

# Leaching of Lead from Computer Printed Wire Boards and Cathode Ray Tubes by Municipal Solid Waste Landfill Leachates

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The proper management of discarded electronic devices (E-waste) is an important issue for solid waste professionals because of the magnitude of the waste stream and because these devices often contain a variety of toxic metals (e.g., lead). While recycling of E-waste is developing, much of this waste stream is disposed in landfills. Leaching tests are frequently used to characterize the potential of a solid waste to leach when disposed in a landfill. In the United States, the Toxicity Characteristic Leaching Procedure (TCLP) is used to determine whether a solid waste is a hazardous waste by the toxicity characteristic. The TCLP is designed to simulate worst-case leaching in a landfill environment where the waste is co-disposed with municipal solid waste (MSW). While the TCLP is a required analysis from a regulatory perspective, the leachate concentrations measured may not accurately reflect the concentrations observed under typical landfill conditions. Another method that can be performed to assess the degree a pollutant might leach from a waste in a landfill is to use actual landfill leachate as the leaching solution. In this study, two lead-containing components found in electronic devices (printed wire boards from computers and cathode ray tubes from computers and televisions) were leached using the TCLP and leachates from 11 Florida landfills. California's Waste Extraction Test (WET) and the Synthetic Precipitation Leaching Procedure were also performed. The results indicated that the extractions using MSW landfill leachates resulted in lower lead concentrations than those by the TCLP. The pH of the leaching solution and the ability of the organic acids in the TCLP and WET to complex with the lead are factors that regulate the amount of lead leached.

## Introduction

Discarded electronic devices, often referred to as "E-waste", represent a growing segment of the solid waste stream. Discarded electronic devices include both home consumer electronics (e.g., televisions, personal computers, laptop computers, radios, video cassette recorders, cellular phones) and commercial and industrial equipment (e.g., computers, industrial electronics). The management of these devices upon disposal has become a major issue of concern for environmental regulators in the solid waste arena. Concern

stems from both the magnitude of the waste stream and the potential environmental impacts associated with the toxic chemicals found in most electronic devices. According to a study prepared for the U.S. EPA, discarded electronic devices have been estimated to comprise approximately 2–5% of the U.S. municipal solid waste (MSW) stream (1). Industry experts have projected that more than 20 million personal computers became obsolete in 1998 and that more than 60 million personal computers will be retired in 2005 (2). Approximately 3 million ton of E-waste was estimated to be disposed in U.S. landfills in 1997 (1).

Concerns have been raised that toxic chemicals will leach from these devices when disposed (3–5). The toxic chemicals commonly used in electronic devices include metals and metalloids (e.g., arsenic, cadmium, chromium, copper, lead, and mercury) and organic chemicals such as brominated flame retardants (BFRs). The printed wire boards (PWBs, also referred to as circuit boards) found in most E-waste, for example, may contain arsenic, cadmium, chromium, lead, and mercury (6, 7). Cathode ray tubes (CRTs) in computer monitors and televisions may contain barium, cadmium, copper, lead, zinc, and several rare earth metals (7). Lead is one heavy metal with known toxic properties that is found in large amounts in many electronic devices (6). Electronic devices, along with lead-acid batteries, are the major contributors of lead in the municipal solid waste stream (8). Lead-based solder (typically a 60:40 ratio of tin to lead), which is used to attach electrical components to PWBs, represents the major solder type used in most PWB applications (6, 7). Typical PWBs have been reported to contain approximately 50 g of tin-lead solder/m<sup>2</sup> of PWB (7) and approximately 0.7% of the total weight of a PWB (9). In CRTs, leaded glass provides shielding from X-rays generated during the picture projection process. Color CRTs contain 1.6–3.2 kg of lead on average (10). The possible effects of lead on human health and the environment are well-documented (11–13).

In the United States, under regulations promulgated as part of the Resource Conservation and Recovery Act (RCRA), solid wastes containing large amounts of leachable lead are regulated as hazardous wastes unless otherwise exempted. Lead leaching is measured using a batch extraction test known as the Toxicity Characteristic Leaching Procedure (TCLP, U.S. EPA Method 1311) (14, 15). In a previous study, color CRTs from televisions and computer monitors were found to leach enough lead using the TCLP to be toxicity characteristic (TC) hazardous wastes in most cases (16). The majority of samples tested exceeded the TC limit of 5 mg/L for lead. Discarded color CRTs are thus considered TC hazardous wastes unless test results show otherwise (17).

