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Leaching of metal pollutants from four well casings used for ground-water monitoring

Alan D. Hewitt

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Polytetrafluoroethylene (PTFE), polyvinylchloride (PVC), stainless steel 304 (SS 304) and stainless steel 316 (SS 316) well casings were tested for suitability for ground-water monitoring. A laboratory experiment, testing for the leaching of Ag, As, Ba, Cd, Cr, Hg, Pb, Se and Cu, was run in triplicate by exposing sections of the well casings to ground water for four periods ranging from 1 to 40 days. The results showed that PTFE did not leach any of the nine analytes studied, while SS 316 and PVC showed significant leaching of Cr, Cd and Pb; SS 316 also leached significant amounts of Ba and Cu. Stainless steel 304 showed significant leaching of Cr and Pb. In every case where contamination was observed, the release of metal analyte, when averaged over all of the exposure periods, was the greatest from either SS 304 or SS 316 Released contaminants were sorbed back onto the well casings in several cases.

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PREFACE

This report was prepared by Alan D. Hewitt, Research Chemist, Geochemical Sciences Branch, Research Division, U.S. Army Cold Regions and Research and Engineering Laboratory. This project was funded by the U.S. Army Toxic and Hazardous Materials Agency (R-90 Multi-analytical Services), Martin H. Stutz, project monitor.

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Leaching of Metal Pollutants From Four Well Casings Used for Ground-water Monitoring

ALAN D. HEWITT

INTRODUCTION

Ground-water monitoring requires the installation of conduits to transfer water to the surface for collection. Four commonly used well casings are made from 2-in. (5-cm) diameter polyvinylchloride (PVC), stainless steel 304 (SS 304), stainless steel 316 (SS 316) and polytetrafluoroethylene (PTFE) pipes. Representative sampling of ground water requires that materials employed in the saturated zone do not influence the concentration of analytes of interest.

Only a few studies have reported the influence of well-casing materials on the concentrations of inorganic substances in ground water during water quality analyses. Several studies have demonstrated that these materials (stainless steel, PVC and PTFE) sorb appreciable quantities of certain ionic species (Eichholz et al. 1965, Miller 1982, Hewitt 1989). Evidence also exists showing that metals are released into ground water from stainless steel and PVC pipes (Houghton and Berger 1984, Barcelona and Helfrich 1986, Hewitt 1989). The release of metal analytes by stainless steel has been associated with its corrosion, which in some instances has been observed to produce a hydrous iron precipitate (Barcelona and Helfrich 1986, Hewitt 1989).

Recently, a laboratory experiment was conducted testing the effects of ground-water composition on the well casings cited above (Hewitt 1989). In this experiment two concentrations of metals (As, Cd, Cr and Pb), pH and total organic carbon were introduced as ground-water solution variables. Results of this experiment indicated that PTFE was inert to the variables, whereas both PVC and stainless steel well casings were affected. These two materials leached and sorbed some of the metals introduced into the ground water. In addition, several stainless steel casing sections developed surface oxidation, which introduced a random source of variation by providing release mechanisms and active sites for sorption. PVC was a low-level source for Cd and provided sorption sites for Pb. Stainless steel 316 was a low-level source for Cd and provided sorption sites for As, Cr and Pb. Stainless steel 304 was also a low-level source for Cd and provided sorption sites for As and Pb. The extent of the sorption or release of metals was often influenced by the solution variables. This study concluded that the stainless steel casings were the least suitable for monitoring the metals studied (As, Cd, Cr and Pb) in the ground water solutions.

A concurrent study done at CRREL (Parker et al. 1989) looked at ground-water solutions spiked with organic compounds exposed to the same four well casings. In contrast to the results for metals, eight (cis and trans-1,2-dichloroethylene, m-nitrotoluene, trichloroethylene, chlorobenzene, o-,p- and m-dichlorobenzene) of the ten organic compounds studied sorbed more quickly and to a greater extent onto PTFE than PVC and did not sorb onto the stainless steels. The same results were obtained when the ground water was treated with 2.0 g NaCl/L to test for effects of ionic strength. These findings support the earlier work of Reynolds and Gillham (1985) who observed rapid sorption of tetrachloroethylene by PTFE well casing. They suggested that PTFE is the least desirable material for a well casing when organic compounds are monitored in ground water.

The results of these two recent CRREL studies (Hewitt 1989, Parker et al. 1989) and supporting evidence in the literature led to the suggestion that PVC may be the best compromise among these four well casings for monitoring ground water for both inorganic and organic analytes (Parker et al. 1988).

The objective of this study is to examine metal leaching characteristics of these four well-casing materials in ground water. Leaching studies that compare these four well casings have not been reported in the literature. The results of this experiment will determine which casings are the most or the least susceptible to leaching the metals. The analytes analyzed included all of the metals on the Environmental Protection Agency's priority pollutant list, along with copper.

