O_2/L (Thomann and Mueller, 1987).

$$c = c_s - L_o^N\left[1 - \exp\left(-K_N \frac{x}{U}\right)\right]$$ Eq. 21

The following equation is a combination of the Streeter Phelps and nitrification equations.

$$c = c_s - \left\{\frac{K_d}{(K_a - K_r)}\left[\exp\left(-K_r \frac{x}{U}\right) - \exp\left(-K_a \frac{x}{U}\right)\right]\right\}L_o - (c_s - c_o)\exp\left(-K_a \frac{x}{U}\right)$$
$$- L_o^N\left[1 - \exp\left(-K_N \frac{x}{U}\right)\right]$$ Eq. 22

where,

c = dissolved oxygen concentration at a distance x downstream of the discharge (mg/L)

c_s = saturation concentration of dissolved oxygen (mg/L)

c_o = initial dissolved oxygen concentration (mg/L)

K_d = effective deoxygenation rate of organic carbon (T^{-1})

K_a = reaeration coefficient (T^{-1})

K_r = overall loss rate of CBOD due to settling and oxidation of soluble BOD (T^{-1})

x = distance downstream (L)

U = average stream velocity (L/T)

L_o = initial concentration of carbonaceous material as BOD (mg/L)

L_o^N = initial concentration of ammonia and organic nitrogen (mg-N/L)

K_N = rate of oxidation of ammonia and organic nitrogen (d^{-1})

The initial concentrations of oxygen, organics, and ammonia should be calculated after dilution of leachate with the receiving water body. The continuity equation may be used for this calculation:

$$Q_L C_L + Q_S C_S = Q_i C_i$$ Eq. 23

where $Q_i = Q_L + Q_S$, and

Q_L = flowrate of leachate (volume/time)

C_L = concentration of parameter in leachate (mass/vol)

Q_S = flowrate of receiving stream (volume/time)

C_S = background concentration of parameter in receiving stream (mass/vol)

Q_i = flowrate downstream of discharge (volume/time)

C_i = concentration of parameter downstream of discharge (mass/vol)

Although the water quality model QUAL2E is not used in this analysis, the manual lists typical values for some of the rate constants used in equations 20 - 22 (Brown and Barnwell, 1987). Thomann and Mueller (1987) also list typical values. Table 16 presents the values as they appear in the literature. Of course, the actual values used in Eq. 22 should be based on knowledge of a specific water body because they depend on stream flow, depth, and morphology.

Table 16. Typical Values For Rate Constants (Brown and Barnwell, 1987)

Rate Constant	Values listed by Brown and Barnwell (1987) d^{-1}	Values listed by Thomann and Mueller (1987) d^{-1}
K_d	0.02 - 3.4	0.1 - 0.5 mean depth > 5' 0.5 - 3.0 mean depth < 5'
K_a	0.0 - 100	0.05 - 12.2
K_r	-0.36 - 0.36	
K_N	na[1]	0.1 - 0.5 mean depth > 5' 0.5 - 3.0 mean depth < 5'

1. Values not listed.

C. Use of the Streeter Phelps/Nitrification Equation

The combined form of the equation can be used to evaluate different scenarios for the release of leachate to a receiving water body. If the model shows that dissolved oxygen levels will fall below the permit level, then pretreatment of the leachate in an aeration basin, lagoon, or constructed wetland may be necessary to remove organics and ammonia prior to discharge.

To illustrate the use of Eq. 22, it was applied to a hypothetical landfill. The following examples use a typical leachate flowrate of 1000 gal/d (~10.7 gal/ac/d for a 100 ac landfill) with an ammonia concentration of 250 mg-N/L and a BOD of 100 mg/L. The leachate is assumed to discharge into a stream with a background ammonia concentration of 0.1 mg-N/L, a background organic nitrogen concentration of 1 mg-N/L, and a background BOD of 5 mg/L. Flowrates from Crabtree Creek in Raleigh, NC are used to compare the effect of using the median flow (Figure 5) and low flow (Figure 6) in the model.

The effects of changing the stream flow (Q_s) and reaeration constant (K_a) are presented in Figures 5 and 6. According to the USGS stream flow data, the long-term median flow for Crabtree Creek at Highway 1 in Raleigh is 36 cfs (2.3×10^7 gal/d). If the leachate described above is discharged to this stream, then the resulting ammonia and BOD concentrations after mixing are 0.11 mg-N/L and 5.00 mg/L, respectively.

Under median flow conditions, we have assumed that the average stream velocity is 1 ft/s, and that K_d, K_r, and K_N are all equal to 0.3 day^{-1}. This common value was chosen because it represents the mid range for deep streams given by Thomann and Mueller (1987). The combined form of the Streeter Phelps and nitrification equations is used along with reaeration coefficients ranging from 0 d^{-1} - 5 d^{-1} to predict downstream oxygen concentrations. Under these conditions, the dissolved oxygen concentration decreases dramatically when $K_a = 0$ d^{-1}, but does not drop below 5 mg/L when reaeration values are above 0.26 d^{-1} (Figure 5). During median flow conditions, the ammonia concentration at the discharge point is also below the standards recommended in the following section. Thus, leachate discharge to this stream will probably not require pretreatment unless the reaeration constant is below 0.26 d^{-1}.

When the hypothetical leachate is discharged into Crabtree Creek during low flow, the stream parameters change. The stream velocity is assumed to be 0.1 ft/s, and K_d, K_r, and K_N are assumed to equal the midrange for shallow streams, 1.5 day^{-1}. According to the USGS (1998), the 7Q10 for Crabtree Creek is 2.0 cfs (1.3×10^6 gal/d). Results are shown in Figure 6. After dilution and mixing at the discharge point, the ammonia concentration is 0.3 mg-N/L, and the BOD is 5.07 mg/L. Under these conditions the DO still does not drop below 5 mg/L unless K_a is less than 1.4 d^{-1}. In addition, the ammonia concentration does not exceed the strictest standards presented in the following section.

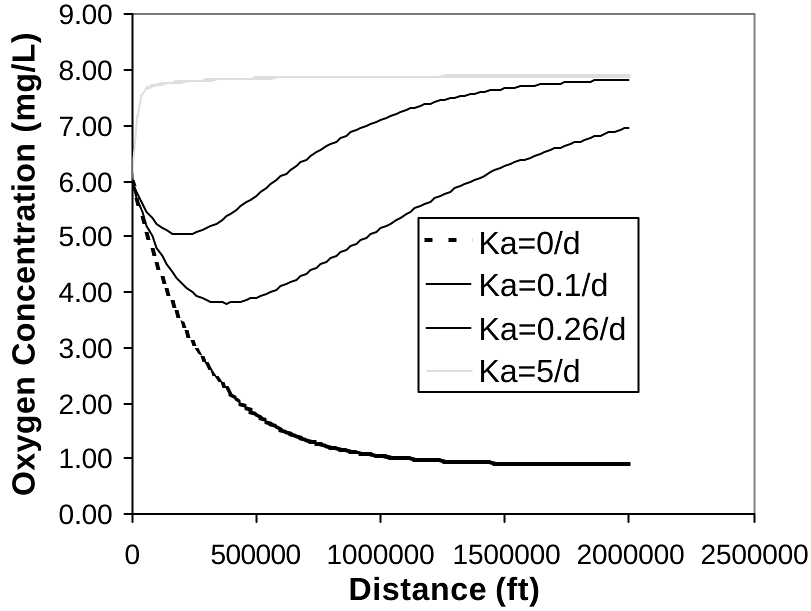

Figure 5. Downstream Oxygen Concentrations in Crabtree Creek During Median Flow

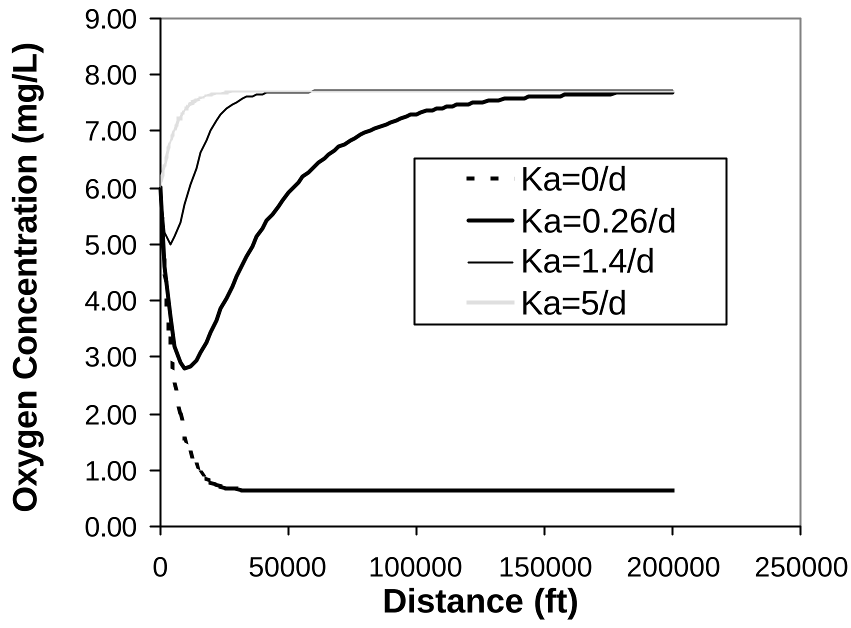

Figure 6. Downstream Oxygen Concentrations in Crabtree Creek During Low Flow

D. Acceptable Ammonia Levels in Landfill Leachate

The Streeter Phelps/Nitrification equation discussed in sections II and III will only predict downstream oxygen concentrations. To predict ammonia concentrations, the continuity equation presented in Section IV B can be used.

Ammonia standards depend on the temperature and pH of the stream, and the presence of early life stages of fish. Table 17 lists guidelines that protect organisms from chronic toxicity at pH 7 and pH 8 which are in the 1999 Update of Ambient Water Quality Criteria for Ammonia (US EPA, 1999b).

Table 17. Ammonia Standards For Fresh Water (US EPA, 1999b)

	Ammonia Standards (mg-N/L)			
Temperature (°C)	Early Life Stages Present		Early Life Stages Absent	
	pH = 7	pH = 8	pH = 7	pH = 8
0	5.91	2.43	9.60	3.95
10	5.91	2.43	7.91	3.26
20	4.15	1.71	4.15	1.71
30	2.18	0.897	2.18	0.897

The lowest ammonia standard presented in Table 17 is 0.9 mg-N/L. Thus, a leachate with an ammonia concentration of 250 mg-N/L would require a dilution factor of 278 to meet the lowest standard. Of course, this calculation neglects any ammonia biodegradation.

E. Nutrient Loading

Thomann and Mueller (1987) define eutrophication as excessive plant and algal growths that interfere with water uses. Though eutrophication is a natural process in surface water systems, its rate is significantly increased by the presence of excess nutrients, especially nitrogen and phosphorous. Increased rates of eutrophication can lead to low DO levels due to night time plant respiration and decay of organic matter, interference with water treatment plant operation, decline of aesthetic and recreational water uses, and the growth of toxic algae that may interfere with shell fish production.

Though ammonia is no longer toxic after it is oxidized to nitrate, the nitrogen still contributes nutrients that may cause excessive plant growth (USGS, 1999). Nitrogen levels in streams that drain agricultural and urban areas are already typically high. Sources of nitrogen include fertilizers, automobile and electric power plant emissions, effluent from sewage treatment plants, and animal wastes. The average natural background concentrations for total nitrogen, nitrate, and ammonia are 1.0, 0.6, and 0.1 mg-N/L, respectively. These background concentrations were measured in streams located in nondeveloped, nonagricultural watersheds (USGS, 1999).

Because nitrogen is readily available in most fresh water streams, phosphorous is usually the limiting nutrient for excessive plant growth (USGS, 1999). Sources of phosphorous include discharges from wastewater treatment plants, fertilizers, soil erosion, and livestock waste. The natural background concentration of phosphorous in the average freshwater stream is 0.1 mg-P/L, and concentrations above this value may lead to eutrophication (USGS, 1999).

In order to improve water quality in the Neuse River Basin in NC, regulators have proposed a ruling that will limit discharges of nitrogen and phosphorous in the watershed. The primary goal is to reduce the nitrogen load at New Bern, NC by 30% (NCDEHNR, 1997). In 1997, New Bern received 8.7 million lbs-N/yr of which 2.1 million lbs can be attributed to permitted dischargers, leaving 6.6 million lbs from nonpoint sources such as agricultural runoff and atmospheric deposition. The following regulations have been proposed (NCDEHNR, 1997):

- Wastewater treatment plants with over 0.5 MGD of effluent should not exceed total nitrogen concentrations of 3.5 mg-N/L or total phosphorous concentrations of 2 mg-P/L.
- Wastewater treatment plants with over 0.05 MGD of effluent should also not exceed total phosphorous concentrations of 2 mg-P/L. There is no nitrogen limit for these facilities.
- Stormwater runoff from agricultural and forested land should not exceed an area weighted loading rate of 5.1 lbs-N/ac/yr.

- Stormwater runoff from new development should not exceed an area weighted loading rate of 3.6 lbs-N/ac/yr.

To compare potential nutrient loads from a landfill to these other sources of nitrogen and phosphorous, annual loads from each source have been calculated for a population of 500,000. According to Tchobanoglous and Burton (1991), the range of typical wastewater flowrates from a typical house is 45 - 90 gal/person/d. Assuming water use of 65 gal/person/d and that the input flowrate to a wastewater treatment plant is the same as the effluent discharge, a treatment plant that serves 500,000 people will discharge 32.5 MGD. If the allowable nitrogen concentration is 3.5 mg-N/L, then this plant will release 346,000 lbs-N/yr. If the same plant releases effluent with a phosphorous concentration of 2 mg-P/L, it will release approximately 19,800 lbs-P/yr. To put this on the scale of a landfill that will receive waste for 20 years, these values become 6.92×10^6 lbs-N and 3.96×10^5 lbs-P, over 20 years.

Based on the analysis in Section III, it is assumed that a landfill will produce 10.7 gal/ac/d of leachate during the post-closure period. If we assume a population of 500,000 people with a landfill acceptance rate of 4 lbs/person/day (after recycling), then this population will produce 1.46×10^{10} lbs of waste over a 20 year period. Dividing this by a typical landfill-waste density of 1300 lbs/yd^3, the volume of waste is 1.12×10^7 yd^3, or 3.03×10^8 ft^3. To take into account the daily cover soil, this volume can be multiplied by a factor of 1.1 to obtain 3.33×10^8 ft^3. Assuming this landfill is 80 ft deep, the surface area will be 95.7 ac, or 100 ac for simplicity. If the landfill produces 10.7 gal/ac/d, and the leachate has an ammonia concentration of 250 mg-N/L and a phosphorous concentration of 1 mg-P/L, then the nutrient loads would be 1.63×10^4 lbs-N and 65.2 lbs-P, over a 20 year period. Thus, the nitrogen released from a typical landfill that serves 500,000 people is 0.2% of that released from a typical wastewater treatment plant serving the same population; the phosphorous released is only 0.02%. These numbers are extremely low considering that a leachate release may never occur, and that the quantity of leachate produced from old, lined sites is often negligible within ten years of receiving the final cover (Bonaparte et al., accepted for publication).

A hypothetical landfill release may also be compared to acceptable stormwater releases from the same area of land. According to the proposed rule to manage nutrient discharges in the Neuse River Basin, a 100 ac site of agricultural or forested land can release 510 lbs-N/yr, or 10,200 lbs-N over a 20 year period. 100 ac of new development can release 360 lbs-N/yr, or 7,200 lbs-N over a 20 year period. Though the landfill releases 1.6 times the nutrient load of the agricultural/forested land and 2.3 times that of the new development, these values are not high considering that the loads are being compared to stormwater runoff and that the landfill release is likely an overestimate.

F. Summary

Ammonia and organics will remain in landfill leachate for decades. If released to the environment, both of these pollutants can decrease the dissolved oxygen available for the survival of aquatic organisms. Ammonia is also toxic to several species, and its nitrogen may contribute to eutrophication. The biodegradable portion of the organics (BOD) in landfill leachate is present in relatively low concentrations and will probably not have adverse impacts on the environment. The impacts of the more stable organics (COD) should be investigated. The concentrations of phosphorous in leachate are not typically high enough to cause a nutrient problem. Ammonia will probably be the most significant long-term problem.

A method to evaluate potential impacts of a leachate release to surface water has been presented in this section. The methodology includes (1) the use of the Streeter Phelps equation with terms added to consider nitrification to calculate the dissolved oxygen concentration downstream of a hypothetical leachate release, (2) nutrient loading, and (3) ammonia toxicity.

In the case of a hypothetical leachate discharge to Crabtree Creek in Raleigh, NC, the DO concentration remains above 5 mg/L under both median and low flow conditions as long as the reaeration coefficient is above 0.26 d^{-1} and 1.4 d^{-1}, respectively. In addition, the ammonia concentration does not exceed the strictest standard in either case.

The Neuse River nutrient management strategy was used to compare nitrogen and phosphorous loads from a landfill to other sources in the basin. When compared on a population basis to a typical wastewater treatment plant, a typical landfill released a small percentage of the nutrient load, 0.2% of the nitrogen and 0.02% of the phosphorous. However, the nutrient load is 1.6 to 2.3 times greater than that from a stormwater discharge from the same area. Nonetheless, the landfill does not contribute the majority of the nutrient load to the basin. More importantly, the wastewater treatment plant is a continuous discharge, whereas the leachate discharge is hypothetical and may never occur.

In summary, several aspects of water quality must be addressed when leachate is released to a surface water body. Dissolved oxygen levels should be modeled, and ammonia toxicity must be considered. In addition, the mass of nutrients delivered to the stream should not contribute to excessive plant growth. Of these three problems, ammonia toxicity will probably be the most common. Simple aeration strategies such as cascading water may alleviate this problem, as the ammonia will be oxidized to nitrate, but aeration will not decrease the nutrient load to the stream.

G. Regulatory Considerations

Landfills that discharge leachate or gas condensate to waters of the US are regulated under the Clean Water Act and require a NPDES permit. The following information is from the Development Document for Final Effluent Limitations Guidelines and Standards for the Landfills Point Source Category (US EPA, 2000) and only applies to landfills that discharge leachate directly to a receiving body.

The goal of the landfill requirements is to limit the amounts of metals, ammonia, and toxic organics to water bodies. After review by the EPA, it was decided that regulations on metals were not necessary because the concentrations are typically low in landfill leachate and are not improved by biological treatment. Limitations for BOD_5, ammonia, and TSS are presented in Table 18. Based on the effluent limitations, a landfill cannot discharge a leachate stream with BOD, NH_3-N, or TSS concentrations in excess of the values listed in Table 18 even if this discharge appears to be acceptable based on the surface water quality modeling approach described above.

Table 18. Effluent Limitations for Landfill Leachate

Pollutant	Maximum for One Day (mg/L)	Monthly Average Shall Not Exceed (mg/L)
BOD_5	140	37
NH_3-N	10	4.9
TSS	88	27

V. Landfill Gas

Landfill gas typically consists of 40-60% methane and the remainder CO_2. Landfill gas models predict maximum methane production in the few years after site closure. This is illustrated in Figure 7 for a landfill that accepts 286,000 tons/yr for 20 years. Potential problems associated with landfill gas include odors, subsurface lateral migration that can lead to an accumulation of landfill gas in underground structures and a subsequent explosion hazard, and the release of methane, a greenhouse gas, to the atmosphere. Subtitle D landfills are required to manage gas such that the concentration at the site boundary does not exceed 25% of the lower explosive limit. Most landfills install either passive vents in which the gas is vented to the atmosphere, or active collection systems, in which the collected gas is either flared or utilized as an energy source. Under the New Source Performance Standards (US EPA, 1996), landfills with a capacity exceeding 2.5 million metric tons are required to collect and either flare landfill gas or use it in an energy recovery project.

To terminate the post-closure monitoring period, it should be safe to discontinue any active gas collection system. It would appear that safety can be considered from three perspectives, odors, explosions, and release of a greenhouse gas. With respect to odors, if the discontinuation of an active landfill gas collection system results in problematic odors, then the gas collection system should be reactivated. With respect to explosions, monitoring wells could be installed in the vadose zone around a landfill. If significant concentrations of methane appear in these wells, it would suggest that there is significant subsurface lateral migration and the potential for this to result in an explosion hazard can be evaluated. Finally, with respect to the release of a greenhouse gas, guidance is required from the US EPA on an acceptable emission rate. Methane production models can then be used to estimate when gas production from a landfill falls below this methane production rate. In suggesting the need for this guidance, the US EPA may require additional regulatory authority to regulate methane releases from landfills.

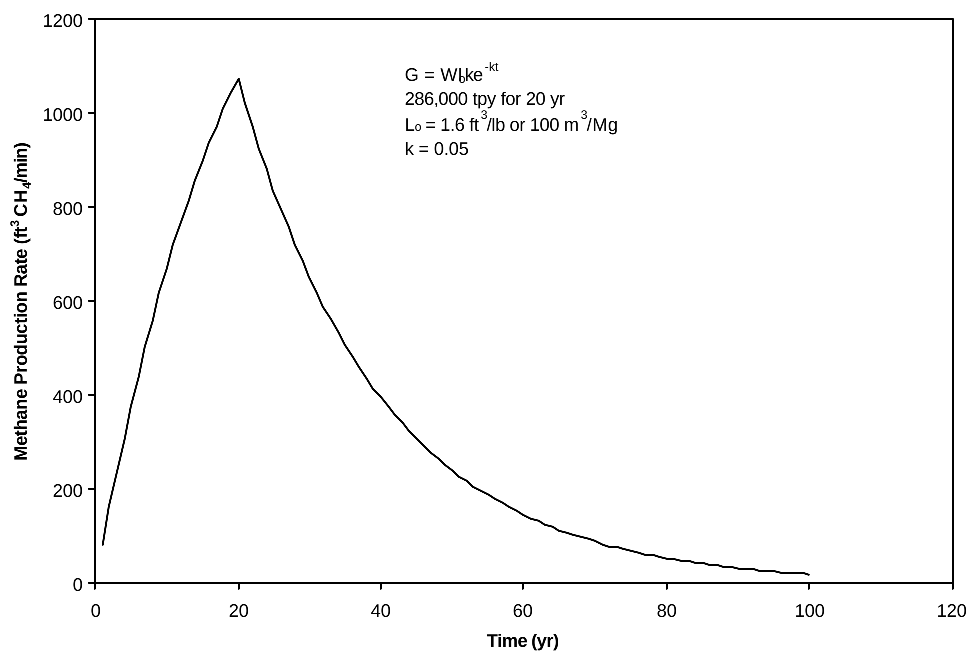

Figure 7. Gas Production Rate

VI. Summary

Landfills approaching termination of the post-closure period present an issue that has not yet been addressed: at what point does a landfill become sufficiently stable to terminate the owner's responsibility to monitor the site? This question has many angles.

First the composition of the leachate must be determined with a particular focus on organics and ammonia. Both of these constituents deplete oxygen resources in aquatic environments and can result in toxicity to several aquatic species. In addition, ammonia contributes nitrogen to the system. Once decomposition has neared completion, the ratio of BOD to COD is typically 0.1 or lower, and the BOD and COD concentrations are near 100 mg/L and 1000 mg/L, respectively. Ammonia levels tend to remain high. Though acquisition of a permit may require that heavy metal concentrations also be monitored, they are not expected to impede termination of the post-closure period.

Second, an estimate is needed of the quantity of leachate produced after closure. Several methods have been evaluated in this paper. For most sites, the flowrate to the leachate collection system should be known. This measurement could be performed during collection and disposal of the leachate. One feasible alternative to direct measurement of leachate flow is to assume a liner efficiency and calculate percolation through the final cover. If this method is used, efficiencies on the lower end of the ranges presented in this paper should be assumed.

Once the estimates have been obtained for the concentrations of BOD and ammonia and the flowrates of the leachate and receiving stream, the Streeter Phelps/Nitrification equation can be used to estimate DO concentrations in a receiving water body. The DO should remain above 5 mg/L. In addition, the continuity equation should be used to determine if ammonia concentrations will exceed the standards for a particular stream. If either of these parameters is not acceptable, then leachate collection and treatment should not be terminated.

In addition to measuring the quantity of leachate produced, gas production rates should also be monitored. A decrease in gas production will signify a lapse in microbial productivity, which can either be attributed to waste stabilization or non-ideal conditions for the microbes. Measuring gas quantities in wells off site will ensure that the liners are performing and that the risk of subsurface lateral migration leading to an explosion is minimal.

Ultimately, it will probably be necessary to look very closely at the first group of landfills to reach the 30-year post-closure state. These sites will require significant assessment of all available data to determine if post-closure can be terminated while protecting human health and the environment.

VII. Conclusions and Recommendations

- A 30-year post-closure period may or may not be adequate for a landfill to reach stability. Technical criteria to define leachate and gas stability need to be established.
- To protect the environment in case of an accidental release, several criteria should be met: (1) the leachate should not contain high concentrations of organics, ammonia, or heavy metals; (2) the leachate should not exhibit toxicity to the organisms in the ecosystem; and (3) the landfill should not release gas that may cause contamination of ground water supplies, odor nuisances, or explosions.
- Use of organic indicators is not adequate to define stability because as leachate from new refuse on top of the landfill percolates through older refuse, the organics will be consumed and the indicators may give a false impression of decomposition.
- Ammonia may be the most significant, long-term pollutant in landfill leachate.
- Heavy metal concentrations in landfill leachate are typically below US Drinking Water Standards. Landfills that have become aerobic need to be identified and analyzed for release of metals. Additional work is required to evaluate the potential release of metals in well-decomposed refuse that has become aerobic. Work is also required to determine whether landfills will actually turn aerobic once the refuse is thoroughly decomposed.
- More work is required before bioassays can be used to monitor landfill leachate.
- Leachate production data is needed from landfills that have received a final cover.

- Landfills do not contribute significant nutrient loads to surface water compared to waste water treatment plants.
- The Streeter Phelps/Nitrification equation should be used to model an actual leachate discharge and should be verified against actual DO concentrations in the receiving stream.

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APPENDICES

Appendix A. Landfill Leachate Composition

Table of Contents

List of Tables

Landfill Leachate Composition

I. Introduction

Landfill leachate typically contains water, heavy metals, organic material in various stages of decomposition, and inorganics such as ammonia, sulfate, and metal cations (Christensen et al., accepted for publication). Its composition depends on several factors including water infiltration, the state of waste decomposition, initial waste composition, codisposal practices, and site hydrology (Reinhart and Grosh, 1998; Pohland and Harper, 1985). Thus it is impossible to predict the stability of a landfill from any one of these factors. However, it may be possible to determine the stability of landfill leachate through a series of chemical indicators.

In order for leachate to safely be released to the environment, several criteria should be met: (1) the leachate should not contain high concentrations of organics, ammonia, or heavy metals; (2) the leachate should not exhibit toxicity to the organisms in the ecosystem. This section presents information on the organic and ammonia contents of landfill leachate and describes the changes in these concentrations from one phase of decomposition to another. Section III presents a table that lists organic indicators that have been used to quantify leachate composition and stability.

II. Waste Decomposition

This section will describe the four major phases of waste decomposition: the aerobic phase, the anaerobic acid phase, the accelerated methane production phase, and the decelerated methane production phase.

Oxygen present in the void spaces and moisture of fresh refuse triggers the aerobic phase of waste decomposition. Oxygen and nitrate serve as electron acceptors in the degradation of soluble organic carbon (Barlaz and Ham, 1993). The aerobic phase in a landfill lasts only a few days because oxygen is not replenished once the waste is covered. Similarly, there is generally not a significant input of nitrate (Barlaz and Ham, 1993). 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. Leachate can contain high concentrations of organics with COD values ranging from tens of thousands to one hundred thousand mg/L (Barlaz and Ham, 1993). As most of this organic carbon is readily degradable, relatively high BOD/COD ratios are expected.

