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NAS KEY WEST
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JUSTIFICATION FOR THE USE OF RIGID POLYVINYL CHLORIDE MONITORING WELL
CASING AND MONITORING WELL SCREEN WITH TRANSMITTAL LETTER NAS KEY WEST
FL
11/7/1991
NAS KEY WEST



DEPARTMENT OF THE NAVY

NAVAL AIR STATION
KEY WEST, FLORIDA 33040-5000

5090
Ser 1883DL/2156
7 Nov 1991

Ms. Michelle Glenn
U. S. Environmental Protection Agency,
Region IV
345 Courtland Street N.E.
Atlanta, GA 30365

Dear Ms. Glenn:

Please find enclosed Naval Air Station Key West's request for a variance from EPA's Standard Operating Procedures for well casing construction materials as discussed September 10, 1991.

Point of contact for any questions or further information is Diane Lancaster at (305)292-2911.

Sincerely,

A handwritten signature in black ink, appearing to read "R. A. Demes", is written over a horizontal line.

R. A. DEMES
Engineering Director
Public Works Department
By direction of
the Commanding Officer

Enclosure:

- (1) Justification for the Use of Rigid Polyvinyl Chloride Monitoring Well Casing and Monitor Well Screen at Naval Air Station Key West, FL

ENCLOSURE (1)

JUSTIFICATION FOR THE USE OF RIGID POLYVINYL
CHLORIDE MONITORING WELL CASING AND
MONITOR WELL SCREEN AT
NAVAL AIR STATION
KEY WEST, FLORIDA

November 1991

EXECUTIVE SUMMARY

The U.S. Navy prepared this document to address questions raised by the Environmental Protection Agency (EPA) in their review of the Resource Conservation and Recovery Act Facility Investigation (RFI) work plan for Naval Air Station (NAS) Key West, Florida. EPA questioned the proposed use of rigid polyvinyl chloride (PVC) as a monitor well construction material during the investigation. The Navy is submitting this document to justify the use of PVC well construction materials.

The analysis performed within this document addresses the EPA information requirements identified below:

- * 1. Data-Quality Objectives
- 2. Concentration ranges of anticipated compounds
- 3. Residence time of groundwater in well
- 4. Reasons for not using a hybrid well
- 5. Literature review of adsorption/desorption characteristics
- 6. Wall thickness/annular space
- 7. Type of PVC to be used

The Navy has reviewed the data-quality objectives, the contaminants being monitored, the site's hydrogeology, the pertinent laws, regulations, literature, and manufacturers' recommendations.

The more recent scientific literature strongly supports the selection of PVC well construction materials. The basis for this decision is the nature of the groundwater and the ability to produce a sufficient quantity of water, minimizing residence time of groundwater in the well prior to sampling. This evaluation supports the use of PVC well construction materials for the RFI investigation to be performed at NAS Key West.

1.0 INTRODUCTION

Recent discussions between the U.S. Navy and the EPA relative to the proposed investigation to be conducted at NAS Key West have centered on the selection of well casing material for monitoring wells. The Navy believes the use of Schedule 40 PVC will be equivalent if not more superior to stainless-steel casings for groundwater monitoring purposes. The following discussion addresses EPA's seven points of required information as requested in comments to the RFI work plan from EPA.

The seven point information requirements are summarized below:

* 1. Use of an alternate well construction material must satisfy the Data Quality Objective(s) (DQO) for the investigation.

2. The constituents and the concentrations anticipated should be compatible with the casing material.

3. The productivity of the aquifer and the anticipated residence time of the sample within the well should minimize the potential of the sample being affected by the casing material.

4. The reasons for not using a hybrid well should be reviewed.

5. Literature on adsorption/desorption characteristics of the constituents and elements of concern for the type of PVC to be used should be reviewed.

6. Determine if there is an anticipated increase in the thickness of the casing wall and whether an increase in the annular space will be required.

7. Present the type of PVC to be used and the manufacturers specifications, if available. Additionally, EPA requests an assurance that the PVC to be used will not leach, mask, react, or otherwise interfere with the contaminants being monitored within the limits of the DQO(s).

The following sections discuss the available information relative to each of these items.

In preparing these discussions numerous technical articles were reviewed, including 'Selection of Groundwater Well Construction Material for Naval Air Station Cecil Field Jacksonville, Florida' (April, 1990) prepared by Robert Moser, P.E. (Southern Division Naval Facilities Engineering Command) and 'Justification For the Use of Rigid Polyvinyl Chloride Monitoring Well Casing and Monitor Well Screen at Operable Unit 1, Naval Air Station Jacksonville, Florida' (September, 1991) prepared by Geraghty and Miller, Inc. These documents provided excellent summaries and discussions of many of the above issues which have been incorporated throughout this document.

2.0 INFORMATION REQUIREMENTS

To adequately address the seven criteria, the Navy has gathered site-specific information concerning the project objectives, the constituents of concern at NAS Key West, the general groundwater quality and the aquifer characteristics. In addition, an extensive literature search was performed by both Robert Moser and Geraghty and Miller to collect current scientific literature on the appropriateness of various well construction materials. The results of this data compilation are presented in the following sections.

2.1 Data Quality Objectives

Data Quality Objectives (DQOs) are statements of the level of uncertainty that a decision maker is willing to accept in results derived from environmental data. DQOs are requirements needed to support decisions relative to the various stages of remedial actions and are developed through a three stage process. Stage 1 defines the types of decisions which will be made regarding site remediation, evaluating available data, developing a conceptual model, and specifying objectives for the project. Stage 2 requires the identification of the data necessary to meet the objectives established in Stage 1 which includes the selection of the sampling approaches and the analytical options for the site. Stage 3 specifies the methods by which data of acceptable quality and quantity will be obtained to make decisions.

2.1.1 Stage 1 DQOs; Problem Summary

Task 1. Decision Types. The purpose of the RFI is to determine if NAS Key West sites pose a threat to public health or the environment and if a threat exists, to determine the most effective means for addressing the perceived threat in a manner that is protective of human health and the environment. With the above stated purposes in mind, the questions pertinent to groundwater at NAS Key West are: (1) Are the constituents of concern present in the groundwater capable of impacting the public and/or valuable environmental resources?, and (2) If an impact is anticipated, what is the most effective means of alleviating the risk due to impacted groundwater?

During the course of the investigation, the Navy will make decisions regarding groundwater issues based upon data generated during the RFI. The Navy's Project Manager will be in close contact with federal and state Project Managers to ensure that the data generated will be sufficient to address the questions presented above consistent with the requirements of the RCRA Hazardous and Solid Waste Amendment (HSWA) permit as well as EPA policy and guidance.

Task 2. Evaluation of Available Information. Available information was reviewed and used in developing the work plan. Investigations and information used in this process included, but were not limited to, report of previous investigations by Geraghty and Miller, Inc. (1987 and 1988) and International Technology Corp. (1990 and 1991).

The review of past reports indicated that usefulness of the data with

respect to groundwater characterizations is not sufficient for making quantitative decisions concerning all of the sites. The available information does provide a basis for developing a conceptual model of potential groundwater impacts.

Task 3. Conceptual Model of Potential Groundwater Impacts. Conceptual models have been developed in the Phase I Remedial Investigation Report based upon available historical information obtained during previous investigations and sampling completed to date. Data indicate that soil, groundwater and surface water have been impacted with metals, volatile organic compounds (VOCs) and pesticide contamination at various sites. The groundwater at all sites is classified as Class III-G, not suitable for drinking water.

a. Truman Annex Refuse Disposal Area This site was an uncontrolled disposal area. Waste encountered during investigation consisted of approximately 50 percent construction debris, 15 percent household refuse and 35 percent scrap metal. Groundwater analytical results for this site indicate the detection of many of the metal compounds. Some metals (barium, iron, and sodium) are probably more related to the soil/rock characteristics or perhaps sea water intrusion, than an indication of contamination. The other detected metals (antimony, arsenic, cadmium, chromium, copper, lead, manganese, mercury, nickel, and zinc) may represent contamination caused by leaching of wastes from within the landfill. One monitoring well had detectable concentrations of pesticides. No organic or semi-organic compounds were detected.

b. Truman Annex DDT Mixing Area This site was used for chemical mixing and handling of pesticides. The building in which operations took place has been demolished, and contamination on site is attributed to unintentional spillage that took place in and around the building. Analytical test results indicated metals (iron, sodium and cadmium) and various pesticide compounds.

c. Boca Chica Open Disposal Area This site was used to burn refuse and an area was used to place debris that could not be burned. Groundwater analysis detected VOCs, base neutral extractable compounds, and metals (mercury, copper, and arsenic).

d. Boca Chica DDT Mixing Area This area was used for that same purpose as the Truman Annex DDT Mixing Area. Compounds detected include metals (sodium and iron), organics (benzene, chlorobenzene, 1,2-dichloroethane, ethylbenzene, 2-methylnaphthalene, and naphthalene) and pesticides.

e. Fleming Key North Landfill This landfill consisted of 60 percent household debris, 30 percent construction debris, 5 percent electrical debris and 5 percent scrap metal. Analytical results of the groundwater indicate metal compounds (antimony, cadmium, chromium, iron, lead manganese, mercury, nickel, and sodium) and one well had detectable concentrations of VOCs.

f. Fleming Key South Landfill This landfill consisted of 40 percent vehicle debris, 30 percent household debris, 25 percent construction debris

and 5 percent electrical debris. One test pit contained liquid oil, gas and antifreeze. Analytical groundwater results detected metals (calcium, magnesium, potassium, iron, sodium, arsenic, chromium, lead and mercury), and organic compounds (chlorobenzene).

g. Boca Chica Fire Fighting Training Area This site is used to burn downgraded fuel while training fire fighters. Analytical test results detected metals (chromium, iron and sodium) and organics (benzene, ethylbenzene and naphthalene).