MATERIALS AND METHODS

Materials

The PVC and stainless steel well casings were obtained from Johnson Well Screen, and the PTFE was obtained from MIP, Inc. All well casings were specifically manufactured for ground-water monitoring. The casings all had approximately a 5-cm inner-wall diameter and were cut in lengths of approximately 2 cm. The exact length of the rings depended on the wall thickness and diameter of the pipe because we wanted to maintain a constant surface area of 80 cm^2. Cut surfaces composed 17% of the area for the PTFE and PVC well casings and 9% for the stainless steels.

Precautions were taken during pipe milling to prevent exposure to grease, dirt, oil and solvents, and to avoid excessive handling. After milling, the individual well-casing rings were rinsed with deionized water (Millipore, Type 1) and air dried before being placed into the ground-water-filled sample containers. During rinsing we made no attempt to remove surface discoloration or ink on the pipes; we used them as we had received them from the manufacturer. This limited cleaning was consistent with commonly employed field protocols.* The well-casing sections were handled with plastic gloves and nylon forceps after milling. Two sections of the SS 316 pipe were not used because excessive surface rust had formed. In general the stainless steel well casings appeared to have developed more rust during the 9-month storage period than they had when first obtained. All experimental work was performed in class 100 cleanrooms.

Polypropylene jars (69 mm od × 62 mm height, 125 mL, Model 6185-E37, Thomas Scientific) served as the sample containers. The jars were soaked in a 10% v/v concentrated, redistilled HNO_3 (G. Fredrick Smith [GFS]) deionized water solution, then rinsed with and soaked for several days in deionized water prior to use. Other materials, such as the 7.5-mL sample aliquot bottles (polyethylene, Nagle), pipette tips (Eppendorf), and the 2-L glass bottles (reagent grade HNO_3 bottles, Baker), were cleaned similarly.

Test design

Tests for the release of metals from PVC, PTFE, SS 304 and SS 316 well casings were done in triplicate by exposing sections of each to ground water for periods of 1, 5, 20 and 40 days. Three sample containers with no well casings served as controls for each of the exposure periods. The containers with and without well casings were filled with 98 mL of ground water collected from a 76-m-deep domestic well system in Weatherfield, Vermont; 60 containers, 12 with a single section of each of the four well-casing candidates (12 × 4) and 12 controls, made up the experimental sample setup. The well-casing rings were submerged in the ground-water-filled sample containers creating a pipe-surface-area-to-aqueous-volume ratio of 0.82 cm^2/cm^3. This experimental design provides a surface-area-to-solution ratio similar to that of well casings in ground-water monitoring wells below the saturated zone; however, the ratio is much lower than that which exists for well screens.

Samples were prepared by transferring weighed amounts of ground water into each jar from a single 2-L glass bottle. The jars were selected randomly for the experiment because the ground water was transported in three separate 2-L glass bottles. The pH and conductivity of the ground water from all of the bottles was 7.8 and 2.40×10^{-2} mho/cm, respectively. Ground water collected from this source previously showed similar pH and conductivity levels (Hewitt 1989). While the well casings were exposed to the ground water, the jars were sealed with a cap and stored in the dark at 24°C. After the well-casing sections had been removed from the jars at the end of each time interval, 2 mL of concentrated HNO_3 (GFS) was added to the ground water to bring the pH below 1.0. Studies have shown that acidification below pH 1.5 is effective in preventing the loss of trace metals from natural waters (Subramanian et al. 1978). The acidified, ground-water-filled jars were recapped, hand-swirled for 10 seconds, then left at rest for at least 72 hours before we transferred a 5-mL aliquot to a 7.5-mL sample vial (polyethylene, Nagle) for the subsequent determination of Ag, As, Ba, Cd, Cu, Cr, Pb and Se.

The entire experimental setup was duplicated for the analysis of Hg, except that we determined Hg immediately after the ground-water-filled jar was acidified.

In a preliminary experiment, ground water stored in the polypropylene jars was spiked with

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* Personal communication with Louise V. Parker, CRREL, 1989.

Cd, Cr and Pb to see if sorption of metals ions on the jar walls would interfere with the test results. These metal ions, added to the ground water and stored for 6 days in the sample jars, were recovered upon acidification (Table 1). The desorption of metal ions from container walls has been reported by Choa et al. (1968). For this preliminary test, 5.00 µg/L of Cd, Cr and Pb was allowed to sit in ground-water-filled jars (100 mL) for 6 days. Then we added 2 mL of concentrated HNO_3 (GFS), hand swirled the solution for 10 seconds, and removed a 5-mL aliquot. A second 5-mL aliquot was removed 72 hours later, following the same procedure. The results in Table 1 show that an average of 95% of the aqueous metal was recovered immediately after acidification, and aliquots removed 3 days later showed only 2% (not significant at the 95% confidence level) additional analyte recovery. Thus the metals either remained in the laboratory ground-water solution or were desorbed from the jar walls quickly upon acidification.

Table 1. Recovery of Cd, Cr and Pb (4.90 µg/L) from ground water stored in the sample jars and allowed to equilibrate for 6 days before being acidified with 2 mL of concentrated HNO_3.