Generators other than households who dispose of more than 100 kg of color CRTs per month must manage them via a permitted hazardous waste facility. Generators who produce less than 100 kg per month (conditionally exempt small quantity generators [CESQGs]) may under RCRA dispose of these wastes in a state-permitted solid waste management facility (e.g., MSW landfill). Many states, however, ban CESQG hazardous waste from landfills. RCRA regulations exclude solid waste produced by households from the definition of hazardous waste; a color television or computer monitor that is disposed by a household is not a hazardous waste. At least one state, California, has not adopted the household waste exclusion. Because many CRTs can still be legally disposed in MSW landfills, state environmental regulators and local communities must determine what additional initiatives, if any, should be enacted to address CRT disposal. One possible action is to provide funding so CRTs can be

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collected at household hazardous waste collection facilities, by curbside collection, or through special collection events. The estimated cost to recycle one CRT has been reported to range from \$9 (computer monitors, small TVs) to \$35 (console TVs) (18). States also have the option of banning the disposal of these devices in landfills. In the United States, both California and Massachusetts have banned CRT disposal in landfills (17). With recent research indicating that other discarded electronic devices (those that contain PWBs with lead solder) will in many cases fail the TCLP for lead (19), similar questions will be raised for E-waste as a whole.

It is important to note that the TCLP was designed to be a conservative regulatory test to rapidly screen which solid wastes warrant more stringent management as hazardous wastes. The TCLP leaching solution uses acetic acid to represent the types of acids produced during the biological decomposition of waste in a landfill; the concentrations prescribed, however, were selected to represent worse-case acid-forming conditions (20). The TCLP extraction fluid pH is 4.9 (2.8 for alkaline wastes), while the pH of typical landfill leachates are much closer to neutral (21, 22). The TCLP was not intended, per se, to be a method for predicting pollutant leachate concentrations from a typical landfill where a potential hazardous waste is disposed. The question "what will the concentration of lead be in a landfill's leachate as result of disposing of lead-containing E-waste?" is a separate question from "is lead-containing E-waste a TC hazardous waste?". There is certainly value to having a better understanding of how the disposal of E-waste in landfills will impact leachate quality. As previously explained, even if a waste is determined to be hazardous by the TC, there will still be times when it is disposed in a landfill (from household waste, CESQG waste). In addition, wastes that do not exceed TC limits using TCLP might still leach sufficient concentrations of pollutants to elevate concentrations in the landfill's leachate. While modern landfills designed for disposal of MSW will be lined to intercept and collect the leachate, elevated pollutant concentrations in a landfill's leachate can pose problems for leachate treatment and disposal and perhaps impact the long-term management of leachate after the landfill is closed. Thus, an understanding of how E-waste might leach in a landfill is beneficial to landfill operators, who must decide which wastes they are willing to accept, and to policy-makers and regulators when weighing the costs and benefits of measures such as landfill bans or funding for household hazardous waste collection of E-waste.

The leaching of pollutants from solid wastes co-disposed with MSW can be evaluated by constructing and operating simulated landfills (20, 23). These tests, however, can be time-consuming and expensive. An alternative approach is to perform a relatively rapid leaching test like the TCLP but substituting MSW landfill leachate as the leaching fluid. Hooper et al. (24) used this approach and found that the TCLP extracted some elements at roughly the same concentration as California MSW landfill leachates, while other elements were extracted to greater extents in the landfill leachates. Similar research has not been reported for discarded electronic devices.

In this study, leaching tests using MSW leachates from lined landfills in Florida were performed to examine lead leachability of PWBs from computers and color CRTs from computer monitors and televisions. The TCLP was performed on the same samples. The objective of the research was to gather information regarding how much lead would leach from PWBs and CRT glass under typical Florida MSW landfill leachate conditions relative to TCLP. The purpose of the research was not to examine the applicability of the TCLP. As previously stated, the TCLP was purposely designed to provide a relatively rapid test that could distinguish solid wastes that warranted specific management as a hazardous

waste. Nor was the research intended to characterize either CRTs or computers for the RCRA TC. Such research has already been published for CRTs (16), and the PWBs tested in this study do not represent a whole computer. In addition to the TCLP, two other regulatory-based leaching tests were performed. The U.S. EPA's synthetic precipitation leaching procedure (SPLP) is a synthetic rainwater leaching test (15) and is sometimes used to assess the leachability of wastes disposed in inert landfills that do not contain large amounts of biodegradable matter. California's Waste Extraction Test (WET) is used to determine hazardous waste status in that state (25).

## Materials and Methods

Experimental methods included preparation of the PWBs and CRTs, collection and analysis of the MSW leachates, performance of the leaching tests, and analysis of extracted leachates.