Areas that have not been characterized include Building A-980, Sand Blasting Area by Building A-990, and Former Hazardous Waste Storage Building A-824. These sites will be characterized in this field investigation.

2.1.2 Stage 2 DQOs; Data Needs

To assess the impact of contaminated groundwater on public health and the environment and to screen potential remedial alternatives, data on the underlying lithology of the site, groundwater flow patterns, and groundwater quality must be collected. The following sections discuss the additional hydrogeologic data that has been collected to satisfy the identified data need.

Task 1. Geologic Data. The Florida Keys were created through eustatic elevation of limestone rock units. All of the Lower Keys are composed of Miami Oolite, which consists of calcium carbonate and tiny ooloids or spherical calcareous grains. Key Largo Limestone underlies the Miami Oolite on all the Lower Keys and consists of cemented remains of ancient coral reefs, fossils, and shells. The Miami Oolite is approximately 20 feet thick at Key West and is considered to contain primary porosity. Primary porosity is porosity that developed during the final stages of sedimentation. The underlying Key Largo Limestone is permeable and yields water but the quality is poor, being close to that of seawater. The Key Largo Limestone is approximately 180 feet thick at Key West.

Task 2. Hydraulic Data. International Technologies Corp (IT) performed two slug tests using the Bouwer and Rice method of analysis. Average hydraulic conductivities were calculated at 72 and 1024 gallons per day per square foot (gpd/ft²), respectively. Transmissivity values range from approximately 72,500 gallons per day per foot (gpd/ft) to 12,400 gpd/ft. Because storativity cannot be determined from slug test data, an estimated value of storativity of an unconfined aquifer is generally considered to be equal to the total porosity. The estimated average porosity of the oolitic limestone ranges from 0.20 to 0.35, the storativity ranges from 0.10 to 0.17. The lower value is representative of silty sand, while the higher value is representative of the oolitic limestone.

Task 3. Groundwater Quality. Although Key West is underlain by highly transmissive limestones, most groundwater is brackish, saline or hypersaline and cannot be used for potable purposes. A fresh water lens does exist and is located in the western interior of the island, however an assessment of the groundwater by the United States Geological Survey (USGS) concluded that the water does not meet State drinking water standards for

several constituents. Drinking water is piped from Miami through an aqueduct system and no known wells are currently being used for drinking water in Key West.

2.1.3 Stage 3 DQOs; Sampling Program

To further characterize the type and extent of the contamination at each site, the sampling program presented in the RFI work plan will be conducted. The following sections will briefly describe the types of activities to be performed as well as the potential impact monitor well construction materials may have on the data collection effort.

Task 1. Geologic Data. The need for additional geologic data will be satisfied by collection of soil borings along with associated geotechnical sampling to include lithologic descriptions of each site. Monitor well construction materials do not have any impact on the collection or quality of lithologic or geotechnical information collected during the sampling program.

Task 2. Hydraulic Data. As indicated in the work plan, additional wells will be installed to obtain hydraulic and contaminant data at each site. Monitor well construction materials can have an impact on the quality of the water level data obtained from the monitor wells. The quality of the data will be impacted if the well construction materials fail, resulting in collapse of the well screen or if the monitor well screen becomes fouled due to corrosion or biological growth. In either of these instances, the monitor well is no longer in connection with the groundwater and the water table data becomes suspect.

Task 3. Groundwater Quality. Further groundwater monitoring is scheduled in Phase II of the RFI work plan to provide additional data concerning groundwater contamination at the various sites at NAS Key West. Chemical data on a more complete list of constituents will be collected at some sites to adequately characterize the contamination. The additional chemical data will also be used to adequately review and screen potential remedial alternatives for the site. Current research indicates that, under certain conditions, various monitor well casing materials can impact the groundwater chemistry in the vicinity of monitor wells and lead to biased data.

2.2 Identification of Constituent Concentrations

The RFI work plan identified constituents and the range of concentrations detected in the groundwater during previous investigations. The concentrations detected have been summarized in Appendix A.

2.3 Aquifer Productivity and Sample Residence Time

International Technologies Corp (IT) performed two slug tests using the Bouwer and Rice method of analysis. Average hydraulic conductivities were calculated at 72 and 1024 gallons per day per square foot (gpd/ft²), respectively. Transmissivity values range from approximately 72,500 gallons per day per foot (gpd/ft) to 12,400 gpd/ft. Because storativity cannot be determined from slug test data, an estimated value of storativity

of an unconfined aquifer is generally considered to be equal to the total porosity. The estimated average porosity of the oolitic limestone ranges from 0.20 to 0.35, the storativity ranges from 0.10 to 0.17. The lower value is representative of silty sand, while the higher value is representative of the oolitic limestone.

Based on calculations previously completed in "Justification For the Use of Rigid Polyvinyl Chloride Monitoring Well Casing and Monitor Well Screen at Operable Unit 1, Naval Air Station Jacksonville, Florida" (September, 1991) prepared by Geraghty and Miller, Inc., the well can be purged at a rate greater than 0.5 L/min without being purged to dryness. The Sampling Plan requires purging of three to five well volumes prior to sampling. All groundwater samples will be collected immediately after well purging has been completed. Consequently, the monitor wells can be adequately purged and the water that will be sampled will represent formation water.

2.4 Hybrid Monitoring Well Usage

The Navy chooses to not utilize hybrid wells due to the corrosion factors present at the sites, the absence of a legal mandate requiring them, and the relative cost of these installations.

2.5 Literature on adsorption/desorption characteristics of the constituents and elements of concern for the type of PVC to be used.

A review of EPA guidance and published literature was conducted in an effort to summarize the current position of the Agency as well as the bulk of the scientific investigations performed to date. EPA Region IV's Engineering Support Branch Standard Operation Procedures and Quality Assurance Manual (1991) recommends the use of stainless steel for temporary shallow monitoring wells and stainless or PVC (where organic compounds are not of concern) for permanent monitoring wells. EPA's RCRA Groundwater Monitoring Technical Enforcement Guidance Document recommends the use of stainless steel well materials for long-term monitoring programs, i.e., 30 or more years, due to its structural integrity. Stainless steel is also indicated to not sorb or leach trace organic constituents to the same degree as other materials. EPA's Handbook of Suggested Practices for the Design and Installation of Groundwater Monitoring Wells (1989) indicates that corrosive conditions can limit the life of metallic wells and result in a bias of groundwater samples. EPA's Groundwater Handbook (1987) recommends that the selection of well casing materials be conducted in light of known groundwater constituents.

Several papers were reviewed to determine the current understanding of the interaction between groundwater and select well construction materials. The papers are summarized below with full text appended.

- "Review of Studies Concerning Effects of Well Casing Materials on Trace Measurements of Organic Compounds", R. M. Dowd, 1987. Appendix B.

Dowd concluded that no difference exists between the effects of stainless steel and PVC among certain VOCs when the groundwater is purged from the wells. Static experiments show no difference between Teflon and PVC. Field experiments suggest that PVC may be more sensitive than Teflon,

and therefore would detect volatile organics better than Teflon. Dowd concludes that sampling variability may have a greater effect on trace organic measurements than the well material.

- "Sorption of Organics by Monitoring Well Construction Materials", A. L. Sykes, R. A. McAllister, and J. B. Homolya, 1986. Appendix C.

The authors concluded that no statistical difference exists in the analytical results for groundwater samples collected from monitoring wells constructed of PVC, Teflon, and stainless steel 316.

- "Well Construction and Purging Effects on Ground Water Samples", M. J. Barcelona and J. A. Helfrich, 1986. Appendix D.

