Toxicity of landfill leachates

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The character of landfill leachates has been described by numerous investigators. Field studies of municipal and other solid waste landfills, as well as controlled experiments using lysimeters and pilot-scale fills, have provided considerable documentation defining both the nature of leachates and their patterns of variability in terms of composition, strength, and environmental impact. These data generally include a wide range of general chemical parameters, along with additional selective information describing nutrient content, organic and inorganic composition, and microbiology. In addition, the presence of numerous other specific compounds and trace substances, both organic and inorganic, including specific bacteria and viruses have been isolated and identified.

One very important parameter, however, is conspicuously absent from information presented in published leachate literature. This parameter is *leachate toxicity*. Seldom, if ever, in the broad range of leachate literature has assessment of leachate toxicity been reported. Presumably this is a reflection of the fact that the toxicity of landfill leachates has not often been studied.

The project described herein was initiated in an attempt to fill what appears to be a considerable gap in existing information by making a review of leachate toxicity based on the results of a number of leachate studies. Toxicity assessments have been made using leachates from a variety of sources. They include "natural" leachates from several existing landfill operations as well as "synthetic" leachates from lysimeter studies.

In addition to reviewing leachate toxicity data, this investigation also attempts to:

- Identify potential agents of toxicity and correlate these with leachate chemical composition,
- Assess the importance of pH as it affects toxicity determinations, and
- Document and explain observed temporal changes in leachate toxicity.

DESCRIPTION OF METHOD

Information from a number of leachate toxicity studies was obtained from several local water quality management agencies. For the most part, toxicity analyses had all been conducted at independent testing laboratories that used two bioassay methods, the standard static 96-hour bioassay and a rapid toxicity assessment test—the residual oxygen bioassay (ROB) technique. Rainbow trout (*Salmo gairdneri*) were used as test organisms in all experiments.

The static 96-hour bioassay is widely used and well-documented as a toxicity test procedure.[1] The ROB test is a newly developed procedure that has recently received considerable attention and favorable reviews,[2] particularly in its application to leachate monitoring.[3, 4] In this test, serial dilutions are set up in sealable containers, along with a control using dilution water only. All solutions are aerated to saturation levels. Pretest dissolved oxygen (DO) measurements are taken and recorded. Test fish are introduced into each container, which is sealed for the duration of the test (normally 6 to 8 hours). Immediately after the death of all fish in a given container, DO is again measured and recorded. In the control test and at very low toxicant concentrations, death results from asphyxiation because of oxygen starvation at DO levels around 1 mg/l. These deaths are exemplified by a horizontal "no effect" line when placed on a log-log plot of residual DO versus wastewater concentration. At higher toxicant concentrations, death occurs prior to oxygen starvation as a result of the toxicant interaction with DO. This yields higher test-end, residual

TABLE I. Composition of solid waste in test lysimeters.

Category	Composition (% by wet weight)
Food waste	11.8
Garden waste	9.8
Paper products	47.6
Plastic, rubber, and leather	5.4
Textiles	3.6
Wood	4.7
Metals	8.7
Glass and ceramics	7.0
Ash, dirt, and rocks	1.4

oxygen levels which, when plotted, define a sloping "effect" or "response" line. At the point of intersection of these two lines, the threshold limit value (TLV) for a particular wastewater is determined.

TLV as a measurement of toxicity has been reported by Vigers and Maynard [2] to compare favorably with computed LC_{50} values as determined by the standard 96-hour static bioassay. In their studies which used pentachlorophenate as a reference toxicant, the mean LC_{50} value was 90 ppb while the TLV was 75.3 ppb. This difference was not found to be statistically significant at the 5% level. It is noted, however, that this comparative testing was based on a very limited set of observations. More recent investigations [5] have shown that the two measures may not in fact be directly comparable.

In this investigation, leachates were categorized according to source as being either "natural" that is, originating from full-scale existing landfill operations, or as "synthetic" when originating from test lysimeters containing a composite of typical solid waste materials (Table I). The lysimeters were operated under a carefully controlled set of temperatures and precipitation rates to simulate actual landfill conditions. Natural leachates were further broken down according to raw, diluted, and treated forms. Leachates that emanated directly from active fill areas of a landfill were considered under the raw leachate category. Collection of these samples was either directly from wells driven into the landfill or from leachate springs. In the diluted leachate category were those samples collected from surface waters, such as creeks and drainage ditches where leachate strength had been diluted by other surface waters and/or attenuated in its passage through the landfill. Treated leachate results are based on a set of raw leachate samples that were analyzed for toxicity after having received varying degrees of physical and combined physical/chemical treatment.