**Sample Collection and Preparation.** In 2001 and 2002, a total of 30 personal computers were collected from individual donations, local electronic repair facilities, and a local household hazardous waste collection facility. In 2001, 20 personal computers were disassembled manually, and the PWBs were separated from the computers (set A). A total of 10 computers were collected and disassembled to retrieve PWBs in 2002 (set B). In both sets, only the large motherboards of the CPUs were collected; future references to PWB analysis in this paper refers to the motherboards. No preference was given in the selection of computers or PWBs to any specific type, manufacturer, or age. Manufacturing dates of the computers ranged from 1985 to 1998. The brand, type, and total weight of each computer were recorded. Following disassembly, the total weight of each PWB was recorded. On average, the PWBs (motherboards) made up approximately 6.0% of the total weight of the computers. After weighing, each wire board was size-reduced to approximately 5 cm by 5 cm using hand-held shears. No further size reduction procedure was conducted due to the difficulties in cutting some of the materials affixed to the boards. It is noted that this is larger than the size required by the TCLP (less than 9.5 mm). Again, the purpose of these experiments was to assess leaching of lead-containing components in MSW leachate relative to other standardized leaching tests and not to characterize for the TC. While the size reduction requirement of the TCLP may result in waste characteristics not representative of actual landfill conditions, it was conservatively designed to minimize the time to reach equilibrium conditions. The pieces of size-reduced PWBs were manually mixed in a 20-L high-density polyethylene (HDPE) bucket.

**CRTs in Computers and Televisions.** A total of 36 CRTs from televisions and computer monitors were collected from individual donations, electronics repair facilities, an electronics manufacturer, and institutional electronics disposal. A CRT consists of three major glass fractions: the neck, the funnel, and the faceplate. The CRT display unit was dismantled, and the different sections of the CRT were separated. Each section was placed in a stainless steel bowl and manually crushed with a hammer. The 36 CRTs tested were the same ones used in the previous study performed to determine if CRTs were TC hazardous wastes (16, 19). Thirty of the CRTs were color and 6 were monochrome. In the previous study, each fraction of each CRT was leached using the TCLP. In this study, all of the crushed glass from the funnel sections of the 30 color CRTs was combined and mixed in a stainless steel bowl. The mixed glass was then separated by size using stainless steel sieves. Figure 1 presents the particle size distribution of the glass. The glass pieces ranging from 4.75 to 6.3 mm were used for the leaching experiments in this study. This size range does meet the requirements of the

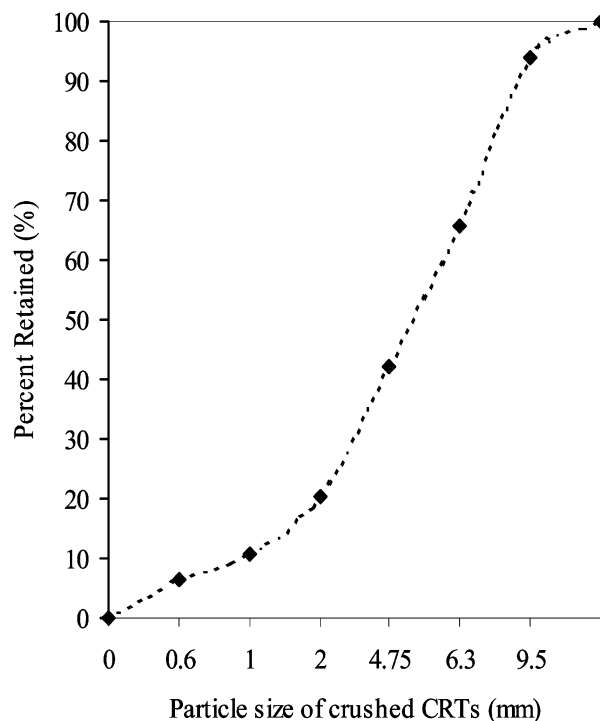


FIGURE 1. Particle size distribution of cathode ray tube glass.

TCLP. Again, it is noted that the purpose was not to perform a TC characterization; the glass tested was not representative of an entire CRT.

**MSW Leachate Sampling and Characteristics.** In 2001, leachates from five lined solid waste landfills in Florida (designated as sites A–E) were collected from leachate collection sumps using Teflon bailers. The sites selected consisted of four MSW landfills and one Class III landfill (site D). In Florida, Class III landfills receive yard trash, construction and demolition debris, carpet, cardboard, furniture, and other materials not expected to produce leachate that poses a threat to public health or the environment. In 2002, six different leachates were collected from six active MSW landfills (designated as sites F–I and 2002 sites A and B). Two landfills (sites A and B) were sampled in both 2001 and 2002.

After the leachate samples were collected, parameters immediately measured in the field included pH and oxidation reduction potential (Accumet portable AP10), dissolved oxygen (DO) (YSI Inc., model 55/12 FT), and specific conductance (HANNA Instruments, model H19033). Three samples from each site were collected in 1-L glass bottles to accommodate different chemical analyses. One sample was preserved with sulfuric acid (pH <2) for organic content analysis (e.g., COD and TOC). Another sample was acidified with nitric acid (pH <2) for metals analysis. A third sample used for conventional water quality parameter analysis was not preserved. All samples were stored on ice and transported to a refrigerated room until a chemical analysis was performed. In addition, approximately 20 L of leachate was collected in a HDPE container. The amount of headspace was minimized by filling the containers as much as possible. The leaching tests with landfill leachates were conducted within 24 h of collection to minimize possible changes in leachate characteristics.