	Acidification period			
	Less than 10 minutes		*72 hours*	
	Amount added (µg/L)	*Percent recovered*	*Amount added (µg/L)*	*Percent recovered*
Cd	4.65	94.9	4.72	96.3
	4.72	96.3	4.85	99.0
Cr	4.48	91.4	4.58	93.5
	4.48	91.4	4.69	95.7
Pb	4.93	100.2	4.72	96.3
	4.72	96.3	5.01	102.2
Average recovery	95.1%		97.2%	

Analysis

Silver, arsenic, barium, cadmium, copper, chromium and lead were determined by Graphite Furnace Atomic Absorption (GFAA) using a Perkin-Elmer (PE) model 403 Atomic Absorption Spectrophotometer (AAS) coupled with a PE model 2200 heated graphite atomizer. Instrumental procedures followed the general guidelines provided in the manufacturer's instrument manual (Perkin-Elmer 1981). Hand injections of either 20, 50 or 100 µL were employed for the analytes mentioned above. For the determination of Se, a matrix modifier—0.015 mg Pd and 0.01 mg $Mg(NO_3)_2$—was added so that the charring temperature could be raised to 1200°C. Of this group, only As and Se determinations required deuterium background correction.

Mercury was determined by Cold Vapor Atomic Absorption (CVAA). We employed a 48-mL aliquot for the Hg determinations, following a modified Hatch and Ott (1968) procedure. Aliquots of 48 mL of ground water were reduced with 2 mL of 10% v/v stannous chloride and then sparged with Hg-free air. The reduced Hg vapor passed through a $Mg(ClO_4)_2$ water vapor trap into an optical cell designed to enhance detection (Tuncel and Atoman 1980). The optical cell was positioned in the light path of the PE model 403 AAS.

Mercury was determined the same day that well casings were removed from ground-water-filled jars to limit volatilization of Hg from solution (Coyne and Collins 1972, Lo and Wai 1975) and to avoid vapor contamination associated with storage in poly containers (Cragin 1979). All of the other metals were determined within 2 weeks after the last exposure period.

Analysis procedures were designed to achieve detection limits of 1% or less of the present domestic water quality levels set by the EPA (Table 2). Selenium, determined by graphite furnace, was the only metal with a detection limit slightly above this level (Table 2). Method Detection Limits (MDLs) were established following the procedure outlined in the Federal Register (1984) for the analysis of a sample in a given solution. The MDL estimate requires that a minimum of seven replicate determinations be made of an analyte concentration that is one to five times the estimated detection level.

Table 2. EPA interim primary drinking water quality levels (1983) and the method detection limits (MDL).

Metal	*EPA primary drinking water levels (µg/L)*	*MDL (µg/L)*
Ar	50	0.48
Ba	1000	2.4
Cd	10	0.059
Cu	1000	4.3
Cr	50	0.16
Pb	50	0.11
Hg	2.0	0.010
Se	10	0.21
Ag	50	0.12

Table 3. Summary of ANOVA and LSD determinations for average analyte concentrations (μg/L). Materials with common underlining are not different at the 95% confidence level as determined by the LSD.

a. Barium

	Days	*Well casing*				
	1	Control	PTFE	PVC	SS304	SS316
		4.5	6.0	6.6	7.1	7.7
(LSD = 1.4)						
	5	PTFE	Control	PVC	SS304	SS316
		5.3	5.8	6.6	7.8	9.9
(LSD = 2.1)						
	20	PTFE	Control	PVC	SS304	SS316
		5.5	5.9	6.1	7.4	11.3
(LSD = 2.2)						
	40	PTFE	PVC	Control	SS304	SS316
		5.2	5.6	5.9	7.0	10.1
(LSD = 2.0)						

b. Chromium

	Days	*Well casing*				
	1	Control	PTFE	PVC	SS304	SS316
		0.20	0.22	1.23	1.60	6.06
(LSD = 9.76)						
	5	Control	PTFE	PVC	SS316	SS304
		0.20	0.22	1.12	1.79	3.34
(LSD = 0.31)						
	20	PTFE	Control	PVC	SS316	SS304
		0.19	0.22	1.20	3.30	4.61
(LSD = 1.17)						
	40	Control	PTFE	PVC	SS316	SS304
		0.21	0.21	1.11	2.53	5.13
(LSD = 1.09)						

c. Lead

	Days	*Well casing*				
	1	Control	PTFE	SS316	SS304	PVC
		0.16	0.35	0.90	1.14	2.46
(LSD = 1.45)						
	5	Control	PTFE	SS316	SS304	PVC
		0.21	0.27	1.27	1.56	2.23
(LSD = 1.53)						
	20	Control	PTFE	SS316	PVC	SS304
		0.14	0.35	1.00	1.04	2.86
(LSD = 2.80)						
	40	PTFE	Control	PVC	SS316	SS304
		0.21	0.33	0.78	1.26	2.06
(LSD = 1.52)						

d. Copper

	Days	*Well casing*				
	1	PVC	SS304	Control	PTFE	SS316
		9.4	10.7	11.9	12.1	35.8
(LSD = 12.0)						
	5	PTFE	PVC	Control	SS304	SS316
		7.8	9.9	10.1	11.0	42.6
(LSD = 12.4)						
	20	PVC	PTFE	Control	SS304	SS316
		6.8	8.3	10.1	26.1	81.2
(LSD = 38.5)						
	40	PVC	PTFE	SS304	Control	SS316
		4.4	5.2	11.5	14.0	82.3
(LSD = 17.2)						