Synthetic leachates are also further broken down according to age of fill. Initial toxicity testing was conducted after the lysimeters had been in operation for approximately 1 year. The leachate at that time corresponded to that coming from a very "young" fill. A second set of tests was run some 5 years after a lysimeter start-up, which corresponds to a landfill of "medium age."

RESULTS

Leachate toxicity (Table II) varies widely with a range of more than three orders of magnitude. Lysimeter leachates were found to be the most highly toxic with average 96-hour LC_{50} values of 0.35% by volume. Raw natural leachates were also highly toxic with 96-hour LC_{50} values being in the range of 4.9 to 7.0%. Corresponding TLVs varied from 4.9 to 7.7%. A considerably reduced toxicity is evident for diluted natural leachates that were collected from drainage ditches surrounding a landfill. This reflects the beneficial effects of simple dilution or attenuation in reducing leachate toxicity in the natural setting. In most instances, however, measurements indicated that diluted leachates were still toxic at the point of discharge to receiving waters.

With the highly toxic synthetic leachates, a marked decrease in toxicity is apparent over time. Test results (Table II) show that leachate toxicity from young fills (< 1 year) was reduced over 80-fold within a 5-year period. More detailed lysimeter leachate toxicity data (Table III) reveal that the most significant decreases occurred in lysimeters that were operated wtih high rates of precipitation loading and leachate recycle (tanks S, M, and U). Hydraulic loading in each case was equivalent to an annual infiltration rate of 2 300 mm (90 in.) per year.

Toxicities of treated leachates varied from 16% to > 100%, with the lowest attainable toxicity consistently achieved being 96-hour LC_{50} values of > 100%. The treatment studies [6] evaluated physical treatment (filtration through columns of natural peat) and physical/chemical treatment (lime coagulation followed by peat filtration). Although both methods were able to produce nontoxic effluents, combined physical/chemical treatment offered an advantage in that additional throughput was possible before breakthrough

TABLE II. Toxicity of leachates.[a,b]

Leachate origins	Description	Toxicity[a] 96 h-LC_{50}	Toxicity[a] TLV	Data source
Natural	Raw	5.6[b] (4.2 –7.0)	6.3 (4.9 –7.7)	(6) (7)
	Diluted	64.3 (17.5 –100)	64.0 (18 –100)	(8) (9)
	Treated[c]	—	Nontoxic (16–100)	(6)
Synthetic	Young (<1 yr)	0.35 (0.062–0.70)	0.61 (0.18–0.96)	(10)
	Medium age (4–6 yrs)	28.8 (0.40 –100)	—	(11)

[a] Toxicity expressed as % V/V.
[b] Toxicities shown are arithmetically averaged results with associated range of observed values (bracketed) except for treated leachates.
[c] With treated leachates the lowest attainable toxicity is shown along with the complete range of toxicities achieved in treatment studies (bracketed).

occurred (Figure 1) although at a higher cost. In addition, treatment processes were capable of achieving considerable reductions (85 to 99%) of most other pollutants (Table IV), most notably, reductions in heavy metal concentrations. As can be seen, in addition to meeting effluent toxicity guidelines, leachates receiving physical and physical/chemical treatment met all other normal requirements, with the single exception being manganese concentrations that were slightly higher than stated objectives.[12]

To investigate the relationship between specific toxic agents and leachate, several sets of linear regression analyses were performed. Toxicants such as heavy metals, ammonia, and various organic substances (such as tannins and lignins) are often present in landfill leachates in relatively high concentrations (Table IV). Thus, potential agents of toxicity were readily identifiable, and it remained to be seen if these could be correlated with measured toxicities. Because of the paucity of chemical data describing the natural leachates of known toxicity, only lysimeter leachate data could be included in the regression analyses.