The leachate samples collected were analyzed for various leachate parameters, including metals, anions, sulfides, alkalinity, total dissolved solids, and organic acids. U.S. EPA methods (15) and other standard methods (26) were employed. After sample filtration (0.45- $\mu$ m membrane filter), an ion chromatograph (Dionex DX-500) was used for ion analysis. Heavy metals in the leachate were determined by

inductively coupled plasma-atomic emission spectroscopy (ICP-AES, Thermo Jarrell Ash Corp. model 95970; U.S. EPA Method 6010B) after metals digestion for liquid samples (U.S. EPA Method 3020A). Volatile fatty acids (VFAs) were measured by gas chromatography (year 2002 leachate samples only) using a Shimadzu gas chromatograph (Shimadzu GC 9-AM) with a flame ionization detector (FID). Samples for VFAs were centrifuged at 10 000 rpm for 10 min, and the resultant supernatant was acidified with 1:9 v/v parts sample to 20%  $H_3PO_4$  containing 1000 mg/L isobutyrate. A total of 2  $\mu$ L of sample was injected onto a 2-m-long, 3.2-mm-wide i.d. glass column packed with 10% SP1000 and 1%  $H_3PO_4$  in Chromosorb WAW 100/120. Table 1 presents the parameters analyzed and the chemical characteristics of the leachates.

**Leaching Test Protocols.** The leaching tests performed in this study included the TCLP, SPLP, and WET. Table 2 summarizes the three standardized batch test protocols used as well as the extraction test protocol that utilized landfill leachates. TCLP, SPLP, and WET were performed on every sample. The same source of CRT glass was leached using the MSW landfill leachates collected in 2001 and 2002. The set A PWBs were leached using the 2001 landfill leachate samples, and the set B PWBs were leached using the 2002 landfill leachate samples. The extraction process included placing 100 g of sample (PWBs or CRTs) into a 2.2-L extraction vessel, adding 2 L (or 1 L for WET) of leaching solution to the vessel, tumbling for  $18 \pm 2$  h (or 48 h for WET), and filtering the extract using a pressurized filtration apparatus with a 0.7- $\mu$ m borosilicate glass fiber filter (Environmental Express TCLP filters). After filtration, the extract was digested (U.S. EPA Method 3020A) and analyzed for lead by the ICP-AES. All samples were leached in triplicate. Laboratory blanks, sample spikes, duplicates, and calibration check samples were performed as appropriate.

## Results and Discussion

**Leaching Results Using MSW Landfill Leachates.** The results of the batch leaching tests for the PWB pieces and the CRT glass using MSW landfill leachates are presented in Figure 2. In the PWB leaching experiments, lead concentrations ranged from 0.53 to 5.0 mg/L, with an average of 2.2 mg/L. For the CRT samples, lead concentrations ranged from 1.7 to 6.0 mg/L, with an average of 4.1 mg/L. Background levels of lead in the landfill leachates ranged from less than detection limit (0.04 mg/L) to 0.07 mg/L, as shown in Table 1. Lead concentrations in the extracts for site D and 2002 site A were corrected for the background lead concentrations.

The MSW leachates varied to some extent in their ability to extract lead from the PWB and the CRT samples, as presented in Figure 2. An analysis of variance (ANOVA) was conducted to determine whether there was any significant difference in lead concentration among the MSW leachate extracts. The critical value of the *F* distribution ( $\alpha = 0.05$ ,  $df_1 = 10$ ,  $df_2 = 22$ ) was 2.30. The computed values of *F* for the PWB and CRT samples were 5.0 and 2.7, respectively. This suggests that at least one of the mean lead concentrations in the MSW leachate extracts differed from the others. Fisher's least significant difference (LSD) test was then performed to determine differences among possible pair-wise combinations. In the case of the PWB data analysis, 2002 site A and 2002 site B were significantly greater than all the others. For the CRT data analysis sites D, E, I, A (2001 and 2002), and B (2001 and 2002) were significantly greater than those obtained by sites H, C, F, and G. For the same sites with different years (sites A and B), lead concentrations were different for the PWB extracts but not significantly different for the CRT samples. While the Class III landfill leachate (site D) extracted lead from the PWB samples at the lowest level, the average lead concentration of the CRT samples using the Class III landfill leachate was among the highest.