The MDL is obtained by multiplying the standard deviation of the replicate measurements by the appropriate one-sided *t*-statistic corresponding to *n*–1 degrees of freedom at the 99% confidence level.

Each sample aliquot with a determined analyte concentration above the MDL was analyzed at least twice. Analyte concentrations were based on the average peak heights from a strip chart recording.

Aqueous calibration standards for Ag, As, Cd, Cr, Cu, Pb, Hg and Se were prepared by diluting 1000-mg/L certified atomic absorption stock solutions (Fisher Scientific Corp.). A Ba stock standard was made by dissolving a weighed amount of $Ba(NO_3)_2$ (Baker, Reagent Grade) in deionized water. Working standards were prepared in deionized water acidified to 2% v/v with HNO_3 (GFS).

Calibrations were established by determining three different concentration standards in triplicate. Standards were randomly introduced throughout the course of sample analysis, and all of the calibration curves were linear over the concentration range examined. To see if the intercepts were significantly different from zero, we compared the residuals for the models with an intercept and for the models with zero intercept using the *F*-ratio at the 95% confidence level. Analyte concentrations in the samples and controls were determined based on the slope and intercept only if the intercept was deemed significant. Otherwise, a zero-intercept linear model was employed.

To assess leaching of metals from the surfaces of the four well-casing materials, an analysis of variance (ANOVA) was performed on those metals (Ba, Cr, Cu and Pb) that had been consistently found above the established MDL for the four well-casing materials and the control. If a significant difference was detected by the ANOVA among the average analyte concentrations in the ground water for a given material, then a Least-Significant-Difference (LSD) analysis was performed. Both analyses (ANOVA and LSD) used the 95% confidence level. The results of these analyses established which well casings contributed particular analytes to the ground water over and above those contributed by other well casings or the control for the different exposure periods (Table 3). In addition the aqueous metal concentrations that exceed 1% of the EPA drinking water quality level were identified. This low-level warning criterion was chosen since this study did not always establish the native levels present in the ground water. Thus, the contribution of metals from the well casings could range from one or more orders of magnitude above the background concentrations.

RESULTS

Barium

The analysis of the 1-day exposure samples showed that all of the ground-water-filled vessels containing pipe sections had aqueous Ba concentrations that were significantly greater than that of the control; however, all of the values were low (Table 3a). The subsequent exposure periods do not follow this pattern but instead established that SS 316 was the only material that consistently contributed significant levels of Ba to the solution relative to the other samples and the controls (Fig. 1a). The average increase in aqueous concentration for the ground water exposed to SS 316 was about 70% compared to the control. After 5 days of exposure, ground water in contact with SS 316 developed aqueous Ba concentrations that exceeded 1% of the drinking water quality level established by the EPA. None of the other well casings tested produced aqueous Ba concentrations that exceeded 1% of the EPA drinking water quality criterion or were significantly different from the control after the initial exposure period.

Cadmium

We did not use ANOVA with Cd since the majority of concentrations determined were below the MDL (Appendix A). After 1 day of exposure, both ground-water solutions containing SS 316 and PVC had aqueous Cd that exceeded 1% of the EPA drinking water quality level.

Figure 1b shows the average Cd concentrations determined for the control and well casings. It appears that Cd is initially released from SS 316 and PVC but becomes resorbed onto the well casing with time. Stainless steel 316 contributes approximately an order of magnitude (more than 10% of the EPA drinking water quality level in some cases) more Cd than PVC for equivalent exposure periods.

Chromium

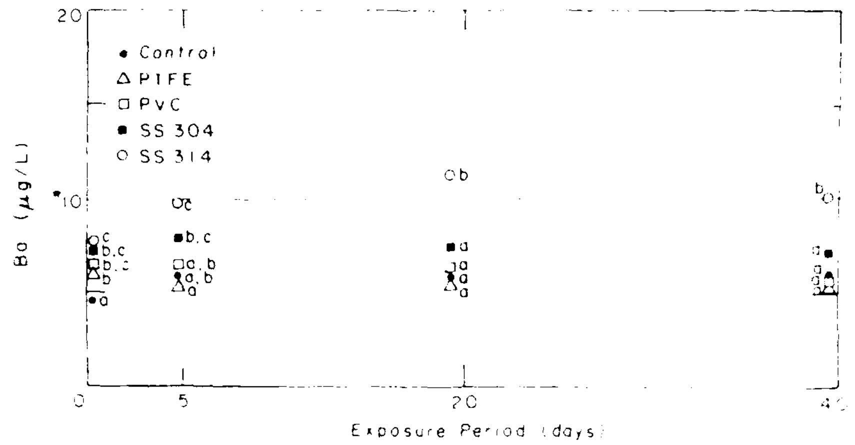
Beyond the 1-day exposure, the analysis consistently demonstrated that both stainless steel well casings contributed significantly greater quantities of Cr to the ground water than the control or the other materials tested (Table 3b). These metal well casings, along with PVC, increased Cr concentrations in the ground water above 1% of the EPA drinking water quality level (Fig. 1c). The extent of the Cr contamination coming from the PVC was three to five times less than that coming from the SS 304, which usually showed the highest average contamination for a given exposure period, the exception being the initial exposure period.