As oxygen sources are depleted, the waste becomes anaerobic, which supports fermentation reactions. This process results in an accumulation of carboxylic acids, and the pH decreases (Barlaz and Ham, 1993). Reinhart and Grosh (1998) report that the highest values of BOD and COD are detected during this phase. The ratio of BOD to COD in the acid phase has been reported to be above 0.4 (Ehrig, 1988) or 0.7 (Robinson, 1995).

The onset of the methane production phase occurs when measurable quantities of methane are produced. Some authors have reported that it may take only a few years to reach this phase (Barlaz and Ham, 1993; Krumpelbeck and Ehrig, 1999). During this phase the acids that accumulate in the acid phase are converted to methane and carbon dioxide by methanogenic bacteria (Reinhart and Grosh, 1998). COD and BOD levels begin to decrease and pH increases (Reinhart and Grosh, 1998) as acids are consumed (Barlaz and Ham, 1993). BOD to COD ratios may drop below 0.1 (Ehrig, 1988; Pacey, 1999; Reinhart and Grosh, 1998).

In the decelerated methane production phase, pH continues to increase as carboxylic acid concentrations decrease to below 100 mg/L (Barlaz and Ham, 1993). Some COD remains in the leachate but is mostly recalcitrant compounds such as humic materials (Barlaz and Ham, 1993; Christensen et al., 1994). Ratios of BOD to COD continue to decline and may approach zero.

The most important factor that governs the rate of waste decomposition is the moisture content of the landfill. Landfills in arid climates will take much longer to stabilize than those in rainy climates, depending on the landfill cover used. Leachate recycle has been found to increase the rate of waste decomposition due to the moisture addition (Pohland and Harper, 1985; Krumpelbeck and Ehrig, 1999; Pohland, 1975; Pacey, 1999). Pacey (1999) reports that leachate recycle may stabilize waste within five to ten years. Pohland and Harper (1985) report that leachate recycle can reduce the time required for stabilization by 90%.

III. Organic Content of Landfill Leachate

Organic materials found in landfill leachate are typically volatile fatty acids, and humic and fulvic compounds (Christensen et al., 1994). The volatile fatty acids are mostly found in the acid and methane production phases while the humic and fulvic compounds comprise most of the organics in older, methanogenic leachate (Christensen et al., 1994).

Landfill leachates with high organic carbon contents are a pollution problem because they exert excessive oxygen demands to receiving water bodies and may cause disruptions to ecosystems.

There are three ways to measure the organic content of an effluent stream. TOC measures the total organic carbon present whether it has been degraded or not, COD indicates the organic content that can be oxidized chemically, and BOD measures the organic content that can be degraded biologically. In any given sample, COD is greater than or equal to BOD. As a landfill ages and waste is degraded, concentrations of BOD and COD both decrease (Reinhart and Grosh, 1998). However, BOD decreases faster, and its concentration can approach zero. COD will remain in the leachate and is comprised of humic and fulvic compounds (Christensen et al., 1994).

A. Organic Indicators of Waste Decomposition

Several parameters have been suggested to determine the degree of waste decomposition: pH, oxidation reduction potential, COD, BOD, BOD/COD, alkalinity, heavy metals, ionic strength, and the sulfate to chlorine ratio (Pohland and Harper, 1985; Chian and DeWalle, 1977; Reinhart and Grosh, 1998). The most commonly used indicators are BOD, COD, and the ratio of the two. Though no standard values exist for these indicators, Table 1 summarizes some suggested ranges that correlate with the phase of decomposition. Several authors assign a BOD/COD value of less than 0.1 to stabile leachate (Reinhart and Grosh, 1998; Pacey, 1999; Pohland and Harper, 1985; Ehrig, 1988). Pacey (1999) has suggested that BOD should be less than 100 mg/L and COD should be less than 1000 mg/L. Table 2 summarizes various leachate composition data from several studies.

Variability may be expected in the values of COD and thus the BOD to COD ratio due to 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. In addition, poor sampling methods that expose anaerobic leachate to oxygen in the air may cause Fe(II) to oxidize to Fe(III) and precipitate out of the leachate. Thus, some of the COD will be lost.

Table 1. The Use of Organic Indicators to Define Waste Decomposition[1]

COD and BOD are expressed in mg/L.

Phase:	Acid	Methanogenic	Stabilized	Reference
Organic Indicator:				
BOD COD BOD/COD	>10,000 --- > 0.7	20% of COD < 2000 ---	--- --- ---	Robinson, 1995
BOD COD BOD/COD	--- 3×10^{-4} - 3×10^{-7} ---	--- < 2000 ---	--- --- < 0.1	Reinhart and Grosh, 1998
BOD COD BOD/COD	--- --- ---	--- --- ---	--- --- 0.02 - 0.013	Pohland and Harper, 1985
BOD COD BOD/COD	--- --- > 0.4	--- 3000 - 4000 < 0.1	--- --- ---	Ehrig, 1988
BOD COD BOD/COD	--- --- ---	--- --- ---	< 100 < 1000 < 0.1	Pacey, 1999

1. These values are suggested ranges to define leachate decomposition.

Table 2. Organic and Ammonia Concentrations (mg/L) in Landfill Leachates from Older, Methanogenic Landfills[1, 2]

BOD	COD	BOD:COD	Ammonia as N	Reference
5.7 - 1100 (50, na)	76 - 6997 (50, na)	--- (na, na)	12.4 - 1571 (50, na)	Range of concentrations from 21-30 year old, German landfills (Krumpelbeck and Ehrig, 1999)
290 (50, na)	1225 (50, na)	0.24 (50, na)	445 (50, na)	Average concentrations from 21-30 year old, German landfills (Krumpelbeck and Ehrig, 1999)
44 (35, 120)	320 (85, 550)	0.11 (35, -)	110 (104, 190)	Average concentrations from old, Danish landfills (Kjeldsen and Christophersen, accepted for publication)
39 (1, na)	398 (1, na)	0.10 (1, na)	233 (1, na)	A sample composition at Sandsfarm Landfill (Robinson, 1995)
11 (1, na)	190 (1, na)	0.06 (1, na)	282 (1, na)	A sample composition at Bishop Middleham Landfill (Robinson, 1995)
38 (1, na)	517 (1, na)	0.07 (1, na)	399 (1, na)	A sample composition at Odsal Wood Landfill (Robinson, 1995)
1.0 (1, na)	53 (1, na)	0.02 (1, na)	42.6 (1, na)	A sample composition at East Park Drive Landfill (Robinson, 1995)
2.5 (1, na)	64 (1, na)	0.04 (1, na)	29.8 (1, na)	A sample composition at Marton Mere Landfill (Robinson, 1995)
20 - 550 (na, na)	500 - 4500 (na, na)	--- (na, na)	30 -3000 (na, na)	Range of concentrations reported in the literature for methanogenic leachate (Ehrig, 1988)
180 (na, na)	3000 (na, na)	0.06 (na, na)	750 (na, na)	Average concentrations reported in the literature for methanogenic leachate (Ehrig, 1988)
33 (3, 5.57)	75 (3, 7.23)	0.44 (3, 0.05)	0.5 (3, 0)	Average values from the last three days of a laboratory experiment with leachate recycle (Pohland, 1975).
34 (3, 9.54)	224 (3, 23.3)	0.15 (3, 0.026)	1.07 (3, 0.40)	Average values from the last three days of a laboratory experiment with leachate recycle and pH adjustment (Pohland, 1975).
36 (3, 4.04)	194 (3, 17.6)	0.19 (3, 0.035)	3.5 (3, 0.91)	Average values from the last three days of a laboratory experiment with leachate recycle, pH adjustment, and nutrient addition (Pohland, 1975).

1. The values represent actual composition results, averages or ranges of concentrations (or ratios) from leachate composition data. The landfills sampled are older, methanogenic landfills. The laboratory experiment data (Pohland, 1975) comes from stabilized waste.
2. Number of samples and standard deviation are given in parentheses (n, std.dev.)

IV. Ammonia Content of Landfill Leachate

Another pollution problem associated with landfill leachate is ammonia nitrogen which includes both ammonia and ammonium ion. In landfill leachates, which typically have pH values ranging from 5-8, the ammonium ion is most prevalent (Robinson, 1995). Several researchers have attributed the toxicity of landfill leachate to not only organics but also ammonia (Wong, 1989; Assmuth and Pentillae, 1995; Cheung et al., 1993; Lambolez et al., 1994; Clement et al., 1997). Ammonia also exerts an oxygen demand and contributes nitrogen that can lead to eutrophication. Others have identified ammonia as the most significant long term pollution problem at landfill sites (Robinson, 1995; Krumpelbeck and Ehrig, 1999; Christensen et al., 1994; Christensen et al., accepted for publication). It is continuously released in leachate until all wastes are degraded. Futhermore, there is no mechanism for its degradation under methanogenic conditions (Robinson, 1995; Burton and Watson-Craik, 1998).

Krumpelbeck and Ehrig (1999) report that in a study of 50 German landfills, ammonia concentrations did not show a significant decrease even 30 years after closure. Ehrig (1988) reports that there is not a significant change in ammonia concentrations from the acidic to methanogenic phase, and that the average value is 750 mg-N/L. According to Krumpelbeck and Ehrig (1999) and Kjeldsen and Christophersen (accepted for publication) average ammonia concentrations for old landfills range from 100 to 450 mg-N/L. Table 2 reports ammonia concentrations from these and other studies.

V. Summary

The stability of a landfill and its leachate composition are difficult to estimate based on factors such as waste composition or age. Criteria must be established that will aid in the management of landfill leachate and prevent harmful releases to the environment. Determining the organic content and ammonia content may be the first step. Organic indicators such as BOD, COD, and BOD/COD have been used by many to estimate waste decomposition (Robinson, 1995; Reinhart and Grosh, 1998; Pacey, 1999; Pohland and Harper, 1985; Ehrig, 1988). Most of these researchers have assigned a

BOD/COD less than 0.1 (Reinhart and Grosh, 1998; Pacey, 1999; Pohland and Harper, 1985; Ehrig, 1988), a BOD less than 100 mg/L, and a COD less than 1000 mg/L (Pacey, 1999) to signify landfill stability. Unfortunately, even if a leachate does not have a high organic strength, ammonia levels may remain high until all waste is degraded and the ammonia is flushed out (Robinson, 1995; Krumpelbeck and Ehrig, 1999; Christensen et al., 1994; Christensen et al., accepted for publication).

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Appendix B. The Presence of Heavy Metals in Landfill Leachate

Table of Contents

List of Tables

The Presence of Heavy Metals in Landfill Leachate

I. Introduction

The objective of this section is to summarize information on heavy metal concentrations in the leachate produced from landfills that contain well-decomposed refuse. Processes that can affect concentrations of metals in leachate are described in the following section, followed by a review of existing data. The final section evaluates the potential for increased metals concentrations in landfills that are so stable as to become aerobic.

II. Heavy Metals in Landfill Leachate

There have been a large number of studies in which researchers have reported metals concentrations from full-scale landfills, test cells, and laboratory studies. The ultimate conclusion of these studies is that metals concentrations are not of concern in landfill leachate (Christensen et al., accepted for publication; Robinson, 1995; Reinhart and Grosh, 1999; Revans et al., 1999; Kjeldsen and Christophersen, accepted for publication; Christensen et al., 1994). Thus, rather than summarize each individual study, a selection of review articles and major field studies is presented and summarized in this section.

As presented in Table 1, most heavy metal concentrations in landfill leachate are at or below US drinking water standards. Only two of the ten studies reported Cd concentrations greater than one order of magnitude higher than the US Drinking Water Standards; two studies also detected Cr concentrations in excess of the drinking water standards. One study showed Zn concentrations that minimally exceeded the standard, 5.31 mg/L compared to 5.0 mg/L. There were no reports of Cu in excess of its drinking water standard (1.3 mg/L). The US Drinking Water Standard for Pb is 0, and all leachates were reported to contain some amount of Pb, ranging from < 0.005 mg/L to 0.19 mg/L. A federal standard for Ni has not been established, but the concentrations found in the 10 studies ranged from 0.0036 mg/L to 0.348 mg/L. Metals are subject to both adsorption and precipitation, and it has been suggested that these attenuation mechanisms are responsible for the relatively low metal concentrations in leachates (Christensen et al., accepted ; Revans et al., 1999; Flyhammar and Hakansson, 1999; Christensen et al., 1994).

Table 1. Heavy Metal Concentrations in Leachate[a]

Study	1	2	3	4	5	6	7	8	9	10	11
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, accepted for publication).
2. Average leachate concentration in 20 German landfills in the methanogenic phase (Christensen et al., accepted for publication).
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., accepted for publication).
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).
11. National primary drinking water regulations, USA (US EPA, http://www.epa.gov/OGWDW/wot/appa.html).

--- Data not available

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

In general, metal species are of concern when they are biologically available at toxic levels to organisms in the receiving ecosystem (Smith et al., 1999). This category includes not only free metal ions in solution, which are usually present in low concentrations, but also organic complexes, inorganic complexes, and metals bound to colloidal sized particles (Christensen et al., accepted for publication).

III. Processes Affecting Metals Concentrations

Four processes have been reported to control heavy metal concentrations in landfill leachates: complexation, oxidation-reduction reactions, sorption, and precipitation (Christensen et al., accepted for publication; Revans et al., 1999; Flyhammar and Hakansson, 1999; Christensen et al., 1994). A short description of each process is presented in this section.

A. Sorption

Christensen et al. (accepted for publication) define sorption to include adsorption, absorption, surface complexation, and ion exchange. Adsorption and absorption occur when the molecules of a gas, liquid, or dissolved substance and a particle "stick" together (Lu et al., 1985). This process generally decreases metals concentrations in landfill leachate.

Divalent metal cations tend to favor sorption with negatively charged sites such as colloidal particles, calcite, clay minerals, organics, and oxides of Fe, Mn, Al, and Si (Christensen et al., accepted for publication).

B. Complexation

Complexation is the combination of metal ions with non-metallic ligands by covalent bonds; it is capable of increasing metal solubilities (Lu et al., 1985). In general, a large fraction of the dissolved metals content is complexed with organics compared to the fraction present as the free metal ion (Christensen et al., accepted for publication; Jensen et al., 1999; Jensen and Christensen, 1999). For example, the humic-like substances formed from waste decomposition can serve as ligands for metal complexes (Reinhart and Grosh, 1998). In a study of leachate polluted groundwater, Christensen et al. determined that free metal ions of Cu and Pb only made up 1-2% of the total metals measured in the leachate; for Cd, Ni, and Zn, the fraction was slightly higher, 7-17%. These numbers are based on a model that does not include inorganic complexes (Christensen et al., accepted for publication).

A study at the North Bay Landfill in Canada compared the capacity of a sample to form organic complexes with the actual concentrations in the leachate. The actual concentrations were orders of magnitude less than those expected from complexation, and the lower concentrations were attributed to sorption and precipitation (Christensen et al., accepted for publication).

C. Precipitation

Precipitation occurs when a metal species falls out of solution as a solid. Sulfides and carbonates are capable of forming precipitates with Cd, Ni, Zn, Cu, and Pb. Occasionally, phosphates and hydroxides will also precipitate metals (Christensen et al., accepted for publication). Hydroxide precipitates form at pH at or above neutral which is typically the case in methanogenic leachates (Reinhart and Grosh, 1998).

Sulfide is formed from sulfate during waste decomposition in landfills, and sulfide precipitation is often cited as an explanation for low concentrations of heavy metals (Christensen et al., accepted for publication; Christensen et al., 1994). Even small concentrations of sulfides will precipitate heavy metals except for Cr, which does not form an insoluble sulfide compound (Christensen et al., accepted for publication). However, Cr does tend to form insoluble precipitates with hydroxide (Christensen et al., accepted for publication; Revans et al., 1999).

While carbonates are abundant in landfill leachate, the solubilities of metal carbonates are generally higher than that of metal sulfides (Christensen et al., accepted for publication). In general, sulfide precipitation is expected to dominate heavy metal attenuation compared to complexation agents (Reinhart and Grosh, 1999).

The equilibrium concentrations of a number of metals in the presence of both sulfides and carbonates are presented in Table 2. Concentrations are calculated for five cases: (1) in distilled water, (2) pure water at pH 7.5, (3) a sulfide concentration of 10 mg/L of sulfide as sulfur, (4) 10 mg/L of sulfur as sulfate, and (5) a carbonate concentration of 23.7 mg/L CO_3^{-2}. The carbonate concentration was calculated based on an open system with a carbon dioxide partial pressure of 0.5 atm.

As presented in Table 2, all solubilities of heavy metals are lower in typical landfill leachate (pH = 7.5 or the presence of sulfide or carbonate) than in pure water. The exception is the hydroxide compounds because the concentration of hydroxide ions at pH 7.5 is insignificant. For both Fe and Cu, the oxidized states form less soluble compounds with sulfide than the reduced states. However, in the presence of sulfide, the Fe and Cu would be present in reduced form. The solubilities of sulfate compounds are orders of magnitude higher than the sulfide compounds. For example, the equilibrium Pb^{+2} concentrations in the presence of a sulfide and sulfate are 1.8 x 10^{-19} mg/L and 10.8 mg/L, respectively. However, no study presented in Table 1 detected Pb concentrations greater than 0.19 mg-Pb/L. As for the solubilities of the carbonate species, none exceeded the US drinking water standards except Pb. Fe can form hydroxide compounds in the +2 or the +3 oxidation state. As with the sulfide compounds, the oxidized form has a lower solubility. Except for Cr^{+3} and Fe^{+3}, the solubility of hydroxides in pure water is similar to that in leachate. Only the solubility of cadmium hydroxide exceeds the US drinking water standard, but none of the studies in Table 1 exhibit concentrations within two orders of magnitude of this theoretical solubility. In general, metals are present in landfill leachate at concentrations in excess of their equilibrium concentrations in the presence of 10 mg/L S^{-2} or 23.7 mg/L CO_3^{-2}, suggesting that complexation plays a role in determining metals concentrations in leachates.

Revans et al. (1999) used a metals solubility model called PHREEQEC to study the presence of Zn, Cd, and Cr in landfilled waste. This model estimated that in an anaerobic landfill, Zn and Cd are often present as sulfide precipitates, while in aerobic landfills they form carbonate precipitates. Cr was predicted to form hydroxide precipitates in anaerobic landfills. In young landfills, organic and inorganic ligands greatly increase solubility, but the metal concentrations are governed by other processes such as sorption and precipitation once the landfill becomes methanogenic and reduced conditions are established (Revans et al., 1999). If a landfill becomes aerobic, these conditions may change, and a release of metals could result (Revans et al., 1999) as discussed in Section IV.

Table 2. Equilibrium Concentrations of Heavy Metals in Distilled Water and Typical Leachate[1]

Oxidation State	Compound	Solubility Product Kso	Molecular Weight g/mol	Solubility Pure Water mg/L	Solubility 10 mg/L sulfide mg/L	Solubility 23.7 mg/L carbonate mg/L	Solubility 10 mg/L sulfate mg/L	Solubility pH of 7.5 mg/L
2	ZnS	3.16E-23	65.38	3.68E-07	6.63E-15			
2	CdS	8.51E-28	112.4	3.28E-09	3.07E-19			
2	PbS	2.69E-28	207.2	3.40E-09	1.79E-19			
2	CuS	7.94E-37	63.54	5.66E-14	1.62E-28			
2	FeS	5.01E-18	55.84	1.25E-04	8.97E-10			
2	HgS	1.00E-50	200.5	2.01E-20	6.43E-42			
2	NiS	1.41E-24	58.7	6.98E-08	2.66E-16			
2	MnS	1.41E-15	54.94	2.06E-03	2.49E-07			
3	Fe_2S_3	1.00E-88	111.68	1.10E-13	1.01E-34			
1	Cu_2S	2.00E-47	127.08	2.17E-11	1.26E-22			
2	$HgSO_4$	4.79E-07	200.5	1.39E+02			3.08E+02	
2	$PbSO_4$	1.62E-08	207.2	2.64E+01			1.08E+01	

Oxidation State	Compound	Solubility Product Kso	Molecular Weight g/mol	Solubility Pure Water mg/L	Solubility 10 mg/L sulfide mg/L	Solubility 23.7 mg/L carbonate mg/L	Solubility 10 mg/L sulfate mg/L	Solubility pH of 7.5 mg/L
2	$ZnCO_3$	2.09E-11	65.38	2.99E-01		3.E-04		
2	$CdCO_3$	2.51E-13	112.4	5.63E-02		7.E-06		
2	$PbCO_3$	3.98E-14	207.2	4.13E-02		2.E-05		
2	$CuCO_3$	1.38E-10	63.54	7.47E-01		2.E-02		
2	$NiCO_3$	1.38E-07	58.7	2.18E+01		2.E+01		
2	$FeCO_3$	3.16E-11	55.84	3.14E-01		4E-03		
1	Cu_2CO_3		127.08	0.00E+00		0.00E+00		
3	$Fe_2(CO_3)_3$		111.68	0.00E+00		0.00E+00		
3	$Cr(OH)_3$	1.58E-30	52	8.09E-04				2.59E-06
3	$Fe(OH)_3$	1.00E-38	55.84	7.75E-06				1.76E-14
2	$Mn(OH)_2$	1.58E-13	54.94	1.87E+00				1.87E+00
2	$Zn(OH)_2$	6.31E-18	65.38	7.61E-02				7.61E-02
2	$Fe(OH)_2$	1.66E-14	55.84	8.97E-01				8.97E-01
2	$Cd(OH)_2$	4.47E-15	112.4	1.17E+00				1.17E+00
2	$Cu(OH)_2$	4.79E-20	63.54	1.45E-02				1.45E-02
2	$Ni(OH)_2$	1.66E-14	58.7	9.43E-01				9.43E-01

D. Redox Conditions

The common oxidation states of several metals often found in landfill leachate are presented in Table 3 (Merck Index, 1989; Holm and Christensen, 1997). Redox reactions will be discussed in Section IV.

Table 3. Common Oxidation States of Some Heavy Metals

Metal	Common Oxidation States
Cd	+2
Cu	+1, +2
Ni	+2
Pb	+2
Cr	+3, +6
Hg	0, +2
Zn	+2

IV. The Effect of Oxidizing Conditions on Metals Concentrations

Once refuse is completely decomposed under anaerobic conditions and the majority of the degradable carbon is consumed, it is theoretically possible that a landfill will gradually become aerobic as air diffuses into the site and oxygen is not consumed in the oxidation of organic matter. To our knowledge, there are no reports of a site turning aerobic, and we expect that this may only occur in near geologic time. However, in considering long term landfill stability, this issue must be considered as a number of biological and chemical reactions will occur that may affect the mobilization of metals.

Several authors express concern that future oxidation of a landfill could result in increased heavy metals concentrations in leachate (Reinhart and Grosh, 1999; Flyhammar et al., 1998; Martensson et al., 1999). There are three mechanisms by which metals concentrations could increase in an aerobic landfill including (1) the oxidation of metal sulfides to metal sulfates that are generally more soluble (Table 2), and (2) the increased complexation capacity of oxidized humic acids relative to reduced humic acids (Martensson et al., 1999). In addition, (3) the oxidation of sulfides would result in sulfuric acid production, and a pH decrease will increase metals solubilities.

Under anaerobic conditions, sulfates in refuse are reduced to sulfides that form insoluble precipitates with most metals (Christensen et al., accepted for publication). However, as the refuse mass turns aerobic, sulfide can be oxidized to sulfate. Sulfate compounds are considerably more soluble than the corresponding metal sulfides which may increase metal mobilization. For example, pK_{so} values for PbS and $PbSO_4$ are 27.6 and 7.73, respectively.

In addition to sulfates, metal solubility will be influenced by the oxidation-reduction potential (E_h). As the E_h of the refuse increases, the solubilities of some metals increase. For example, Charlatchka and Cambier (2000) showed that the concentrations of Pb, Zn, and Mn decreased 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.

Under aerobic conditions, additional refuse decomposition will occur as some of the lignocellulosic substrate is more degradable under aerobic conditions relative to anaerobic conditions. The resulting CO_2 production has been reported to result in a pH decrease in both refuse (Revans et al., 1999) and harbor sediments (Calmano et al., 1993), and it is well known that metal dissolution is enhanced at low pH. Another factor that will exert downward pressure on refuse pH is the oxidation of reduced S, N, and Fe 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 provide additional buffer capacity, although carbonate release has not been demonstrated experimentally (Calmano et al. 1993).

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 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 as refuse is oxidized. Finally, iron (III) hydroxides and oxyhydrates will be 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 metals release from landfills. Their model considered the oxidation of refuse, organic matter, humic substances, and metal sulfide precipitates, as well as pH buffering associated with calcite dissolution. They considered oxygen diffusion into landfills with a range of landfill geometries and degrees of saturation and predicted that heavy metals mobilization will not occur for thousands of years. The current challenge is to verify this result experimentally.

There have been a limited number of studies on metals concentrations in decomposed refuse that has become aerobic. To test the effects of changing conditions on leachate composition, Flyhammar et al. (1998) used four experimental test cells that were constructed in 1975 on top of an old landfill. The test cells were originally used to monitor changes in leachate composition in a leachate-recycle reactor. In 1993, Flyhammar et al. began to sample the leachate from the anaerobically degraded waste and to collect data on Fe, Mn, Zn, Cu, Pb, Cr, Ni, and Cd. Table 4 presents metals concentrations in the leachate in 1975 compared to 1993. Each concentration decreased by a factor of 5 or more except for Cu.

Table 4. Leachate Concentrations Reported by Flyhammar et al., 1998[1]

	Concentrations mg/L	
Metal	1975 (23)	1993 (1)
Zn	10	2.2
Cu	0.07	0.04
Pb	0.2	0.02
Cr	0.25	0.01
Ni	0.5	0.05
Cd	0.015	0.006

1. The concentrations represent the mean leachate concentrations during the year. The number of samples is given in parentheses next to the year.

To quantify the amount of reactive metal species in the twenty-year-old waste, the samples were boiled in concentrated HNO_3 for two hours. Only 5% of Pb, Cr, Ni, and Cd were released, while less than 15% and 50% of Cu and Zn were released, respectively (Flyhammar et al., 1998). Thus, acid-extractable metals remain after 20 years, but their release to leachate does not appear to increase.

Revans et al. (1999) studied the effects of changing redox conditions on heavy metal concentrations. Six test cells, 500 mm in length and 100 mm in diameter, were constructed with previously decomposed, anaerobic, domestic waste to which Cd, Zn, and Cr were added. Once anaerobic conditions were reestablished in the columns, three of the six were converted to aerobic conditions by aeration with moist air. Water was added weekly to all six reactors to produce leachate that was collected and analyzed, but not recirculated. No significant increase in metals concentrations was detected despite the heavy loads applied at the beginning. In fact, less than 1% of each metal was leached by any test column, anaerobic or aerobic. Although metal sulfides would have been oxidized by the aerobic conditions that were established, processes such as sorption, carbonate precipitation, and hydroxide precipitation apparently retained the metals (Revans et al., 1999). In a similar experiment, Flyhammar and Hakansson (1999) reported no evidence of Zn or Fe sulfide dissolution upon aeration though changes in solid species of Zn and Fe did occur.

To test the effects of aerobic conditions on landfill leachate, Martensson et al. (1999) conducted an aeration experiment similar to Revans et al. (1999). Samples of degraded waste from a codisposal landfill in Sweden were placed in reactors and treated with either nitrogen gas or air to establish anaerobic and aerobic conditions. After 82 days, leachate was produced by adding 100 mL of distilled water to the waste followed by rotation and centrifugation. Aeration did result in a decrease in pH from 8.52 to 8.18. An increase in Cd, Ca, Cr, and Zn concentrations did occur in aerobic waste, but Al, Fe, and Mo decreased (Martensson et al., 1999).