TABLE 1. Landfill Leachate Characteristics

parameters	sites sampled in 2001					sites sampled in 2002					
	site A	site B	site C	site D	site E	site A	site B	site F	site G	site H	site I
pH	7.74	7.74	6.80	7.30	7.48	7.97	7.89	6.53	8.10	8.22	7.92
conductivity (mS/cm)	18.0	9.19	26.9	3.4	10.1	19.4	12.3	4.62	20.1	3.84	13.3
TDS (mg/L)	6320	4610	16830	2450	3780	7960	5880	2620	9610	1960	5540
alkalinity (mg/L) <sup>a</sup>	5210	3120	7400	1800	3450	8050	4350	1250	6850	1550	5300
COD (mg/L)	2770	1320	2650	2230	1020	3080	2530	792	9570	1260	2150
Anions and Cation (mg/L)											
sulfide	0.20	0.09	0.07	0.03	0.04	0.225	7.75	<0.008	31.7	0.072	0.132
chloride	1610	748	9160	406	1070	1950	1270	296	2150	1500	2402
sulfate	<1.0	<1.0	<1.0	<1.0	<1.0	17.6	141	7.91	33.0	11.0	2.42
ammonia	814	441	437	97.3	614	2630	1160	20.6	1490	125	833
calcium	59.5	89.9	1190	278	110	44.1	128	275	177	174	67
potassium	418	268	1460	256	365	746	356	24	813	123	604
sodium	1100	1190	3410	274	1010	1990	1550	532	1920	367	1390
Organic Acid (mg/L)											
acetate	— <sup>b</sup>	—	—	—	—	40	366	35.0	582	273	13.5
propionate	—	—	—	—	—	19	13	22.6	176	32	11.3
isobutyrate	—	—	—	—	—	100	126	74.0	172	129	101
butyrate	—	—	—	—	—	23	7.5	<1.0	82	29	1.8
isovalerate	—	—	—	—	—	13.3	59	<1.0	159	56	0.4
valerate	—	—	—	—	—	3.0	<1.0	<1.0	52	36	0.26
Metal (mg/L)											
iron	30.7	4.2	26.4	7.4	3.6	5.1	5.0	13.1	5.8	16.2	13.1
lead	<0.04	<0.04	<0.04	0.07	<0.04	0.06	<0.04	<0.04	<0.04	<0.04	<0.04
zinc	0.11	<0.1	<0.1	<0.1	<0.1	0.12	<0.1	0.24	<0.1	0.23	0.24

<sup>a</sup> As CaCO<sub>3</sub>. <sup>b</sup> A dash (—) indicates that the parameter was not measured.

TABLE 2. Summary of Leaching Test Procedures

	standardized leaching test			MSW leachate
	TCLP	WET	SPLP	
pH of leaching solution	4.93 ± 0.05 (acetic acid and sodium hydroxide)	5.00 ± 0.05 (citric acid and sodium hydroxide)	4.20 ± 0.05 (sulfuric and nitric acids)	7.60 <sup>a</sup>
no. of CRTs and PWBs used	30/10 <sup>b</sup>	30/10 <sup>b</sup>	30/10 <sup>b</sup>	30/(20 or 10) <sup>c</sup>
solid to liquid ratio (g of waste to L of solution)	100 g/2 L	100 g/1 L	100 g/2 L	100 g/2 L
extraction period	18 ± 2 h	48 h	18 ± 2 h	18 ± 2 h

<sup>a</sup> Average of 11 landfill leachates. <sup>b</sup> Set A, 10 PWBs. <sup>c</sup> Set A, 10 PWBs in 2001; set B, 20 PWBs in 2002.

Several factors can impact how a given leachate will extract lead. There were not enough different leachate samples tested to perform a detailed analysis of all the factors impacting lead leaching. Some factors, however, will be discussed further. Previous leaching studies have shown that lead exhibits a marked increase in leachability at both low and high pH values. Minimum leachability is observed at neutral pH values, and an increase in leachability is observed at pH values less than 6 and greater than 10 (27–30). In this study, the leachate pH values of all the leachates fell within the range of 6–8; the change in pH during the leaching test was minor, with only slight increases noted from the original leachates. When lead concentrations were compared to final leachate pH values to evaluate a possible relationship, no significant correlation was found. This is largely a result of the very narrow pH range encountered among the leachate samples. The presence of organic acids (e.g., volatile fatty acids) in landfill leachate may also enhance metal mobility due to complexation (23, 31). VFAs were measured only for sites F–I, 2002 site A, and 2002 site B. No clear evidence of a correlation between organic acid concentrations and the amounts of lead extracted was found. Other factors that could impact metal leaching include dissolved organic matter, hardness, oxidation–reduction potential, ionic strength, and species that might result in precipitation (sulfides, hydroxides). Additional experimentation using more leachate

samples with a more complete analysis is needed to explore the possible relationships further.