The ANOVA and LSD tests failed to distinguish any difference between the materials for the 1-day exposure because of the large variation among the the three SS 316 samples. If SS 316 is removed, the analysis shows both PVC and SS 304 to contribute significantly greater quantities of Cr to the ground water than do the control and PTFE. The only material that showed a consistent trend was SS 304 (Fig. 1c), which created increasing concentrations of Cr with time. Throughout the experiment there was no significant difference for Cr between the control and the PTFE well casing.

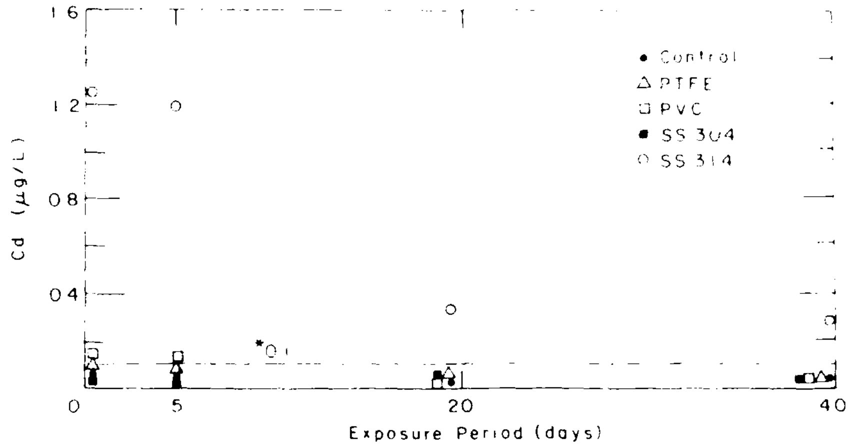
Lead

The first two exposure periods showed PVC to leach the greatest amount of Pb and to be significantly different from the control and PTFE. The two longest exposure periods showed that SS 304 leached the greatest amount of Pb to ground water; however, the levels observed in solution for SS 304 were only statistically different from the rest for the 40-day exposure period (Table 3c). The average levels obtained for both of the stainless steels and for PVC consistently exceeded 1% of the EPA drinking water quality standard (Fig. 1d).

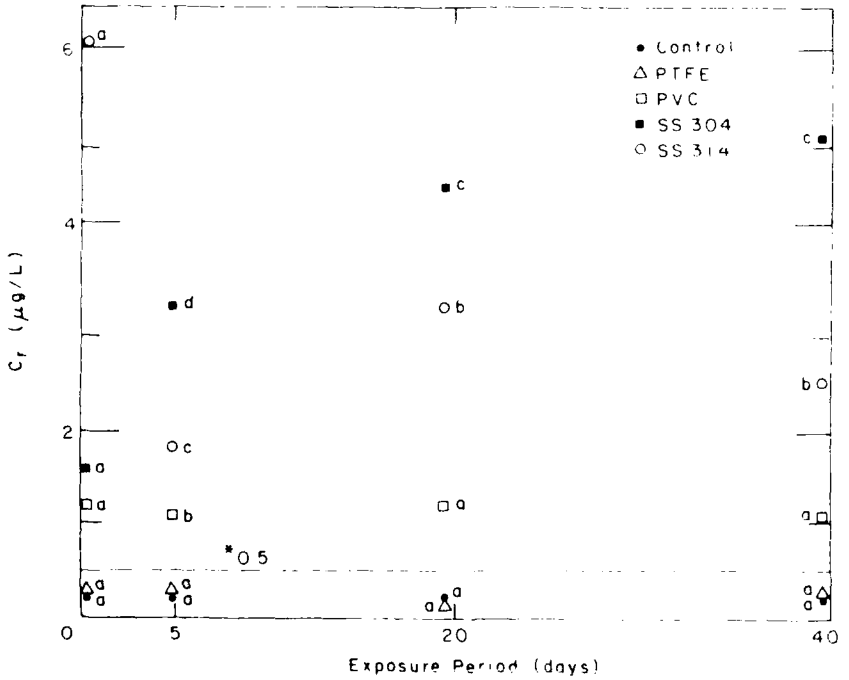
The most distinctive trend was the decrease in Pb with increasing time of exposure for PVC (Fig. 1d). Both stainless steel well casings showed slight decreases in Pb levels after aqueous concentration maxima were obtained. The Pb that was initially released was resorbed by the PVC and stainless steel well casings. Throughout the experiment, there was no significant difference among the control, SS 316 and PTFE.



a. Ba.