Metals in landfills are generally assumed to be bound by sulfides. However, the landfill sampled by Martensson et al. (1999) was reported to contain only enough sulfur to bind to 5% of metals present which could explain why the results differed from those reported by Revans et al.

(1999). In Martensson's work, other mechanisms such as complexation, carbonate precipitation, and hydroxide precipitation apparently attenuated the metals, as aeration did not result in a significant metals release. Another potential explanation for the difference between the results of Martensson et al. and Revans et al. is that the bottom of the columns used by Revans may have remained anaerobic.

In conclusion there is little evidence to date to indicate that the concentrations of heavy metals will increase as a landfill becomes aerobic though the results of two studies on this issue are not consistent. Processes such as sorption, precipitation, and increased cation-exchange capacity are believed to inhibit the mobility (Martensson et al., 1999). Revans et al. (1999) suggest that "as the environment changes from methanogenic to oxic, the speciation of the metals may change, but the aqueous concentrations in leachate are not anticipated to increase significantly."

V. Conclusions and Future Research Needs

Metals in both the colloidal and dissolved fractions contribute to overall solubility (Christensen et al., accepted for publication; Jensen et al., 1999; Jensen and Christensen, 1999). The interaction of four physical and chemical processes determines heavy metal concentrations in landfill leachate. Complexation could increase solubility, but apparently competing processes such as sorption and precipitation are sufficient to maintain low metals concentrations in landfills (Christensen et al., accepted for publication; Revans et al., 1999; Reinhart and Grosh, 1998). Once a landfill reaches the methanogenic phase, heavy metal concentrations are generally low (Revans et al., 1999).

Theoretically, metals could be released if a landfill becomes aerobic. Evidence of this has only occurred in some laboratory experiments, while others indicate that the problem is insignificant. To our knowledge, there are no reports in the literature of a landfill becoming aerobic, and field studies are required to identify any such landfills. Given the conflicting experimental evidence on the effects of aeration on heavy metals, additional research is recommended to conclusively address the potential for higher metals concentrations in leachate after long periods of time.

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Appendix C. Toxicity of Landfill Leachate

Table of Contents

List of Tables

List of Appendices

Toxicity of Landfill Leachate

Contents

I. Introduction

The objective of this report is to identify and compare available toxicity tests and their applicability to monitoring landfill leachate. Such tests may be useful as part of a testing program to determine whether a landfill can be considered stable. In the case of landfill leachate, stability should infer that it can be safely released to the environment. Leachate should not deplete oxygen sources, inhibit species survival, or emit dangerous chemicals.

To date, there are no federal requirements that landfill leachate undergo toxicity testing before it is released to the environment. Research has shown, however, that many landfill leachates do contain toxic and sometimes carcinogenic compounds (Baun et al. 1999, 1998; Brown and Donnelly, 1988; Christensen et al., accepted for publication). Other studies have shown that leachate from a municipal solid waste landfill can be more toxic than leachate from hazardous waste landfills (Brown and Donnelly, 1988; Clement et al., 1996; Schrab et al., 1993). The higher toxicity may be attributed to the mixture of compounds present in a municipal landfill, as opposed to the uniform waste streams that a particular hazardous waste landfill will normally receive. Even though large scale co-disposal is no longer practiced, small quantity generators such as small businesses and households do continue to dispose of hazardous chemicals in municipal landfills (Brown and Donnelly, 1988).

Releasing toxic leachate into the environment may potentially contaminate groundwater supplies, harm aquatic ecosystems, and transfer toxic compounds into other media such as air and soil. Chemical analyses can identify the presence of specific dangerous compounds, but the identification is not enough to assess the danger (Baun et al., 1999; Christensen et al., accepted for publication; Clement et al., 1996; Cheung et al., 1993). In addition, chemical analyses only detect a small percentage of the toxic compounds present (Baun et al., 1999; Galassi et al., 1988; Bihari and Batel, 1994). Bioassay toxicity tests that measure effects on organisms can detect toxicity even when the contaminants are not identified by chemical analysis. They reflect the cumulative and synergistic effects of all the compounds (Baun et al., 1999; Christensen et al., accepted for publication; Lambolez et al., 1994).

This review begins with a discussion of general considerations for bioassay tests. Next, individual assays with aquatic invertebrates, bacteria, algae, fish, and duckweed are discussed and compared. Studies in which bioassays were used to measure the toxicity of landfill leachate are then summarized. This is followed by a discussion of future research requirements for assessing the toxicity of landfill leachates.

II. General Principles of Bioassays

In this section, some general principles applicable to bioassay testing are described. This is followed by descriptions of specific bioassays that utilize a number of test species.

Bioassays use organisms to measure the potential adverse effects of pollutants on the environment. They are capable of measuring not only organism death, but also physiological, metabolic, and reproductive changes. Fish, crustaceans, bacteria, and algae are among the most common organisms used for testing complex environmental samples such as landfill leachate for toxicity (Christensen et al., accepted for publication). Many authors suggest using a battery of tests with organisms from different trophic levels to increase toxicity detection (Baun et al., 1999; Christensen et al., accepted for publication; Lambolez et al., 1994; Maciorowski and Clarke, 1980; Clement et al., 1996; Rand, 1980; Schrab et al., 1993; Cheung et al., 1993). Bacteria are typically used for genotoxicity testing. Genotoxicity should be analyzed if downstream ground water supplies are used for human consumption (Baun et al., 1999).

Most bioassays report toxicity as LC50 or EC50, the lethal or effective concentrations that affect 50% of the population (Rand, 1980). A lower number for the LC50 or EC50 means a leachate is more toxic relative to another sample because a lower concentration will show lethal or adverse effects in 50% of the test population (Baun et al., 1999). Results may also be expressed as toxic units, 1 / (LC50 or EC50). The advantage of using the toxic unit is that a higher value signifies a higher toxicity.

Different types of tests are available to assess different environmental impacts. Acute toxicity tests expose organisms to at least four different concentrations of a sample for 24 to 96 hours to determine the concentration that causes death in some fraction of organisms. These tests may be conducted under static or flow through conditions. The static tests are easier to perform but have a characteristic disadvantage: the toxicant may volatilize, degrade, or react with metabolic products. Thus, the concentration of a toxicant may change with time. For flow-through conditions, water is only cycled through once, so the concentrations of toxicant and dissolved oxygen remain constant (Rand, 1980).

There are several limitations to bioassay testing. First, organisms with special ecological needs should not be used if these requirements cannot be met in the laboratory. For example, test organisms may build up a tolerance to a toxicant and could pass this acclimation on to future generations. Therefore, some organisms should be isolated from pollutant exposure for breeding purposes. Second, it is often difficult to distinguish organism death from immobility. Immobile organisms may recover, and other organisms may die after the test is complete (Maciorowski and Clarke, 1980).

A third limitation to acute bioassays is that in the environment, organisms are usually exposed to sublethal concentrations due to dilution of the water body. Because acute tests measure lethal concentrations, they will not reflect chronic or latent effects due to lower concentrations. Chronic tests expose the organisms for their entire life cycle and measure effects on hatching, growth, reproduction, and survival. Chronic tests use the flow through method and test a minimum of four concentrations (Rand, 1980).

Once the toxicity of a complex mixture has been established, it may be difficult to determine which compound or compounds are contributing to the toxicity. Fractionation divides the sample into groups of similar compounds. Separate bioassays can then be performed to identify the responsible fraction. A toxic group can either be analyzed for chemical constituents or refractionated and tested first (Parkhurst, 1982).

An additional limitation of bioassays is that results may reflect harmful interactions between test organisms and test conditions rather than toxic organic compounds (Baun et al., 1999). For example, the high conductivity of landfill leachate may contribute a significant amount of toxicity to test organisms and may mask the toxicity of organic chemicals. One solution is sample fractionation by solid phase extraction. This method extracts organic chemicals onto a resin while other constituents such as inorganic ions remain in the effluent. The organics are removed from the resin with acetone, which is then evaporated from the sample with either nitrogen gas or atmospheric air. Toxicity tests can then be conducted on the organic fraction separately (Baun et al., 1999). If ions are the only contributor to toxicity, then dilution may suffice.

A. Frequently Used Bioassay Toxicity Tests

To date, bioassays have typically been used as a research tool rather than a regulatory device. They have proven useful for testing wastewater treatment plant effluents, leachate and hazardous wastes, groundwater, and oil drilling waste fluids (Bulich, 1984). The California State Pollution Control Laboratory uses the acute 96 hr, fathead minnow test to measure the toxicity of an extract of solid hazardous wastes. If the waste exhibits a 96 hr LC50 of less than or equal to 750 mg/L, then the substance is considered toxic. Another laboratory reportedly has designated a 15 minute Microtox EC50 or a 96 hr fathead minnow LC50 of 750 mg/L or less as toxic (Bulich, 1984). Some municipalities monitor the toxicity of industrial waste streams prior to treatment at wastewater plants with bacterial assays (Anderson et al., 1984).

Bailey et al. (2000) used *Ceriodaphnia dubia* and *Selenastrum capricornutum* for whole effluent toxicity tests of sewage treatment plants in Australia. The toxicity to these organisms was attributed to pesticides and ammonia. 15 of the 18 treatment plants exhibited acute or chronic toxicity to *Ceriodaphnia dubia*, and 2 decreased growth rates of *Selenastrum capricornutum*. Vlaming et al. (2000) used *Ceriodaphnia dubia, Selenastrum capricornutum,* and fathead minnows (*Pimephales promelas*) for whole effluent toxicity tests in California. Goodfellow et al. (2000) used *Ceriodaphnia dubia, Daphnia magna*, and fathead minnows to test the ion toxicity of 2900 test solutions. They found that *Ceriodaphnia dubia* was more sensitive than *Daphnia magna* and fathead minnows to all the salt solutions. Rutherford et al. (2000) used rainbow trout (*Oncorhynchus mykiss*), *Daphnia magna, Vibrio fischeri, Ceriodaphnia dubia,* and

Selenastrum capricornutum to test the toxicity of landfill leachate after pretreatment in a wetland treatment system.

Bioassay toxicity tests generally use aquatic invertebrates, bacteria, microalgae, fish, and higher plants to estimate the effects of a contaminated waste stream on ecological habitats and humans. This section will describe each test and give examples of its application.

1. Aquatic Invertebrates

Aquatic invertebrates include crustaceans such as *Daphnia magna*, *Daphnia pulex, Thamnocephalus platyurus,* and *Ceriodaphnia dubia*. Toxicity tests use these animals to assess the toxicity of various pollutants by exposing the organisms to solutions of varying concentrations and counting the number of dead or immobile organisms (Buikema et al., 1980). There are many advantages to using these species in toxicity testing. They have relatively short life spans, 2-4 weeks, so it is possible to test for chronic, reproductive, and life cycle effects. It is also possible to produce genetically uniform cultures, and hatching times can be controlled to produce a stock of organisms with the same age. Also, sensitivity varies throughout the life cycle, and once the most sensitive stage is identified, it may be advantageous to use this form in the toxicity test (Maciorowski and Clarke, 1980).

One factor supporting the use of invertebrates is their niche in aquatic ecosystems. If aquatic invertebrates are disturbed by a toxicant, then the surrounding ecosystem may also feel the effects either directly or indirectly by loss of a food source. Invertebrates occupy a low trophic level, and their disturbance may exhibit a chain effect throughout the ecosystem. The pollutants that affect aquatic invertebrates may also disturb aquatic food industries such as oysters, mussels, and clams or destroy the food source for marketable fish and waterfowl. Also invertebrates are diverse, and they will show a range of toxic effects. Another advantage of the use of invertebrates is that they are large enough to eliminate the need for microscopes but small enough so that laboratory space for organism growth is inexpensive relative to fish. (Maciorowski and Clarke, 1980).

a. *Daphnia magna* and *Daphnia pulex*

Daphnia magna and *Daphnia pulex* are the two most common aquatic invertebrates used in toxicity testing. *Daphnia magna* is usually chosen for its larger size, which is easier to see, but it is not as common in the waters of North America as *Daphnia pulex*. According to Buikema et al. (1980), *Daphnia magna* should be used to test waters with a calcium carbonate concentration of greater than 120 ppm while *Daphnia pulex* is better suited for waters with a calcium carbonate concentration of around 45 ppm.

Problems with variation in *Daphnia* tests may be a result of experimental design factors such as available food, nutrients, light, and organism age. Four types of food are commonly used, but the algae *Chlamydomonas reinhardi* is the most nutritional for *Daphnia*. *Daphnia* that are fed this algae produce more young, and their test results are less variable. Even post exposure to this algae can increase survival rates after exposure to certain toxicants. Poor nutrition in general may increase sensitivity. Also, if a toxicant is absorbed onto food before consumption, then the exposure rate will increase (Buikema et al., 1980). Nutrition conditions of females will affect the survival of her offspring. Vitamins that enhance egg production are vitamin B12, thiamine, and pantothenic acid. Measuring the health of the culture by reproductive observations can comprehensively rate the overall test conditions. High ratios of the number of eggs per female or the number of young to number of adults signify a healthy population (Buikema et al., 1980).

Infant *Daphnia* cannot survive more than 48 hours without food. Buikema states that animals of 2.0 - 2.5mm in length are the least susceptible to starvation. Any test where food is not administered for more than 24 hours should be tested on adults within this size range. Also, infant *Daphnia* are more sensitive to toxicants because they are smaller and molt frequently (Buikema et al., 1980).

Daphnia are particularly sensitive to nitrates and amines. Resin columns used for deionizing water may contribute amines, and metal distillation columns may introduce toxic metals. Thus, water used for bioassays with *Daphnia* should be glass distilled (Buikema et al., 1980).

Daphnia have been used to analyze various water samples. St. John et al. (1994) analyzed silicon grout leachate with *Daphnia magna* to measure the LC50. Baun et al. (1999, 1998) used

Daphnia magna to assess the toxicity of leachate polluted groundwater. To test the acute toxicity of industrial solid wastes, Lambolez et al. (1994) used a 24hr *Daphnia magna* test. They also used *Daphnia magna* for a 28 day chronic toxicity test to measure reproductive inhibition. Assmuth and Penttilae (1995) and Atwater et al. (1983) tested landfill leachates with *Daphnia magna* acute toxicity tests.

b. *Thamnocephalus platyurus* and *Ceriodaphnia dubia*

In contrast to tests with *Daphnia*, less information is available on the use of *Thamnocephalus platyurus* and *Ceriodaphnia dubia* in toxicity testing. Clement et al. (1997, 1996) used both organisms to test landfill leachate and found that both were more sensitive than *Daphnia magna*. Fuick et al. compared the acute toxicity of *Ceriodaphnia dubia* to brook trout and fathead minnow. *Ceriodaphnia dubia* were more sensitive to mine runoff than both species of fish (Fuick et al., 1991).

2. Luminescent bacteria

Two types of luminescent bacteria are typically used in acute toxicity tests. Each test measures bioluminescent suppression of the bacteria *Photobacterium phosphoreum* (Microtox) or *Vibrio fischeri* after exposure to different concentrations of toxicants. These tests are particularly sensitive because they measure suppression of metabolic activity along with organism death. The tests measure the light output versus sample dilution which is then used to calculate the EC50, the sample concentration that yields a 50% reduction in light output after 15 or 30 minutes of exposure. Baun et al. (1999) used the EC20 value to signify the "beginning toxicity."

Ribo and Kaiser (1983) compared the toxicities of chlorophenols, chlorobenzenes, and 3-para substituted phenols using Microtox and several other bioassays. They found that Microtox testing times of 15 to 30 minutes gave the most consistent results. Statistical correlations between Microtox assays and assays with other organisms are presented in Table 1. The results show that Microtox results are similar to toxicities measured with several species of fish and crustaceans.

Table 1. Correlations Between Microtox Toxicity and Alternate Toxicity Assays[1]

Chemical	Species	r^2

chlorophenols	brown trout	0.92
	bluegill	0.77
	guppy	0.89
	shrimp	0.68
chlorobenzenes	rainbow trout	0.74
	minnows	0.80
	guppy	0.86
	Daphnia magna	0.83
3-para substituted phenols	rainbow trout	0.93

1. Data represent the correlation coefficient between the specified assay and the Microtox assay for a series of comparisons.

3. Algae

Algae are the primary producer in many aquatic habitats and therefore an important organism to test. *Raphidocelis subcapitata*, also referred to as *Selenastrum capricornutum*, is the most frequently used algae in toxicity tests. Mayer et al. (1998) suggest using low algal densities in the test for more accurate results. One method of testing microalgae is by measuring chlorophyll fluorescence, which determines photosynthetic activity by measuring the light emitted from the surface of a leaf after a plant has been exposed to potential toxicants (Judy et al., 1991). This method does not require prefiltering of waste water, so the chance of loosing toxicity is reduced greatly, nor does it require the expensive equipment associated with other methods (Mayer et al. 1997).

As with *Daphnia*, several variables have been found to increase variability in algal test results, including pH, light, nutrients, and nitrogen. Mayer et al. (1998) reported that pH affects toxicity by governing the types of some species present: generally neutral species are more toxic than charged species. The nitrogen source, NO_3^- or NH_4^+, had no observed effect, and neither did temperature. It was also concluded that the light should be strong enough to produce light-saturated photosynthesis to reduce test variability and that nutrients should be present in excess (Mayer et al. 1998). If these variables are not standardized, then results of different tests will be difficult to compare (Mayer et al., 1998; Mayer et al., 1997; Arensberg et al., 1995).

4. Fish

In the natural environment, acute effects will usually be seen in fish in approximately four days (Rand, 1980). Acute effects can be estimated with bioassays by counting the number of dead organisms. Chronic tests, which are more complex, measure effects on reproduction, respiration, locomotion, or metabolism. Several problems arise with fish bioassays. Fish require more lab space than the smaller organisms discussed previously not only because they are larger, but also because their toxicity tests require more organisms. Another problem is that fish do not demonstrate toxicity responses to compounds with low toxicity. Fish should therefore only be used with highly toxic effluents (Birge et al., 1982).

5. Duckweed

The duckweed plant (*Lemna minor*) consists of one small, leaflike frond with a single root. The frond floats either on the water surface or just below (Bishop and Perry, 1979). Bioassays are performed by subjecting the plants to water with varying concentrations of a test material, usually for seven days (Cowgill and Milazzo, 1989).

Lemna minor is an important organism to test because of its universal presence and importance in the food chains of several aquatic species and waterfowl (Cowgill and Milazzo, 1989). The advantages of this plant are its small size, ease of growth, high surface area to volume ratio, and ease of cloning. The bioassay may be operated under static or flow through conditions. Flow through tests should maintain a constant light intensity and flow rate of nutrients and test material (Bishop and Perry, 1979).

Several growth factors can be measured to determine seven day EC50 values: root elongation, frond number, plant number, and dry mass (Cowgill and Milazzo, 1989). Bishop and Perry (1979) describe the advantages and disadvantages of each method. Even though the root is usually the most sensitive part of a plant, it is a very small fraction of duckweed biomass. Also the upper part of the duckweed plant can still grow even when root growth has stopped. A disadvantage of both root length and dry weight measurements is they can only be measured at the end of the test. Also, dry weight measurements reflect not only plant growth but also starch content, a parameter that does not reflect sensitivity. The advantage of using frond count to measure growth rate is that it can be measured everyday. However, they suggest only using data from days four through seven to allow for stabilization of the growth rate (Bishop and Perry 1979). Cowgill and Milazzo (1989) however, suggest measuring the dry mass after the test duration because it is less expensive and time consuming than counting fronds, numbers of plants, or measuring root lengths.

B. Comparison of Bioassays

Several authors have compared the sensitivities of a number of bioassays concurrently. Each species varies in its costs, simplicity, and tolerance to different constituents.

Baun et al. (1999, 1998) found that bioassays using algae were more sensitive than crustaceans or luminescent bacteria, but that bacterial tests are easier and faster. Lambolez et al. (1994) used the microalgae *Raphidocelis subcapitata / Selenastrum capricornutum* for a growth inhibition test on industrial solid waste leachates and report that this algae is more sensitive than bacteria and daphnids by 2 orders of magnitude. An advantage of algal tests is they are not as sensitive to ammonia toxicity and particulates as animal bioassays. They are cost effective, sensitive, "rapid," and ecologically important (Arensberg et al., 1995). Also, as reported above, algae may be more sensitive than animals to chemical pollutants (Mayer et al., 1997).

In general, bacteria may not be as sensitive to toxicants as other organisms, but their assays are easier, faster, and less expensive to test (Ronnepagel et al., 1995; Baun et al., 1998; Kaiser et al., 1994; Ribo and Kaiser, 1993). Kaiser et al. found results of Microtox tests to correlate well with tests on several fish species such as the fathead minnow, guppy, and bluegill, and the water flea, *Daphnia magna*. Clement et al. (1997) report that COD is toxic to luminescent bacteria but does not describe the COD composition.

Algal and bacteria tests are relevant to environmental damage studies because they may reveal the impact of leachate on surface water systems (Baun et al., 1999). In studies by Baun et al.(1999, 1998), algae and luminescent bacteria were the most sensitive organisms to the leachate polluted groundwater; *Daphnia magna* however were not sensitive to the samples tested (1999).

Daphnia are reported to be more sensitive than most species of fish (Birge et al., 1982; Stubblefield and Maki, 1982). For instance, a discharge from an organic chemical plant caused five fish kills from 1970 to 1975. The EPA tested the toxicity of the effluent using *Daphnia magna* and *Lepomis macrochirus* (bluegill sunfish) to establish permit levels (Peltier, 1982). The LC50 values presented in Table 2 show that Daphnia were more sensitive to lower concentrations of effluent than the bluegill sunfish.

Table 2. LC50 Values of Chemical Plant Effluent

Species	Time/Condition	LC50 mg/L
Lepomis macrochirus	96 h, flow through	0.8
Lepomis macrochirus	96 h, static	0.39
Daphnia magna	48 h, static	0.25
Daphnia magna	48 h, static, duplicate	0.24

Atwater et al. compared toxicities of *Daphnia* to rainbow trout and sockeye salmon and reported that *Daphnia* tests are more cost effective, faster, and may be more sensitive to certain constituents than both types of fish (1983). *Daphnia* are also less sensitive to pH toxicity than fish, but more sensitive to zinc (Atwater et al., 1983). Clement et al. (1997) report that alkalinity is also toxic to *Daphnia*.

Only Clement et al. (1997) compared the sensitivity of the crustaceans *Daphnia magna*, *Thamnocephalus platyurus*, and *Ceriodaphnia dubia*. *Daphnia magna* was the least sensitive of the three species.

Duckweed is not widely used in bioassays, but it is relatively sensitive. Bishop and Perry (1979) found duckweed to be more sensitive than both bluegill sunfish (*Lepomis macrochirus*) and *Daphnia magna* by two orders of magnitude. Clement et al. (1996) show that in general duckweed is more sensitive than algae.

Table 3 ranks the Microtox, *Daphnia magna*, fish, duckweed, and algal bioassays according to cost, simplicity, and sensitivity. While two reports have shown duckweed to be among the most sensitive test species, tests with luminescent bacteria and *Daphnia* are more commonly used. This is justified by their relative cost, their simplicity, and reasonably good correlations between these assays and assays with higher organisms. While the use of algae in bioassays is less common, it does allow an assessment of photosynthetic activity which is otherwise neglected.

Table 3. Bioassay Comparisons of Cost, Simplicity, and Sensitivity[1]

Cost	Simplicity	Sensitivity
Microtox	Microtox	Duckweed
Daphnia magna	*Daphnia magna*	Algae
Algae	Algae	Microtox
Duckweed	Duckweed	*Daphnia magna*
Fish	Fish	Fish

1. Cost is ranked least expensive to most expensive, simplicity is ranked easiest to most difficult, and sensitivity is ranked from most sensitive to least sensitive.

C. Genotoxicity

Genotoxic compounds, or mutagens, cause DNA mutations to either the chromosome or the gene. If these mutations cause cancer, then the compounds are classified as carcinogens. Usually the initial compound is not genotoxic: it becomes genotoxic only after it has been metabolized by some organism (Zahn et al., 1994) After conversion, the compound will either be released to the environment or bioaccumulated (Bihari and Batel, 1994). If the test species is not capable of metabolizing the compound into its toxic form, the toxicity will not be detected. If other organisms in the ecosystem are able to biologically transform the compound, then toxicity will be exerted. Therefore, if a bioassay species is not capable of metabolizing a chemical into the mutagenic or carcinogenic compounds, then an enzyme such as liver extract must be added to the bioassay (WHO, 1985).

Chromosomal damage is a change in the number or structure of chromosomes. Damage to the gene is a mutation of the gene itself (WHO, 1985). The primary damage tests measure changes in the DNA such as single or double strand breaks, strand unwinding, protein crosslinks, or strand crosslinks. Secondary tests measure chromosomal changes and micronucleus formation (Bihari and Batel, 1994).

The DNA repair test assumes that if cells are undergoing repair, some damage must have occurred. This repair, also called unscheduled DNA synthesis, can be detected by measuring increased levels of a biological compound called thymidine (WHO, 1985). Another DNA repair test compares two strains of a bacteria: one is capable of DNA repair; the other is not. If a chemical alters the DNA of the bacteria, then the strain that is not capable of repair will die. Bacteria commonly used in this test are *Escherichia coli*, *Bacillus subtilis*, *Proteus mirabilis*, and *Salmonella typhimurium*. The DNA repair tests using *Escherichia coli* and *Salmonella typhimurium* are documented to be 79% and 65% effective at detecting mutations, respectively (Davis et al., 1980).

The secondary tests are relatively simple, but they are not accurate enough to rely on independently (WHO, 1985).

Bacteria are the most commonly used organisms to detect mutations (WHO, 1985). They are capable of detecting forward and reverse mutations and DNA repair. To increase sensitivity, more than one strain is analyzed per test. The assays consist of the bacteria, the test chemicals, and an enzyme such as the S-9 liver extract (WHO, 1985).

There are several advantages of using bacteria in genotoxicity assays. For instance, their DNA is a single, circular molecule that can be easily accessed by chemical penetration through the cell wall (WHO, 1985). Because bacteria are genetically less complex than other organisms, it is easier to manipulate their DNA to produce more sensitive organisms (Davis et al., 1980). Also bacteria are capable of testing complex mixtures, they are typically 90% effective, and millions of organisms can be tested at once. These tests are relatively simple and may only require a few hours to administer (Davis et al., 1980).

The disadvantages are that it is difficult to extrapolate the mutagenicity of bacteria to the dangers for humans or animals. Also, bacteria cannot detect the carcinogens that do not cause DNA damage. Some chemicals only cause chromosomal damage so bacteria, which do not contain chromosomes, will not be useful for testing these mutagens (WHO, 1985). Another disadvantage is that there is no standard cut-off for whether or not a test is positive or negative. Therefore results from different laboratories will vary. Other sources of variation include the use of different strains and different methods of storage, culture, and enzyme addition. Interpretation of data is also a problem. If the organisms die due to acute toxicity, then the mutagenic effects will not be apparent and the sample should be fractionated. Positive tests may reflect problems with the test conditions and not genotoxicity. Also a negative test result does not assure that a chemical is not mutagenic (Brusick, 1979).