Simple regression between toxicity and concentration for the young lysimeter leachate

TABLE III. Toxicity of synthetic leachates.

Lysimeter	Initial Testing: Leachate Age (year)	Initial Testing: Toxicity (% V/V)	Second Testing: Leachate Age (year)	Second Testing: Toxicity (% V/V)
X	0.78	0.062	5.18	0.40
H	0.06	0.27	4.29	0.70
W	0.71	0.185	5.18	0.75
D	0.80	0.083	5.18	2.0
A	0.66	0.25	5.10	2.4
B	0.61	0.44	5.05	10.5
S	0.41	0.20	4.87	24.0
K	0.45	0.70	4.89	34.5
E	0.27	0.50	4.87	42.0
M	0.41	0.46	5.05	>100
U	0.76	0.65	5.15	>100

data yielded the regression coefficients (r^2) shown in Table V. Even the highest r^2 values indicated only marginally significant relationships. Most notable was the fact that the more toxic substances except for tanninlike compounds and nickel appear in the lower half of the ranking with coefficients in all cases being less than 0.3.

A stepwise multiple linear regression analysis was performed on these data. The most significant correlation using 95% confidence limits (F test) was obtained by the following equation:

$$\text{toxicity} = 3.09 - 0.966 \log (\text{iron}) \quad (1)$$

which had an r^2 value of 0.661. Again, while marginally accurate in accounting for observed toxicity, the above relationship is entirely unsatisfactory when it comes to accounting for contributory factors. In the presence of lethal concentrations of known toxicants (NH_3, As, Cr, Ni, and Zn), it is doubtful that iron was the only contributing factor.

With reduced confidence limits (90%), the following relationship was found:

$$\text{toxicity} = 3.27 - 0.687 \log (\text{acidity}) + 0.628 \log (\text{total dissolved solids}) - 1.077 \log (\text{iron}) \quad (2)$$

which had an r^2 value of 0.800.

While Equation 2 is more appealing because of its additional terms and the fact that more of the toxicity is explained, the reduced confidence limits make it less attractive. Thus,

TABLE IV. Chemical composition of landfill leachates.[a,b]

Parameter	Raw Leachate	Diluted Leachate	Treated Leachate	Synthetic Leachate (young)	Synthetic Leachate (medium age)	PCB Class "A A" Effluent Guidelines
pH	6.3–7.8	6.6–7.5		4.8–5.2	5.0–6.6	6.5–8.5
5-day biochemical oxygen demand (BOD$_5$)	120–2 980			9 100–20 600	11–26 000	45
Chemical oxygen demand (COD)	730–4 720	110–1 900	100–117	19 700–45 300	137–34 900	
Total carbon	930–1 830	154–620				
Total organic carbon (TOC)	810–1 600	30–300		7 300–16 350	83–9 150	
Total solids	3 190–6 490	750–4 800		10 000–33 000	718–18 400	
Total volatile solids	1 092–2 930			5 350–20 330	124–10 300	
Total dissolved solids (TDS)				9 810–32 670	389–17 000	
Acidity (as $CaCO_3$)	185–790	72		2 400–7 700	65–6 750	
Alkalinity (as $CaCO_3$)	1 350–3 510	337–1 280	53–80	4 100–7 700	184–7 600	
TKN	8.7–494		11.6–16.8			
NH_3-N	<0.3–427	7.5–38		312–711	<0.3–368	
Total N				459–1 200	6.5–630	
Total P	0.8–4.7	0.1–0.5	0.02–0.07	4.0–49.6	0.9–9.8	
Chloride	125–2 400	190–2 200	203–250	620–1 880	5.3–730	
Sulfide	<0.02–30			<0.2	—	0.5
Fluoride	0.27	0.14–0.22		0.03–0.98	<0.1–0.29	5.0
Cyanide				<0.05	—	0.10
Sulfate	5.3–250	7.6–80		192–1 320	<2–620	50
Nitrate		0.01–0.03		<0.5	<0.05	
Tanninlike compounds	62.4			402–950	4.6–1 117	
Na	128–840	120–1 200	40–44	400–1 360	3.1–524	
K	51–600	12–63	16–18	350–1 440	2.1–421	
Ca	164–1 065	86–410	19–22	920–2 450	63.8–1 280	
Mg	39–128	20–150	6.3–7.9	84–310	3.9–120	
Al	0.27–1.26			1.2–108	0.1–2.5	2.0
As	0.006–0.038			<0.006–2.75	<0.006–1.75	1.0
Ba	0.08			0.11–1.01	—	
Be	0.025			<0.05–0.09	—	
B	4.5–7.4			2.6–73	0.19–4.14	5.0
Cd	<0.001–0.004	<0.01		0.005–0.89	<0.001–0.162	0.005
Cr	0.025–0.085	<0.02–0.04		0.09–16.8	0.003–0.410	0.10
Cu	<0.01–0.05			0.03–0.12	0.009–0.09	0.20
Fe	1.6–30.3	1.8–42	0.07–0.13	308–1 136	195–1 820	0.3
Pb	0.023–0.065	<0.02–0.04	0.025–0.036	0.077–3.15	0.003–0.082	0.05
Mn	0.6–7.8	2.0–4.4	0.05–0.06	12.1–35.7	1.1–19.5	0.05
Hg				<0.001–0.008 8	<0.000 1–0.000 5	0.000 6
Mo	0.013			<0.01–0.6	—	0.2
Ni	<0.002–0.069	<0.01		0.15–0.79	<0.005–0.342	0.3
Ti	—			<0.2	—	
V	—			<0.05	—	
Zn	0.43–1.32	0.08–0.82	0.14–0.16	46–298	0.18–75	0.5
Toxicity	7.7–4.2	100–17.5	100	0.70–0.062	100–0.40	100