**Comparison of MSW Leachate Results to Standardized Tests Results.** Figure 3 presents a comparison of the average lead concentrations measured using MSW landfill leachate with the concentrations measured using the standardized leaching tests. The MSW landfill leachate results represent an average of all 11 samples tested. In all cases, the TCLP extracted more lead from the size-reduced PWBs and the CRT glass than did the MSW leachate. The greater lead concentrations measured using the TCLP can be attributed to two primary factors. The higher affinity of acetate ions in TCLP for lead complexation resulted in higher levels of lead in the extracts. Acetate, a component of the TCLP leaching solution, is a monodentate ligand with an affinity for transition metal ions such as lead. Acetate, which chelates strongly with lead, enhances dissolution and complexation of metals (32–34). The concentration of acetate in the TCLP fluid is 5990 mg/L. This compares to the measured acetate concentrations in the range of 13–580 mg/L in the 2002 Florida MSW landfill leachate samples.

As discussed previously, pH also plays an important role in the leachability of lead. The initial and final pH values for the MSW landfill leachates were higher than those in the TCLP and WET. The initial pH of all MSW leachates was neutral, ranging from 6.5 to 8.2 (Table 1). This is typical for

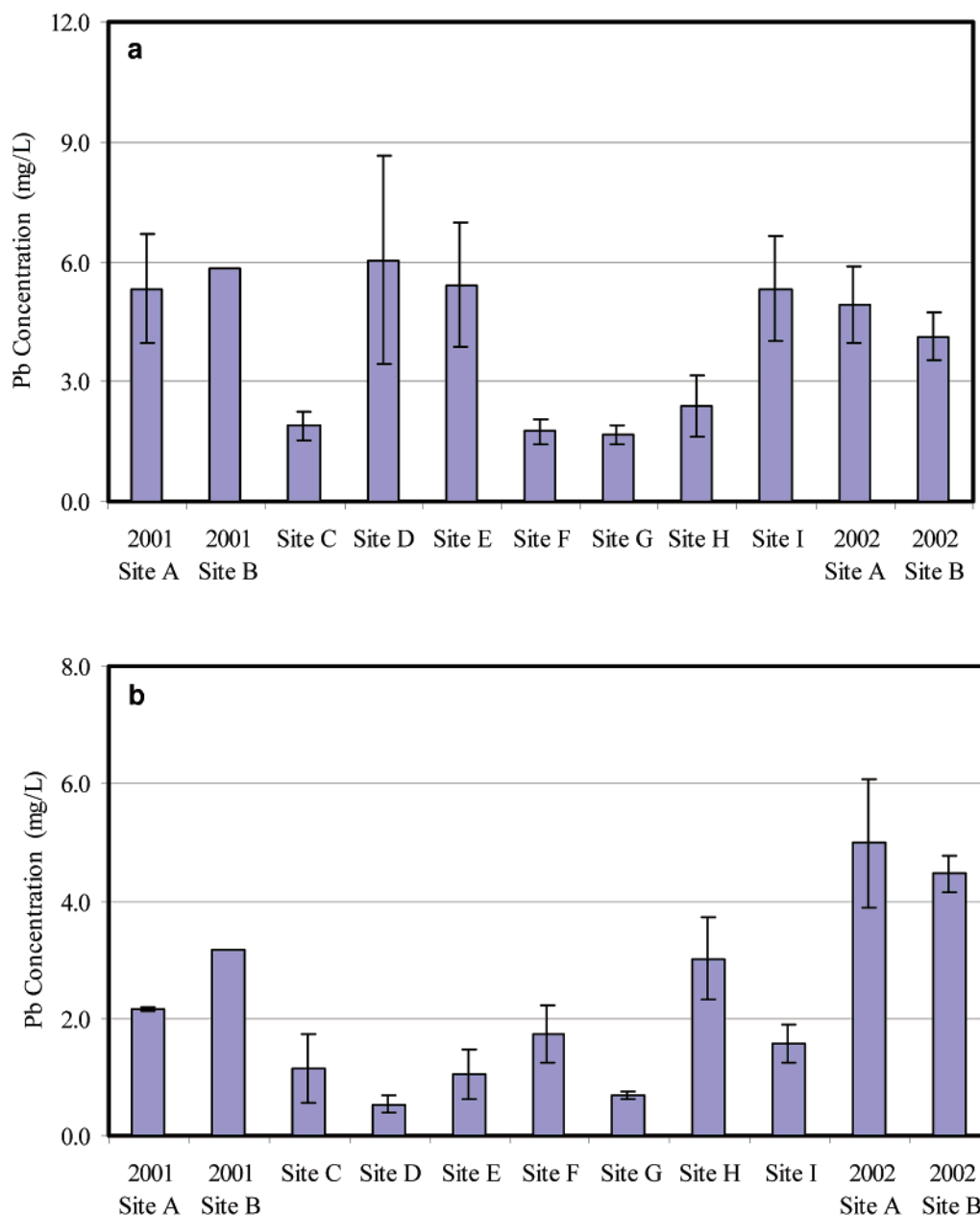


FIGURE 2. Pb leaching from CRTs (a) and PWBs (b) using MSW landfill leachates.