Genetic mutations can occur in the forward or reverse direction, to a new form or back to a previous form. The Ames test is a three day bioassay that uses four strains of the bacteria *Salmonella typhimurium*. Prior to testing, the four strains are mutated so they can no longer produce histidine, an amino acid essential for their survival. When the test begins, the bacteria are exposed to the test material along with enough histidine for survival. Once the small amount of histidine is used up, only those bacteria that mutate back to the original form will be capable of producing their own histidine. These mutated bacteria will continue to grow and form colonies while the others will not (Davis et al., 1980). At the end of the test, the number of colonies are counted (WHO, 1985). The test is positive according to Davis et al. (1980) if one of the four strains mutates. The Ames test is reported to be 85% - 90% effective at distinguishing between genotoxic and nongenotoxic compounds (Zakrzewski, 1991; WHO, 1985). *Escherichia coli* can also be used in a similar test to measure reverse mutation (Davis et al., 1980).

The *umu* test only uses one strain of *Salmonella typhimurium.* When a DNA-damaging agent "turns on" the *umu* operon, cancer and mutation can result, and β-galactosidase is produced. The *umu* test screens for this process by measuring the increased levels of β-galactosidase after a 2 hour incubation period (Oda, 1985).

1. Comparison of Genotoxicity Tests

Bacterial genotoxicity assays are the most widely used because the organisms give consistent results and are easy to alter genetically. In various experiments, the Ames test has shown a 90% agreement with known carcinogens and non carcinogens. The 10% uncertainty can be alleviated by using a battery of tests that will detect damage to both chromosomes and genes (WHO, 1985).

Oda chose the *umu* test as an alternative to the Ames test because it is easier and less expensive. He evaluated 38 chemicals; 31 of which were known animal carcinogens which had undergone prior testing using the Ames test. Of the 38 chemicals tested, 26 that had previously been found positive by the Ames test were also detected by *umu*. Six had negative toxicities to both tests. Of the remaining six, the Ames test detected one and the *umu* test detected five. Thus the *umu* test detected more toxic samples than the Ames test, with the advantage of only using one strain of bacteria (Oda, 1985).

For complete genotoxic assessment, Hansen (1994) suggests using DNA damage tests on both fish and mussels along with an *umu* test. Oda (1985) suggests that the *umu* test is better for screening environmental samples that contain amino acids and nutrients. *Umu* is also faster than the Ames test (Zahn et al., 1994).

III. The Applicability of Toxicity Bioassays to Landfill Leachate

Landfill leachate is a complex mixture that may contain thousands of chemicals. Chemical analysis of all constituents is not a feasible method to determine environmental dangers. Bioassays are a more practical alternative. Several authors have demonstrated the toxicity of landfill leachates to bioassay organisms (Baun et al., 1999; 1998; Brown and Donnelly, 1988; Christensen et al., accepted for publication; Clement et al., 1996; Schrab et al., 1993; Wong, 1989; Assmuth and Penttilae, 1995; Atwater et al., 1983; Cheung et al., 1993). Clement et al. (1996) state that "leachates of domestic wastes were significantly more toxic than those of pure industrial wastes" and that "the most toxic leachates were found for landfills receiving hazardous industrial wastes mixed with domestic wastes." This section of the report summarizes some of the experiments that have tested landfill leachate toxicity with bioassays.

Lambolez et al. (1994) and Clement et al. (1997, 1996) both used a large battery of toxicity tests to evaluate various leachates. Lambolez et al. (1994) evaluated leachates from 15 industrial solid wastes, and Clement et al. (1997, 1996) tested 27 samples of landfill leachates from 14 landfills. Clement et al. (1997, 1996) found that though toxicity varied between test species, landfills, and types of wastes, the various tests did give similar toxicity rankings.

Lambolez et al. (1994) used *Photobacterium phosphoreum*, a luminescent bacteria; *Daphnia magna*, an aquatic invertebrate; and *Raphidocelis subcapitata* / *Selenastrum* *capricornutum,* an algae, to test 15 industrial solid waste leachates that he divided into five categories, C1-C5, defined below.

C1 wastes from industrial waste water treatment plants

C2 paint wastes

C3 metal industry wastes

C4 incinerator wastes

C5 soil contaminated with PCB's

Each test only used two out of the three organisms. Table 4 compares the sensitivities of the organisms by listing a ratio of how many times each organism was the most sensitive out of the tests run. If results could not be compared, then the data was not included in the ratio. For instance if one of the tests was not performed or if the ranges of EC50's given for the organisms overlapped.

Table 4. Ratios of Most Sensitive Test Organism[1]

	C1	**C2**	**C3**	**C4**	**C5**
Test Species					
Acute Tests					
Microtox	2:3	2:2	1:2	1:1	1:1
Daphnia magna	1:3	0:2	1:2	0:1	0:1
Chronic Tests					
Algae	3:5	3:3	2:2	4:4	0:0
Daphnia magna	0:5	0:3	0:2	0:4	0:0

1. Data represent the number of times an organism was most sensitive out of the total number of tests conducted.

Lambolez et al. (1994) reported that algae are more sensitive than the bacteria or *Daphnia,* but no test directly compared algae to bacteria. It is evident however, that both the algae and bacteria were more sensitive than *Daphnia* for the solid waste leachates tested (Lambolez et al., 1994).

Clement et al. (1997) studied landfill leachate from 14 landfills. He also divided the wastes into categories.

C1 domestic wastes

C2 domestic wastes mixed with non-hazardous industrial wastes

C3 non-hazardous industrial solid wastes

C4 hazardous industrial solid wastes

C5 hazardous industrial solid wastes plus domestic wastes

C6 artificial leachates generated from common household wastes

Species from three trophic levels were compared and the results are presented in Table 5. Again, the table only compares complete data sets. If a test was not performed on one of the species in the trophic level, that data was omitted. Both Tables 4 and 5 only list which tests were the most sensitive for the leachate tested. Almost every leachate exhibited an EC50 to each species.

Table 5. Ratios of Most Sensitive Test Organism[1]

	C1	**C2**	**C3**	**C4**	**C5**	**C6**
Test Species						
Acute Tests						
Decomposers:						no
Vibrio fischeri	1:11	0:2	0:2	2:2	0:5	sufficient
Spiristomum ambiguum	10:11	2:2	2:2	0:2	5:5	data
Consumers:						
Brachionus calyciflorus	0:11	0:2	0:2	0:4	0:5	no
Thamnocephalus platyurus	6:11	1:2	1:2	2:4	3:5	sufficient
Daphnia magna	0:11	0:2	0:2	0:4	0:5	data
Ceriodaphnia dubia	5:11	1:2	1:2	2:4	2:5	
Chronic Tests						
Producers:						
Scenedesmus subspicatus	1:12	0:2	0:2	2:2	1:5	0:2

Lemna minor	11:12	2:2	2:2	0:2	4:5	2:2

1. Data represent the number of times an organism was most sensitive out of the total number of tests conducted.

Table 5 shows that the algae, *Scenedesmus subspicatus,* was less sensitive than duckweed, *Lemna minor,* for all types of wastes tested except C5, a combination of hazardous industrial solid wastes and domestic wastes. The aquatic invertebrates *Thamnocephalus platyurus* and *Ceriodaphnia dubia* were consistently more sensitive than *Daphnia magna* and *Brachionus calyciflorus,* a rotifer, for all categories of waste tested. The protozoa, *Spiristomum ambiguum,* were more sensitive than the luminescent bacteria, *Vibrio fischeri,* for all wastes except C4, the hazardous, industrial solid wastes (Clement et al., 1997). More tests are necessary to verify these sensitivity rankings.

Wong (1989) measured the LC50 of landfill leachate using the freshwater fish tilapia (*Sarotherodon mossambicus*). He tested five concentrations and used a 96 hour incubation period. He attributed the toxicity to ammonia and organics such as phenols and humic and fulvic acids. He purposefully chose tilapia because they are resistant to polluted environments; bioassays utilizing this organism may underestimate the toxicity to other organisms in the ecosystem (Wong 1989).

Assmuth and Penttilae (1995) compared *Daphnia magna* results from a 24 hr test without dilution to a 48 hr standard test in the evaluation of leachate from 35 mixed-waste disposal sites. Chemical analyses were also performed. The average 24 hr test gave slightly lower toxicity results than the 48 hr test and therefore may underestimate acute toxicity. There was no consistent correlation between toxicity and chemical composition, but in general toxicity correlated with increases in chloride content, ammonia, and hardness. In some cases, the metals concentrations reported to be toxic to *Daphnia magna* did not exhibit toxic effects (Assmuth and Penttilae, 1995). Though hardness is an indicator of leachate strength, it is not toxic, and it can decrease the toxicity of some substances such as heavy metals (Assmuth and Penttilae, 1995).

In a summary table, Assmuth and Penttilae (1995) present the EC50 values for *Daphnia magna* exposed to landfill leachate in five studies. The EC50's range from 2.3% to 89% leachate.

Thus, landfill leachates from different sites show a wide range of toxicity.

Atwater et al. (1983) measured the acute toxicity of landfill leachate using *Daphnia pulex*, rainbow trout (*Salmo gairdneri*), and sockeye salmon (*Oncorhynchus nerka*). Results from *Daphnia* compared well to fish. Rainbow trout are highly sensitive to municipal solid waste landfill leachates, and they found *Daphnia* to be nearly as sensitive with an R^2 value of 0.87. Increasing the pH of acidic leachates decreased the toxicity for fish by 100 times and *Daphnia* by only 10 times. The 24 hr *Daphnia* test was not as accurate as the 48 hr test. However, the 48 hr test determined slightly higher toxicities than the 96 hour test. Atwater et al. conclude that zinc is more toxic to *Daphnia* than fish and that *Daphnia pulex* is a good indicator of leachate toxicity (Atwater et al., 1983).

Schrab et al. (1993) tested leachate from four municipal solid waste landfills. Landfill 1 was an old codisposal site that is now a Superfund site. Landfills 2 and 3 are engineered, municipal solid waste landfills; Landfill 2 was closed in 1988. Landfill 4 is an operating landfill that receives both municipal and non-hazardous industrial wastes. Groundwater adjacent to Landfill 3 was also tested. They used three genotoxicity tests: the Ames test with *Salmonella*, a DNA repair bioassay with *Bacillus subtilis*, and a chromosome damage test with *Aspergillus nidulans* as well as a Microtox assay. Schrab et al. (1993) state that two of these municipal solid waste landfills were as toxic as co-disposal or hazardous waste landfills reported in literature.

The Ames test did not detect any mutagenic leachates. However, chemical analyses verified that three of the four samples contained compounds in concentrations usually detected positive by the Ames test. This discrepancy can be attributed to the acute toxicity of the leachate to the *Salmonella* organism. Schrab et al. (1993) suggest that the acute toxicity masked the mutagenic effects by killing the organisms before they were genetically altered and that sample fractionations prior to testing may eliminate this effect. The *Bacillus* test only detected one positive mutagenic leachate which was Landfill 4, the operating landfill that receives both municipal and industrial waste. The *Aspergillus* had two positive genotoxic responses: Landfills 1 and 2. Landfill 2, the municipal landfill that receives primarily residential waste, was more toxic than Landfill 1, the codisposal hazardous waste landfill. These tests suggest that the municipal landfill is potentially more genotoxic than the codisposal landfill that receives hazardous waste (Schrab et al., 1993).

The Microtox showed acute toxicity of all four landfill leachates: 30 minute EC50 values were 2.0%, 3.6%, 6.0%, and 3.9% leachate for landfills 1 - 4, respectively. Landfill 1 was twice as toxic as 2 and 4 which were both two times as toxic as 3. Chemical analysis identified "chronically toxic volatile, semi-volatile, or non-volatile organic compounds" in all of the leachate and groundwater samples (Schrab et al., 1993).

Cheung et al. (1993) studied the acute toxicity of two landfill leachates to four algal species. 96 hr EC50 values are presented in Table 6. The species used in order of increasing sensitivities were *Chlorella pyrenoidosa*, *Scenedesmus sp.*, *Chlorella vulgaris*, *and Dunaliella tertiolecta.* The Junk Bay Landfill leachate was more toxic to all four species than Gin Drinker's Bay Landfill. Cheung et al. attributed the toxicity to organic compounds and ammonia. If algal systems are used for leachate treatment, comparing the sensitivities of algaes to leachate is important. For example, *Chlorella* and *Scenedesmus* have high growth and nutrient removal rates. They are commonly used for wastewater treatment and various laboratory experiments (Cheung et al., 1993).

The Gin Drinker's Bay leachate was less toxic than Junk Bay leachate. The growth of *Chlorella pyrenoidosa* and *Scenedesmus* was not inhibited. *Chlorella vulgaris* and *Dunaliella tertiolecta* showed growth inhibition at 50% and 25% leachate dilutions, respectively. The Junk Bay leachate showed growth suppression at 5% leachate for *Dunaliella tertiolecta*, 20% for *Chlorella vulgaris* and *Scenedesmus*, and at 50% all three organisms ceased growth. Only *Chlorella pyrenoidosa* could grow in the 50% Junk Bay leachate. Cheung states that "landfill leachates in general are toxic to algae" (Cheung et al., 1993).

Table 6. Acute Toxicity to Four Algae Species

Algae Species	96 hr EC50 (% volume)	
	Junk Bay	Gin Drinkers Bay

Chlorella pyrenoidosa	23	>50
Chlorella vulgaris	7	33
Scenedesmus sp.	20	>50
Dunaliella tertiolecta	<5	21

A study in 1989 at the Highway 101 Landfill in Nova Scotia, determined that the landfill leachate was toxic to rainbow trout (*Oncorhynchus mykiss*), brook trout (*Salvelinus fontinalis*), and *Daphnia magna* (Rutherford et al., 2000). To remedy the situation, two aerobic lagoons and a constructed wetland system were used to pretreat the leachate. In 1994, acute toxicity test were used to test the effectiveness of the new treatments: a 96 hr, static rainbow trout test; a 48 hr, static *Daphnia magna* test; and a 15 min Microtox test with *Vibrio fischeri*. To test chronic effects, 7 day *Ceriodaphnia dubia* and 76 hr *Selenastrum capricornutum* test were performed. The Ames test was also used to test for genotoxicity. No acute effects were detected. Chronic effects were only seen in *Selenastrum capricornutum*. The Ames test detected slightly mutagenic effects. Though the leachate did exhibit toxicity, the treatment systems did reduce the effects (Rutherford et al., 2000).

In another study, Brown and Donnelly (1988) tested leachates for specific contaminants and used toxicity data for each contaminant to estimate the risk. Though they did not use bioassays, "toxic and cancer causing chemicals were found in the leachates from all 58 landfills... irrespective of the type of waste which they receive."

A common problem with testing landfill leachate and leachate-polluted ground water is that they have different characteristics than the industrial and municipal waste water that are commonly tested with bioassays: leachate is anaerobic, has high alkalinity, chloride, ammonium, and reduced iron. Most toxicity tests employ aerobic organisms, so the leachate must be aerated prior to testing. Aeration can lead to a loss of compounds through oxidation, volatilization, or co-precipitation with manganese and iron oxides (Baun et al., 1999). However, aerobic organisms are useful because they reveal how leachate may affect surface and ground water ecosystems (Baun et al., 1999).

Leachate generally has a high concentration of ions that can contribute toxicity to organisms. Ionic strength calculations can compare ion concentrations of leachate to the growth medium of a specific organism and can be calculated two ways (Snoeyink and Jenkins, 1980). The first equation incorporates species concentration and charge:

$$\mu = \frac{1}{2}\sum_{i}(C_i Z_i^{2})$$

where C_i equals the concentration of ionic species i (moles/liter), Z_i equals the charge of species i, and μ is unitless.

For example, *Daphnia* growth media contain 192 mg/L $CaSO_4$, 192 mg/L $NaHCO_3$, 8 mg/L KCl, 120 mg/L $MgSO_4$ (St. John, 1994) which has an ionic strength of 0.012 according to the above equation. Baun et al. determined ionic concentrations in the leachate polluted groundwater closest to the Vejen Landfill to have ionic concentrations of 400 mg/L Na^+, 205 mg/L K^+, 118 mg/L Ca^{+2}, 44 mg/L Mg^{+2}, 189 mg/L NH_4^+, and 722 mg/L Cl^-. This corresponds to an ionic strength of 0.036 (Baun et al., 1999). This polluted groundwater has an ionic strength three times that of the growth medium. Assuming this groundwater has diluted the leachate strength, leachate will have an even greater ionic strength. Tolerance experiments need to be conducted on various organisms to determine the maximum ionic strength that will not adversely affect tests results.

A second calculation converts measured conductance to ionic strength (Snoeyink and Jenkins, 1980):

$$\mu = 1.6 * 10^{-5} * (\text{specific conductance } (\mu\text{mho/cm}))$$

At two landfills in Denmark, groundwater aquifers polluted with leachate were sampled at various distances from the landfills. At 15m from the Grindsted Landfill the specific conductance was 1900 µS/cm, or 1900 µmho/cm. At 10m from the Vejen Landfill the specific conductance was 3210 µS/cm, or 3210 µmho/cm (Baun et al., 1999). These convert to ionic strengths of 0.0304 and 0.0514, respectively. The equation using specific conductance estimates a higher ionic strength for the Vejen Landfill than the first equation.

In addition to ionic strength, alkalinity should also be compared. Baun et al. found that clean water samples with an alkalinity of 6.6 meq/L inhibited algal growth (*Selenastrum capricornutum*) by 10%. An alkalinity of 24.7 meq/L suppressed growth by 50%. Groundwater samples collected close to Grindsted and Vejen Landfills had alkalinities of 22.1 meq/L and 21.8 meq/L, respectively (Baun et al., 1999). These alkalinities would not be suitable for this algal test. Other organisms should also be tested for tolerance to alkalinity.

In summary, Lambolez et al. (1994) reported that leachate toxicity was dependent on ammonia, alkalinity, and COD. Clement et al. (1997) suggested that ammonia especially affects duckweed and daphnids, and that alkalinity enhances ammonia toxicity and may even exert direct toxicity to daphnids. Wong (1989) attributed tilapia toxicity to ammonia and organics. Assmuth and Penttilae (1995) report that in general toxicity correlated with chloride content, ammonia, and hardness. A study by Atwater et al. (1983) asserts that zinc is more toxic to *Daphnia* than fish, and that *Daphnia* are less sensitive to pH toxicity. Cheung et al. (1993) attribute the algal toxicity of the leachate to organic compounds and ammonia. Ammonia, organic matter, and heavy metals found in leachate may be toxic to fish, *Daphnia*, and algae (Wong, 1989). Finally, Christensen et al. (accepted for publication) summarized a number of toxicity assays on landfill leachate, and these tests are presented in Appendix 1. Relationships between leachate composition and EC50's as presented by a number of researchers, are presented in Appendix 2. Appendix 3. lists references used in Appendix 1 and 2.

IV. Future Research Needed

Much research is still needed in the area of leachate toxicity testing. Standard methods need to be developed that will incorporate the general qualities of leachate into variables such as ion concentrations, alkalinity, and pH. For bioassays where feeding is acceptable, standard methods need to be developed for the concentration and frequency of food and nutrients.

There is a lack of information on the aquatic invertebrates *Thamnocephalus platyurus* and *Ceriodaphnia dubia*. More testing is needed to compare their ion and alkalinity tolerances and organic compound sensitivities to *Daphnia*. Similar tolerance tests are also needed for luminescent bacteria and algal species since these organisms are widely used in bioassays. Marine organisms may be the most tolerant to ion and alkalinity toxicity, and their sensitivity to landfill leachate should also be investigated.

Sample fractionation can ease the determination of which constituents in leachate are responsible for toxicity and point to the most effective treatment options. This technique may be the only way to mechanistically evaluate a complex mixture such as leachate.

Finally, the use of bioassay data in evaluating environmental releases must be established. For instance, which assays should be used to test landfill leachate? What is the maximum EC50 concentration for each assay that can be safely released? If a leachate sample exceeds the EC50, what procedures will be followed? How will dilution into the receiving water body be taken into consideration? Once a leachate passes the toxicity bioassays, how often should it be monitored? These questions must be answered before bioassay testing can be used in landfill management.

V. Summary

Landfill leachates can be harmful to the environment even though toxic compounds may be present in low concentrations. A number of organic compounds that are present in leachate are known to exert toxicity and some are carcinogenic (Baun et al., 1998). Chemical analyses alone cannot identify all micropollutants present, their combined effects, or the environmental risks (Baun et al., 1999; Christensen et al., accepted for publication; Clement et al., 1996; Cheung et al., 1993). In addition, toxicity cannot be consistently correlated with chemical analyses or predicted from the nature of the wastes (Assmuth and Penttilae, 1995; Lambolez et al., 1994; Atwater et al., 1983). Bioasssays may be helpful in identifying cumulative risks and ecological hazards easily and inexpensively. However, the results of bioassays must be analyzed to determine the extent to which a positive result represents a significant threat to human health and the environment.

Clement et al. suggested that three trophic levels should be represented in a bioassay test battery. Producers such as the algae *Scenedesmus subspicatus*, consumers such as the invertebrate *Daphnia magna*; and decomposers such as the bacteria *Vibrio fischeri* are an example. This combination will theoretically detect 90% of toxic samples (Clement et al. 1996).

Maciorowski and Clarke (1980) suggested including organisms present in the receiving water body and imitating the environmental conditions such as current, temperature, alkalinity, and salinity. In addition to testing multiple species, separate bioassays are needed to determine acute, chronic, and genotoxic effects.

Several authors attributed toxicity of landfill leachate to ammonia and organic compounds (Wong, 1989; Cheung et al., 1993; Lambolez et al., 1994; Clement et al., 1997; Assmuth and Penttilae, 1995). Organisms that can tolerate high ion and alkalinity contents will best reflect organic toxicity. Sample fractionation may help identify which classes of compounds are contributing to toxicity, the appropriate treatment methods, and the significance of any potential risks.

VI. Conclusions

- The Microtox assay has proven to be relatively simple and cost effective. It has shown strong correlations with other organisms such as fish and has been found more sensitive than tests with *Daphnia magna*.
- The algal assays are more complicated and expensive, but could be a strong indicator of leachate effects on photosynthetic organisms and are generally more sensitive than bacteria (Microtox) or daphnids.
- The relative sensitivity of fish bioassays does not justify the associated increase in cost.
- If a landfill leachate is discharged to a potential source of drinking water, then a genotoxicity test such as the Ames test should also be conducted.
- Because leachate generally contains high concentrations of ions, alkalinity, and ammonia, the sensitivity of the bioassays to these factors must be determined.
- Research is required to understand how to relate bioassay test data to the release of leachate with low concentrations of organic compounds and ammonia.

Appendix 1. A Summary of Toxicity Assays on Landfill Leachate

Biological species (number of species) used in toxicity tests for assessment of toxicity of landfill leachate.

Type of landfill	No. of landfills	F	C	D	A	B	G/M	Other	Study
Municipal	1	2	1						Ernst et al. (1994)
Municipal	1					1*			Nohava et al. (1995)
Municipal	4					1*	3		Schrab et al. (1993)
Municipal	7	2	1						Cameron and Koch (1980) Atwater et al. (1983)
Municipal/mixed	2						4		Helma et al. (1996)
Municipal/mixed	3			1		1*		2	Devare and Bahadir (1994)
Municipal/industrial	14		3	1	1	1*		2	Clément et al. (1996)
Municipal/industrial	14			1					Clément and Merlin (1995)
Mixed	35		1						Assmuth and Penttilae (1995)
Mixed	1	1							Kaur et al. (1996)
Mixed?	2				4				Cheung et al. (1993)
Mixed?	1	1							Wong (1989)
Sanitary	19	1	1			1*			Kross and Cherryholmes (1993)
Sanitary	1	1**							McBride et al. (1979)
Sanitary	1	1	1		1	1*			Plotkin and Ram (1984)

F: Fish, C: Crustacean, D: Duckweed, A: Algae, B: Bacteria, G/M: Genotoxicity/mutagenicity, Other: Protozoans, Rotifers, Plants. *: Luminiscent bacteria (Microtox), **: Acute and chronic/sublethal tests

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Appendix 2. Relationships Between Leachate Composition and EC50

Relationships between toxicity of different landfill leachates and physico-chemical parameters. estimated by stepwise multiple linear regression.

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

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

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Appendix D. A Review of Methods to Estimate Leachate Production From Closed Landfills

Table of Contents

List of Tables

List of Figures

A Review of Methods to Estimate Leachate Production Rates From Post-Closure Landfills

I. Introduction

Both the quantity and composition of leachate must be defined to predict the impact of a leachate release on groundwater or surface water systems. The objective of this section is to estimate leachate production from a traditional landfill after placement of the final cover. It should be noted that the data presented in this section are based on traditional landfills. Once estimates of leachate production rates have been developed, they may be used in concert with leachate quality data for evaluation of contaminants such as BOD, COD, and ammonia and to analyze the environmental impact of a leachate release during and after the post-closure monitoring period.

Several methods to estimate leachate quantity are presented and compared in this chapter. The first method uses data that have been published on the quantity of leachate present in leachate collection systems and in leachate detection systems underlying a liner system. These data were used to estimate leachate collection efficiencies for a number of liner configurations. Additional studies that have provided estimates of the number of defects per area of liner were also reviewed. These data were then used to calculate flowrates through liner systems based on an assumed head on liner. The HELP model (Schroeder and Peyton, 1995) was also used to estimate leachate production based on the local climate and characteristics of a landfill.

II. Flowrates through Landfill Liners

A. A. Typical Flowrates and Liner Efficiencies

Liners or liner systems may be composed of one or more of the following materials: flexible membrane liner (FML), geosynthetic clay liner (GCL), or compacted clay layer (CCL). A Subtitle D liner, which is the basis for the estimates developed here, includes a composite liner consisting of compacted clay and a FML. Under Subtitle D, the final cover may be no more permeable than the liner system. Thus, a final cover typically includes a vegetative layer, a drainage layer, and a composite liner. Water that is not removed by evapotranspiration or the drainage layer passes through the cover, percolates through the waste, and becomes leachate. A leachate collection and removal system (LCRS) is used to remove most of the leachate. The liner system below the LCRS protects the environment. However, some leachate may permeate this liner and enter either the leachate detection system (LDS), if present, or the environment. Landfills that use a LDS are referred to as double lined landfills because there is typically a secondary liner below the LDS.

Gross et al. (1997) used the flowrate of liquid into the LDS to infer cover efficiencies and to estimate infiltration rates as a percent of annual precipitation. Data were presented for three liner systems: (1) FML, (2) FML/GCL, and (3) FML/CCL. Although liner system (3) is most applicable for this analysis because it conforms to Subtitle D requirements, data on all three liners are analyzed. There is a lack of data focusing on cover efficiencies that incorporate FML's or GCL's, but it is evident that they limit infiltration compared to a CCL. This is mainly because a CCL tends to desiccate and crack (Gross et al., 1997).

Gross et al. (1997) studied a database that included 194 cells at 54 double lined landfills. The primary components of the double liner systems that were studied included a FML, FML/GCL, or FML/CCL. The authors defined the apparent efficiency of the liners as

$$E_a\% = (1 - (\text{LDS flowrate} / \text{LCRS flowrate})) * 100\% \qquad \text{Eq. 1}$$

and listed representative values for various liners as presented in Table 1.

Table 1. Liner Efficiencies (Gross et al., 1997)

Liner Type	Efficiency %	Representative Value
FML	97.2 - 99.9	99, n = 15
FML/GCL	99.3 - 100	99.9, n = 24
FML/CCL	na[1]	99.9, estimated value

1. Gross et al. (1997) suggest that consolidation water may interfere with liner efficiency calculations because it contributes excess water to the LDS, so the efficiency calculation is not applicable.

These efficiencies are for a liner at the bottom of a landfill that typically has smaller slopes than final cover systems. The larger slopes will increase efficiency, so surface efficiencies may be higher. It is conservative to assume that they are the same (Gross et al., 1997).