[a] All parameters expressed as mg/l except for pH and toxicity (96-hr TL_m expressed as % V/V).
[b] Metals expressed as total metal concentration.

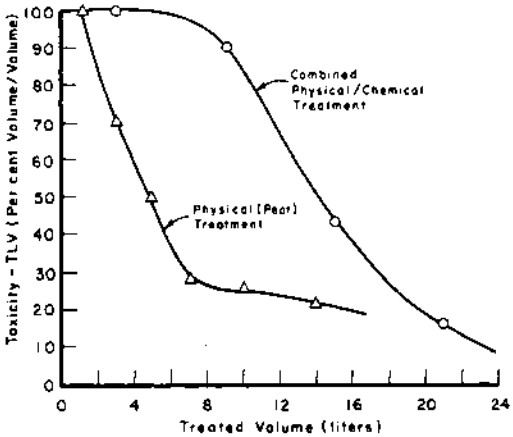

FIGURE 1. Leachate toxicity versus treated volume throughput.

for young leachates the equations obtained using multiple regression analysis do not represent much improvement over those obtained using simple regression. In terms of adequately explaining observed toxicity, results were unsatisfactory.

The reasons behind this are postulated as follows. First, the leachates considered in the foregoing analysis were all extremely toxic. The high toxicities could have resulted in considerable measurement errors, which would have had a severe effect on the results of regression because of increased variance as a result of measurement errors. Secondly, the extreme toxicity may have been a reflection of some highly complex, completely unknown synergistic and/or antagonistic phenomena that could not be considered in the analysis. Thirdly, the narrow range of toxicity values over which to regress made it difficult to obtain a good correlation. Thus, in spite of the comprehensive nature of chemical data available to characterize the young lysimeter leachates, more data were needed to enable a better understanding of the factors contributing to leachate toxicity.

TABLE V. Simple regression (r^2) coefficients—young leachate study.

Fe	0.661	TDS	0.388	Cr	0.176
Acidity	0.577	pH	0.374	Chloride	0.175
Tannins	0.561	Na	0.339	Ba	0.164
Ca	0.492	Mg	0.339	Pb	0.128
COD	0.479	K	0.334	As	0.079
Mn	0.451	Zn	0.280	B	0.005
Sulfate	0.410	Al	0.221	P	0.004
Ni	0.395	Fluoride	0.213	NH_3-N	0.001
		Cu	0.207		

TABLE VI. Simple regression (r^2) coefficients—medium age leachate study.