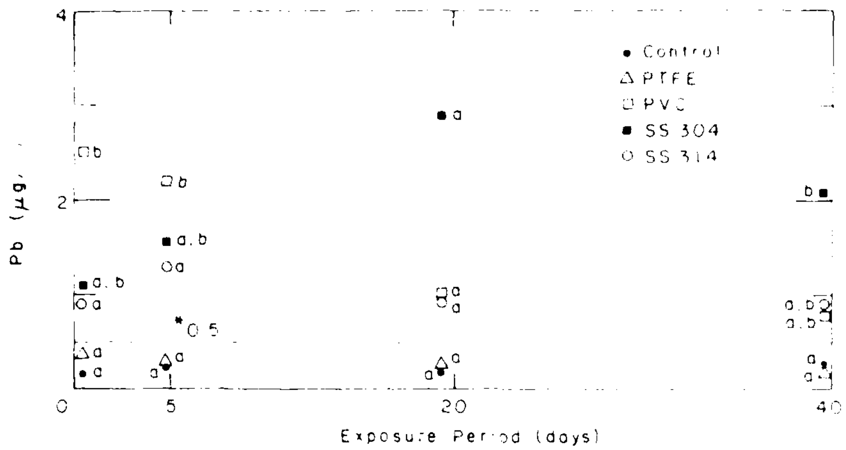


b. Cd.

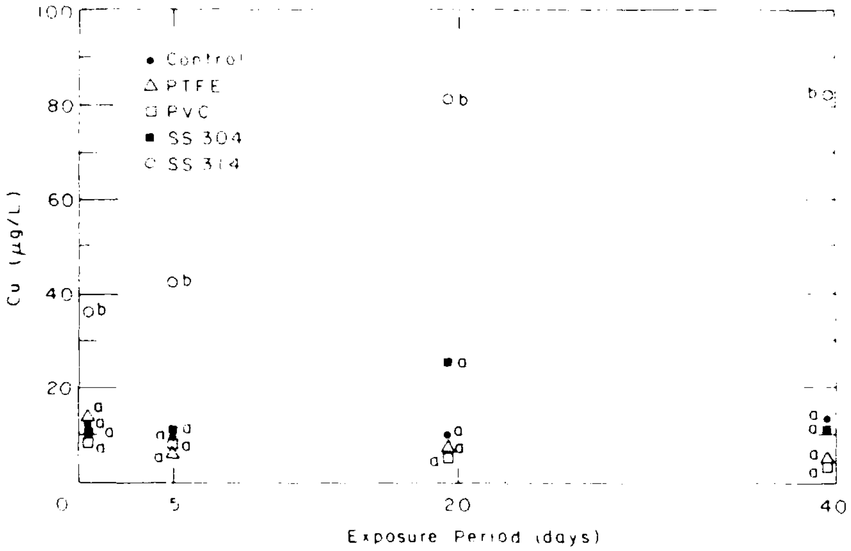


c. Cr.

Figure 1. Ground-water leaching of metals from well casings being examined. Common letters next to points denote no significant difference at the 0.05% confidence interval as determined by ANOVA and LSD.



d. Pb



e. Cu.

Figure 1 (cont'd).

Copper

The statistical analysis distinguished SS 316 as the only material that contributed significantly more Cu when exposed to ground water than the other materials tested (Table 3d). Aqueous concentrations exceeded 80 µg/L for SS 316 versus about 10 µg/L for the other materials and the control.

Both PTFE and PVC well casing showed a general trend of decreasing Cu with increasing time of exposure and often showed concentrations below the control (Fig 1e). This trend, along with the lack of any trend with respect to the control, demonstrates that these two plastic pipes provided the *substrate for sorption*.

Arsenic, mercury, selenium and silver

The determinations for As, Hg, Se and Ag were not statistically analyzed because the majority of the concentrations were at or below the established MDLs. Based on the analysis methods employed, none of the well casings consistently contributed As, Hg or Ag above 1% or Se above 2% of the EPA drinking water quality level.

DISCUSSION

Ground water was collected from a domestic well system and stored in sealed 2-L glass bottles for approximately 24 hours prior to being transferred into test jars. Ground water collected in this fashion is aerated at the faucet and exposed to an oxygen-rich environment every time the lid of the container is removed. Handling the ground water in this manner most likely changed the oxidation potential, facilitating oxidation reactions (Stumm and Morgan 1970). We made no attempt to simulate the natural ground-water redox state or to quantitatively assess the chemical equilibria that existed during the course of this experiment.