Gross et al. (1997) and Bonaparte et al. (accepted for publication) suggest that other sources of LDS flow are clay consolidation, ground water infiltration, drainage layer compression, and water that enters during construction. They define E_a% as the apparent liner efficiency because these other sources of water may affect the calculated efficiency. If the only source of LDS flow is liner leakage, then the calculated value is the true efficiency E_t%. E_t% is site specific and typically changes with time.

Given the typical construction of landfill liners, which specifies 4 ft between the top of the groundwater table and the bottom of the liner system, most of the sources of water that could enter the LDS are likely to be insignificant. Ground water infiltration will only occur in sites where the liner is below the ground water table. Construction water and compression water will only enter the LDS during the early stages of landfill operation (Bonaparte et al., accepted for publication). Finally, consolidation water is not likely to yield a significant volume of water. Construction of CCL's will most likely specify an in-place water content (w) that is wet of optimum to achieve a low hydraulic conductivity. However, the degree of saturation of the CCL will be less than 100%. Consolidation of the clay layer first requires compression of the air voids to attain a 100% saturation condition, or the introduction of water under a high enough gradient to overcome capillary pressures and saturate the voids. Subsequently, the expulsion of water from the clay voids under the imposed waste loads can occur and consolidation can take place. Even if such consolidation were to occur, the volume of water reaching the LDS would be insignificant compared to flows measured in the field. For example, assuming 3mm diameter holes, a wetted zone of 10 times the hole diameter, a hole density of 10 holes per hectare, and consolidation settlement of 30%, we calculate that the clay will yield 1.27 L/ha of total water volume. Assuming this settlement takes place over 4 years after the CCL around the pinholes is saturated, the rate is 0.0009 L/ha/d. This calculation is presented in detail below. The result is several orders of magnitude less than the flowrate data summarized below (Bonaparte et al., accepted for publication).

Clay Consolidation Calculation

- Assume 3 mm diameter holes
- Assume 10 holes/hectare
- Assume a wetted diameter of 10 times the hole diameter = 30 mm
- Assume 2 ft of compacted clay thickness = 610 mm
- Assume a 30% volume change due to consolidation

The volume of water expelled, V_w, is

$$V_w = \text{wetted area} * \text{clay thickness} * \%\,\text{consolidation} * \text{number of holes/hectare} \qquad \text{Eq. 2}$$

$$V_w = (0.003\,\text{m} * 10)^2 * \frac{\pi}{4} * (0.6\,\text{m}) * (0.3) * (10\,\text{holes/hectare})$$

$$V_w = 0.00127\,\text{m}^3/\text{ha} = 1.27\,\text{L/ha}$$

- If the consolidation takes place over 4 years and the coefficient of consolidation is 0.848 ft^2/yr, then the rate is 0.0009 L/ha/d. Compared to LCRS flowrates of 100 L/ha/d, this value is orders of magnitude smaller.

Bonaparte et al. (accepted for publication) studied 31 double-lined cells at 14 landfills; all but four contain MSW. All 31 have FML primary liners, and the secondary liner is either FML or FML/CCL. Only 6 cells received final cover, each with a FML cap. They state that the efficiencies can be as high as 99%, but typically range from 90% to 99%. They also reviewed three other studies with FML primary liners; the secondary liner type was not specified for the three studies. Although double lined, the FML primary liner is not as restrictive a flow barrier as an FML/CCL used at a Subtitle D landfill.

Bonaparte and Gross (1993) summarize average LDS flowrates during periods of operation and compare liner systems with and with out QA/QC programs, as presented in Table 2. Data are presented as percent of landfills in the database with the given range of LDS flowrates. The database consists of 25 cells at 10 landfills from the East, Northcentral, Gulf Coast, and Midwest regions of the US. The impact of the QA/QC implementation is apparent here as 80% of landfills with QA/QC had less than 200 lphd of leachate production versus more than 1000 lphd for approximately 70% of landfills constructed without a QA/QC program.

Table 2. Percent of Landfills in the Bonaparte and Gross (1993) database with given LDS flowrates.

With QA/QC	With out QA/QC
40% < 50 lphd (5.3 gal/ac/d)	70% > 1000 lphd (107 gal/ac/d)
80% < 200 lphd (21.4 gal/ac/d)	
0% > 1000 lphd (107 gal/ac/d)	

Maule (1993) studied three upper Midwest landfills, each with a FML primary liner and formal QA/QC program, for up to 17 months. The average monthly LDS flowrates were 10, 11, and 21 lphd (1, 1.2, 2.2 gal/ac/d, respectively). The peak flowrates in the LDS were 31, 34, and 66 lphd (3.3, 3.6, 7 gal/ac/d, respectively). Here too, the liner is not as restrictive as a Subtitle D liner.

Tedder (1997) studied 22 cells at 8 double lined landfills in Florida with FML primary liners and formal QA/QC programs, monitored for 3 - 64 months. Data are presented in Table 3. They do not specify the periods of data collection.

Table 3. Leachate Flowrates for Landfills With FML Liners (Tedder, 1997)

Average LCRS	1100 - 91,000 lphd (118 - 9700 gal/ac/d)
Average LDS	14 - 4300 lphd (1.5 - 460 gal/ac/d)
Percent of landfills with given LDS flow ranges	23% < 50 lphd (5.3 gal/ac/d), 59% < 200 lphd (21.4 gal/ac/d), 14% > 1000 lphd (107 gal/ac/d)

Table 4 presents the average monthly LCRS and LDS flowrates for the 31 landfill cells during three periods: i) initial cell filling, ii) active cell filling, and iii) post Bonaparte et al., accepted for publication). These data are for cells with FML primary liners and from sites with no groundwater intrusion. Sites with and without QA/QC are included. Periods are defined as i) initial - the first few months of cell filling when there is not enough waste to impede precipitation flow directly into the LCRS, ii) active - cell filled with waste and daily and intermediate covers, iii) post closure - after the cell has received its final cover. The sites with 99.99% efficiency in period i, the maximum efficiency, 100% efficiency in period ii), and 99.71% efficiency in period three are three different sites that have a FML primary liner and a FML/CCL secondary liner. All three had a QA/QC program. The site with the lowest efficiency in period i, 83.08%, was a MSW site with a CSPE (chlorosulfonated polyethylene) primary liner, a PVC/CCL secondary liner, and no QA/QC. The site with the lowest efficiency in period ii), 61.3%, was a MSW site with a QA/QC that had a FML primary liner and a FML/CCL secondary liner. The site with the lowest efficiency in period iii, 2.8%, had a CSPE (chlorosulfonated polyethylene) primary liner, a PVC secondary liner, and no QA/QC.

Table 4. Average Monthly LCRS and LDS Flowrates From 31 Cells with FML Primary Liners (Bonaparte et al., accepted for publication)

Period	LCRS flowrate lphd (gal/ac/d)	LDS flowrate lphd (gal/ac/d)	Efficiency %
i	1475 - 111,031 (158 - 11,869)	4 - 2144 (0.43 - 229)	83.08 - 99.99
ii	103 - 16,224 (11 - 1734)	0 - 1603 (0 - 171)	61.3 - 100
iii	317 - 1306 (34 - 140)	1 - 328 (0.11 - 35)	2.80 - 99.71

LDS flowrate data were also presented according to LDS material and whether or not a QA/QC plan was implemented during construction. Tables 5, 6, and 7 present flowrate data and liner efficiencies. The efficiencies vary between periods because they are based on several sites, not one site over time.

Table 5. Data for a Sand LDS System With a Formal QA/QC and a FML Liner (Bonaparte et al., accepted for publication)

Period	LDS Flow lphd [gal/ac/d]	Efficiency %
i	6 - 290 (mean = 114, median = 90, n = 14) [0.64 - 31 (mean = 12.2, median = 9.6, n = 14)]	98.16 - 99.94 (mean = 99.29, median = 99.26, n = 11)
ii	1 - 140 (mean = 36, median = 17, n = 13) [0.11 - 15 (mean = 3.8, median = 1.8, n = 13)]	78.92 - 99.98 (mean = 97.91, median = 99.14, n = 13)
iii	2 -60 (mean = 25, median = 19, n = 4) [0.2 - 6.4 (mean = 2.7, median = 2.0, n = 4)]	92.25 - 99.97 (mean = 96.14, median = 96.29, n = 4)

Table 6. Data for a Sand LDS System Without a Formal QA/QC and a FML Liner (Bonaparte et al., accepted for publication)

Period	LDS Flow lphd [gal/ac/d]	Efficiency %
i	122 - 2144 (mean = 911, median = 740, n = 6) [13 - 229 (mean = 97, median = 79, n = 6)]	83.08 - 99.31 (mean = 94.15, median = 97.10, n = 4)
ii	66 - 1603 (mean = 681, median = 689, n = 7) [7 - 171 (mean = 72.8, median = 73.6, n = 7)]	73.56 - 99.23 (mean = 86.95, median = 86.45, n = 7)
iii	217 - 241 (mean = 229, median not given, n = 2) [23.2 - 25.8 (mean = 24.5, median not given, n = 2)]	41.93 - 83.49 (mean = 62.71, n = 2)

Table 7. Data for a Geonet LDS System With a Formal QA/QC and a FML Liner (Bonaparte et al., accepted for publication)

Period	LDS Flow lphd [gal/ac/d]	Efficiency %
i	5 - 439 (mean = 128, median = 34, n = 4) [0.5 - 46.9 (mean = 13.7, median = 3.6, n = 4)]	94.14 - 99.87 (mean = 98.31, median = 99.62, n = 4)
ii	2 - 358 (mean = 106, median = 23, n = 6) [0.2 - 38.3 (mean = 11.3, median = 2.4, n = 6)]	91.32 - 99.95 (mean = 96.86, median = 98.40, n= 6)

These efficiencies are for FML liners only; composite liners that incorporate CCL or GCL may have higher efficiencies. It is difficult to compare the efficiencies of FML and composite liners directly because the conditions at each site vary. Also, most efficiencies decrease with time because the LCRS flowrates decrease faster than the LDS flowrates. According to the Bonaparte et al. (accepted for publication), FML liners with QA/QC had efficiencies ranging from 91% - 100%. FML's should not be used alone if an efficiency of 90% is not acceptable. Rather, they should be supplemented with suitable clay as typical for Subtitle D landfills (Bonaparte et al., accepted for publication).

Bonaparte (1995) reports that average monthly leachate flows in the LCRS were up to 3400 lphd (363.5 gal/ac/d) for double lined landfills during the active period. Three years after closure and placement of a FML cover, the average was 70 lphd (7.5 gal/ac/d).

Table 8 describes the design of 187 cells considered in an EPA report by Bonaparte et al., (accepted for publication). Only 43 of the 187 cells have received final cover. The landfills in this database include MSW, HW, MSW and ash, coal ash, and C&D waste types, at Northeast, Southeast, and Western US locations. The efficiencies of liners made of FML, FML/GCL, and FML/CCL are evaluated (Bonaparte et al., accepted for publication).

Table 8. Percentage of 187 Cells With the Following Characteristics (Bonaparte et al., accepted for publication)

Liner	LDS
FML, 37%	Sand or gravel, 48%
FML/CCL or FML/GCL/CCL, 48%	Geonet, 52%
FML/GCL, 15%	No data

The efficiency data for FML liners was presented in Tables 4 through 7. This report does add that the 8 cells without QA/QC had average monthly LDS rates 1 to 2 orders of magnitude greater than the others (Bonaparte et al., accepted for publication).

The flowrate data for FML/GCL liners is based on a study of 28 cells at 7 landfills with up to 83 months of monitoring data. All of these cells had QA/QC, and the data are presented in Table 9. They do not include mean apparent efficiencies for these cells, only LDS flowrates. However, they suggest that FML/GCL liners can achieve 99.9% efficiency but will typically range from 99% to 99.9% (Bonaparte et al., accepted for publication).

Table 9. LDS Flowrates for Double Lined Landfills with a FML/GCL Liner (Bonaparte et al., accepted for publication)

Period	LDS Flowrate lphd (gal/ac/d)
i	0 - 290 (0 - 31)
ii	0 - 11 (0 - 1.2)
iii	0 - 2 (0 - 0.2)

The FML/CCL and FML/GCL/CCL liners are represented by 13 cells at 9 landfills with up to 121 months of monitoring data; all have QA/QC. For these liners it is suggested that consolidation water from the clay may affect the volume of LDS liquid, so the apparent efficiency might not estimate the true efficiency accurately (Bonaparte et al., accepted for publication). LDS flowrates are given in Table 10.

Table 10. LDS Flowrates for Double Lined Landfills with a FML/CCL or FML/GCL/CCL Liner (Bonaparte et al., accepted for publication)

Period	LDS Flowrate lphd (gal/ac/d)
i	10 - 1400 (1.1 - 150)
ii	0 - 370 (0 - 39.6)
iii	5 - 210 (0.5 - 22.4)

To evaluate the performance of composite clay liners, chemical concentrations had to be compared between the LCRS and LDS leachates. Bonaparte et al. (accepted for publication) state that most of the LDS flow for the sites, summarized in Table 10, can be attributed to consolidation water. This study examined the constituents sulfate, chloride, benzene, toluene, and xylene and estimated that the true liner efficiency ranged from 99% to 99.9% based on mass fluxes from the LCRS and LDS data. However, clay can act as a reactive barrier where cation exchange could take place as the leachate passes through. Thus, sulfate and chloride may not be suitable tracers.

This report also presents leachate generation data, which is expressed as LCRS flowrate (lphd). Table 11 presents the LCRS flowrates along with leachate as a percent of rainfall for various geographic locations in the US. (Bonaparte et al., accepted for publication).

LCRS flowrates decreased by a factor of 4 one year after closure, and by one order of magnitude 2 to 4 years after closure. Six years after closure, LCRS flowrates ranged from 5 - 1200 lphd (0.5 - 128.3 gal/ac/d) with a mean of 180 lphd (19.2 gal/ac/d). Nine years after closure these flowrates were negligible. These values are for the closed landfills in the study and are not specific to a certain liner type (Bonaparte et al., accepted for publication).

Table 11. LCRS Flowrates and Percent Precipitation for MSW Landfills, by Geographic Location (Bonaparte et al., accepted for publication)

Period	Geographic Location	LCRS Flow[1] lphd; [gal/ac/d]	Percent Precipitation[1]
i	NE SE W	1050 - 39,900 (10,200); [112 - 4260 (1090)] 1480 - 43, 700 (10,400); [158 - 4670 (1110)] no data	4 -160 (39) 5 - 157 (33) no data
ii	NE SE W	41 - 17,700 (3,530); [4.4 - 1890 (377)] 300 - 10,900 (2,930); [32 - 1160 (310)] 55 - 110 (83)[2];[5.9 - 11.8 (8.9)]	0.1 - 54 (13) 1 - 23 (8) 0.5 - 1 (0.7)[2]
iii	NE SE W	55 - 680 (400); [5.8 - 73 (43)] No data No data	0.2 - 3 (1)

1. Values are presented as range (mean).
2. Values only represent two cells from one landfill.

Tables 1 though 11 present field data from several studies; many do not specify whether or not a Subtitle D liner was used. The study that does focus on sites where a FML/CCL was present had LDS flowrates that ranged from 5 - 210 lphd (0.5 - 22.4 gal/ac/d) in the post closure period. Liner efficiencies are not given with this data (Bonaparte et al., accepted for publication). As a comparison, New York State has a 20 gal/ac/d operational limit through the primary liner system (Phaneuf and Vana, 2000). Other studies suggest that achieving liner efficiencies of 99% is possible although efficiencies of 90 - 99% are more reasonable. A sample calculation assuming 99% efficiency in the cover and liner follows.

Liner Efficiency Calculation

The amount of leachate reaching a LDS, or the environment in the absence of a LDS, can be calculated based on assumed liner efficiencies. A sample calculation follows, and the results are presented in Table 12.

Table 12. Landfill Water Balance

Annual Precipitation (in/yr)	40
Efficiency of Landfill Cover %	99
Efficiency of Bottom Liner %	99
Loss due to ET and RO %	60
Conversion factor (inches of rain to gal/ac/d)	74.395
Calculated Values:	
Annual Precipitation gal/ac/d	2975.8
Infiltration through vegetative layer gal/ac/d	1190.32
Infiltration through cap gal/ac/d	11.903
Flow into LCRS gal/ac/d	11.784
Flow into LDS (or the environment) gal/ac/d	0.119

The flow into the LDS, or the environment, is 0.12 gal/ac/d under the specified conditions. If the concentration of BOD is 100 mg/L, COD is 1000 mg/L, and NH_3-N is 250 mg/L, then the mass released will be 45.4 mg/ac/d of BOD, 454 mg/ac/d of COD, and 113 mg/ac/d of NH_3-N. If NH_3-N concentrations remain elevated at 750 mg/L, then the mass released will be 341 mg/ac/d. These concentrations will be analyzed in the chapter on surface water quality to look at the effects of leachate on surface water systems.

B. Liner Defects

Another approach to estimating leachate release is to estimate the number of liner defects and then calculate the resulting flowrate. Murray et al. (1995) estimate liner leakage rates through low permeability soil and composite liners. They also report leak estimates that have been found by other researchers and use them as input data to the HELP model. Based on modeling results, they estimate that FML liners reduce leakage through CCL liners by 62% - 73% with poor installation or 99% with good installation. Their data are presented in Table 13 as hole frequency per area.

Table 13. Measured Frequency of Holes in FML Liners (Murray et al., 1995)

Leak Estimate (holes / ha)	Reference
3 - 5; liner with good installation and QA/QC program	Giroud and Bonaparte, 1989
2.5 - 55 (mean = 35, n = 61)	Laine and Miklas, 1990
16 in bottom liners, 12 on side slope liners	Saleh, 1992
Mean = 26, range of 1 std dev = 5 - 127	Laine and Darilek, 1993
Mean = 3, range of 1 std dev = 0.61 - 14.8; with good QA/QC Mean = 125, range of 1 std dev = 25 - 612; with poor QA/QC These values are the result of the "Latin-Hypercube" statistical analysis on the values reported above.	Murray et al., 1995

Giroud et al. (1992) describe equations that estimate leakage rates through composite liners with a variety of defects, liquid heads, and coefficients of permeability. A composite liner consists of a geomembrane liner above a low permeability soil, which will be referred to as clay for convenience. The assumptions and final equations for estimating leakage rates are summarized below.

When water enters a defect in a geomembrane, it flows radially around the hole for some distance and then begins to flow through the clay. The radial area covered by the spread of water is called the wetted area. The radius of the wetted area is largely dependent on the quality of contact between the geomembrane and the clay. Criteria that describe good contact conditions are few wrinkles in the geomembrane, well-compacted clay, and smooth-surfaced clay. Poor contact conditions are the presence of a critical number of wrinkles in the geomembrane, clay that does not have a smooth surface and was not well compacted (Giroud et al., 1992).

The factors that most significantly affect flow through a composite liner are size and number of defects, hydraulic conductivity of the clay, and the head of liquid on top of the geomembrane (Giroud et al., 1992).

These equations assume that (1) the liquid head above the defect remains constant and operates under hydrostatic conditions, (2) the wetted area remains saturated, and (3) the liquid is water. Flow through the geomembrane due to permeation is neglected (Giroud et al., 1992).

Table 14 lists the equations for flowrate and wetted area under specified conditions. The equations for small holes are based on both theoretical analyses and large-scale model tests. The equations for long defects are based on equations derived for small defects and have not been verified with actual data (Giroud et al., 1992).

The head of liquid above the wetted area is at a maximum below the defect because the soil carries the same head as the geomembrane itself. It decreases radially toward the perimeter of the wetted area. Thus the hydraulic gradient also decreases radially from the center. The values i_{avg} and i^*_{avg} account for this decrease (Giroud et al., 1992).

To calculate the flowrate when the liquid head is large, a value of R, the radius of wetted area, is needed. For the case of small liquid head, R was calculated based on the assumption that hydraulic gradient equals one. For larger liquid heads, the hydraulic gradient is greater than one. Using the value of R for small liquid heads will give a larger, conservative estimate of flow (Giroud et al., 1992).

Table 14. Equations for Calculation of Leakage Through Composite Liners

Defect and liquid head	Quality of Contact between Geomembrane and Soil Liner	Equations	
Small, circular defect; Small liquid head	Good	$Q = 0.21 a^{0.1} h_w^{0.9} k_s^{0.74}$ $R = 0.26 a^{0.05} h_w^{0.45} k_s^{-0.13}$	Eq. 3 Eq. 4
Small, circular defect; Small liquid head	Poor	$Q = 1.15 a^{0.1} h_w^{0.9} k_s^{0.74}$ $R = 0.61 a^{0.05} h_w^{0.45} k_s^{-0.13}$	Eq. 5 Eq. 6
Small, square defect; Small liquid head	Good	$Q = 0.21 b^{0.2} h_w^{0.9} k_s^{0.74}$ $R = 0.26 b^{0.1} h_w^{0.45} k_s^{-0.13}$	Eq. 7 Eq. 8
Small, square defect; Small liquid head	Poor	$Q = 1.15 b^{0.2} h_w^{0.9} k_s^{0.74}$ $R = 0.61 b^{0.1} h_w^{0.45} k_s^{-0.13}$	Eq. 9 Eq. 10
Small, circular defect; Large liquid head	All Qualities	$i_{avg} = (1 + h_w/(2H_s \ln(R/R_o)))$	Eq. 11
Small, square defect; Large liquid head	All Qualities	$i_{avg} = (1 + h_w/(2H_s \ln(2R/b)))$	Eq. 12
Small, circular defect; Large liquid head	Good	$Q = 0.21 i_{avg} a^{0.1} h_w^{0.9} k_s^{0.74}$	Eq. 13
Small, circular defect; Large liquid head	Poor	$Q = 1.15 i_{avg} a^{0.1} h_w^{0.9} k_s^{0.74}$	Eq. 14
Small, square defect; Large liquid head	Good	$Q = 0.21 i_{avg} b^{0.2} h_w^{0.9} k_s^{0.74}$	Eq. 15
Small, square defect; Large liquid head	Poor	$Q = 1.15 i_{avg} b^{0.2} h_w^{0.9} k_s^{0.74}$	Eq. 16
Long defect	All Qualities	$i^*_{avg} = (1 + h_w/(H_s \ln(2R/b)))$	Eq. 17
Long defect	Good	$Q = 0.52 i^*_{avg}(B-b) b^{0.1} h_w^{0.45} k_s^{0.87} +$ $0.21 i_{avg} b^{0.2} h_w^{0.9} k_s^{0.74}$	Eq. 18
Long defect	Poor	$Q = 1.22 i^*_{avg}(B-b) b^{0.1} h_w^{0.45} k_s^{0.87} +$ $1.15 i_{avg} b^{0.2} h_w^{0.9} k_s^{0.74}$	Eq. 19

Variables:

Q = Rate of Leakage (m^3/s) a = Geomembrane hole area (m^2)

h_w = Head of liquid (m) k_s = Hydraulic conductivity of the clay (m/s)

R = Radius of wetted area (m) R_o = Radius of circular defect (m)

B = Length of long crack (m) b = Length of square defect or width of long crack (m)

H_s = Thickness of clay layer (m)

i_{avg} = Average hydraulic gradient in the soil beneath the circular portion of the wetted area (at the ends of the long defect and under wetted area of small defects)

i^*_{avg} = Average hydraulic gradient in the soil beneath the rectangular portion of the wetted area (along a long defect)

The equations for small liquid head assume that the hydraulic gradient in the soil is one and that leakage rates do not depend on the thickness of the soil layer. This assumption is valid for cases where the head on top of the soil approaches zero. In other cases, the average hydraulic gradient is used to calculate flow. Flow is assumed to be vertical (Giroud et al., 1992).

Figures 1 through 4 show flowrates under changing conditions for 3 mm and 10 mm diameter holes. These hole sizes were obtained from the mean values under good and poor conditions, respectively, as determined by Murray et al. (1995). All four figures assume a clay thickness of 2 ft (0.6 m) and a saturated hydraulic conductivity of 1 E-9 m/s. According to Giroud and Bonaparte (1989), landfills that do not accept liquid waste will have liquid heads less than 1 ft if a granular leachate collection system is used, or a few millimeters if a geosynthetic is used. Figures 1 and 3 show flowrates that result from 1 ft of head. Figures 2 and 4 show flowrates that result from 2 ft of head. As shown in the graphs, the flowrate is linear with respect to the number of defects due to the method of calculation, and it increases drastically from good to poor contact conditions. These data also illustrate the sensitivity of the equations to the head on the liner.