The authors reported that field studies indicated that PVC consistently detected higher levels of purgeable organics than did either stainless steel or Teflon when purged. Volatile organics were also reported to be present at higher levels in PVC wells than stainless steel wells when the wells were not purged. They conclude that purging the stagnant water from the monitoring wells is essential to the collection of reliable groundwater sampling. The potential exists for improper well purging to present greater errors in the sample results than either the sampling mechanism or well casing material.

- "Leaching of Metal Pollutants from Four Well Casings Used for Groundwater Monitoring", A. D. Hewitt, 1989. Appendix E.

Within the experimental design, Teflon well casing materials were identified as the most suitable material since it did not leach any of the nine metals that were examined. PVC, stainless steel 304, and stainless steel 316 all contributed barium, cadmium, chromium, lead, and copper to water sampled from wells constructed with these materials. The author concludes that in investigations where trace concentrations of metals are of concern Teflon well construction materials should be used. PVC is the appropriate second choice since its influence appears to be minimal and predictable.

- "Influence of Well Casing Composition on Trace Metals in Ground Water", A. D. Hewitt, 1989. Appendix F.

In this paper, the author reported that stainless steel 304 and stainless steel 316 are both susceptible to oxidation and are not appropriate for use where trace metal concentrations are involved. PVC was reported as having some potential effect on cadmium at low levels and possibly lead where increasing concentrations of total organic carbon are present. PVC is, however, recommended as the second choice due to cost as well as the strong possibility that the observed effects are of less concern when residence time within the PVC well casing is less than 24 hours.

- "Evaluation of Four Well Casing Materials for Monitoring Selected Trace Level Organics in Ground Water", L. V. Parker, T. F. Jenkins, and P. B. Black, 1989. Appendix G.

The authors reported that of the four well casing materials considered (Teflon, PVC, stainless steel 304, and stainless steel 316), Teflon was clearly the poorest choice for monitoring low levels of organic constituents. In testing with PVC, several compounds exhibited some loss; however, the rate of loss was always much slower than for the Teflon casings. Usually 24 hours lapsed before significant losses occurred. Neither stainless steel material exhibited loss of organic constituents; however, rusting of the material occurred very quickly - sometimes overnight in water amended with sodium chloride. The authors conclude that, in a monitoring situation where the well is purged and then sampled within 8-24 hours, PVC cased wells are probably suited for sampling most organics.

Reported compatibility of PVC with various constituents is provided in Appendix H. Most compounds are compatible with PVC materials. Several compounds, specifically solvent type compounds, are not compatible with PVC in a concentrated solution. Conditions which are expected to be encountered in the groundwater at NAS Key West are not believed to represent a threat to the integrity of the PVC well casing material, however the corrosive nature of the soils and hypersalinity of the groundwater may present a problem with stainless steel casing.

2.6 Anticipated increase in annular space due to increased thickness of the casing wall.

No anticipated increase in the borehole annular space is anticipated since the outside diameter of the two-inch PVC well screen and casing is identical to the stainless steel materials.

2.7 Type of PVC to be used and the manufacturers' specifications. Assurance that the PVC to be used will not leach, mask, react, or otherwise interfere with the contaminants being monitored within the limits of the DQOs.

Appendix H provides the manufacturers specifications for the PVC casing material that will be used for the NAS Key West investigation. Southern Division Naval Facilities Engineering Command Guidelines for Groundwater Monitoring Well Installation, Appendix I, specifies Schedule 40 flush threaded joints meeting ASTM F480 and ASTM D1785.

The Navy cannot assure EPA Region IV that PVC will not leach, mask, react, or otherwise interfere with the contaminants that will be monitored. Likewise, the Navy cannot assure EPA Region IV that stainless steel will not leach, mask, react, or otherwise interfere with the contaminants that will be monitored.

3.0 RATIONALE FOR SELECTION OF MATERIALS

The basis for the selection of the well casing material includes an evaluation of the compatibility of each material with the groundwater chemistry, the cost of the material and the generation of data of sufficient quality to meet the DQOs for the project.

3.1 Chemical Compatibility

The groundwater is known to contain VOCs, pesticides, and metals. Several of the compounds are known to be compatible with PVC according to manufacturers' literature, Appendix H. VOCs in concentrated amounts are not compatible with PVC according to manufacturer's recommendations; however, previous groundwater analyses at these sites do not indicate that VOCs will be encountered in the groundwater at concentrated levels. Therefore, PVC casing and screen sections should be suitable at this facility.

The use of stainless steel screens and casing material is traditionally not recommended at locations where the groundwater is known to have strong oxidation qualities since stainless steel well materials are known to contain chromium and iron, all which are leachable under non-oxidizing conditions. Hypersalinity of the groundwater increases the oxidizing conditions, in turn, increasing the leaching of the metals. Studies have also shown that stainless steel rusts rapidly under hypersaline conditions.

3.2 Federal Acquisition Requirements

The Federal Acquisition Regulations (FAR) require the government to select materials which meet the minimum needs of the investigation. If a less expensive material is available that can provide comparable data, the less expensive material will be specified. Based upon known site conditions and the compatibility of the well construction materials with the site contaminants and groundwater conditions, PVC is the well construction material that should be purchased. The material cost for a fifteen foot, two inch diameter PVC well with a five foot screen is approximately \$23 per linear foot. The same diameter stainless steel well costs approximately \$46 per linear foot. A hybrid PVC and stainless steel well would cost approximately the same as a stainless steel well.

The more costly well construction materials are sometimes considered as necessary investments in long-term monitoring programs, i.e., 30 year monitoring programs. The Navy intends to use the monitoring wells for the investigation phase only, which is anticipated to last approximately two years. The suitability of the wells for use during Remedial Design and Remedial Action will be determined after the Record of Decision is signed.

3.3 Quality of Chemical Data

PVC is the most commonly used material for well casings and screens. The Water Well Journal (May 1988) reported the results of a January 1988 survey of well drillers which indicated that 85.7 percent of the monitoring wells constructed are of PVC. Stainless steel and low carbon steel accounted for 10.8 percent of the wells installed while other types of well materials

accounted for the remaining 3.5 percent. From this survey one can either conclude that VOCs are not present at 88.7 percent of the sites or that PVC is an acceptable well construction material and is the industry standard.

Review of recent literature indicates that PVC is a suitable material for use in the construction of monitoring wells. The literature concludes that PVC may be more sensitive to the detection of VOCs than Teflon. Field studies indicate that Teflon and stainless steel are more variable than PVC for reactive organics and inorganics. One author concluded that no statistical differences exist between analytical results of water samples exposed to PVC, stainless steel, and Teflon.

Current research indicates that the quality of the analytical data collected from adequately purged PVC monitor wells will accurately represent the vertical and areal extent of contamination as well as provide quantitative chemical data for support of the risk assessment.

4.0 CONCLUSIONS

The Navy has undertaken the preparation of this document to ensure that the concerns of EPA Region IV have been considered in detail. The Navy has reviewed the data quality objectives, the contaminants being monitored, the site hydrogeology, the pertinent literature, and manufacturers' recommendations.

The most recent work cited in Section 2.5 performed by Parker, et al and Hewitt strongly supports the selection of PVC well construction materials. This conclusion is arrived at by considering the nature of the groundwater and the ability to produce a sufficient quantity of water to minimize the residence time of the groundwater in the well prior to sampling. Other factors supporting the use of PVC well construction materials include the corrosivity of the soil and environment in the Key West area, as well as federal acquisition requirements relative to the cost comparison of PVC versus stainless steel well casing. This evaluation, therefore supports the use of PVC well construction materials for the investigations to be performed at NAS Key West.