Parameter	r^2	Parameter	r^2
COD	0.748	Ni	0.580
Tannin	0.729	Organic N	0.567
Acidity	0.729	Cl	0.549
Log total hardness	0.722	K	0.523
TOC	0.701	Cu	0.513
Ca	0.677	Mg	0.490
BOD_5	0.677	Zn	0.475
H+	0.671	Fe	0.315
COD-BOD	0.656	Pb	0.250
Total N	0.643	Al	0.198
Mn	0.640	Cr	0.137
Total solids	0.628	Flouride	0.137
TDS	0.615	Total P	0.083
Volatile solids	0.602	B	0.039
Na	0.582		

Simple regression analyses were next performed on medium-age lysimeter leachates yielding considerably improved results (Table VI), mainly because these data covered a wider range of toxicities. Also, in this case the logarithm of toxicity was regressed with concentration. Of the known toxicants, tannin, nickel, and copper all had r^2 values greater than 0.5.

Stepwise multiple regression of medium-age leachate data also yielded improved results. The following relationship having an r^2 value of 0.943 was established using 95% confidence limits.

$$\log \text{ toxicity} = 1.427 - 0.386 \text{ (unionized } NH_3) - 101\,400\ [H^+] - 0.000\,539 \text{ (tannin)} - 4.074 \text{ (Cu)} \quad (3)$$

Each term in Equation 3, with the exception of the constant and the hydrogen ion concentration, can be rationalized in terms of toxicity to fish. The constant term describes a residual toxicity in these data that cannot be accounted for by the independent variables. The hydrogen ion concentration term shows that pH plays an important role as a determinant of toxicity, particularly at low pH values where this term is of singular importance. With pH values greater than 8, there is almost no change in toxicity attributable to pH. In summary, the second set of regression analyses showed good correlation between toxicity and specific, known toxicants. This included the observation that pH effects contributed substantially to leachate toxicity.

In further consideration of pH effects on leachate toxicity (Figure 2), these effects were found to play an extremely important role in all toxicity determinations, which often dramatically altered measured values of toxicity. For example, the toxicity of a sample adjusted to pH 7, a procedure normally used in bioassay testing, may be radically different from that measured at the pH of the sample as collected (in some cases by even more than one order of magnitude). With lysimeter and wood waste leachates, which tend to be quite acidic, a considerable increase in measured toxicity is evident when bioassays are run without pH adjustment. With slightly basic natural landfill leachates, the trend is reversed with measured toxicity being greater at high pH values. Thus, it is evident that toxicity measured at the standard pH value of 7 may not accurately reflect the true nature or full extent of leachate toxicity.

There are several possible explanations for the effects of pH change on toxicity. If ammonia nitrogen is present at pH values above 7, toxicity is known to increase as a result of increases in the more highly toxic undissociated ammonia fraction.[13] Precipitation of iron and zinc hydroxides and other insoluble compounds at these pH values have also been reported to result in increased toxicity, the mechanism being related to coating of fish gill structures.[13] Increased toxicity at lower pH values is also likely a result of changes in ionic form, in this case, of the weakly dissociated acids and bases. Sulfides and cyanides have been shown to be considerably more toxic at low pHs. For example, toxicity increases of 1 000-fold with a pH decrease from 8.0 to 6.5 have been reported for nickel cyanide.[13] Similar, although less dramatic increases have been reported for sodium sulfide. Another possible explanation, albeit speculative, is that the mechanisms inherent in synergistic/antagonistic phenomena are intensified at low pH values, thus further complicating the understanding of observed changes.

One final observation related to pH is that with wood waste leachates (Figure 2), although the effects of pH change are still apparent, they are not as severe. Thus, with wastes comprised mainly of organics, pH changes do not appear to play nearly as important a role as with wastes such as the