MSW landfill leachate pH (21, 22). Leaching tests with MSW leachate yielded slightly higher pH values than the initial pH values. As stated previously, the TCLP leaching solution was buffered with sodium hydroxide at a pH of 4.93. The TCLP final pH values were close to the initial pH values: 4.97 for printed wire boards and 5.08 for CRTs (Table 3). This indicated that the alkalinity released by the PWB pieces and CRT glass was not large enough to greatly change the pH during the 18-h TCLP extraction. The MSW leachate leaching tests were conducted at a pH that was higher than the pH value for the TCLP, resulting in less lead leaching (Table 3). In the California study that compared the leaching of multiple metals from a number of solid wastes using the TCLP and MSW leachates, the pH values of the MSW leachate extraction solution were also generally higher than the TCLP (24), but the pH effect on lead leaching was not as clearly observed.

When the average concentration of lead in MSW leachate extracts was compared to that of the WET (Figure 3), the results for CRT glass were similar to those obtained with the TCLP. Citrate, a component of the WET leaching solution,

is a tridentate ligand that chelates with metal ions such as lead (35). The final pH value of the WET for CRTs was close to the initial pH of 5.0 (Table 3). As was the case with the acetic acid-based TCLP, the use of citric acid in the WET coupled with lower pH values resulted in higher concentrations of lead from the CRT glass when compared to MSW leachate extracts. In the case of computer PWBs, the leaching results from the WET differed; the average lead concentration was close to that of the MSW leachate extracts. It is uncertain whether the mechanism for relatively low leachability of lead was precipitation or interference with other chemicals in the circuit boards. Further study is needed to determine the cause of the lower concentrations of lead in the WET extract. Hooper et al. (24) found lead to leach from several waste streams at significantly greater concentrations using the WET relative to those extracted by California MSW leachates.

The lead concentrations measured in the SPLP leachates were lower than the TCLP, WET, and the MSW landfill leachates. When comparing the SPLP results with those from the TCLP and WET, a similar rationale as previously