APPENDIX A

CONSTITUENTS DETECTED AND RANGE OF CONCENTRATIONS FOUND IN GROUNDWATER
COLLECTED AT NAVAL AIR STATION KEY WEST
1990

TABLE 3-11

DATA SUMMARY - SITE 1
 Truman Annex, Refuse Disposal Area
 NAS-Key West
 Key West, Florida
 IT Project No. 595392

MEDIA	CLASS	PARAMETER	CSC	MINIMUM* CONCENTRATION	MAXIMUM CONCENTRATION
Groundwater	Inorganics	Antimony	14	95.2	557
		Arsenic	50	---	62.2
		Barium	1,000	1,318	1,380
		Cadmium	10	22.2	54.5
		Chromium	50	61.1	657
		Copper	1,000	3,360	10,200
		Iron	300	793	155,000
		Lead	50	57.1	5,700
		Manganese	50	65.6	2,940
		Mercury	2	2.4	16.2
		Nickel	70	88.2	303
		Sodium	160,000	800,000	7,840,000
		Zinc	5,000	7,320	15,200
		Pesticides/PCB	Alpha-chlordane	.027	---
Gamma-chlordane			.027	---	1.1
Sediment Sample	Pesticides/PCB	Aroclor-1260	45	---	2,300
		Aroclor-1254	45	---	210

NOTE:

* = Minimum values represent the smallest concentration level above CSC
 --- = Present when only one value above CSC exists
 CSC = Concentrated standards for comparison

TABLE 3-17

DATA SUMMARY - SITE 3
 Truman Annex, DDT Mixing Area
 NAS-Key West
 Key West, Florida
 IT Project No. 595392

MEDIA	CLASS	PARAMETER	CSC	MINIMUM* CONCENTRATION	MAXIMUM CONCENTRATON
Groundwater	Inorganics	Cadmium	10	11.4	13.9
		Iron	300	425	895
		Sodium	160,000	534,000	1,140,000
	Pesticides/PCB	Alpha-BHC	.05	---	.11
		Beta	.05	.91	7.00
		Dieldrin	.05	.47	1.80
		4,4-DDD	.15	.77	2.10
		4,4-DDE	.10	---	.19
		4,4-DDT	.10	---	.21
		Heptachlor epoxide	.0039	---	.14
Soil Sample	Pesticides/PCB	4,4-DDT	1,000	1,800	220,000
		4,4-DDD	1,500	2,000	83,000
		4,4-DDE	1,000	8,600	33,000

NOTE:

- * Minimum values represent the smallest concentration level above CSC
- Present when only one value above CSC exists
- CSC Concentration standards for comparison

TABLE 3-32

DATA SUMMARY - SITE 4
 Boca Chica, Open Disposal Area
 NAS-Key West
 Key West, Florida
 IT Project No. 595392

MEDIA	CLASS	PARAMETER	CSC	MINIMUM* CONCENTRATION	MAXIMUM CONCENTRATION
Groundwater	Inorganics	Arsenic	50	---	59
		Chromium	50	---	106
		Iron	300	1,110	14,300
		Lead	50	54.8	377
		Manganese	50	---	53.6
		Sodium	160,000	5,530,000	11,000,000
		Mercury	2	---	8.4
	Pesticides/PCB	Heptachlor epoxide	38	---	120
Soil Samples	Pesticides/PCB	Heptachlor epoxide	38	---	120
		Aroclor-1260	45	---	940
Surface Water	Inorganics	Cadmium	10	---	13.7
		Sodium	160,000	---	13,100,000
Sediment Sample	Inorganics	Chromium	85	---	118
	Pesticides/PCB	Aldrin	21	---	84
		Heptachlor epoxide	38	---	71

NOTE:

- * Minimum values represent the smallest concentration level above CSC
- Present when only one value above CSC exists
- CSC Concentration standards for comparison

TABLE 3-45

DATA SUMMARY - SITE 5
 Boca Chica, DDT Mixing Area
 NAS-Key West
 Key West, Florida
 IT Project No. 595392

MEDIA	CLASS	PARAMETER	CSC	MINIMUM* CONCENTRATION	MAXIMUM CONCENTRATION
Soil Samples	Pesticides/PCB	4,4-DDT	1,000	1,000	2,800,000
		4,4-DDT	1,500	23,000	1,800,000
		4,4-DDE	1,000	---	8,400
	Inorganics	Silver	51	---	386
Groundwater	Inorganics	Iron	300	465	1,700
		Sodium	160,000	1,460,000	1,620,000
	Volatiles	Chlorobenzene	10	57	210
		1,2-dichloroethene	4.2	1,500	1,800
		Ethylbenzene	2	---	38
		Naphthalene	10	40	46
		Xylenes	50	---	76
	Pesticides/PCB	Alpha-BHC	.05	14	16
		Beta-BHC	.05	0.54	6.1
		Delta-BHC	.05	0.10	15
		4,4-DDE	.01	.16	22
4,4-DDT		.01	.16	34	
4,4-DDD		.15	---	.76	
Surface Water	Inorganics	Lead	50	---	53.6
		Sodium	160,000	6,410,000	6,590,000
	Pesticides/PCB	Beta-BHC	.05	---	.066
		4,4-DDD	.15	---	.24

TABLE 3-45

DATA SUMMARY - SITE 5
 Boca Chica, DDT Mixing Area
 NAS-Key West
 Key West, Florida
 IT Project No. 595392

MEDIA	CLASS	PARAMETER	CSC	MINIMUM* CONCENTRATION	MAXIMUM CONCENTRATION
Sediment Samples	Pesticides/PCB	4,4-DDD	1,500	6,000	13,000
		4,4-DDE	1,000	1,800	2,800
		4,4-DDT	1,000	1,900	2,500

NOTE:

- *• Minimum values represent the smallest concentration level above CSC
- Present when only one value above CSC exists
- CSC Concentration standards for comparison

TABLE 3-53

DATA SUMMARY - SITE 7
 Fleming Key, North Landfill
 NAS-Key West
 Key West, Florida
 IT Project No. 595392

MEDIA	CLASS	PARAMETER	CSC	MINIMUM* CONCENTRATION	MAXIMUM CONCENTRATION
Groundwater	Inorganics	Antimony	29	90.4	141
		Cadmium	10	10.3	21.7
		Chromium	50	115	384
		Iron	300	549	121,000
		Lead	50	125	1,430
		Manganese	50	56.9	656
		Mercury	2	12.4	73
		Nickel	70	---	91.2
		Sodium	160,000	6,190,000	14,300,000
Surface Water	Inorganic	Iron	300	---	556
		Lead	50	---	72.2
		Sodium	160,000	---	11,400,000

NOTE:

- * Minimum values represent the smallest concentration level above CSC
- Present when only one value above CSC exists
- CSC Concentration standards for comparison

TABLE 3-63

DATA SUMMARY - SITE 8
Fleming Key, South Landfill
NAS-Key West
Key West, Florida
IT Project No. 595392

MEDIA	CLASS	PARAMETER	CSC	MINIMUM* CONCENTRATION	MAXIMUM CONCENTRATION
Groundwater	Inorganic	Antimony	14	---	95.4
		Arsenic	50	50.5	109
		Cadmium	10	11.4	71.8
		Chromium	50	55.9	115
		Copper	1,000	---	1,780
		Iron	300	3,340	70,600
		Lead	50	59	1,870
		Manganese	50	85	508
		Mercury	2	2.2	11.5
	Sodium	160,000	7,090,000	10,400,000	
	Volatiles	Chlorobenzene	10	---	63
Sediment Samples	Inorganic	Antimony	6.8	20.3	20.7
Surface Water	Inorganic	Arsenic	50	---	57.3
		Cadmium	10	---	19.8
		Iron	300	---	305,000
		Lead	50	---	155
		Manganese	50	---	294
	Sodium	160,000	---	9,390,000	
	Pesticides/PCB	Aroclor-1242	.0046	---	1.1

NOTE:

- * Minimum values represent the smallest concentration level above CSC
- Present when only one value above CSC exists
- CSC Concentration standard for comparison

TABLE 3-75

**DATA SUMMARY - SITE 10
 Boca Chica, Fire Fighting Training Area
 NAS-Key West
 Key West, Florida
 IT Project No. 595392**

MEDIA	CLASS	PARAMETER	CSC	MINIMUM* CONCENTRATION	MAXIMUM CONCENTRATION
Groundwater	Inorganic	Cadmium	10	---	13.5
		Chromium	50	53	73.5
		Iron	300	1,230	4,940
		Manganese	50	---	62.2
		Sodium	160,000	1,330,000	9,340,000
	Volatiles	Benzene	1	---	11
		Ethylbenzene	2	---	15
		Naphthalene	10	---	39

NOTE:

- * Minimum values represent the smallest concentration level above CSC
- Present when only one value above CSC exists
- CSC Concentration standard for comparison

APPENDIX B

REVIEW OF STUDIES CONCERNING EFFECTS OF WELL CASING MATERIALS
ON TRACE MEASUREMENTS OF ORGANIC COMPOUNDS

REVIEW OF STUDIES CONCERNING EFFECTS OF WELL CASING MATERIALS
ON TRACE MEASUREMENTS OF ORGANIC COMPOUNDS

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ABSTRACT

This report analyzes the results of laboratory and field studies that allow a direct experimental comparison among commonly used monitoring well casing materials (stainless steel, Teflon, rigid PVC) in terms of their potential effects on measurements of trace organic compounds. Each of the studies analyzed attempts to determine experimentally how much -- if any -- sorption occurs, or what differences result among measured concentrations of a series of organic compounds.

Because the compounds tested were not consistent among all the studies -- although some of the same compounds were represented in several of them -- the analysis compares effects of the casing materials on sorption of different chemicals. The laboratory studies analyzed in this report all relate the measurements taken to a control, and the field investigation to measurements of the same trace compounds in adjacent wells constructed of different casing materials.