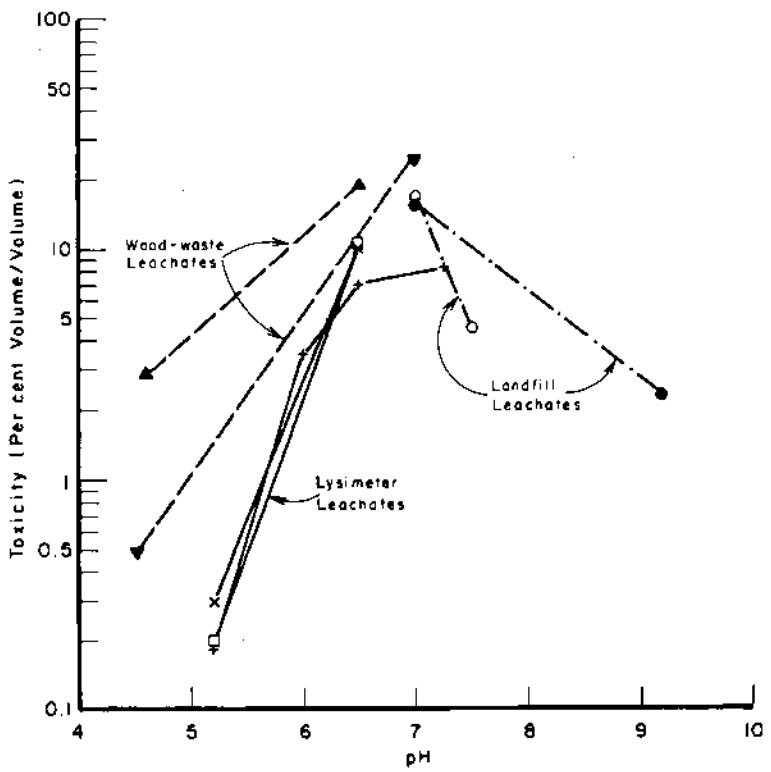

FIGURE 2. Leachate toxicity versus pH.

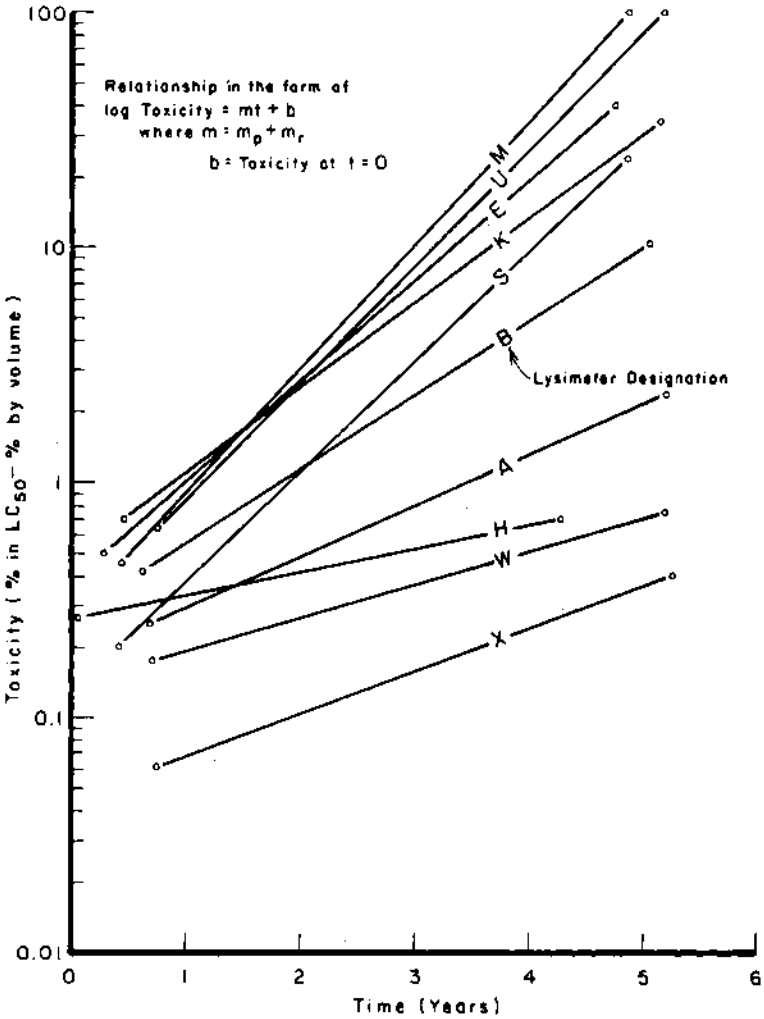

FIGURE 3. Plot of lysimeter leachate toxicity versus time.

lysimeter leachates that are more inorganic in nature.