There was visible rust on 11 of 24 sections of the stainless steel pipes (Table 4). Each pipe section was carefully examined prior to submersion and after removal from the ground water. In this experiment and in a previous one (Hewitt 1989), oxidation on the stainless steel was predominantly found on the wall. If oxidation is providing sites for sorption or release mechanisms, then the freshly cut surfaces were most likely not a major factor in the behavior of these two materials. Fresh surfaces on the PVC pipe is not an experimental artifact since PVC well screen is made by slotting the pipe.

It was apparent from the values determined for the control samples that the three 2-L glass bottles used to transport the ground water had different concentrations of aqueous Cu (Appendix A). The range of aqueous Cu concentrations most likely reflect the residence time of the ground water in the household and well plumbing. Three distinct populations of Cu concentration were determined for the controls at the 99% confidence level ($\bar{X}_1$ = 9.8±0.0, n = 4; $\bar{X}_2$ = 10.3±0.22, n = 5; $\bar{X}_3$ = 16.0±0.0, n = 3). Differences between adjacent mean concentrations were established by testing against the maximum variance determined for all of populations (i.e., $\bar{X}_1\bar{X}_2$ and $\bar{X}_2\bar{X}_3$).

The groups established by the three Cu populations were then tested to determine if any of the other metals found above its MDL were also significantly different. Table 5 presents the averages and standard deviations for the metal groups based on the Cu populations for the controls. Only Pb shows the same increasing mean concentration trend as the Cu groupings; however, the averages for the adjacent Pb groups were not significantly different at the 95% or even the 80% confidence level when analyzed in the same manner as the Cu populations. This analysis establishes that only Cu was significantly influenced by the sample preparation procedure. Mixing the ground water from the three collection bottles would have eliminated

Table 4. Physical state of stainless steel pipes after exposure to ground water.

	Section											
	1	2	3	4	5	6	7	8	9	10	11	12
SS 304	WR	WR	—	—	WR	—	WR	WR	ER	WR	WR	WR
SS 316	—	—	—	—	ER	—	—	—	—	—	—	ER

KEY: WR = rust on wall, ER = rust on edge; dash means no rust.

Table 5. Average metal concentrations for the controls based on the groups established by the Cu populations (µg/L).

Populations		*Cu*	*Cr*	*Pb*	*As*	*Ba*
1	Avg.	9.8*	0.22	0.11	0.55	5.2
	Std. Dev.	0.0	0.021	0.00	0.114	0.92
2	Avg.	10.3*	0.20	0.25	0.48	5.8
	Std. Dev.	0.49	0.032	0.154	0.00	0.29
3	Avg.	16.0*	0.22	0.28	0.48	5.6
	Std. Dev.	0.0	0.021	0.048	0.00	0.75

* Statistically different at the 99% confidence level.

Table 6. Summary of results.

	Ba	*Cd*	*Cr*	*Pb*	*Cu*
Materials that leached >1% of the EPA drinking water quality level in ground-water solutions.	SS 316 PVC	SS 316 PVC	SS 304 SS 316 PVC	SS 304 PVC SS 316	NA*
Material that showed the highest average overall amount of analyte leached	SS 316	SS 316	SS 304	SS 304	SS 316

* Does not apply.

this artifact. The level of Cu leached from the SS 316 far exceeded the difference between the established populations.

The results of this study support our previous work (Hewitt 1989) showing that PTFE is the least reactive material, whereas both PVC and stainless steel well casings influence aqueous concentrations of metals in laboratory ground-water solutions. As in the first study, the variance among the stainless steel replicates was often the greatest, indicating that this material is susceptible to producing random error. Both studies found that SS 316 and PVC leach and sorb Cd; in addition, these two materials, along with SS 304, sorb Pb. Studies in the future should be conducted under anoxic conditions to see if oxidation of the stainless steel is simply an artifact of these experiments. If corrosion of stainless steel is absent under reducing conditions, then we might expect less random variation and less of an influence on the metal analytes in ground-water solutions.

A summary of the results (Table 6) clearly shows that the stainless steels were the greatest sources of contamination under these experimental conditions. When PVC leached metals (Pb, Cr and Cd) that exceeded 1% of the EPA drinking water quality specifications into solution, there was always a trend showing a decrease in concentration with time of exposure. This would suggest that the leaching of Pb, Cr and Cd from PVC is a surface process and is small. Most likely the initial release could be decreased by more extensive cleaning before the pipes are used. The same statement does not apply to the stainless steel well casings. In the cases of leached Cu from SS 316 and leached Cr from SS 304, the concentrations of these metals continually increased with time over 40 days. It is possible that stainless steels could supply these analytes to ground water over an extended period of time, perhaps the entire life of the casing.

CONCLUSION

Among the four types of well casings tested, PTFE was the only material that did not leach any of the nine metals examined. The other materials tested in this experiment (PVC, SS 304 and SS 316) compromised laboratory ground-water samples by contributing analytes of interest (Ba, Cd, Cr, Pb and Cu). Investigations where only trace metals are of interest should use PTFE below the saturated zone. PVC would be the appropriate second choice since its influence on metal analytes appears predictable and small. In contrast, the two stainless steel materials should be avoided.