In comparing the measured trace concentrations to determine whether the well casing materials cause significant differences in results, a ratio was formulated to reflect the relative sorption effects of each of the materials; sensibly constant ratios over a reasonable range of trace concentrations would indicate few, or relatively minor, differences between the various materials, while varying ratios would indicate larger differences.

The report first reviews the methodology and results of each individual investigation analyzed; these results are then compared across studies through the averaged ratios; and conclusions are drawn about similarities and differences in sorption behavior. Additional observations about sample variation, effects of well purging, and limited measurements of non-volatile compounds are noted.

INTRODUCTION

This review compares the results of four studies that allow a direct experimental comparison of the potential effects of commonly used well casing materials (stainless steel, Teflon, and rigid PVC) on measurements of trace levels of organic compounds.^{1,2,3,4,5}

Each of the four studies was designed to determine experimentally, for a water solution in contact with well casing materials (or coupons made from them), how much sorption occurs, or what differences results among measured concentrations, using several organic compounds. The tested organics were not identical in each of the studies, although several compounds are represented more than once. Three of these studies are laboratory experiments that relate the measurements to a control, while the fourth is a field study that compares measurements of the same trace concentrations on different casing materials in wells located close together.

If well materials affect trace level measurements significantly, then a ration can be formulated to reflect the relative sorption effects of the materials. Such a value should be sensibly constant over a reasonable range of trace level concentrations.

Although the exact procedures and the organic compounds measured differ among the studies, nevertheless any superiority of one well casing material over another ought to be observable if the studies are sufficiently sensitive. It is possible, of course, that at the trace levels measured -- 100 ppb to 20 ppb for the laboratory studies and subparts per billion for the field study -- other sources of variation so overwhelm the results that no meaningful differences can be observed, and this information is in itself useful.

LABORATORY STUDIES

Some preliminary observations about the varying lengths of the test periods in the laboratory studies are in order. The Reynolds and Gillham study tested only the effects on virgin materials over times up to 7 days. This approach is important in addressing the initial effect of a material on trace level measurements and on the mechanism of sorption. However, this approach does not mimic actual field protocol, such as followed by Barcelona. Barcelona et al. show that purging is essential to the correct operation of monitoring wells.

Both the ChemWaste and the Radian laboratory studies also incorporate a 7-day exposure period; however, beyond that they each add 1-hour and 24-hour re-exposures to represent results from both initial and much more closely calibrated samplings. These differences should be kept in mind throughout this review.

1. The Reynolds & Gillham Study. This laboratory study compared effects of six organic polymer materials -- PVC, Teflon, nylon, polypropylene, polyethylene, and latex rubber -- on a series of different trace level organics, ranging from 20 ppb to 45 ppb of 1,1,1 trichloroethane, 1,1,2,2 tetrachloroethane, hexachloroethane,

bromoform, and tetrachloroethylene. Samples were withdrawn from exposure at times varying from 10 minutes to 7 days and analyzed.

From the published results, the ratios of the concentrations of the chemicals in contact with Teflon and PVC coupons respectively can be calculated and compared for the five different compounds measured. These ratios, representing the relative effects of sorption on virgin materials, are presented in Table 1. This table indicates that, for four of the compounds measured, the Teflon/PVC ration is very close to 1, implying little difference among the materials' effects. The exception is tetrachloroethylene, which shows much greater sorption, and hence less sensitivity, for Teflon.

Table 1

RATIO OF CONCENTRATION MEASUREMENTS
OVER TIME FOR DIFFERENT COMPOUNDS

Reynolds Study: TEF/PVC

<u>Compound/Time</u>	<u>10 min.</u>	<u>Average</u>	
		<u>100 min. - 7 days</u>	<u>7 day</u>
1,1,1 Trichloroethane	1.06	0.93	0.68
1,1,2,2 Tetrachloroethane	1.05	1.01	0.94
Hexachloroethane	1.08	1.07	1.30
Bromoform	1.06	1.17	1.68
Tetrachloroethylene	0.9	0.46	0.10
Average	1.03	0.93	0.94

In general, this shows typical differences at the 7-day time of + 30%. Some of the chemicals are detected more easily with PVC (1,1,1 trichloroethane, 1,1,2,2 tetrachloroether, tetrachloroethylene) and some more easily with Teflon (hexachlorethane, bromoform). Based on this study alone, it would be difficult to establish clear superiority of either Teflon or PVC.

2. The ChemWaste Management Study. This study measured effects of coupons of stainless steel, Teflon, and rigid PVC on six organic compounds: methylene chloride, 1, 2 dichloroethane, trans-1,2 dichloroethylene, trichloroethylene, chlorobenzene, and toluene. The test solutions were prepared in the same way as for the Reynolds study; the organics were dissolved in a concentrated methane solution and then exposed to the coupons at two diluted concentrations of 50 ppb and 100 ppb.

Each casing material was first exposed for an initial 7 days (similar to Reynolds) after which the solutions from each coupon and control were sampled. The coupon materials were then re-exposed for

TABLE 2
CWM Report
RATIO OF CONCENTRATION MEASUREMENTS
OVER TIME FOR DIFFERENT COMPOUNDS
(50ppb nominal)
SS/PVC

Compound/Time	Reexposed		Initial 7days
	1 hour	24 hour	
Meth Chl	1.02	1.07	0.92
1,2-DCE	1.05	1.08	0.92
1,2-DCY	0.98	1.12	1.00
Trichloroethylene	1.00	1.10	1.02
Toluene	1.05	1.13	0.88
Chlben	1.18	1.14	0.95
avg	1.05	1.11	0.95

Compound/Time	TEF/PVC Reexposed		Initial 7days
	1 hour	24 hour	
Meth Chl	1.02	0.94	0.89
1,2-DCE	1.03	0.94	0.90
1,2-DCY	0.98	0.93	0.76
Trichloroethylene	1.02	0.92	0.73
Toluene	1.03	1.06	0.79
Chlben	1.00	1.19	0.84
avg	1.01	1.00	0.82

TABLE 3
CWM Report
RATIO OF CONCENTRATION MEASUREMENTS
OVER TIME FOR DIFFERENT COMPOUNDS
(100ppb nominal)
SS/PVC

Compound/Time	Reexposed		Initial 7days
	1 hour	24 hour	
Meth Chl	1.03	0.93	1.06
1,2-DCE	1.03	1.03	1.03
1,2-DCY	0.96	1.04	1.13
Trichloroethylene	1.03	1.03	1.11
Toluene	1.05	1.03	1.07
Chlben	1.06	1.04	1.13
avg	1.03	1.01	1.09

Compound/Time	TEF/PVC Reexposed		Initial 7days
	1 hour	24 hour	
Meth Chl	0.79	0.90	1.12
1,2-DCE	0.81	0.92	1.14
1,2-DCY	0.78	0.88	0.91
Trichloroethylene	0.80	0.87	0.89
Toluene	0.79	0.90	1.01
Chlben	0.97	0.90	1.04
avg	0.83	0.90	1.02

one hour and resampled. A third sampling was performed following a final 24-hour re-exposure.

Results from the initial 7-day conditioning period -- which may have very little significance since groundwater sampling protocols require purging wells prior to drawing samples -- show that PVC is a better measuring materials 12 times, while the Teflon or stainless steel is better 12 times.

To compare the remainder of the results, the concentrations of the other experimental exposure was used to calculate ratios. Table 2 shows the SS/PVC and TEF/PVC ratios for the 50 ppb nominal concentration, and Table 3 shows the same calculations for the 100 ppb nominal concentration, for all six of the tested chemicals.

There are not enough control samples or replicates in this study to estimate the standard deviation. The best that can be done is to compare the ratios. Inspection of Tables 2 and 3 shows that PVC and Teflon exhibit very similar behavior; the ratios are close to 1, a result similar to that from the Reynolds study (although the latter shows wider variation). Where the coupons were re-exposed for one hour, Teflon is slightly better (2-3%) at the 50 ppb nominal case for four chemicals and PVC slightly better (2-3%) for one chemical; at the 100 ppb nominal, PVC is better (20%) for all six chemicals. After 24 hours re-exposure, at the 50 ppb nominal PVC is better (6-8%) for four chemicals and Teflon better (6 to 20%) for two, while at the 100 ppb nominal PVC appears about 10% better for all six chemicals.

The re-exposure samples show stainless steel performing better once better (2-10%) than PVC ten times and PVC performing better once for each of the two nominal concentrations. For the 1-hour re-exposures, which are likely to most nearly represent a well monitoring protocol after purging, only one of the ratios (chlorobenzene for SS/PVC) is more than 5% greater than 1.