DISCUSSION

Results from several aspects of this investigation deserve further discussion. The first is the importance of pH effects on measured toxicity. The implication of this finding is that leachate toxicity will almost certainly be underestimated if measurements are made solely on neutralized samples. A suitable alternative toxicity assessment procedure would be to conduct bioassays on samples with and without pH adjustment. By this procedure, a better understanding would be gained of the true nature of a particular leachate's toxicity and the extent to which it changes over a range of pH values. Also, if natural receiving waters were used for dilution purposes in all toxicity testing, some account would be taken of the buffer capacity of both the waste and receiving water, thereby enabling a better, more realistic assessment of the potential environmental impact of a particular leachate discharge to a specific receiving environment.

Another important aspect of this investigation is the observed temporal changes in leachate toxicity (Figure 3), in particular as it relates to precipitation rates and leachate recycle rates (Table VII).

A relationship was developed that quantified observed changes in terms of precipitation and recycle rates. For tanks with no recycle (Figure 4), this can be expressed as:

$$m_p = 0.00153p^{0.70} \qquad (4)$$

TABLE VII. Description of lysimeters and operating conditions.

Tank No.	Depth and Type of Waste	Depth and Type of Final Cover	Initial Condition	Annual Hydraulic Loading (rainfall equivalent)	Leachate Recycle	Recycle Rate (rainfall equivalent)	Precipitation Rate (rainfall equivalent)
A	2.73 m garbage	0.68 m hog fuel	Saturated	1 143	No	—	1 143
B	2.73 m garbage 378.5 l septic tank sludge	0.68 m soil	Saturated	2 286	No	—	2 286
E	2.73 m garbage 75.7 l septic tank sludge	0.68 m soil	Field moisture	1 143	Yes	424	719
H	2.73 m garbage	0.68 m soil	Field moisture	381	No	—	381
K	2.73 m garbage	0.68 m soil	Field moisture	2 286	No	—	2 286
M	2.73 m garbage	0.68 m hog fuel	Field moisture	2 286	Yes	897	1 389
S	2.73 m garbage 75.7 l septic tank sludge	0.68 m soil	Field moisture	2 286	Yes	1 613	673
U	2.73 m garbage	0.51 m hog fuel	Saturated	2 286	Yes	1 212	1 074
W	2.73 m garbage 378.5 l septic tank sludge	0.68 m soil	Saturated	381	Yes	86	295
X	2.73 m garbage	0.68 m hog fuel	Saturated	381	Yes	91	290

[a] Annual hydraulic loadings, recyle rates, and precipitation rate are expressed in rainfall equivalent units as mm/y (mm × 0.03937 = in.).

where m_p is the change in toxicity attributable to p millimetre of annual precipitation. The coefficient of determination for this power curve relationship is 0.99, signifying an excellent fit.

If the effect of leachate recycle on reducing toxicity is considered to be distinct and separate from that caused by precipitation, a relationship can be established for the data from tanks with recycle (E, M, S, O, W, X) that relates recycle rate to the portion of toxicity change not accounted for by precipitation (Figure 5):

$$m_r = 0.00851r^{0.51} \quad (5)$$

where m_r is the change in toxicity attributable to r mm of annual leachate recycle. Although the degree of fit ($r^2 = 0.86$) is not quite as good as the previously described relationship, it still signifies a good power curve fit.

Combining the foregoing two components of slope, observed changes in toxicity over time can be described by the following general relationship:

$$\log \text{toxicity} = (0.00153\ p^{0.70} + 0.00851r^{0.51})\ t + b \quad (6)$$

where toxicity is expressed as % (V/V), p is the annual precipitation rate (millimeter per year), r is the annual recycle rate (millimeter per year), t is time (years), and b is initial leachate toxicity, determined at $t = 0$.

Thus, at least for the data collected by this study, a reasonable relationship can be developed that describes toxicity changes with time according to initial leachate toxicity, precipitation, and recycle rates. The utility of such a relationship is that it may be used as a predictive model for temporal changes in toxicity.