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APPENDIX A: LEVELS OF CD, PB, CR, BA AND CU DETERMINED IN GROUND-WATER SOLUTIONS (MG/L).

Pipe	*Replicate*	*Time (days)*	*Number*	*Cd*	*Pb*	*Cr*	*Ba*	*Cu*
Cntrl	1	1	1	< D*	0.11	0.21	4.2	9.8
Cntrl	2	1	2	< D	0.11	0.19	4.6	9.8
Cntrl	3	1	3	< D	0.26	0.21	4.8	16.0
Cntrl	1	5	4	< D	0.40	0.15	5.5	10.5
Cntrl	2	5	5	< D	0.11	0.24	5.8	9.8
Cntrl	3	5	6	< D	0.11	0.20	6.1	10.1
Cntrl	1	20	7	< D	0.11	0.24	6.1	10.1
Cntrl	2	20	8	< D	0.11	0.22	6.1	9.8
Cntrl	3	20	9	< D	0.19	0.20	5.5	10.5
Cntrl	1	40	10	< D	0.42	0.20	5.8	10.1
Cntrl	2	40	11	< D	0.24	0.20	5.7	16.0
Cntrl	3	40	12	< D	0.33	0.24	6.3	16.0
PTFE	1	1	1	< D	0.40	0.19	5.9	10.8
PTFE	2	1	2	< D	0.26	0.28	6.1	9.8
PTFE	3	1	3	< D	0.40	0.19	5.9	15.7
PTFE	1	5	4	0.117	0.40	0.24	4.9	9.1
PTFE	2	5	5	< D	0.30	0.21	5.5	8.4
PTFE	3	5	6	< D	0.11	0.21	5.5	5.9
PTFE	1	20	7	0.117	0.40	0.21	5.8	15.3
PTFE	2	20	8	< D	0.30	0.16	5.2	4.9
PTFE	3	20	9	< D	0.36	0.19	5.5	4.6
PTFE	1	40	10	< D	0.11	0.21	4.9	4.3
PTFE	2	40	11	< D	0.11	0.16	5.5	7.0
PTFE	3	40	12	< D	0.40	0.26	5.2	4.3
PVC	1	1	1	0.109	2.19	1.13	6.7	9.4
PVC	2	1	2	0.125	3.09	1.40	7.0	9.8
PVC	3	1	3	0.175	2.11	1.15	6.1	9.1
PVC	1	5	4	0.075	2.39	1.15	7.3	8.0
PVC	2	5	5	0.142	2.43	1.30	6.4	13.2
PVC	3	5	6	0.109	1.87	0.91	6.1	8.4
PVC	1	20	7	< D	2.11	1.30	6.4	11.9
PVC	2	20	8	< D	0.66	1.40	5.8	4.3
PVC	3	20	9	< D	0.34	0.91	6.1	4.3
PVC	1	40	10	< D	0.93	1.08	6.1	4.3
PVC	2	40	11	< D	0.75	1.03	5.2	4.3
PVC	3	40	12	< D	0.66	1.22	5.5	4.6
SS 304	1	1	1	< D	0.48	1.22	6.7	9.8
SS 304	2	1	2	< D	0.88	1.13	7.0	13.2
SS 304	3	1	3	< D	2.05	2.45	7.6	9.1
SS 304	1	5	4	< D	1.25	3.33	7.3	9.8
SS 304	2	5	5	< D	0.96	3.21	8.2	8.0
SS 304	3	5	6	< D	2.47	3.48	7.9	15.3
SS 304	1	20	7	0.092	0.80	4.36	7.3	9.1
SS 304	2	20	8	< D	2.81	3.87	7.6	49.7
SS 304	3	20	9	< D	4.98	5.59	7.3	19.6
SS 304	1	40	10	< D	2.97	5.10	6.7	9.1
SS 304	2	40	11	< D	1.47	4.56	6.4	9.4
SS 304	3	40	12	< D	1.73	5.73	7.9	15.9
SS 316	1	1	1	2.629	0.41	1.48	7.0	27.5
SS 316	2	1	2	0.209	0.60	1.29	7.3	35.5
SS 316	3	1	3	0.926	1.70	15.36	8.7	44.5
SS 316	1	5	4	0.217	0.96	1.81	8.5	37.2
SS 316	2	5	5	2.930	2.30	1.69	11.5	37.6
SS 316	3	5	6	0.451	0.55	1.86	9.6	52.9
SS 316	1	20	7	0.326	1.50	3.45	12.3	102.2
SS 316	2	20	8	0.376	0.76	2.96	9.3	56.7
SS 316	3	20	9	0.326	0.75	3.50	12.3	84.8
SS 316	1	40	10	0.267	0.67	3.18	11.5	97.3
SS 316	2	40	11	0.384	0.75	1.94	9.0	77.2
SS 316	3	40	12	0.209	2.36	2.45	9.8	72.3

* Less than MDL.