Without any estimate of uncertainties, it is impossible to know if the differences shown in this experiment (on the order of 10%) are significant.

3. The Radian Study. This study followed the same general protocol as the ChemWaste study, with several significant differences. The same six chemical compounds were used in the experiment but were dissolved to a nominal 100 ppb concentration in a water solution as a carrier, instead of methanol as for Reynolds and ChemWaste. The exposure periods were similar: an initial 7 days exposure of the well casing coupons, followed by 1-hour and 24-hour re-exposures. The 7-day and 24-hour re-exposures were held at 5°C, while the 1-hour re-exposure was at room temperature. Table 4 shows the

results of these analyses, with trichloroethylene deleted since it was not stable.

Table 4 also presents the results of the nine control samples, with averages and standard deviations. An analysis of variations showed that the controls were not drawn from different populations and thus can be averaged. Therefore the standard deviation gives an estimate of the variation likely to be seen in any set of measurements.

Inspection of Table 4 shows that, for the re-exposure experiments for the 20 paired differences possible between concentrations (of PVC and Teflon, and PVC and stainless steel) with the various coupons, no difference exceeds two standard deviations, and five exceed one standard deviation. For the initial 7-day exposure, none of the differences between PVC and Teflon are greater than two standard deviations.

However, the stainless steel results are quite unusual. The stainless steel concentrations are more than two standard deviations greater than the controls, as well as more than two standard deviations larger than both PVC and Teflon for all of the chemicals. It seems likely that some contamination has entered the system. The stainless steel coupons were used just as received from the manufacturer and may have contained cutting fluids or other organics which affected the spiking solutions; there was no control using a coupon without a spike to check for this. In any event, it appears that the 7-day stainless steel concentrations are highly suspect and should be redone.

As before, the ratios (SS/PVC and TEF/PVC) can be calculated from these data to establish relative sorption effects between the various compounds. The results, presented in Table 5, are similar to those from the ChemWaste Management study, yielding ratios very close to 1, except for the 7-day SS/PVC ones.

Table 5 also presents an estimated standard deviation for the ratios, based on the standard deviation of the control samples, rather than on the small number (3) of the replicates. While this is not a satisfactory statistical analysis, it gives an order of magnitude to the variations in the ratios which may be present due to uncertainties in the experiment. (A more complete statistical analysis will be performed.)

The results of the re-exposure show that, after one hour, PVC appears somewhat better than both Teflon (2-10%) and stainless steel (10-30%); however, the difference does not appear significant, never exceeding two standard deviations. After 24 hours, PVC appears better than Teflon (0 to 6%) but not as good as stainless steel (8-12%), although again neither set of differences appears significant.

TABLE -
Radian Study
CONCENTRATIONS REMAINING IN SOLUTION
ppb

	ONE HOUR		S.S.	AVG CONTRLS	STD
	PVC	TEFLON			
Meth Chl	129	127	119	127	16
1,2-DCE	68	64	48	57	10
1,2-DCY	109	100	96	103	9
Toluene	42	38	32	37	5
Chlben	67	60	55	58	8

	24 HOUR		S.S.	AVG	STD
	PVC	TEFLON			
Meth Chl	117	118	131	127	16
1,2-DCE	53	51	57	57	10
1,2-DCY	103	97	104	103	9
Toluene	34	34	38	37	5
Chlben	56	54	62	58	8

	7 DAY		S.S.	AVG	STD
	PVC	TEFLON			
Meth Chl	114	136	189	127	16
1,2-DCE	47	56	82	57	10
1,2-DCY	95	102	136	103	9
Toluene	31	35	69	37	5
Chlben	40	56	84	58	8

TABLE 5
Radian Study
RATIO OF CONCENTRATION MEASUREMENTS
OVER TIME FOR DIFFERENT COMPOUNDS
SS/PVC

	Reexposed		24 hour	std dev	Initial 7days	std dev
	1 hour	std dev				
Meth Chl	0.92	0.16	1.12	0.19	1.66	0.29
1,2-DCE	0.71	0.18	1.08	0.28	1.74	0.45
1,2-DCY	0.88	0.11	1.01	0.12	1.43	0.17
Toluene	0.76	0.15	1.12	0.22	2.23	0.45
Chlben	0.82	0.17	1.11	0.23	2.10	0.43
avg	0.82		1.09		1.83	

	Reexposed		24 hour	std dev	Initial 7days	std dev
	1 hour	std dev				
Meth Chl	0.98	0.17	1.01	0.18	1.19	0.21
1,2-DCE	0.94	0.24	0.96	0.25	1.19	0.31
1,2-DCY	0.92	0.11	0.94	0.11	1.07	0.13
Toluene	0.90	0.18	1.00	0.20	1.13	0.23
Chlben	0.90	0.18	0.96	0.20	1.40	0.29
avg	0.93		0.98		1.20	

It is difficult to conclude from this experiment that there is a consistent difference between stainless steel and PVC or Teflon and PVC. Based on the Radian study alone, if a well were purged and subsequently sampled within one hour, PVC would seem to be somewhat more sensitive than either stainless steel or Teflon, but not significantly so. Twenty-four hours after purging, the PVC is still somewhat more sensitive than Teflon but less sensitive than stainless steel -- again, not significantly so. While the initial 7-day stainless steel experiment appears flawed, Teflon is more sensitive after 7 days -- again, not significantly so.

C. Barcelona's Field Experiment

The experiment by Michael Barcelona and John Helfrich was designed to provide a comprehensive field study which tests the differences between three different well casing materials: PVC, stainless steel, and Teflon.⁶ The experiment has potential advantages over the lab studies in that it investigates the detection of chemicals actually in the groundwater at two different contaminated sites.

At each site there were six wells, with one each of the three different casing materials in a cluster upgradient and one each of the three materials clustered downgradient of the site. Each of the wells at a given cluster was installed within two meters of the other two, thereby attempting to assure that each cluster was sampling the same groundwater.

Each site was sampled monthly, six times starting in May and extending through October. At each of the sites, samples were taken prior to purging the stagnant water.⁷ The wells were purged until parameters such as pH stabilized, and then samples were taken for an extensive list of groundwater parameters that included pH, conductivity, temperature, alkalinity, and total iron, and for a series of organic compounds that included total non-volatile organic compounds (NVOC), methylene chloride, 1,1-dichloroethane (1,1-DCE), cis-1,2 dichloroethylene (c-1, 2-DCY), trichloroethylene, 1,1,1 TCE, and chlorobenzene.

The largest concentrations of total volatile halocarbons were detected at Site 2 down-gradient at a few parts per billion. However, the concentrations are so low that, in many cases, clear differences between concentrations at the different wells cannot be seen.

As Barcelona points out in the paper, there were problems at Site 1 with apparent grout contamination at both up- and down-gradient wells. This was a factor in the abnormally large pH levels seen in five out of the six wells at this site. The only well which apparently did not have high pH levels was the down-gradient PVC well. Such grout contamination obviously is of concern if it could

affect the measurement of organic constituents, since those are crucial to the determination of sorption effects. A fairly simple correlation of total NVOC with the pH values for the stainless steel and Teflon wells at Site 1 suggests that there may be a direct relationship between pH and the non-volatile organic compounds. The correlation coefficient was 0.5, with a possibility of 6%. Therefore, in reviewing the Site 1 data, the problems with pH must be kept in mind, since it is apparent that grout contamination occurred and the NVOC values may be affected as well. The wells at Site 2 apparently were constructed in such a way that there was no grout contamination, and the purged pH was as expected.

With respect to well casing material, no definitive conclusions can be drawn because the well casing effect is confounded with spatial variability and, at Site 1, with grout contamination. Each type of well casing is used only once for each experimental sampling. As a result, the differences seen could be a result of either well casing or spatial differences. A so-called mixed model analysis may be used to help disentangle the effects.

Prior to that, however, the data can be inspected to observe if any consistent differences across sites are apparent.

Table 6 shows the results of ratios calculated, as before, for SS/PVC and TEF/PVC for each chemical measured and a group of NVOC chemicals and Total Volatile Halocarbons (TVOC) for Sites 1 and 2 up- and down-gradient.

As can be seen, there are no consistent results indicating that PVC is inferior to the other two materials. For example, for NVOC at Site 1 down-gradient (with grout contamination) stainless steel and Teflon have superior detection capability, but at Site 1 up-gradient, where all wells have grout contamination, PVC is superior. Since, presumably, the differences should be the same, it is likely either spatial variability or grout contamination cause such a large variation.

If, therefore, attention is focused on Site 2 where volatile organics were detected, it can be seen that PVC is consistently superior to both Teflon and stainless steel, except for the non-volatiles. These values, however, have a significant variation and only the measurements for 1,1 DCE suggest that PVC is significantly (more than two standard deviations) superior to stainless steel and Teflon.