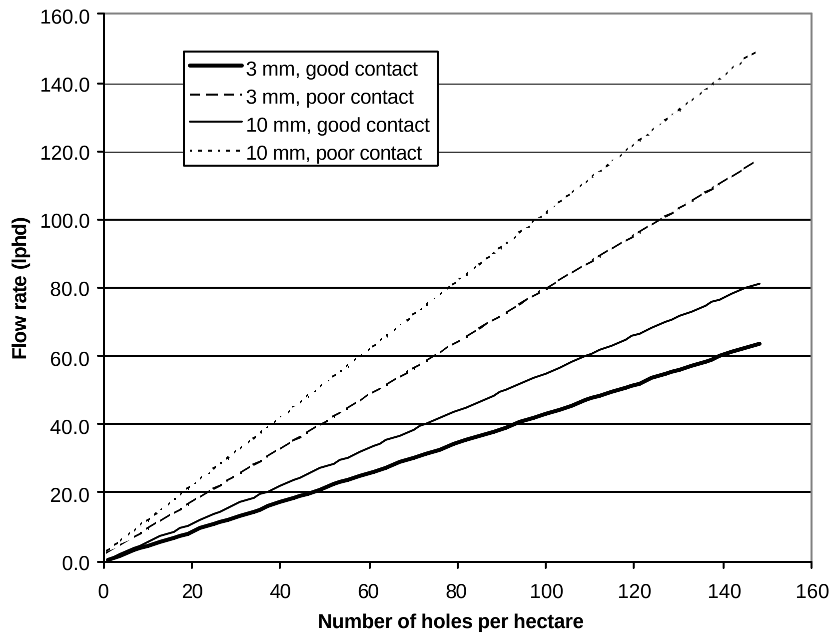

Figure 1. Flowrate Versus Small Hole Density (Head on Liner = 0.3 m)

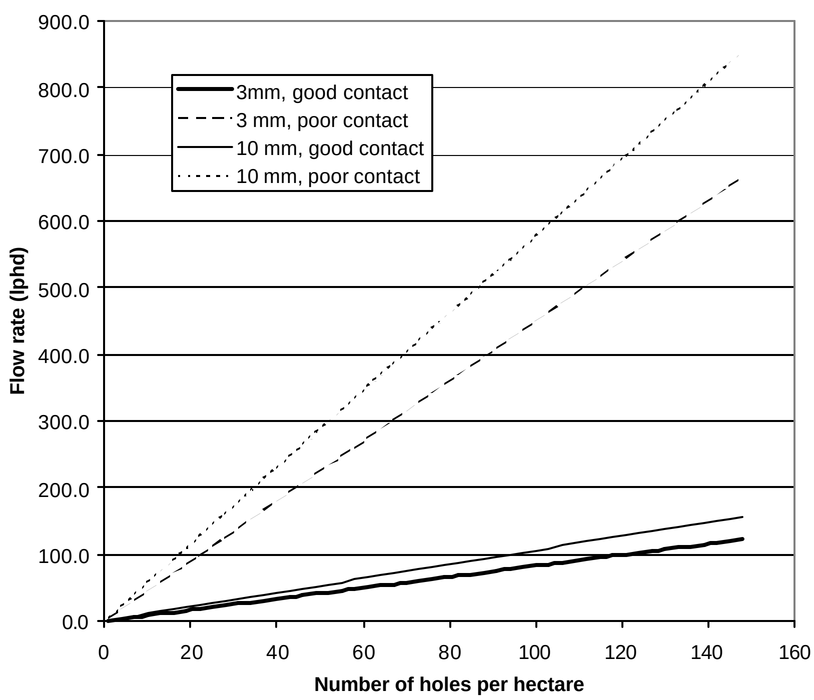

Figure 2. Flowrate Versus Small Hole Density (Head on Liner = 0.61 m)

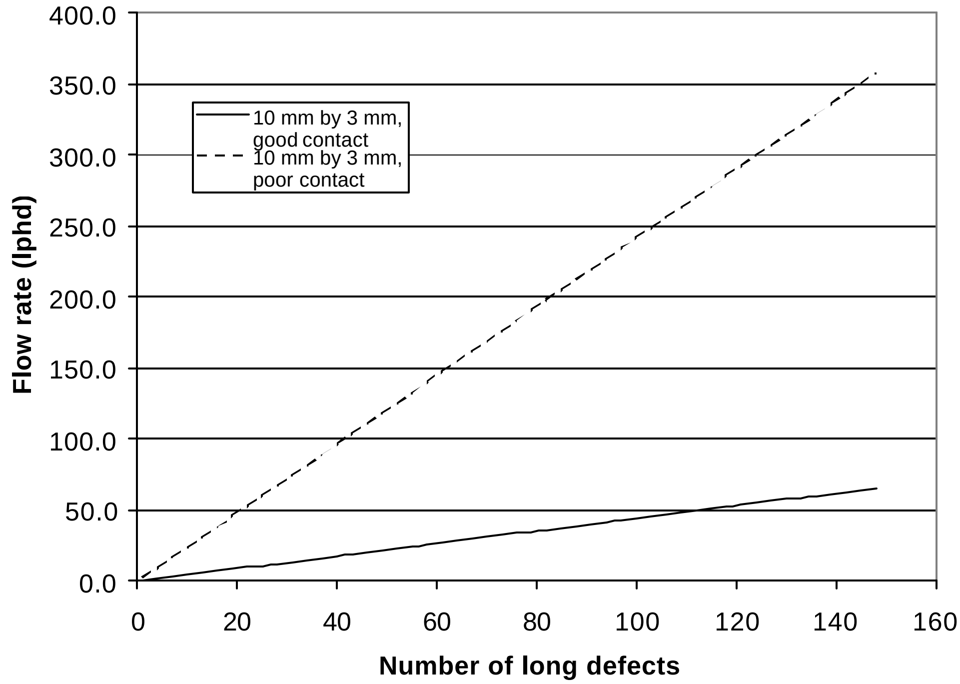

Figure 3. Flowrate Versus Long Defect Density (Head on Liner = 0.3 m)

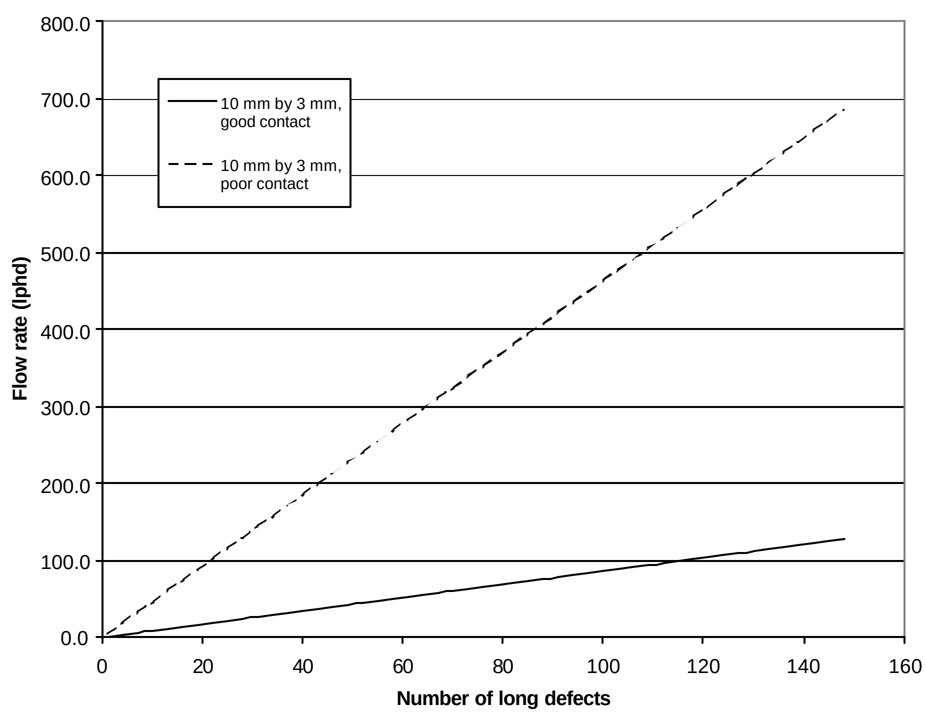

Figure 4. Flowrate Versus Long Defect (Head on Liner = 0.61 m)

C. Head on Liner

Few studies have measured liquid head on liners in landfills. One equation used to calculate maximum leachate head is

$$h = \frac{L}{2} * \left(\frac{e}{k}\right)^{\frac{1}{2}} \qquad \text{Eq. 20}$$

where h is the maximum leachate head (ft), L is the drainage length (ft), e is the average percolation rate into the landfill (in/yr), and k is the hydraulic conductivity of the refuse (in/yr) (Oweis et al., 1990). For example, if the drainage length is 100 ft, the hydraulic conductivity of the refuse is 12,415.7 in/yr (10^{-3} cm/s), and the average percolation rate is 1.6 in/yr, then the resulting head h = 0.56 ft. This value for average percolation rate is based on an annual precipitation of 40 in/yr, a 60% loss due to evapotranspiration and runoff, and a 90% efficiency of the post closure cap. If the cap has not yet been installed, then average percolation is 16 in/yr, and the calculated head is 1.8 ft. These values of leachate head roughly correspond to those used in the Giroud et al. (1992) equations that were used to generate data for Figures 1 through 4.

Oweis et al. (1990) estimate the hydraulic conductivity of fresh refuse to be 10^{-3} cm/s (12,415.7 in/yr) based on empirical, laboratory data. However, this value is expected to change as decomposition and settlement take place.

Oweis et al. (1990) studied the use of leachate wells for leachate discharge and measured a maximum leachate height of 35 ft above the bottom of the waste. Whether or not the landfill was lined was not stated, and this head seems atypically high.

McEnroe (1993) presents a set of analytical equations to estimate the maximum saturated depth over a liner based on the value of R, defined below.

For R < 1/4,

$$Y_{max} = (R - Y_L + Y_L^2)^{\frac{1}{2}} * [(1 - A - 2R) * (1 + A - 2Y_L) / (1 + A - 2R) / (1 - A - 2Y_L)]^{\frac{1}{2A}} \qquad \text{Eq. 21}$$

For R = 1/4,

$$Y_{max} = R * \frac{(1 - 2Y_L)}{(1 - 2R)} * \exp\left[\frac{2(Y_L - R)}{(1 - 2Y_L)(1 - 2R)}\right] \qquad \text{Eq. 22}$$

For R > 1/4,

$$Y_{max} = (R - Y_L + Y_L^{2})^{\frac{1}{2}} * \exp\left[\frac{1}{B} * \tan^{-1}\left(\frac{(2Y_L - 1)}{B}\right) - \frac{1}{B} * \tan^{-1}\left(\frac{2R - 1}{B}\right)\right] \qquad \text{Eq. 23}$$

Where,

$$Y_{max} = \frac{y_{max}}{(L * \tan \acute{a})}$$

y_{max} = maximum saturated depth, (L)

$$R = \frac{r}{(K * \sin 2\acute{a})}$$

r = vertical inflow per unit horizontal area, ($L^3/T/L^2$)
K = hydraulic conductivity of the drainage layer, (L/T)

$$Y_L = \frac{y_L}{L * \tan \acute{a}}$$

y_L = saturated depth at the downstream boundary, (L)
L = horizontal drainage distance, (L)
α = liner angle, measured from the horizontal

$$A = (1 - 4R)^{\frac{1}{2}}$$

$$B = (4R - 1)^{\frac{1}{2}}$$

For example, a collection system with a drainage length of 100 ft (L = 1200 in), a slope of 2 % (α = 1.14°), a downstream depth at the sump of 0 ft (y_L = 0 in), and a hydraulic conductivity of 1 cm/s (K = 12.42 E6 in/yr), has a $Y_L = 0$. If the vertical inflow (average percolation) is 1.6 in/yr, then R = 3.25E-4 which is less than 1/4, so Eq. 21 will be used with A = 0.999. Note that because R is so small, the downstream depth was set to zero to avoid taking the square root of a negative number in the first term of the following equation.

$$Y_{max} = (R - Y_L + Y_L^{2})^{\frac{1}{2}} * [(1 - A - 2R) * (1 + A - 2Y_L) / (1 + A - 2R) / (1 - A - 2Y_L)]^{\frac{1}{2A}}$$

$$Y_{max} = (3.25*10^{-4} - 0 + 0^{2})^{\frac{1}{2}} * \left[\frac{(1 - 0.999 - 2*3.25*10^{-4}) * (1 + 0.999 - 2*0)}{(1 + 0.999 - 2*3.25*10^{-4}) *(1 - 0.999 - 2*0)]^{\frac{1}{2*0.999}}}\right]$$

$$Y_{max} = 0.018 * \left[\frac{(3.5 * 10^{-4})*(1.999)}{(1.998) * (0.001)]}\right]^{0.4995}$$

$$Y_{max} = 0.010 \text{ (unitless)}$$

$$Y_{max} = \frac{y_{max}}{(L * \tan á)}$$

$$y_{max} = Y_{max} * L * \tan\alpha$$
$$y_{max} = 0.0106 * (1200 \text{ in} * \tan 1.14°)$$

$$y_{max} = 0.24 \text{ in} = 0.02 \text{ ft}$$

When this procedure is repeated for r = 16 in/yr, then R = 3.25×10^{-3}, A = 0.993, and y_{max} = 0.03 ft. The McEnroe (1993) equations yield much lower results than the Oweis et al (1990) for maximum leachate head. Although neither equation can be supported with field observations, the Oweis et al (1990) equation is easier to use and appears to give a more conservative estimate of leachate head to use in the flowrate equations.

The HELP model will be used in the following section to calculate changes in liquid head when the hydraulic conductivity of the LCRS changes.

D. Landfill Performance in New York State

New York is one of a few states in which double liners are required for MSW landfills. Electronic copies of 1999 annual reports for operating landfills were provided by the NY State Department of Environmental Conservation. These reports contained data on flowrates both in leachate collection and detection systems as well as the applicable acreage. These data are summarized in Table 21.

Table 21. Landfill Performance in NY

Landfill Type	LCRS flowrate (gal/ac/yr)	LDS flowrate (gal/ac/d)	Liner Efficiency %[a]
Ash	1571	2.6	99.8
Ash	2142	18.8	99.1
C&D	1103	22.5	98.0
C&D	1059	16.8	98.4
Ind/Comm	1822	0.8	99.9
Ind/Comm	2718	1.4	99.9
MSW	256	1.8	99.3
MSW	1722	1.3	99.9
MSW	1083	7.4	99.3
MSW	421	2.5	99.4
MSW	1304	19.6	98.5
MSW	851	24.7	97.1
MSW	222	4.2	98.1
MSW	359	1.0	99.7
MSW	152	0.6	99.6
MSW	666	6.2	99.1
MSW	450	8.2	98.2
MSW	284	2.8	99.0
MSW	2651	4.4	99.8
Max	2718	24.7	99.9
Min	152	0.6	97.1
Mean	1097	7.8	99.1

a. $E_a\% = (1 - (\text{LDS flowrate} / \text{LCRS flowrate})) * 100\%$

Liner efficiencies for the LCRS and LDS flowrates are also presented in Table 21. The mean efficiency is 99.1%. This leachate collection efficiency was then used to calculate a hypothetical leachate flowrate in both the LCRS and the LDS based on the following assumptions:

- annual percipitation of 36.7 in (average of the available 1999 precipitations for counties represented by a landfill in Table 21)
- evapotranspiration and runoff losses of 60%

The results of the liner efficiency calculation are reasonably consistent with the mean values reported in Table 21. The actual mean LCRS flowrate is 1097 gal/ac/d, and the calculated value is 1092 gal/ac/d. The actual mean LDS flowrate is 7.8 gal/ac/d, and the calculated value is 9.8 gal/ac/d.

E. HELP Model

The Hydrologic Evaluation of Landfill Performance (HELP) Model, Version 3.04 (Schroeder and Peyton, 1995) was used to estimate leachate generation and release rates from landfills. A generic 100 ac landfill was modeled. The surface has 100% available area for runoff, a slope of 3%, default soil texture for Greensboro, NC, and a vegetation rating of 3 for fair grass. Default data for Greensboro, NC were chosen to represent the evapotranspiration, precipitation, temperature, and solar radiation data at the site. The evaporative zone depth is 21 in, and the maximum leaf area index is 2.0 for a good stand of grass. When available, the default values provided by the model were used, for instance, to calculate the curve number of the landfill surface or the initial water content of the soils. The HELP model simulates 5 years of data, so a range of flowrates, precipitations, and efficiencies are presented for each run.

This landfill has a composite cover and single composite liner. There are 8 different layers as presented in Table 15. From the top of the landfill down, they are a vegetative layer, drainage layer, FML, compacted clay layer, compacted municipal solid waste, leachate drainage layer, FML, and compacted clay layer.

Table 15. Layer Data From the Top of the Landfill Down

Type[1]	Thickness (in)	Porosity (vol/vol)	Field Capacity (vol/vol)	Wilting Point (vol/vol)	Saturated Hydraulic Conductivity (cm/s)
1	24	0.4370	0.0620	0.0240	5.8×10^{-3}
2	18	0.4170	0.0450	0.0180	1.0×10^{-2}
4	0.06	na[2]	na	na	2.0×10^{-13}
3	24	0.4270	0.4180	0.3670	1.0×10^{-7}
1	961	0.6710	0.2920	0.0770	1.0×10^{-3}
2	24	0.4170	0.0450	0.0180	1.0×10^{-2}
4	0.06	na	na	na	2.0×10^{-13}
3	24	0.4270	0.4180	0.3670	1.0×10^{-7}

1. The model defines four layer types: 1) vertical percolation, 2) lateral drainage, 3) barrier soil liner, and 4) flexible membrane liner.
2. Flexible membrane liners do not require input for porosity, field capacity, or wilting point.

Both of the flexible membrane liners are assumed to have worst-case contact conditions (level 5), a thickness of 0.06 in, and defect hole diameters of 1 mm. The HELP model does not allow the user to specify the diameter of the defects. Each run specifies a different number of defects. Only pinholes are studied, and it is assumed that no long defects are present. The default values for a HDPE liner were chosen. Tables 16, 17, 19, and 20 present the average head on the liner. These values are hundredths to tenths of an inch and are much smaller than those predicted by the Oweis et al. (1990) and McEnroe (1993) equations presented in Section II C. They are also much smaller than the values used to generate Figures 1-4.

The drainage layers have maximum drainage lengths of 100 ft, drainage slopes of 2%, and 0% leachate recirculation. The LCRS initially has a saturated hydraulic conductivity of 1E-2 cm/s, which is the default value for poor-graded sand.

During the initial run, Run 1, the FML's in the cover and liner were deleted from the system to simulate CCL layers alone. In this case, the HELP model assumes that flow through the clay approached saturated flow, and the water or leachate that remained was collected in the drainage systems. These results are shown in Table 16. The calculated liner efficiencies range from ⁻324% to ⁻1464%. Negative efficiencies result when the flow through the clay layer is greater than the flow collected in the LCRS.

These results are not feasible because saturated flow through the entire clay layer does not necessarily occur as collection systems are designed specifically to reduce the amount of leachate that passes through barrier layers. However, the flowrate of leachate through the CCL liner and into the environment remains relatively constant according to the HELP model and is approximately equal to the flow predicted by Darcy's Law, presented below:

$$v = ki \qquad \text{Eq. 24}$$

$$v = (1*10^{-9}\ \frac{m}{s}) * 1 * (3.28\ \frac{ft}{m}) * (28.317\ \frac{L}{ft^{3}}) * (43560\ \frac{ft^{2}}{ac}) * (2.471\ \frac{ac}{ha}) * (3600\ \frac{s}{hr}) * (24\ \frac{hr}{d})$$

$$v = 864 \text{ lphd}$$

Table 16. HELP Run 1, CCL Liner

Year	Precipitation lphd (gal/ac/d)	LCRS Flowrate lphd (gal/ac/d)	Flowrate Through the CCL lphd (gal/ac/d)	% Efficiency[1]	Average Head on the Liner (in)
1974	31610 (3379)	140 (15.0)	865 (92.5)	-518	0.05
1975	39280 (4199)	204 (21.8)	866 (92.6)	-324	0.07
1976	24080 (2574)	55 (5.9)	860 (91.9)	-1464	0.02
1977	23820 (2546)	105 (11.2)	862 (92.1)	-721	0.04
1978	38910 (4159)	203 (21.7)	866 (92.6)	-327	0.07

1. Efficiency = (1 - (LDS flow rate / LCRS flow rate)) * 100%

In Runs 2 - 5, the FML is included and analyzed with a various number of defects. The precipitation data is the same as for Run 1. Table 17 summarizes these runs. As expected, the flowrates into the LDS increase as the number of defects increase.

In Run 2, the FML liner and cover are included and assigned depths of 0.06 in, with a pinhole density of 0 holes/ac. Under these conditions, the liner efficiencies are 98% to 99%. Darcy's Law would predict the flow through the FML in the cover or liner to be:

$$v = ki \qquad \text{Eq. 24}$$

$$v = (2*10^{-13}\ \frac{m}{s}) * 1 * (3.28\ \frac{ft}{m}) * (28.317\ \frac{L}{ft^{3}}) * (43560\ \frac{ft^{2}}{ac}) * (2.471\ \frac{ac}{ha}) * (3600\ \frac{s}{hr}) * (24\ \frac{hr}{d})$$

$$v = 0.17 \text{ lphd}$$

The flowrates into the LCRS predicted by the model are 0.07 to 0.14 lphd. When the LCRS flowrate is 0.07 lphd, the annual precipitation is approximately 10,000 lphd less than the years when the flow is 0.14 lphd. Precipitation varied annually in the default data set. The flowrates through the cover in the years with higher annual precipitation are close to those predicted by Darcy's Law.

Run 3 assumed a pinhole density of 10 holes/ac; the liner efficiencies are ⁻159% to ⁻160%. The model predicted that 645 to 772 lphd would pass through the liner, a value approaching the saturated flow that occurred in Run 1. The FML does inhibit saturated flow slightly. In contrast, the flowrate through the FML and CCL liner that would be predicted by the Giroud et al. (1992) equations under poor contact conditions, with ten 1 mm diameter holes, and a small liquid head (0.1 in) is only 0.60 lphd:

$$Q = 1.15 a^{0.1} h_w^{0.9} k_s^{0.74} \qquad \text{Eq. 25}$$

$$Q = 1.15 * \left(\pi\left(\frac{0.001}{2}\right)^2\right)^{0.1} * \left((0.1\,\text{in}) * (1\,\text{m} / 39.4\,\text{in})\right)^{0.9} * \left(1 * 10^{-9}\,\frac{\text{m}}{\text{s}}\right)^{0.74} * (10\,\text{holes/ac})$$

$$Q = 2.8 * 10^{-9}\,\frac{\text{m}^3}{\text{ac} - \text{s}} = 0.60\ \text{lphd}$$

It is apparent that HELP model is overestimating flow through the clay layer by three orders of magnitude when compared to the Giroud et al. (1992) equations. Evidently 5 holes/ac is enough for the model to assume that almost all of the clay layer is saturated, and the higher flows result. Runs 4 and 5 examine the change in flow through the clay layer when a smaller pinhole density is assumed.

Run 4 assumed a pinhole density of 1 hole/ac, poor contact conditions, and hole diameters of 1mm. The efficiencies in Run 4 increased to 74%. Run 5 analyzed 5 holes/ac, and resulted in efficiencies of ⁻30%. The model results move in the expected direction when fewer pinholes per acre are assumed. However, the collection efficiencies still appear to be unrealistically low.

Table 17. HELP Runs 2-5, FML/CCL Liner, Poor Contact Conditions

Number of Pinholes	LCRS Flowrate lphd (gal/ac/d)	Flowrate Through the CCL lphd (gal/ac/d)	% Efficiency	Average Head on the Liner (in)
0	0.07 - 0.14 (0.007 - 0.015)	1.4×10^{-3} (1.5×10^{-4})	98 - 99	0.0000 - 0.0001
1	482 - 850 (51.15 - 90.9)	125 – 221 (13.4 - 23.6)	74	0.2 - 0.3
5	373 - 464 (39.9 - 49.6)	486 – 604 (51.9 - 64.6)	$^{-}30$	0.02
10	248 - 298 (26.5 - 31.8)	645 – 772 (68.9 - 82.5)	$^{-}159$ - $^{-}160$	0.09 - 0.1

To test if the model was assuming saturated flow through the liners, a leachate detection system was added to the model in Run 6. Like the LCRS in Run 1, the LDS is lined with CCL alone. With the LDS in place, the flowrates into the LCRS are the same as in Run 1. If the liner efficiency calculation is performed, the liners appear to be 88.8% to 98.8% efficient because the LDS flowrates are so small. However, the flowrates into the environment are still approximately 860 lphd. These results are shown in Table 18.

Table 18. HELP Run 6, CCL Double Liner

Year	LCRS Flowrate lphd (gal/ac/d)	LDS Flowrate lphd (gal/ac/d)	Flowrate Into Environment lphd (gal/ac/d)
1974	140 (15.0)	15.7 (1.7)	860 (91.9)
1975	203 (21.7)	2.5 (0.27)	863 (92.2)
1976	56 (6.0)	2.2 (0.24)	858 (91.7)
1977	106 (11.3)	2.3 (0.24)	860 (91.9)
1978	203 (21.7)	2.3 (0.24)	863 (92.2)

It appears that the model is underestimating flow through the LCRS. The actual capacity of these drainage layers is shown in the following calculation.

LCRS Capacity Calculation

- Assume k = 0.01 cm/s
- Assume a slope of 2% and a drainage length of 100 ft
- Assume a depth d = 2 ft
- Assume i = 0.02 (2 ft / 100 ft)
- Assume a 100 ac, square landfill with side lengths of 2087 ft
- Assume there are 20 drainage pipes that span 2087 ft across the landfill

According to Darcy's Law, the flow per unit length of drainage is

$$q = k * i * d \qquad \text{Eq. 26}$$

$$q = (1*10^{-2} \frac{\text{cm}}{\text{s}}) * 0.02 * 2 \text{ ft} * 12 \frac{\text{in}}{\text{ft}} * 2.54 \frac{\text{cm}}{\text{in}}$$

$$q = 0.012192 \text{ cm}^2/\text{s}$$

The flow capacity of the LCRS is then:

$$Q = (0.012192 \frac{\text{cm}^2}{\text{s}}) * 2087 \frac{\text{ft}}{\text{pipe}} * 20 \text{ pipes} * 30 \frac{\text{cm}}{\text{ft}} * 3600 \frac{\text{s}}{\text{hr}} * 24 \frac{\text{hr}}{\text{d}} * 365 \frac{\text{d}}{\text{yr}} * \frac{1 \text{ ft}^3}{28{,}316.8 \text{ cm}^3}$$

$$Q = 1.7 * 10^7 \frac{\text{ft}^3}{\text{yr}}$$

For a 100 ac site, this converts to a collection system capacity of 32,500 lphd. In Run 1, the flowrate in the LCRS over the five years that were simulated, ranged from 105 to 204 lphd, a flowrate 2 orders of magnitude less than the capacity of the system.

Changing the permeability and slope of the LCRS may decrease the flow through the clay. Three more runs were performed with the CCL liner with various permeabilities and slopes. Results are presented in Table 19. Run 7 has the permeability of the LCRS equal to 10 cm/s and the slope equal to 2%. Run 8 has the permeability of the LCRS equal to 0.01 cm/s and the slope equal to 3%. Run 9 has the permeability of the LCRS equal to 10 cm/s and the slope equal to 3%.

Table 19. HELP Runs 7-9, CCL Double Liner

LCRS Conditions	LCRS Flowrate lphd (gal/ac/d)	Flowrate Through the CCL lphd (gal/ac/d)	% Efficiency	Average Head on the Liner (in)
k_{LCRS} = 10 cm/s Slope = 2%	243 - 276 (26.0 - 29.5)	77 - 80 (8.2 - 8.6)	68 - 71	0.0004 - 0.0005
k_{LCRS} = 0.01 cm/s Slope = 3%	40 - 142 (4.3 - 15.2)	830 - 864 (88.7 - 92.4)	$^{-}$2057 - $^{-}$508	0.009 - 0.03
k_{LCRS} = 10 cm/s Slope = 3%	227 - 266 (24.3 - 28.4)	54 - 56 (5.8 - 6.0)	76 - 79	0.0004

Changing the slope of the LCRS from 2% to 3% does not increase the efficiency of the collection system, but increasing the saturated hydraulic conductivity from 0.01 cm/s to 10 cm/s does. The default conductivity for poor-graded sand (0.01 cm/s) should not be used to represent a collection system. Higher values of k_{LCRS} are more reasonable because they account for the pipes in the collection system. The resulting efficiencies are positive but still well below the values reported in Section II A.

In order to achieve efficiencies of 90-99%, a FML must be added to the liner. Runs 10 - 14 show the effects of changing the number of holes in the FML when k_{LCRS} = 10 cm/s and the slope of the LCRS is 2%. The results are presented in Table 20.

Table 20. Runs 10-14, FML/CCL, k_{LCRS} = 10 cm/s, slope$_{LCRS}$ = 2%

Number of Pinholes	LCRS Flowrate lphd (gal/ac/d)	Flowrate Through the CCL lphd (gal/ac/d)	% Efficiency	Average Head on the Liner (in)
0	0 (0)	0 (0)	na[1]	0.00
1	2.1 - 4.7 (0.22 - 0.50)	0.0007 - 0.0014 (7.5×10^{-5} - 1.5×10^{-4})	99.96 - 99.97	0.00
5	8.9 - 17.6 (0.95 - 1.9)	0.012 - 0.023 (0.0013 - 0.0024)	99.86 - 99.87	0.00
10	16.4 -28.8 (1.8 - 3.1)	0.043 - 0.075 (0.004 - 0.008)	99.74	0.00
100	74.7 - 104.3 (8.0 - 11.1)	1.94 - 2.7 (0.21 - 0.29)	97.4	0.0001

1. Efficiency calculation is not applicable with zero LCRS flowrates.

Changing the k_{LCRS} to 10 cm/s has increased the efficiencies to those reported in the literature, but the flowrates through the CCL into the environment or LDS are much lower than the field data. Runs 15 through 17 determine how many holes are needed to achieve flowrates through the clay of 100 lphd (10.7 gal/ac/d) if k_{LCRS} varies from 0.1 cm/s to 10 cm/s. The efficiencies of the liners must decrease to approximately 74%. Results for the three runs are shown in Table 23.

Table 23. Runs 15-17, Number of Holes Required to Reach 100 lphd (10.7 gal/ac/d) of Flow Through the CCL

k_{LCRS} cm/s	Number of holes/ac	Range of Predicted Flow lphd (gal/ac/d)	% Efficiency
0.1	10	83 - 153 (8.9 - 16.3)	74
1	100	76 - 132 (8.1 - 14.1)	74
10	1000	29 - 31 (3.1 - 3.3)	76

Though 100 lphd is the median value of typical post-closure flowrates, the actual range reported by Bonaparte et al., (accepted for publication) is 5 - 210 lphd. Listed below are the ranges of defects for a given k_{LCRS} that will yield the same range of flowrates.