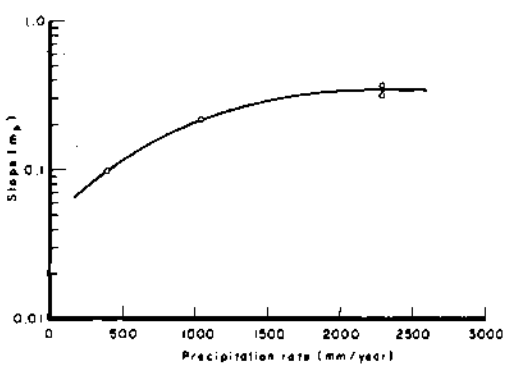

FIGURE 4. Plot of toxicity changes versus precipitation.

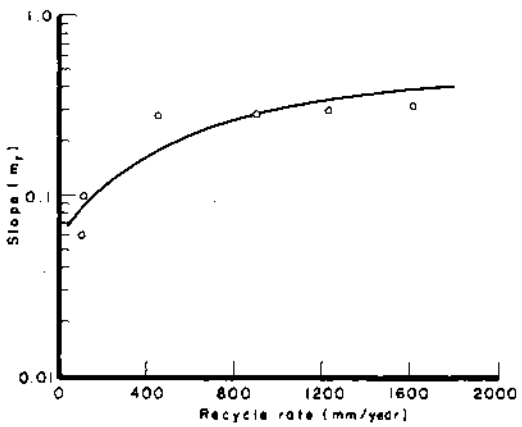

FIGURE 5. Toxicity change not accounted for by precipitation versus recycle rate.

For example, considering a leachate with strength comparable to that from lysimeter H, in the absence of dilution and without treatment, it would take an estimated 15 years to become nontoxic. If the leachate in this low rainfall example were to be completely contained and recycled, a nontoxic effluent would be achieved in less than half that time.

It is recognized that this rather simplistic model may not accurately reflect the more complex situation that exists in most field-scale landfills. There, many additional factors, such as temperature variation, precipitation variation, and particle size of refuse, come into play that could not be considered by this analysis. It is anticipated, however, that it is at least illustrative of the situation in an existing landfill.

In terms of understanding the mechanisms governing temporal changes in leachate toxicity, the fact that both recycle and precipitation rates are similarly related suggests that observed changes might be simply because of flushing action. However, the higher value of the coefficient in Equation 5 suggests that there may be additional benefits attributable to leachate recycling. The lower value of the exponent suggests that these benefits become marginally less important with increasing recycle rates.

The accelerated reduction of toxicity as a result of leachate recycling parallels the findings of other investigators [14] who have found that recycling results in pollutant attenuation and a general improvement of leachate quality. In contrast to those findings, however, which also showed significant reductions in organic pollutants resulting from the addition of wastewater sludges, this research has indicated that there is no detectable reduction in toxicity as a result of sludge additions. However, some attenuation of toxic substances attributable to the addition of wastewater sludges, such as heavy metals, has been noted during long-term lysimeter leachate monitoring studies.[11]

CONCLUSIONS

From a review of a broad range of data presented to describe leachate toxicity, it is concluded that landfill leachates can be, and usually are, highly toxic. Specifically:

- "Natural" leachates are highly toxic within the confines of landfills and usually remain so even at the point of receiving stream discharge, in spite of being attenuated somewhat by that time because of the considerable dilution afforded by surface drainage and groundwaters.
- Agents of toxicity are readily identifiable in landfill leachates and can be correlated with measured toxicity. In multiple regression analysis, over 94% of observed toxicity could be explained by variations in un-ionized ammonia, tannin, copper, and hydrogen ion concentration.
- The role of pH is important in toxicity measurement. Test conditions have a definite bearing on results obtained, which can often be misleading. To alleviate these problems and better understand the full extent of toxicity, it is recommended that, in addition to testing at pH 7, toxicity be measured at the original pH of the leachate using the actual receiving water for any dilution required in the testing.
- Temporal changes in toxicity parallel the documented patterns of variation that characterize other leachate parameters, namely an apparent logarithmic decay over time as the fill "ages." This attenuation is accelerated under conditions of high precipitation.
- Leachate recycling was shown to be an effective means of attenuating toxicity. Recycling reduced toxicity five times as fast per unit equivalent precipitation as precipitation alone did.
- Physical treatment of leachate with peat and combined physical/chemical treatment were both shown to be effective in reducing leachate toxicity.

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