Indeed, the combination of time series data, the fixed spatial distances, and the material differences may allow a determination of errors due to spatial variability, which seem likely to be larger than those due to the analytical variability.

TABLE 6
 Barcelona et. al.
 Ratio of Groundwater Measurements
 for Two Wells
 SS/PVC

	Ratio Site 1 down	STD	Ratio Site 1 up	STD
NVOC	1.83	0.66	0.80	0.40
Meth.Chl.	1.98	0.95	4.25	5.65
	Site 2 down		Site 2 up	
NVOC	1.11	0.35	3.85	6.36
Meth.Chl.	0.90	0.44	0.91	0.39
TVOC	0.61	0.24		
1,2DCE+1,1,1TCE	0.70	0.39		
1,1DCE	0.43	0.16		
Cl,2DCY	0.63	0.22		
average	0.73	0.30		

TEF/PVC

	Ratio Site 1 down	STD	Ratio Site 1 up	STD
NVOC	1.52	0.69	0.92	0.47
Meth.Chl.	1.35	0.92	1.46	1.49
	Site 2 down		Site 2 up	
NVOC	0.89	0.31	12.99	10.35
Meth.Chl.	0.80	0.31	0.98	0.59
TVOC	0.43	0.16		
1,2DCE+1,1,1TCE	0.58	0.12		
1,1DCE	0.62	0.13		
Cl,2DCY	0.72	0.17		
average	0.67	0.20		

Table 7

COMPARISON OF SPECIFIC VOLATILE CHEMICALS
 BETWEEN LABORATORY AND FIELD STUDIES

RATIOS OF CONCENTRATION MEASUREMENTS

SS/PVC

	<u>Radian</u>	<u>CWM</u>	<u>CWM</u>	<u>Barcelona</u>
1,2 DCE	0.71	1.03	1.05	0.70 + .39*
c 1,2 DCY	0.88	0.96	0.98	0.63 + .22

TEF/PVC

	<u>Radian</u>	<u>CWM</u>	<u>CWM</u>	<u>Barcelona</u>
1,2 DCE	0.94	0.81	1.03	0.58 + .12*
c 1,2 DCY	0.92	0.78	0.98	0.72 + .17

* 1,2 DCE & 1,1,1 TCE

D. Comparison of Results

In comparing the different studies, this review has analyzed the ratios of SS/PVC and TEF/PVC derived from the experiments to assess whether one material is consistently better than another in measuring compounds common to the experiments. Table 7 compares 1,2 DCE and cis-1,2 DCY, two compounds measured (at 1-hour re-exposures) in the ChemWaste, Radian and Barcelona experiments (two laboratory and one field experiment). The results are generally consistent, although the Barcelona experiment suggests that PVC is somewhat better than is suggested by the Radian or ChemWaste studies. This implies that PVC would detect concentrations of the chemicals more efficiently than would the other two materials.

A second way to compare the study results is to average all of the chemicals in each of the experiments. Obviously, it is necessary to be very careful about averaging different chemical compounds with different sorption behaviors. But, since it is generally not known which compounds are likely to be in groundwater, averaging the ratios indicates what effects might occur with an unknown compound or suite of compounds in a groundwater situation. Table 8 shows average ratios for SS/PVC and TEF/PVC for two categories for the four studies reviewed. It assumes that the laboratory 1-hour re-exposures are roughly comparable to the Barcelona field experiment when the well is purged and that the 7-day exposures for Reynolds, ChemWaste and Radian are roughly comparable to the Barcelona field experiment when samples are taken under stagnant conditions.⁸ If the average for the 1-hour re-exposure is considered for both SS/PVC and TEF/PVC, the ratios are determined by all four experiments are very similar.

There is somewhat more variation when looking at the 7-day stagnant situation for stainless steel and PVC. One reason for this may be the possibility of contamination in the Radian 7-day stainless steel experiment.

E. Conclusion

Based on this review of the existing studies, several conclusions are possible relating to the effects of well casing materials on the measurement of organic compounds, to apparent sample variations that occur in an actual measurement situation, and to judging the efficacy of various well materials.

- o These four experiments suggest that, when groundwater is purged from a well, there are no consistent differences between the effects of stainless steel and PVC on volatile organic compound measurements. The laboratory experiments also show no significant differences between Teflon and PVC. In the field experiment, there is a small difference that may be significant,

Table 8

RATIO OF WELL CASING MATERIAL MEASUREMENTS
 (average of all volatile organic compounds, laboratory & field tests)

	<u>Radian</u>	<u>CWM</u>	<u>CWM</u>	<u>Reynolds</u>	<u>Barcelona</u>
1 hr. SS/PVC re-exposure (purged)	0.82	1.03	1.05	-	0.93* (1,2)
1 hr. TEF/PVC re-exposure (purged)	0.93	0.83	1.01	-	0.81* (1,2)
7 day SS/PVC (stagnant)	1.64+	1.09	0.95	-	0.46** (2)
7 day TEF/PVC (stagnant)	1.16	1.02	0.82	0.94	0.63** (2)

* Purged; numbers indicate sites

** Stagnant waters; number (2) indicates site 2

+ Possible contamination of the SS values

showing that PVC may be more sensitive; if further work confirms this, PVC would detect volatile organics better than Teflon. If stagnant water is sampled, the comparison between studies is not so clear. Non-volatile chemicals, tested only in the Barcelona study, also do not show a significant difference across sites for any of the three casing materials.

- o The effects of well materials on measurements of trace organic compounds need to be disentangled from sample variability. This variability may contribute an error larger than any error from analytical variability. Further investigation could shed light on whether this is an artifact of this particular field study -- which seems unlikely -- or is consistent at other waste sites.
- o It seems clear that, in order to judge the efficacy of various well materials, comparisons must be made in the context of normal experimental variation. If it is to be judged that well material A has a different sorption than well material B, the difference in concentrations (reflecting different absorption behaviors) between them must be greater than the normal variation in the samples themselves. Further experiments could determine whether effects associated with the well casing material are larger than the 15% to 25% sample variation that seems likely.

Finally, a conservative position at the present may be to allow a choice of any of these three well casing materials: Teflon, stainless steel and PVC, at this time, excluding PVC could result in less detection of organic compounds.

FOOTNOTES

¹Reynolds and Gillham, "Absorption of Halogenated Organic Compounds by Polymer Materials Commonly Used in Groundwater Monitors." Proceedings, Second [1985] Canadian-American Conference on Hydrogeology, Banf, Alberta, 1986, pp. 125-132.

²ChemWaste Management, Inc., "Absorption of Organics by Monitoring Well Construction Materials," unpublished technical note.

³Barcelona, Michael J. and John A. Helfrich. "Well Construction and Purging Effects on Ground-Water Samples," Environmental Science & Technology, Vol. 20, No. 11, 1179-84.

⁴Sykes, McAllister and Homolya. "Sorption of Organics by Monitoring Well Construction Materials." Radian Corporation (to be published).

⁵The review does not include a separate paper by Barcelona et al. that reported an investigation of the relationship between Teflon tubing and PVC tubing because, as Barcelona indicated, flexible PVC

tubing differs from rigid PVC pipe, and the sorption behavior is likely to differ greatly.

⁶The Teflon well at Site 2 was actually a Teflon/aluminum oxide dedicated sampler.

⁷With regard to well purging, Barcelona concluded that the experiment shows that purging is essential in order to eliminate spurious results from stagnant water. The investigation provides a useful data base for studying the necessity of purging.

⁸The stagnant conditions are not exactly the same, since the Barcelona experiment allowed 30 days exposure to the well casing materials, while the laboratory experiments had only 7 days; in addition, the Barcelona casings were not virgin materials.

APPENDIX C

SORPTION OF ORGANICS BY MONITORING WELL CONSTRUCTION MATERIALS

FIELD REPORT

Sorption of Organics by Monitoring Well Construction Materials

by A.L. Sykes, R.A. McAllister
and J.B. Homolya

Introduction

In August of 1985, the Environmental Protection Agency released a draft guidance document titled "RCRA Groundwater Monitoring Technical Enforcement Guidance Manual," which was intended to help EPA and state enforcement officials decide whether specific elements of an owner/operator's ground water monitoring system satisfy the RCRA requirements. The guidance document states that polytetrafluoroethylene or Type 316 stainless steel are the materials of choice as screen or casing in new well installations where volatile organics are the parameters of interest. Since no guidance was provided on appropriate use of casing material at the time the RCRA regulations were promulgated, most operators installed PVC casing in both the original and subsequent monitoring systems because this material had been used for years in the water well industry, is readily available, and is fairly inexpensive.