- k_{LCRS} = 0.1cm/s, 1 to 20 defects per acre
- k_{LCRS} = 1 cm/s, 15 - 500 defects per acre
- k_{LCRS} = 10 cm/s, > 200 defects per acre

In summary, it is possible to predict reasonable flowrates through barrier layers with the HELP model by manipulating the saturated hydraulic conductivity of the collection system and the number of pinholes per acre in the FML. Both of these parameters are generally unknown. The number of pinholes per acre cannot be measured directly, and the conductivity of the collection system changes with time as clogging and deterioration occur. Based on the guess work involved, the HELP model does not appear to be the most efficient method to estimate leachate flowrates into the environment.

III. Conclusions

Four methods have been presented to estimate leachate quantities released to the environment: field data, liner efficiency, hole density, and the HELP model. However, the methods do not show compatible results.

As presented in Section II A, the actual measurements from landfills around the country with Subtitle D liners, show that typical LDS flowrates in the post closure period range from 5 to 210 lphd (0.5 to 22 gal/ac/d). To compare the four methods listed above, a target flowrate of 100 lphd (10.7 gal/ac/d) was chosen to represent the midrange of typical LDS flowrates observed in the field. However, this number may overestimate the actual flowrates that will result after a final cap has been placed.

As seen in the sample water balance calculation in Section II A, if the cover and liner are both 99% efficient, then the resulting LDS flowrate is 1 lphd (0.12 gal/ac/d). In order for the efficiency method to approach 100 lphd (10.7 gal/ac/d), both the cover and liner must be less than 91% efficient. Some of the field data represent situations where a cover is not in place. If the efficiency of the cover is 0%, and the efficiency of the liner is 99%, then the resulting flowrate is 111 lphd (11.9 gal/ac/d).

A comparison of the liner efficiency calculation to the New York State landfill data shows that the efficiency method is fairly accurate when a reasonable estimate of annual precipitation and liner efficiencies can be made. For 1999, the average efficiency of the landfills for which data were available was 99.1%. When this efficiency is used to calculate a LDS flowrate, it results in 9.8 gal/ac/d, which is close to the measured value of 7.8 gal/ac/d.

Section II B presents typical hole densities estimated by Murray et al. (1995) which are 3 holes/ha (1.2 holes/ac) for FML's with good QA/QC and 125 holes/ha (50.6 holes/ac) for FML's with poor QA/QC. If the holes are 3 mm in diameter, and the head above the liner is 1 ft, then the resulting flowrates would be 1.3 lphd (0.14 gal/ac/d) for 3 holes/ha and 100 lphd (10.7 gal/ac/d) for 125 holes/ha, under good and poor contact conditions, respectively. Note that the estimates based on 3 holes/ha are close to the value calculated based on cover and liner efficiencies of 99%. The hole density method approaches 100 lphd when the hole density exceeds 100 holes/ha (40.5 holes/ac).

However, this hole density should rarely be seen in the field as quality control is a standard part of liner construction. Flowrates predicted by the HELP model approach 100 lphd when k_{LCRS} = 0.1 cm/s with 10 holes/ac, k_{LCRS} = 1 cm/s with 100 holes/ac, or k_{LCRS} = 10 cm/s with 1000 holes/ac. The most likely field scenario is probably k_{LCRS} = 0.1 cm/s with 10 holes/ac. This estimate of hole density is at least the same order of magnitude as the estimates of Murray et al. (1995).

While it is not clear which of the three methods will give the most realistic approximation of leachate flowrates from closed landfills, it is clear encouraging that the LDS flowrates in the NY landfill were quite low and consistent with LCRS efficiencies of about 99%. Ultimately, it will be necessary to obtain field data from closed landfills to verify that there is little to no flow in the LCRS several years after placement of the final cap.

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Appendix E. Surface Water Quality Impacts of Landfill Leachate

Table of Contents

List of Tables

List of Appendices

Figures

Surface Water Quality Impacts of Landfill Leachate

I. Introduction

In Appendix A, the composition of leachate associated with well-decomposed refuse was described. Two components of leachate, ammonia and some amount of dissolved organic carbon, will persist even after the refuse mass has nearly reached a state of complete decomposition. The objective of this section is to discuss a methodology that can be used to assess the potential impacts of a leachate release to surface water. Clearly, if a release of leachate to surface water at a known rate and composition does not threaten a surface water body (or an aquifer), then the landfill can be considered stable with respect to the threat of a leachate release.

Studies have indicated that leachate from a stable landfill will have a BOD/COD ratio less than 0.1 (Reinhart and Grosh, 1998, Pacey, 1999, Pohland and Harper, 1985, Ehrig, 1988), and Pacey (1999) has suggested that BOD should be less than 100 mg/L, and COD should be less than 1000 mg/L. Both the organics and ammonia in landfill leachate have the potential to decrease oxygen levels in receiving streams. In addition, ammonia is highly toxic to aquatic organisms. Average ammonia concentrations often range from 100 - 450 mg-N/L in leachate from old landfills (Krumpelbeck and Ehrig, 1999; Kjeldsen and Christophersen, accepted for publication). Even after dilution with the receiving water body, ammonia concentrations may exceed safe levels. Ammonia also contributes nitrogen that may contribute to an oversupply of nutrients.

To evaluate the impacts of leachate on a surface water system, both flowrates and pollutant concentrations must be known for both the leachate and the receiving stream. Direct measurement of these parameters is ideal but expensive and time consuming. During post closure, a more practical alternative is needed. The method presented in this section combines the Streeter Phelps equation, which predicts the depletion of dissolved oxygen from organic matter, with nitrification terms, which estimate the effect of ammonia on dissolved oxygen concentrations. This method may be used to develop an estimate of the effect of a leachate release on surface water. The results may then warrant the collection of additional data to reduce uncertainty in projected dissolved oxygen concentrations.

To begin, data on the flowrate and quality of the potential receiving stream is required. The stream parameters should remain relatively constant over time unless significant changes in the watershed occur. Stream data obtained during landfill operation or from USGS gage stations, along with statistical analysis, can be used to estimate flowrates and concentrations. Most permitting procedures require that the 7Q10 be used as the receiving stream's low flow. The USGS defines the 7Q10 as " the annual minimum 7-day consecutive low flow, which on average, will be exceeded in 9 out of 10 years" (USGS, 1993). Leachate flowrates and concentrations are assumed to be available for landfills during the post-closure monitoring period.

The first step in the process of protecting aquatic environments is to designate a use for the water body. Potential designations include recreation, water supply, fishery, and ecological balance (Thomann and Mueller, 1987). Once a water use has been selected, it is necessary to assign criteria, or concentrations, for the water body that will protect that use. Water quality standards are the regulations on mass discharge or effluent concentration that enforce the criteria. Finally, a cost-benefit analysis could be performed to determine the most efficient waste load allocations in the case of multiple dischargers (Thomann and Mueller, 1987).

This section of the report will discuss surface water quality issues associated with organics and ammonia and present equations to estimate dissolved oxygen and ammonia concentrations downstream of a hypothetical leachate discharge. A few case studies that illustrate the impacts of landfill leachate on surface water systems are then presented.

II. Dissolved Oxygen in Streams

Low dissolved oxygen (DO) concentrations are a common problem in surface water systems that receive effluent discharges with high organic loads. Acceptable minimum concentrations depend on the designated water use, temperature of the water, and presence of early life stages of fish. As a general rule, concentrations below 5 mg/L are stressful to aquatic organisms (Thomann and Mueller, 1987). Many states do allow mixing zones where specified standards are not required. Both standards for DO and limitations on mixing zones vary from state to state.

Sources of DO include photosynthesis, atmospheric reaeration, and inputs from tributaries or effluents. Sinks of DO include oxidation of organic carbon, ammonia, and nitrite; sediment oxygen demand; and the respiration of aquatic organisms. The oxidation of organic carbon requires approximately 2.7 mg-O_2/mg-C, and the oxidation of ammonia-nitrogen and nitrite-nitrogen combined requires about 4.57 mg-O_2/mg-N (Thomann and Mueller, 1987).

Atmospheric reaeration depends on the saturation value of dissolved oxygen in the stream, wind speed, stream turbulence, temperature, surface films, and hydraulic structures. Sediment oxygen demand results from the settlement of organic material (Thomann and Mueller, 1987).

The Streeter Phelps equation is often used to predict oxygen concentrations downstream of an organic carbon discharge (Thomann and Mueller, 1987). This equation is presented below. Note that the term (x/U) may be replaced by time t.

$$c = c_s - \left\{ \frac{K_d}{(K_a - K_r)} \left[\exp\left(-K_r \frac{x}{U}\right) - \exp\left(-K_a \frac{x}{U}\right) \right] \right\} L_o - (c_s - c_o) \exp\left(-K_a \frac{x}{U}\right) \qquad \text{Eq. 1}$$

The following equation may be used to account for oxygen depletion during nitrification. Nitrification only occurs at DO concentrations greater than or equal to 1 - 2 mg-O_2/L (Thomann and Mueller, 1987).

$$c = c_s - L_o^N \left[1 - \exp\left(-K_N \frac{x}{U}\right) \right] \qquad \text{Eq. 2}$$

The following equation is a combination of the Streeter Phelps and nitrification equations. Work to validate Eq. 3 mathematically is presented in Appendix 1.

$$\begin{aligned} c = c_s &- \left\{ \frac{K_d}{(K_a - K_r)} \left[\exp\left(-K_r \frac{x}{U}\right) - \exp\left(-K_a \frac{x}{U}\right) \right] \right\} L_o - (c_s - c_o) \exp\left(-K_a \frac{x}{U}\right) \\ &- L_o^N \left[1 - \exp\left(-K_N \frac{x}{U}\right) \right] \end{aligned} \qquad \text{Eq. 3}$$

where,

c = dissolved oxygen concentration at a distance x downstream of the discharge (mg/L)

c_s = saturation concentration of dissolved oxygen (mg/L)

c_o = initial dissolved oxygen concentration (mg/L)

K_d = effective deoxygenation rate of organic carbon (T^{-1})

K_a = reaeration coefficient (T^{-1})

K_r = overall loss rate of CBOD due to settling and oxidation of soluble BOD (T^{-1})

x = distance downstream (L)

U = average stream velocity (L/T)

L_o = initial concentration of carbonaceous material as BOD (mg/L)

L_o^N = initial concentration of ammonia and organic nitrogen (mg-N/L)

K_N = rate of oxidation of ammonia and organic nitrogen (d^{-1})

The initial concentrations of oxygen, organics, and ammonia should be calculated after dilution of leachate with the receiving water body. The continuity equation may be used for this calculation:

$$Q_L C_L + Q_S C_S = Q_i C_i \qquad \text{Eq. 4}$$

where $Q_i = Q_L + Q_S$, and

Q_L = flowrate of leachate (volume/time)

C_L = concentration of parameter in leachate (mass/vol)

Q_S = flowrate of receiving stream (volume/time)

C_S = background concentration of parameter in receiving stream (mass/vol)

Q_i = flowrate downstream of discharge (volume/time)

C_i = concentration of parameter downstream of discharge (mass/vol)

Although the water quality model QUAL2E is not used in this analysis, the manual lists typical values for some of the rate constants used in equations 1 - 3 (Brown and Barnwell, 1987). Thomann and Mueller (1987) also list typical values. Table 1 presents the values as they appear in the literature. Of course, the actual values used in Eq. 3 should be based on knowledge of a specific water body because they depend on stream flow, depth, and morphology.

Table 1. Typical Values For Rate Constants (Brown and Barnwell, 1987)

Rate Constant	Values listed by Brown and Barnwell (1987) d^{-1}	Values listed by Thomann and Mueller (1987) d^{-1}
K_d	0.02 - 3.4	0.1 - 0.5 mean depth > 5' 0.5 - 3.0 mean depth < 5'
K_a	0.0 - 100	0.05 - 12.2
K_r	$^{-}$0.36 - 0.36	na[1]
K_N	na[1]	0.1 - 0.5 mean depth > 5' 0.5 - 3.0 mean depth < 5'

1. Values not listed.

III. Use of the Streeter Phelps/Nitrification Equation

The combined form of the equation can be entered into a spreadsheet, as in Appendix 2 discussed below, and used to evaluate different scenarios for the release of leachate to a receiving water body. If the model shows that dissolved oxygen levels will fall below the permit level, then pretreatment of the leachate in an aeration basin, lagoon, or constructed wetland may be necessary to remove organics and ammonia prior to discharge.

To illustrate the use of Eq. 3, it was applied to a hypothetical landfill. The following examples use a typical leachate flowrate of 1000 gal/d (~10.7 gal/ac/d for a 100 ac landfill) with an ammonia concentration of 250 mg-N/L and a BOD of 100 mg/L. The leachate is assumed to discharge into a stream with a background ammonia concentration of 0.1 mg-N/L, a background organic nitrogen concentration of 1 mg-N/L, and a background BOD of 5 mg/L. Flowrates from Crabtree Creek in Raleigh, NC are used to compare the effect of using the median flow (Figure 1) and low flow (Figure 2) in the model.

The effects of changing the stream flow (Q_s) and reaeration constant (K_a) are analyzed in Appendix 2 and presented in Figures 1 and 2. According to the USGS stream flow data, the long-term median flow for Crabtree Creek at Highway 1 in Raleigh is 36 cfs (2.3×10^7 gal/d). If the leachate described above is discharged to this stream, then the resulting ammonia and BOD concentrations after dilution are 0.11 mg-N/L and 5.00 mg/L, respectively.

Under median flow conditions, we have assumed that the average stream velocity is 1 ft/s, and that K_d, K_r, and K_N are all equal to 0.3 day^{-1}. This common value was chosen because it represents the midrange for deep streams given by Thomann and Mueller (1987). The combined form of the Streeter Phelps and nitrification equations is used along with reaeration coefficients ranging from 0 d^{-1} - 5 d^{-1} to predict downstream oxygen concentrations. Under these conditions, the dissolved oxygen concentration decreases dramatically when $K_a = 0$ d^{-1}, but does not drop below 5 mg/L when reaeration values are above 0.26 d^{-1} (Fig.1). During median flow conditions, the ammonia concentration at the discharge point is also below the standards recommended in the following section. Thus, leachate discharge to this stream will probably not require pretreatment unless the reaeration constant is below 0.26 d^{-1}.

When the hypothetical leachate is discharged into Crabtree Creek during low flow the stream parameters change. The stream velocity is assumed to be 0.1 ft/s, and K_d, K_r, and K_N are assumed to equal the midrange for shallow streams, 1.5 day^{-1}. According to the USGS (1998), the 7Q10 for Crabtree Creek is 2.0 cfs (1.3×10^6 gal/d). Results are shown in Figure 2. After dilution and mixing at the discharge point, the ammonia concentration is 0.3 mg-N/L, and the BOD is 5.07 mg/L. Under these conditions the DO still does not drop below 5 mg/L unless K_a is less than 1.4 d^{-1}. In addition, the ammonia concentration does not exceed the strictest standards presented in the following section.

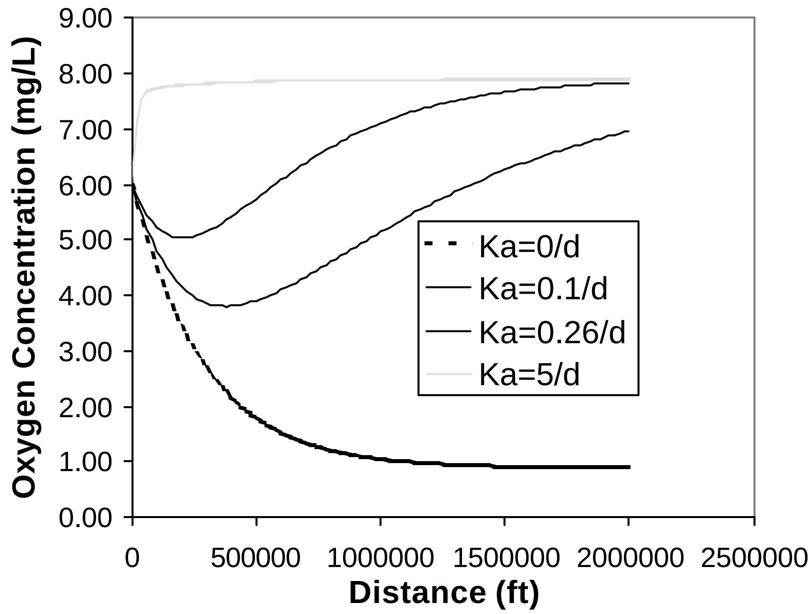

Figure 1. Downstream Oxygen Concentrations in Crabtree Creek During Median Flow

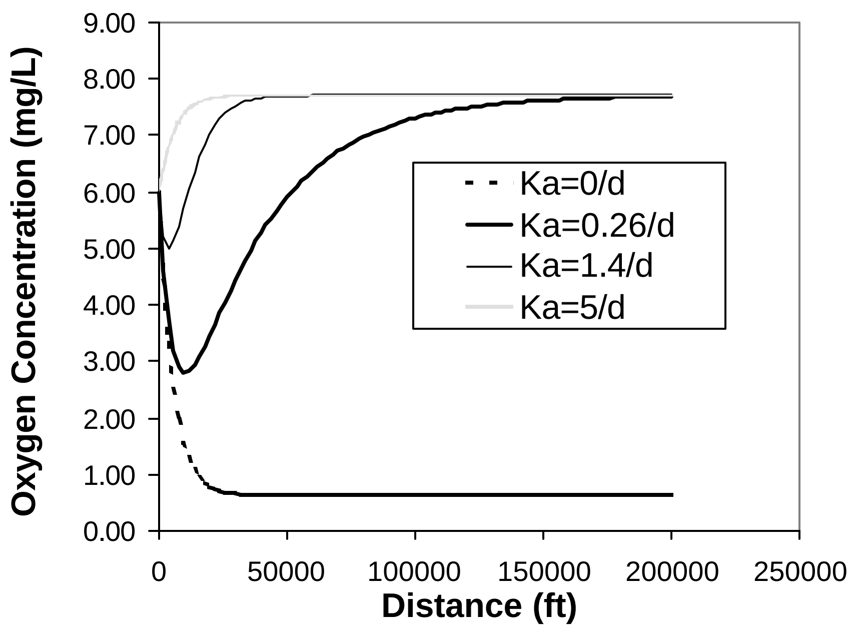

Figure 2. Downstream Oxygen Concentrations in Crabtree Creek During Low Flow

IV. Acceptable Ammonia Levels in Landfill Leachate

The Streeter Phelps/Nitrification equation discussed in sections II and III will only predict downstream oxygen concentrations. To predict ammonia concentrations, the continuity equation presented in Section II can be used.

Ammonia standards depend on the temperature and pH of the stream, and the presence of early life stages of fish. Table 2 lists guidelines that protect organisms from chronic toxicity at pH 7 and pH 8 which are in the 1999 Update of Ambient Water Quality Criteria for Ammonia (US EPA, 1999).

Table 2. Ammonia Standards For Fresh Water (US EPA, 1999)

	Ammonia Standards (mg-N/L)			
Temperature (°C)	Early Life Stages Present		Early Life Stages Absent	
	pH = 7	pH = 8	pH = 7	pH = 8
0	5.91	2.43	9.60	3.95
10	5.91	2.43	7.91	3.26
20	4.15	1.71	4.15	1.71
30	2.18	0.897	2.18	0.897

The lowest ammonia standard presented in Table 2 is 0.9 mg-N/L. Thus, a leachate with an ammonia concentration of 250 mg-N/L would require a dilution factor of 278 to meet the lowest standard. Of course, this calculation neglects any ammonia biodegradation.

V. Nutrient Loading

Thomann and Mueller (1987) define eutrophication as excessive plant and algal growths that interfere with water uses. Though eutrophication is a natural process in surface water systems, its rate is significantly increased by the presence of excess nutrients, especially nitrogen and phosphorous. Increased rates of eutrophication can lead to low DO levels due to night time plant respiration and decay of organic matter, interference with water treatment plant operation, decline of aesthetic and recreational water uses, and the growth of toxic algae that may interfere with shell fish production.

Though ammonia is no longer toxic after it is oxidized to nitrate, the nitrogen still contributes to the nutrient load (USGS, 1999). Nitrogen levels in streams that drain agricultural and urban areas are already typically high. Sources of nitrogen include fertilizers, automobile and electric power plant emissions, effluent from sewage treatment plants, and animal wastes. The average natural background concentrations for total nitrogen, nitrate, and ammonia are 1.0, 0.6, and 0.1 mg-N/L, respectively. These background concentrations were measured in streams located in nondeveloped, nonagricultural watersheds (USGS, 1999).

Because nitrogen is readily available in most fresh water streams, phosphorous is usually the limiting nutrient for excessive plant growth (USGS, 1999). Sources of phosphorous include discharges from wastewater treatment plants, fertilizers, soil erosion, and livestock waste. The natural background concentration of phosphorous in the average freshwater stream is 0.1 mg-P/L, and concentrations above this value may lead to eutrophication (USGS, 1999).

In order to improve water quality in the Neuse River Basin in NC, regulators have proposed a ruling that will limit discharges of nitrogen and phosphorous in the watershed. The primary goal is to reduce the nitrogen load at New Bern, NC by 30% (NCDEHNR, 1997). In 1997, New Bern received 8.7 million lbs-N/yr of which 2.1 million lbs can be attributed to permitted dischargers, leaving 6.6 million lbs from nonpoint sources such as agricultural runoff and atmospheric deposition. The following regulations have been proposed (NCDEHNR, 1997):

- Wastewater treatment plants with over 0.5 MGD of effluent should not exceed total nitrogen concentrations of 3.5 mg-N/L or total phosphorous concentrations of 2 mg-P/L.
- Wastewater treatment plants with over 0.05 MGD of effluent should also not exceed total phosphorous concentrations of 2 mg-P/L. There is no nitrogen limit for these facilities.
- Stormwater runoff from agricultural and forested land should not exceed an area weighted loading rate of 5.1 lbs-N/ac/yr.

- Stormwater runoff from new development should not exceed an area weighted loading rate of 3.6 lbs-N/ac/yr.

To compare potential nutrient loads from a landfill to these other sources of nitrogen and phosphorous, annual loads from each source have been calculated for a population of 500,000. According to Tchobanoglous and Burton (1991), the range of wastewater flowrates from a typical house is 45 - 90 gal/person/d. Assuming water use of 65 gal/person/d and that the input flowrate to a wastewater treatment plant is the same as the effluent discharge, a treatment plant that serves 500,000 people will discharge 32.5 MGD. If the allowable nitrogen concentration is 3.5 mg-N/L, then this plant will release 346,000 lbs-N/yr. If the same plant releases effluent with a phosphorous concentration of 2 mg-P/L, it will release approximately 19,800 lbs-P/yr. To put this on the scale of a landfill that will receive waste for 20 years, these values become 6.92×10^6 lbs-N and 3.96×10^5 lbs-P, over 20 years.

Based on the analysis in Appendix D, it is assumed that a landfill will produce 10.7 gal/ac/d of leachate during the post-closure period. If we assume a population of 500,000 people with a landfill generation rate of 4 lbs/person/day (after recycling), then this population will produce 1.46×10^{10} lbs of waste over a 20 year period. Dividing this by a typical landfill-waste density of 1300 lbs/yd^3, the volume of waste is 1.12×10^7 yd^3, or 3.03×10^8 ft^3. To take into account the daily cover soil, this volume can be multiplied by a factor of 1.1 to obtain 3.33×10^8 ft^3. Assuming this landfill is 80 ft deep, the surface area will be 95.7 ac, or 100 ac for simplicity. If the landfill produces 10.7 gal/ac/d, and the leachate has an ammonia concentration of 250 mg-N/L and a phosphorous concentration of 1 mg-P/L, then the nutrient loads would be 1.63×10^4 lbs-N and 65.2 lbs-P, over a 20 year period. Thus, the nitrogen released from a typical landfill that serves 500,000 people is 0.2% of that released from a typical wastewater treatment plant serving the same population; the phosphorous released is only 0.02%. These numbers are extremely low considering that a leachate release may never occur, and that the quantity of leachate produced from old, lined sites is often negligible within ten years of receiving the final cover (Bonaparte et al., accepted for publication).

A hypothetical landfill release may also be compared to acceptable stormwater releases from the same area of land. The mass discharged from runoff over 100 ac of land over a 20-year period will be calculated. According to the proposed rule to manage nutrient discharges in the Neuse River Basin, a 100-ac site of agricultural or forested land can release 510 lbs-N/yr, or 10,200 lbs-N over a 20-year period. 100 ac of new development can release 360 lbs-N/yr, or 7,200 lbs-N over a 20-year period. Though the landfill releases 1.6 times the nutrient load of the agricultural/forested land and 2.3 times that of the new development, these values are not high considering that the loads are being compared to stormwater runoff and that the landfill release is likely an overestimate.

VI. Water Quality Case Studies

A. Case Study 1 - Orange County, Florida

Schwartz et al. (1999) used the QUAL2E model to evaluate the impacts of a leachate discharge in Orange County, Florida. Leachate was treated in a constructed wetland prior to discharge. The Landfill Outfall Canal (LOC) carries the discharge from the landfill to the Little Econlockhatchee River. LEC1 is just downstream of the point on the LOC where the leachate is discharged from the landfill.

To calibrate a water quality model for the canal, flowrates and constituent concentrations were measured at 13 points along the LOC during a low-flow and high-flow event. They were used to calibrate the QUAL2E model, which was then used to predict the allowable outflow concentrations at LEC1. If concentrations did not meet those allowed, further treatment would be required before the leachate entered the LOC.

Historical data were used to determine the appropriate low and high-flow events. There is one USGS station with flow data on the river and "limited leachate flow data at LEC1." The low-flow event from the landfill is the 7Q10 (the 7 day minimum flow that occurs once every 10 years), which ranges from 0.45 cfs to 0.53 cfs. There was no discharge from the LOC to the river during low flow. High-flow events correlated with rainfall. They assumed the 2 yr, 24 hr storm would result in the average high-flow event. The flow at the gaging station during this storm is 107 cfs. The flow from the landfill was estimated to be 3.6 cfs based on its percent of the total watershed area. The measured leachate flow at LEC1 was 8.1 cfs. They used the actual flow in the model.

The QUAL2E model was used to establish the maximum allowable BOD_{ult} concentration that would not decrease dissolved oxygen concentrations along the LOC. The canal has a total length of 4.4 miles and was divided into 8 reaches for modeling purposes. Reach 1 starts at LEC1, which is the point of compliance for surface water discharge from the landfill. Flow and water level data were gathered for each reach.

The model includes non-point source inflows along the first 7 reaches, four point source discharges from offsite, and one headwater at LEC1 that includes stormwater from a detention pond at the landfill as well as the effluent from the wetland treatment system upstream of LEC1.

The concentrations measured during the low flow event were used to calibrate the model by adjusting parameters such as reaction rate coefficients within allowable ranges. The model was then used to predict concentrations of organic nitrogen, ammonia, nitrite, nitrate, ortho-phosphate, organic phosphorous, chlorophyll a, DO, and BOD_{ult}. The results of the model showed that reaeration of the stream increased in the direction of flow in the LOC and that deoxygenation and sediment oxygen demand were both low during low-flow events.