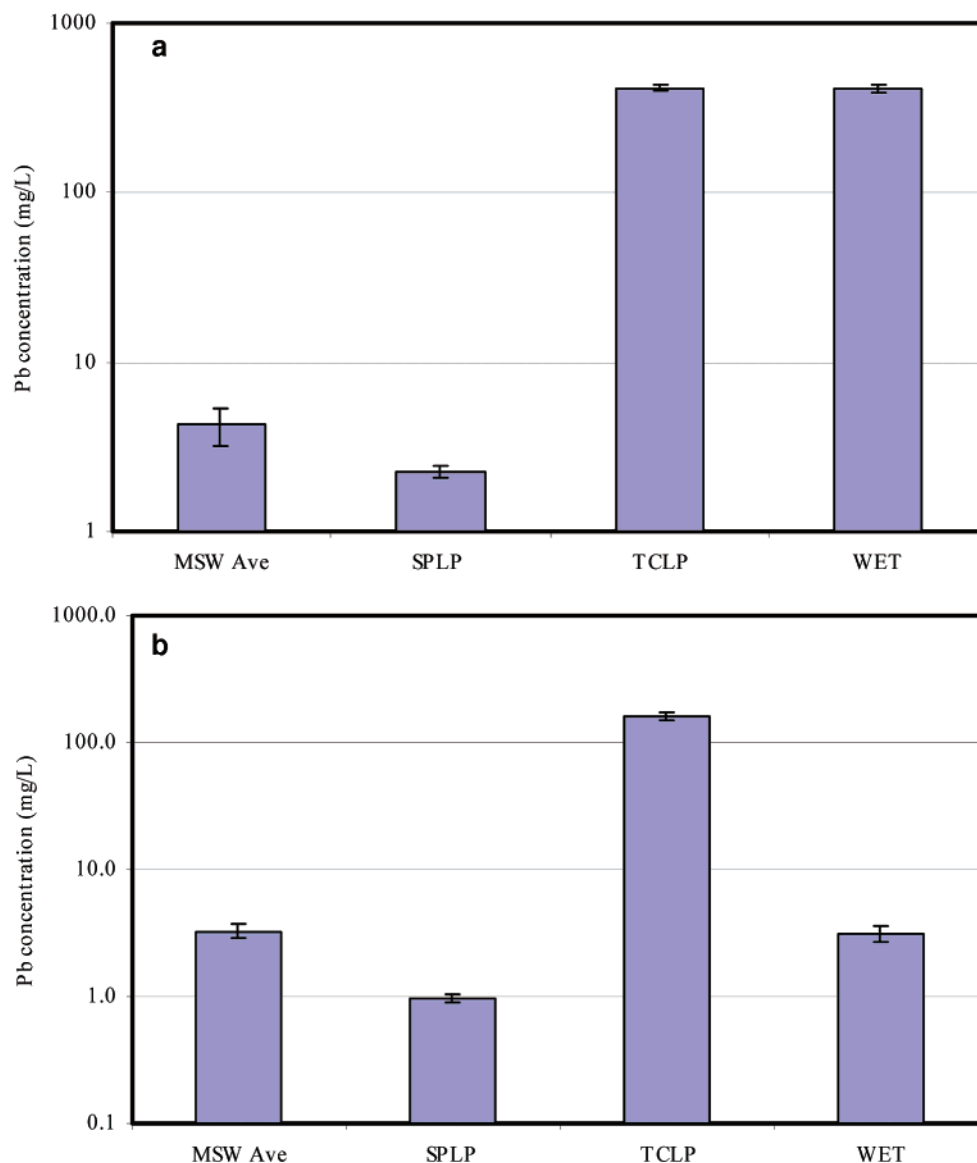


FIGURE 3. Comparisons of Pb leaching from CRTs (a) and PWBs (b) using MSW leachates with SPLP, TCLP, and WET results.

TABLE 3. Summary of Leaching Test Results

		MSW leachate	standardized leaching test		
			TCLP	WET	SPLP
initial pH		7.60 <sup>a</sup>	4.93	5.00	4.20
CRT	avg Pb concn (mg/L)	4.06	413	350	2.27
	avg final pH	7.67	5.08	5.07	9.73
PWB	avg Pb concn (mg/L)	2.23	162	3.15	0.95
	avg final pH	7.69	4.97	5.14	7.70

<sup>a</sup> Average of 11 landfill leachates.

mentioned can be used to explain the lower concentrations of lead in the SPLP. The synthetic acid rain used in the SPLP was not buffered; this resulted in an increase in pH during the leaching test. The pH increased from an initial value of 4.2 to final values of 7.7 for PWBs and 9.7 for CRTs (Table 3). The E-waste components, especially the CRT glass, neutralized the acid found in the SPLP leaching solution. The higher pH values along with the absence of chelating or complexing agents (as found in the TCLP and WET) resulted in the lower lead concentrations in the SPLP extract. The

MSW landfill leachates extracted somewhat more lead than SPLP, especially in the CRT glass. The final pH of the MSW leachates was similar to that in the SPLP for the leaching of the PWBs. The increased lead concentration in the MSW leachate at the same pH as the SPLP indicates that some additional factors in the landfill leachates contributed to extracting the lead beyond pH. The difference was less pronounced with the CRT glass because of the increased pH of the SPLP leachates. Both lead concentrations extracted from the PWBs and CRTs by the SPLP were statistically lower than the average concentrations of lead extracted by the MSW leachates ( $\alpha = 0.05$ ).

**Implications and Limitations.** The dramatic difference between the lead concentrations measured using the TCLP and those measured using Florida landfill leachates merits further discussion. The TCLP was designed to simulate worst-case leaching conditions in a municipal waste landfill. The amount of acetic acid used in the TCLP solution corresponds to the maximum amount expected to be produced under a given co-disposal scenario. Acetic and other fatty acids are produced during anaerobic waste decomposition in landfills but in well-balanced systems they are utilized by other microorganisms and ultimately result in CH<sub>4</sub> and CO<sub>2</sub>. Landfills with leachates containing large amounts of fatty

acids and lower pH values are often classified as being in the acid phase, while well-balanced landfills are classified as being in the methanogenic phase (36). For reasons discussed earlier, lead would tend to leach less in leachates from well-balanced landfills. When compared to leachate concentrations summarized in the literature (36), the Florida landfills were similar to landfills characterized as being in the methanogenic phase. Results using leachates from acid-phase landfills would probably show lead leaching at levels closer to TCLP. In our experience, most landfills in Florida are in the well-balanced phase for the majority of their existence and may never show any evident acid phase. A closer examination of how long modern sanitary landfills remain in the acid phase would be a useful exercise.

From a regulatory perspective, the TCLP was designed to be a conservative test, and thus one would expect such a test to result in greater leaching levels. The very large difference in lead leaching suggests that the test might, however, be over-conservative (especially in light of other conservative features built into the test such as size reduction). On the other hand, in Hooper et al. (24), many elements leached more when using landfill leachate relative to the TCLP. The TCLP may thus be more conservative for some elements (and some wastes) and less conservative for others. The U.S. EPA has been examining the applicability and short-comings of TCLP for a number of years, and this research adds to this complicated issue. For those state and local governmental agencies wrestling with whether to ban discarded electronics from landfills, the results of this work suggest that lead leaching from PWBs and CRTs will be less than might be estimated using TCLP results. It is also important to note that other factors affect the migration of leached lead from a disposed device to the leachate collection system of a landfill (e.g., sorption, reduction, precipitation). A valuable follow-on study would be to critically examine lead concentrations in existing lined landfill leachates, especially comparing landfills that are expected to have different amounts of discarded electronic devices (if such facilities can be identified).

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