After the authors used the calibrated model to predict DO and BOD during the high-flow event, they compared the results to actual measurements. The model was shown to be a reliable predictor for DO and BOD. The model was then used to study the effects of varying DO during low-flow and high-flow events. Eight low-flow simulations were studied: two with LEC1 flow = 0.01 cfs and six with LEC1 flow = 3.4 cfs. All seven of the high flow simulations used 10.26 cfs, the maximum discharge at LEC1. They varied DO values in the canal to represent current conditions as well as values that would occur if the LOC was improved with reaeration along the canal, at LEC1, or both. The low-flow target concentration for BOD_{ult} was set to 10 mg/L if LOC improvements were made along the canal (reaeration) and if an aeration system was installed at LEC1. The high flow limit was set to 20 mg/L if the same improvements were made. The researchers stated that both improvements were necessary because the LOC receives discharges from other wetlands in addition to the headwater from the landfill. Wetland discharges typically have low DO concentrations so installing the improvements will assure the landfill does not further decrease DO.

B. Case Study 2 - Monument Hill Landfill, England

Monument Hill Landfill in southern England is an old, unlined, closed landfill that was built in a valley. The stream that flows through the valley and under the landfill, the Stert Watercourse, was culverted to protect it from the landfill's leachate, but the structure eventually failed, and a "minor but persistent" contamination resulted. Eventually, the Stert Watercourse was diverted around the landfill in a new culvert, but the old culvert continued to release some contaminants. Eventually a reed bed system was installed to treat the leachate. (Robinson et al., 1999).

Before the reed bed system was installed, site monitoring occurred for 1 year at three locations: the leachate discharge point, the new culvert upstream of its merger with the leachate, and 80 m downstream of their junction. Data included continuous flowrates for these streams as well as local rainfall, which was recorded at the landfill site and at four Meteorological Office Weather Stations within a 5-km radius of the site. By comparing rainfall and flowrate data, it was determined that rainfall infiltration only contributed 20% of the flow from the culvert that passed under the landfill. The majority of the flow was ground water and leachate. Leachate flowrates ranged from 16,000 to 79,000 gal/d, and the flows from the new culvert before merging with the leachate ranged from 264,000 to 1,000,000 gal/d.

Contaminants of concern were mainly iron, suspended solids, ammonia, and a pesticide, Mecoprop. The iron was not a health risk, but rather caused orange stains 10 m downstream of the leachate discharge. The mean of the suspended solids concentration, partially due to the iron, was 57.5 mg/L (14 samples). The mean ammonia level was 25.5 mg/L as N (21 samples), and the Mecoprop was 5.34 μg/L (15 samples). The mean BOD_5 level was less than 5 mg/L (21 samples). No significant heavy metals concentrations were detected in the diluted leachate or downstream of its discharge. Chloride concentrations were also monitored at each location, and it was determined that the leachate was diluted by a factor of 6 to 25 in the stream. Expected permit levels for the contaminants mentioned above as well as COD are listed in Table 3.

Table 3. Expected Permit Levels (Environment Agency, UK) For the Treated Leachate at the Monument Hill Landfill

Constituent	Concentration (mg/L)
COD	60
BOD_5	15
Ammonia-N	5
Iron	2
Suspended Solids	25

In another monitoring period after completion of the reed bed, the mean ammonia-N concentration in the effluent from the landfill was 19.4 mg/L. After passing through the reed bed, the concentration decreased to 11.8 mg/L, and after dilution, the concentration was 1.8 mg/L. The reed bed treatment and subsequent dilution achieved ammonia concentrations less than the expected permit level which was 5 mg/L (Robinson et al., 1999). Because of the dilution factor, permitted ammonia levels from the reed beds were increased to 23 mg/L (Robinson et al., 1999).

VII. Summary

Ammonia and organics will remain in landfill leachate for decades. If released to the environment, both of these pollutants can decrease the dissolved oxygen available for the survival of aquatic organisms. Ammonia is also toxic to several species, and its nitrogen may contribute to eutrophication. The biodegradable portion of the organics (BOD) in landfill leachate is present in relatively low concentrations and will probably not have adverse impacts on the environment. The impacts of the more stable organics (COD) should be investigated. The concentrations of phosphorous in leachate are not typically high enough to cause a nutrient problem. Ammonia will probably be the most significant long-term problem.

A method to evaluate potential impacts of a leachate release to surface water has been presented in this section. The methodology includes (1) the use of the Streeter Phelps equation with terms added to consider nitrification to calculate the dissolved oxygen concentration downstream of a hypothetical leachate release, (2) nutrient loading, and (3) ammonia toxicity.

In the case of a hypothetical leachate discharge to Crabtree Creek in Raleigh, NC, the DO concentration remains above 5 mg/L under both median and low flow conditions as long as the reaeration coefficient is above 0.26 d^{-1} and 1.4 d^{-1}, respectively. In addition, the ammonia concentration does not exceed the strictest standard in either case.

The Neuse River nutrient management strategy was used to compare nitrogen and phosphorous loads from a landfill to other sources in the basin. When compared on a population basis to a typical wastewater treatment plant, a typical landfill released a small percentage of the nutrient load, 0.2% of the nitrogen and 0.02% of the phosphorous. However, the nutrient load is 1.6 to 2.3 times greater than that from a stormwater discharge from the same area. Nonetheless, the landfill does not contribute the majority of the nutrient load to the basin. More importantly, the wastewater treatment plant is a continuous discharge, whereas the leachate discharge is hypothetical and may never occur.

The landfill leachates from the two case studies both exhibited surface water quality impacts and required pretreatment before discharge. The landfill in Orange County Landfill, Florida required additional aeration because other wetlands discharge to the same river, and wetlands in general have low dissolved oxygen concentrations. The landfill in England required a pretreatment reed bed because it had high ammonia concentrations. Neither case study focused on nutrient loads.

In summary, several aspects of water quality must be addressed when leachate is released to a surface water body. Dissolved oxygen levels should be modeled, and ammonia toxicity must be considered. In addition, the mass of nutrients delivered to the stream should not contribute to excessive plant growth. Of these three problems, ammonia toxicity will probably be the most common. Simple aeration strategies such as cascading water may alleviate this problem, as the ammonia will be oxidized to nitrate, but aeration will not decrease the nutrient load to the stream.

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Appendix 1 - Verification of the Streeter Phelps/Nitrification Equation

The purpose of Appendix 1 is to verify that combining the Streeter Phelps Equation with nitrification terms will yield the same results as performing these two operations separately.

1. Generate data for the Streeter Phelps Equation.
2. Generate data for the nitrification Equation.
3. Sum the two data sets.
4. Generate data for the combined equation.
5. Compare 3 and 4. Are they the same?

1. Model the dissolved oxygen in a stream with the Streeter Phelps Equation (Thomann and Mueller, 1987). Parameter values are listed next to their descriptors.

	Value
c = dissolved oxygen concentration at a distance x downstream of the discharge (mg/L)	variable
c_s = saturation concentration of dissolved oxygen (mg/L)	8
c_o = initial dissolved oxygen concentration (mg/L)	6
K_d = effective deoxygenation rate of organic carbon (d^{-1})	0.3
K_a = reaeration coefficient (d^{-1})	2
K_r = overall loss rate of CBOD due to settling and oxidation of soluble BOD (d^{-1})	0.3
x = distance downstream (ft)	variable
U = average stream velocity (ft/d)	86400
L_o = initial amount of carbonaceous material (mg/L)	5

$$\mathbf{c = c_s - \{K_d/(K_a - K_r)\ [\exp(-K_r(x/U)) - \exp(-K_a(x/U))]\}L_o - (c_s - c_o)\exp(-K_a(x/U))}$$

Distance x (ft)	Results c mg-O/L
0	6.00
20000	6.47
40000	6.79
60000	7.00
80000	7.16
100000	7.27
120000	7.35
140000	7.41
160000	7.47
180000	7.51
200000	7.55
220000	7.58
240000	7.61
260000	7.64
280000	7.66
300000	7.69
320000	7.71
340000	7.73
360000	7.75
380000	7.76
400000	7.78
420000	7.79
440000	7.81
460000	7.82
480000	7.83
500000	7.84
520000	7.85
540000	7.86
560000	7.87

Distance x (ft)	Results c mg-O/L
580000	7.88
600000	7.89

2. Model the dissolved oxygen in a stream with the nitrification terms (Thomann and Mueller, 1987). Parameter values are listed next to their descriptors.

	Value
K_N = rate of oxidation of ammonia and organic nitrogen (d^{-1})	0.3
N_{01} = the initial value of organic nitrogen at x = 0 (mg-N/L)	1
N_{02} = the initial value of ammonia at x = 0 (mg-N/L)	0.3
x = distance downstream (ft)	variable
U = average stream velocity (ft/d)	86400
L_o^N = initial amount of ammonia and organic nitrogen (mg-N/L)	1.3
c = dissolved oxygen concentration at a distance x downstream of the discharge (mg/L)	variable
c_s = saturation concentration of dissolved oxygen (mg/L)	8

$c = c_s - L_o^N * [1-\exp(-K_N(x/U))]$

Distance x (ft)	Results c mg-O/L
0	8.00
20000	7.91
40000	7.83
60000	7.76
80000	7.68
100000	7.62
120000	7.56

Distance x (ft)	Results c mg-O/L
140000	7.50
160000	7.45
180000	7.40
200000	7.35
220000	7.31
240000	7.26
260000	7.23
280000	7.19
300000	7.16
320000	7.13
340000	7.10
360000	7.07
380000	7.05
400000	7.02
420000	7.00
440000	6.98
460000	6.96
480000	6.95
500000	6.93
520000	6.91
540000	6.90
560000	6.89
580000	6.87
600000	6.86

3. Sum the data sets for BOD = 5 mg/L, and NH_3 =0.3 mg-N/L.
The saturation concentration must be the same in parts 1. and 2.

Distance x (ft)	1. Streeter Phelps c mg-O/L BOD = 5 mg/L	2. Nitrification c mg-O/L NH_3 = 0.3 mg-N/L	3. Combined Data c mg-O/L
0	6.00	8.00	6.00
20000	6.47	7.91	6.39
40000	6.79	7.83	6.62
60000	7.00	7.76	6.76
80000	7.16	7.68	6.84
100000	7.27	7.62	6.88
120000	7.35	7.56	6.91
140000	7.41	7.50	6.91
160000	7.47	7.45	6.91
180000	7.51	7.40	6.91
200000	7.55	7.35	6.90
220000	7.58	7.31	6.89
240000	7.61	7.26	6.88
260000	7.64	7.23	6.87
280000	7.66	7.19	6.86
300000	7.69	7.16	6.85
320000	7.71	7.13	6.84
340000	7.73	7.10	6.83
360000	7.75	7.07	6.82
380000	7.76	7.05	6.81
400000	7.78	7.02	6.80
420000	7.79	7.00	6.80
440000	7.81	6.98	6.79

Distance x (ft)	1. Streeter Phelps c mg-O/L BOD = 5 mg/L	2. Nitrification c mg-O/L NH_3 = 0.3 mg-N/L	3. Combined Data c mg-O/L
460000	7.82	6.96	6.78
480000	7.83	6.95	6.78
500000	7.84	6.93	6.77
520000	7.85	6.91	6.77
540000	7.86	6.90	6.76
560000	7.87	6.89	6.76
580000	7.88	6.87	6.76
600000	7.89	6.86	6.75

4. Model DO with the Streeter Phelps and nitrification terms in a combined equation. Use the c_o and c_s from part 1.
BOD = 5 mg/L, and NH_3 = 0.3 mg-N/L.

$$\mathbf{c = c_s - \{K_d/(K_a - K_r)\ [\exp(-K_r(x/U)) - \exp(-K_a(x/U))]\}L_o - (c_s - c_o)\exp(-K_a(x/U))}$$
$$\mathbf{- L_o^N * [1 - \exp(-K_N(x/U))]}$$

5. Compare 3. and 4. Are they the same?

Distance x (ft)	4. Equation c mg-O/L	3. Combined Data c mg-O/L
0	6.00	6.00
20000	6.39	6.39
40000	6.62	6.62
60000	6.76	6.76
80000	6.84	6.84

Distance x (ft)	4. Equation c mg-O/L	3. Combined Data c mg-O/L
100000	6.88	6.88
120000	6.91	6.91
140000	6.91	6.91
160000	6.91	6.91
180000	6.91	6.91
200000	6.90	6.90
220000	6.89	6.89
240000	6.88	6.88
260000	6.87	6.87
280000	6.86	6.86
300000	6.85	6.85
320000	6.84	6.84
340000	6.83	6.83
360000	6.82	6.82
380000	6.81	6.81
400000	6.80	6.80
420000	6.80	6.80
440000	6.79	6.79
460000	6.78	6.78
480000	6.78	6.78
500000	6.77	6.77
520000	6.77	6.77
540000	6.76	6.76
560000	6.76	6.76
580000	6.76	6.76
600000	6.75	6.75

Conclusion: The combined form of the Streeter Phelps Equation and nitrification terms gives the same results as performing the operations separately. Use of the combined equation may be seen in Appendix 2.

Appendix 2 - Use of the Streeter Phelps/Nitrification Equation

Use of the Streeter Phelps/nitrification equation in a sample calculation.
Data for the flow rates and concentrations of the stream and leachate are given below.
A sensitivity analysis for the reaeration coefficient is included.

First the continuity equation should be used to calculate the concentrations of ammonia and BOD after mixing with the receiving stream (C_i).

Continuity Equation	Figure 1 Median Flow		Figure 2 Low Flow	
$Q_L C_L + Q_S C_S = Q_i C_i$, where $Q_i = Q_L + Q_S$	Ammonia	BOD	Ammonia	BOD
	mg-N/L	mg/L	mg-N/L	mg/L
Q_L = flow rate of leachate (gal/d)	1000	1000	1000	1000
C_L = concentration of parameter in leachate (mg/L)	250	100	250	100
Q_S = flow rate of receiving stream (gal/d)	2.30E+07	2.30E+07	1.28E+06	1.28E+06
C_S = background concentration of parameter in receiving stream (mg/L)	0.1	5	0.1	5
Q_i = flow rate downstream of discharge (gal/d)	2.30E+07	2.30E+07	1.28E+06	1.28E+06
C_i = concentration of parameter downstream of discharge (mg/L)	0.11	5.00	0.30	5.07

Next the Streeter Phelps/nitrification equation can be used with the downstream concentrations to predict downstream dissolved oxygen concentrations.

$$\mathbf{c = c_s - \{K_d/(K_a - K_r)\ [\exp(-K_r(x/U)) - \exp(-K_a(x/U))]\}L_o - (c_s - c_o)\exp(-K_a(x/U))}$$
$$\mathbf{- L_o^N * [1-\exp(-K_N(x/U))]}$$

	Figure 1 Median Flow Value	Figure 2 Low Flow Value
c = dissolved oxygen concentration at a distance x downstream of the discharge (mg/L)	variable	
c_s = saturation concentration of dissolved oxygen (mg/L)	8	8
c_o = initial dissolved oxygen concentration (mg/L)	6	6
K_d = effective deoxygenation rate of organic carbon (d^{-1})	0.3	1.5
K_a = reaeration coefficient (d^{-1})	0, 5, 10	0, 5, 10
K_r = overall loss rate of CBOD due to settling and oxidation of soluble BOD (d^{-1})	0.3	1.5
x = distance downstream (ft)	variable	variable
U = average stream velocity (ft/d)	86400	43200
L_o = initial amount of carbonaceous material (mg/L)	5.00	5.07
K_N = rate of oxidation of ammonia and organic nitrogen (d^{-1})	0.3.	1.5
L_o^N = initial amount of ammonia and organic nitrogen (mg-N/L)	0.11	0.30

Crabtree Creek, Median Flow = 36 cfs

Figure 1. K_a graph, Q_L = 1000 gal/d, Q_S = 2.3 E7 gal/d, C_{Lamm} = 250 mg-N/L, C_{Lbod} = 100 mg/L, C_{Samm} = 0.11 mg-N/L, C_{Sbod} = 5 mg/L.

Distance x (ft)	c mg-O/L K_a = 0 d^{-1}	c mg-O/L K_a = 0.1 d^{-1}	c mg-O/L K_a = 0.26 d^{-1}	c mg-O/L K_a = 5 d^{-1}
0	6.00	6.00	6.00	6.00
20000	5.66	5.71	5.78	7.17
40000	5.34	5.44	5.60	7.54
60000	5.04	5.21	5.45	7.67
80000	4.76	4.99	5.33	7.72
100000	4.50	4.81	5.23	7.74
120000	4.26	4.64	5.16	7.75
140000	4.03	4.49	5.10	7.76
160000	3.82	4.36	5.06	7.77
180000	3.63	4.25	5.04	7.78
200000	3.44	4.15	5.03	7.79
220000	3.27	4.07	5.04	7.79
240000	3.11	4.00	5.05	7.80
260000	2.96	3.94	5.08	7.81
280000	2.82	3.90	5.11	7.81
300000	2.69	3.86	5.15	7.82
320000	2.57	3.84	5.19	7.82
340000	2.46	3.82	5.24	7.83

Crabtree Creek, Low Flow = 2.0 cfs

Figure 2. K_a graph, Q_L = 1000 gal/d, Q_S = 1.28E6 gal/d, C_{Lamm} = 250 mg-N/L, C_{Lbod} = 100 mg/L, C_{Samm} = 0.3 mg-N/L, C_{Sbod} = 5.07 mg/L.

Distance x (ft)	c mg-O/L K_a = 0 d^{-1}	c mg-O/L K_a = 0.26 d^{-1}	c mg-O/L K_a =1.4 d^{-1}	c mg-O/L Ka = 5 d-1
0	6.00	6.00	6.00	6.00
2000	4.42	4.59	5.21	6.43
4000	3.31	3.70	5.00	6.78
6000	2.52	3.18	5.12	7.04
8000	2	2.91	5.39	7.23
10000	2	2.81	5.71	7.37
12000	1	2.83	6.04	7.47
14000	1	2.93	6.34	7.54
16000	1	3.07	6.61	7.58
18000	1	3.25	6.83	7.62
20000	1	3.44	7.02	7.64
22000	1	3.65	7.17	7.66
24000	1	3.85	7.29	7.67
26000	1	4.05	7.38	7.68
28000	1	4.25	7.46	7.69
30000	1	4.44	7.51	8
32000	1	4.62	7.56	8
34000	1	4.79	7.59	8

Distance x (ft)	c mg-O/L K_a = 0 d^{-1}	c mg-O/L K_a = 0.1 d^{-1}	c mg-O/L K_a = 0.26 d^{-1}	c mg-O/L K_a = 5 d^{-1}	Distance x (ft)	c mg-O/L K_a = 0 d^{-1}	c mg-O/L K_a = 0.26 d^{-1}	c mg-O/L K_a =1.4 d^{-1}	c mg-O/L Ka = 5 d-1
360000	2.35	3.81	5.30	7.83	36000	1	4.96	7.62	8
380000	2.26	3.80	5.35	7.83	38000	1	5.12	7.64	8
400000	2.16	3.81	5.41	7.84	40000	1	5.27	7.65	8
420000	2.08	3.82	5.48	7.84	42000	1	5.41	7.67	8
440000	2.00	3.83	5.54	7.84	44000	1	5.54	7.67	8
460000	1.92	3.85	5.61	7.85	46000	1	5.66	7.68	8
480000	1.86	3.88	5.68	7.85	48000	1	5.78	7.69	8
500000	1.79	3.90	5.74	7.85	50000	1	5.89	7.69	8
520000	1.73	3.94	5.81	7.86	52000	1	6.00	7.69	8
540000	1.67	3.97	5.88	7.86	54000	1	6.10	7.69	8
560000	1.62	4.01	5.95	7.86	56000	1	6.19	7.70	8
580000	1.57	4.05	6.01	7.86	58000	1	6.28	7.70	8
600000	1.53	4.09	6.08	7.86	60000	1	6.36	7.70	8
620000	1.48	4.14	6.14	7.87	62000	0.63	6.44	7.70	7.70
640000	1.44	4.19	6.21	7.87	64000	0.63	6.51	7.70	7.70
660000	1.41	4.23	6.27	7.87	66000	0.63	6.58	7.70	7.70
680000	1.37	4.28	6.33	7.87	68000	0.63	6.65	7.70	7.70
700000	1.34	4.33	6.39	7.87	70000	0.63	6.71	7.70	7.70
720000	1.31	4.39	6.45	7.87	72000	0.63	6.77	7.70	7.70
740000	1.28	4.44	6.51	7.87	74000	0.63	6.82	7.70	7.70
760000	1.26	4.49	6.56	7.88	76000	0.63	6.87	7.70	7.70
780000	1.23	4.55	6.62	7.88	78000	0.63	6.92	7.70	7.70
800000	1.21	4.60	6.67	7.88	80000	0.63	6.97	7.70	7.70
820000	1.19	4.65	6.72	7.88	82000	0.63	7.01	7.70	7.70

Distance x (ft)	c mg-O/L $K_a = 0\ d^{-1}$	c mg-O/L $K_a = 0.1\ d^{-1}$	c mg-O/L $K_a = 0.26\ d^{-1}$	c mg-O/L $K_a = 5\ d^{-1}$	Distance x (ft)	c mg-O/L $K_a = 0\ d^{-1}$	c mg-O/L $K_a = 0.26\ d^{-1}$	c mg-O/L $K_a = 1.4\ d^{-1}$	c mg-O/L Ka = 5 d-1
840000	1.17	4.71	6.77	7.88	84000	0.63	7.05	7.70	7.70
860000	1.15	4.76	6.82	7.88	86000	0.63	7.09	7.70	7.70
880000	1.13	4.82	6.87	7.88	88000	0.63	7.12	7.70	7.70
900000	1.11	4.87	6.91	7.88	90000	0.63	7.16	7.70	7.70
920000	1.10	4.93	6.95	7.88	92000	0.63	7.19	7.70	7.70
940000	1.09	4.98	6.99	7.88	94000	0.63	7.22	7.70	7.70
960000	1.07	5.03	7.03	7.88	96000	0.63	7.25	7.70	7.70
980000	1.06	5.09	7.07	7.88	98000	0.63	7.27	7.70	7.70
1000000	1.05	5.14	7.11	7.88	100000	0.63	7.30	7.70	7.70
1020000	1.04	5.19	7.14	7.88	102000	0.63	7.32	7.70	7.70
1040000	1.03	5.24	7.18	7.88	104000	0.63	7.34	7.70	7.70
1060000	1.02	5.30	7.21	7.88	106000	0.63	7.37	7.70	7.70
1080000	1.01	5.35	7.24	7.89	108000	0.63	7.38	7.70	7.70
1100000	1.00	5.40	7.27	7.89	110000	0.63	7.40	7.70	7.70
1120000	0.99	5.45	7.30	7.89	112000	0.63	7.42	7.70	7.70
1140000	0.99	5.50	7.33	7.89	114000	0.63	7.44	7.70	7.70
1160000	0.98	5.54	7.36	7.89	116000	0.63	7.45	7.70	7.70
1180000	0.97	5.59	7.38	7.89	118000	0.63	7.47	7.70	7.70
1200000	0.97	5.64	7.41	7.89	120000	0.63	7.48	7.70	7.70
1220000	0.96	5.69	7.43	7.89	122000	0.63	7.49	7.70	7.70
1240000	0.96	5.73	7.45	7.89	124000	0.63	7.51	7.70	7.70
1260000	0.95	5.78	7.47	7.89	126000	0.63	7.52	7.70	7.70
1280000	0.95	5.82	7.49	7.89	128000	0.63	7.53	7.70	7.70
1300000	0.95	5.86	7.51	7.89	130000	0.63	7.54	7.70	7.70

Distance x (ft)	c mg-O/L K_a = 0 d^{-1}	c mg-O/L K_a = 0.1 d^{-1}	c mg-O/L K_a = 0.26 d^{-1}	c mg-O/L K_a = 5 d^{-1}	Distance x (ft)	c mg-O/L K_a = 0 d^{-1}	c mg-O/L K_a = 0.26 d^{-1}	c mg-O/L K_a =1.4 d^{-1}	c mg-O/L Ka = 5 d-1
1320000	0.94	5.91	7.53	7.89	132000	0.63	7.55	7.70	7.70
1340000	0.94	5.95	7.55	7.89	134000	0.63	7.56	7.70	7.70
1360000	0.94	5.99	7.57	7.89	136000	0.63	7.56	7.70	7.70
1380000	0.93	6.03	7.58	7.89	138000	0.63	7.57	7.70	7.70
1400000	0.93	6.07	7.60	7.89	140000	0.63	7.58	7.70	7.70
1420000	0.93	6.11	7.61	7.89	142000	0.63	7.59	7.70	7.70
1440000	0.92	6.15	7.63	7.89	144000	0.63	7.59	7.70	7.70
1460000	0.92	6.18	7.64	7.89	146000	0.63	7.60	7.70	7.70
1480000	0.92	6.22	7.65	7.89	148000	0.63	7.61	7.70	7.70
1500000	0.92	6.26	7.66	7.89	150000	0.63	7.61	7.70	7.70
1520000	0.92	6.29	7.67	7.89	152000	0.63	7.62	7.70	7.70
1540000	0.91	6.33	7.69	7.89	154000	0.63	7.62	7.70	7.70
1560000	0.91	6.36	7.70	7.89	156000	0.63	7.63	7.70	7.70
1580000	0.91	6.40	7.71	7.89	158000	0.63	7.63	7.70	7.70
1600000	0.91	6.43	7.72	7.89	160000	0.63	7.63	7.70	7.70
1620000	0.91	6.46	7.72	7.89	162000	0.63	7.64	7.70	7.70
1640000	0.91	6.49	7.73	7.89	164000	0.63	7.64	7.70	7.70
1660000	0.91	6.52	7.74	7.89	166000	0.63	7.64	7.70	7.70
1680000	0.90	6.55	7.75	7.89	168000	0.63	7.65	7.70	7.70
1700000	0.90	6.58	7.76	7.89	170000	0.63	7.65	7.70	7.70
1720000	0.90	6.61	7.76	7.89	172000	0.63	7.65	7.70	7.70
1740000	0.90	6.64	7.77	7.89	174000	0.63	7.66	7.70	7.70
1760000	0.90	6.67	7.78	7.89	176000	0.63	7.66	7.70	7.70
1780000	0.90	6.70	7.78	7.89	178000	0.63	7.66	7.70	7.70

Distance x (ft)	c mg-O/L $K_a = 0\ d^{-1}$	c mg-O/L $K_a = 0.1\ d^{-1}$	c mg-O/L $K_a = 0.26\ d^{-1}$	c mg-O/L $K_a = 5\ d^{-1}$	Distance x (ft)	c mg-O/L $K_a = 0\ d^{-1}$	c mg-O/L $K_a = 0.26\ d^{-1}$	c mg-O/L $K_a = 1.4\ d^{-1}$	c mg-O/L Ka = 5 d-1
1800000	0.90	6.72	7.79	7.89	180000	0.63	7.66	7.70	7.70
1820000	0.90	6.75	7.79	7.89	182000	0.63	7.67	7.70	7.70
1840000	0.90	6.77	7.80	7.89	184000	0.63	7.67	7.70	7.70
1860000	0.90	6.80	7.80	7.89	186000	0.63	7.67	7.70	7.70
1880000	0.90	6.82	7.81	7.89	188000	0.63	7.67	7.70	7.70
1900000	0.90	6.85	7.81	7.89	190000	0.63	7.67	7.70	7.70
1920000	0.90	6.87	7.82	7.89	192000	0.63	7.67	7.70	7.70
1940000	0.90	6.89	7.82	7.89	194000	0.63	7.68	7.70	7.70
1960000	0.90	6.92	7.82	7.89	196000	0.63	7.68	7.70	7.70
1980000	0.90	6.94	7.83	7.89	198000	0.63	7.68	7.70	7.70
2000000	0.89	6.96	7.83	7.89	200000	0.63	7.68	7.70	7.70