The EPA has cited a number of reasons why PVC is not an acceptable material for well construction. These include:

- Potential for casing attack and fatigue by exposure to high concentrations of certain organic compounds
- Desorption of plasticizers and additives from the well casing to otherwise uncontaminated ground water (false positive)
- Sorption of organic compounds into the well casing exposed to contaminated ground water (false negative).

Sorption was cited as the major problem in the guidance document, since the possibility of a false negative is of prime concern to the EPA and other regulatory agencies.

In August of 1985, Waste Management Inc. (Jarke 1986) conducted a preliminary research program designed as a practical and realistic evaluation of the potential for sorption to occur for PVC and other materials of construction expected in monitoring wells. This study was designed to address the potential for sorption of previously exposed casing surfaces. Proper monitoring well sampling protocols require that the stagnant water (in equilibrium with the casing) be pumped from a well prior to sampling. The recharge of formation water would

then be representative of the ambient organic concentration and would be in contact with the saturated surface of the well casing for a period of between one and 24 hours before sampling. Therefore, a series of experiments were conducted to investigate the potential of exposed casing materials to further sorption by recharge of the well. Results of those experiments demonstrated that for all materials exposed for both the one and 24 hour cases, the net sorption was nominally zero. These results, however, were only preliminary because the study was limited to a small number of exposure samples, making a statistical interpretation of the data impossible.

The major technical question resulting from WMI's preliminary research program was the use of methanol as a means of dissolving sorbates of interest into water for material exposure studies. Since the final level of methanol in water was significantly greater than the sorbates, it is possible that the exposed material surfaces became saturated with a mono-layer of methanol, preventing any sorption of other organics. Radian's studies were designed to address the criticisms of the WMI work. Consequently, methanol was not used as a vehicle for introducing sorbates to the water matrix. Each component was spiked directly into pure water. Also, each experiment was done in triplicate with full quality assurance and control procedures followed throughout.

Technical Approach

Well Material Coupons: All materials were obtained from Brainard-Kilman Drill Co. (Stone Mountain, Georgia). The PVC was "TriLoc Monitor Pipe," 2-inch (51mm) I.D. by 2 $\frac{3}{8}$ -inch (60mm) O.D. The stainless steel was "Armco Welded 2-inch Type 316," 2 $\frac{1}{4}$ -inch (56mm) I.D. by 2 $\frac{3}{8}$ -inch (60mm) O.D. The polytetrafluoroethylene (virgin PTFE) was 2 $\frac{1}{8}$ -inch (52mm) I.D. by 2 $\frac{3}{8}$ -inch (60mm) O.D. All coupons were cut to a length of 53mm, which produced a surface area of 100cm² per coupon. Each coupon was then cut once lengthwise to allow placement into a 237 mL jar. The tube edges were not considered to be a factor in this study.

Exposure Jars: The exposure jars were "Quorpak Clear with TFE-Lined" screw caps 237 mL capacity (actual 260 mL with no head space). These jars were

obtained from Fisher Scientific, catalog #03-230-7D. All jar cap liners were covered with foil to eliminate possible jar liner effects.

Solvents: All solvents were purchased as chromatography grade and were used without further purification. A stock spiking solution containing methylene chloride (MeCl₂), 1,2-dichloroethane (1,2-DCE), trans-1,2-dichloroethylene (DCEE), toluene and chlorobenzene at a concentration of 10 ppm each was prepared in distilled/deionized water. A second spiking solution was prepared containing 0.5 ppm trichloroethylene (TCEE) in distilled/deionized water. This separate spike was prepared because of TCEE's much lower solubility in water (1.1 ppm) than the other compounds.

Water: Distilled/deionized carbon filtered water was used.

Procedure

Preparation of Glass Jars: The glass jars and lids were each cleaned with soap and water followed by distilled/deionized water rinse. Each jar and lid was then dried at 100 C.

Solution Stability Studies: Prior to the material exposure studies, the 10 ppm and 0.5 ppm spiking solutions were evaluated for stability and the suitability of the test protocol design. Six 260 mL jars with foil-lined caps were filled (without head space) with aliquots of the spiking solutions and pure water to yield approximately 100 ppb of each component. An additional six jars were filled with pure water and represented water blanks for the study. After a one-hour period, an aliquot from each jar was transferred to a Volatile Organic Analysis (VOA) vial (polytetrafluoroethylene-lined cap) and stored in a refrigerator at 5 C for a seven-day period. At the end of the seven days, the original stock-spiking solution was used to prepare a 100 ppb component solution mixture which was then aliquoted to 2 VOA vials for analysis. Additional VOA aliquots were taken from the original 260 mL jars and used for analysis. On day nine, all VOA vial samples were again analyzed and compared to freshly prepared spiked water. Originally, these exposures were to be at two levels, 10 ppb and 100 ppb, but due to the unacceptable variance of compound recoveries from the spiked water at 10 ppb, only the 100 ppb level was used for the exposures.

Well Material Coupon Exposure Studies: Well material coupons were placed in foil-covered glass jars and filled with spiked water solution so that no head space remained. The water solutions were spiked at levels to yield component concentrations between 90 and 150 ppb. The jars were stored at 5 C in a refrigerator for seven days and agitated daily. After the seven-day conditioning period, an aliquot was pipetted into a 40 mL VOA vial with zero head space for analysis. A second aliquot was also pipetted into a 40 mL vial and stored at 5 C as a preserved sample. The remaining solution in the jar was discarded. The jars, with the original coupons, were then refilled with organic-spiked water of the same concentration as in the seven-day conditioning period. After a contact time of one hour, aliquots were again

taken for analysis and preservation. The jars were refilled with organic-spiked water again for a contact time of 24 hours, after which a third aliquot was taken for analysis. Control samples, consisting of spiked water with no well material coupons were carried through the entire procedure. Blank samples, consisting of pure, unspiked water and no coupons were carried through as well.

Sample Analysis

The analytical procedure followed was EPA Method 602 (EPA 1984), which uses gas chromatography with flame ionization detection. The procedure incorporates a purge and trap technique to concentrate the volatile organics from water samples. The instrument used was a Varian 3700, and the data system was a Varian Vista 402. The column was 1.3m x 2mm, stainless steel, packed with 1 percent SP-1000 on Carboxpack B 60/80 mesh. The temperature of the oven was initially at 45 C for three minutes, then programmed to 200 C at 15 C per minute. The nitrogen carrier gas was set at 30 mL per minute through the column and 40 mL per minute through the Tekmar LCS-1 Purge and Trap. Each sample analyzed was transferred to a 5 mL gas tight syringe equipped with a sample valve. Ten microliters of a three-component internal standard mix (15 ng/ μ L) was added to each sample through the syringe to produce a concentration of 30 ppb. The three internal standards were bromochloromethane, 1-chloro-2-bromopropane and 1,4-dichlorobutane.

One exposure level for each compound was studied. These ranged between 87 and 150 ppb. The concentrations varied because the same mass of each compound was used to prepare the stock-spiking solution of each compound. The density was then used to calculate exact concentrations.

The following is a list of each compound studied and the concentration level prepared in the exposure medium:

methylene chloride	133 ppb
1,2-dichloroethane	126
trans-1,2-dichloroethylene	128
trichloroethylene	147
toluene	87
chlorobenze	110

Calibration and Quality Control

The calibration procedure used for the volatile organic analysis was the external standard technique, with internal standards added to each standard and sample for quality control of the analysis. A calibration curve was constructed by preparing three concentration levels of each compound at approximately 10 ppb, 50 ppb and 100 ppb levels. A system blank of pure water was used as a zero point on the curve. A stock solution was prepared from the pure materials in chromatography grade methanol by accurately measuring microliter portions into a known volume. The concentration of each component was then calculated based on its density (mg/ μ L). The

working standards were prepared fresh each day by dilution of the stock into pure water. The three-point curve was prepared initially at the beginning of the stability study and again at the beginning of the exposure study. Linear regression equations were calculated for each curve, then plotted for visual agreement with linearity. Subsequently, only the 100 ppb standard was used to ensure that the calibration curve was within the EPA protocol of <10 percent CV (percent coefficient of variation). The linear regression plots are shown in Figure 1. In addition, three internal standards were used to verify system control. Each 5 mL sample and standard received 10 μ L of the three-component mix (15 μ g/mL), which was equivalent to 30 μ g/L (ppb), immediately before analysis. Figure 2 is a plot of the internal standards with the calculated percent CVs. This data shows that throughout the study the percent CVs were 5 percent, which, according to EPA Method 602 <10 percent is acceptable. The mean concentration of all the blanks analyzed during the exposure study shows that methylene chloride was 13 ppb; 1,2-DCE was 3 ppb; DCEE and TCEE were 2 ppb; and toluene and chlorobenzene were 1 ppb.

Results and Discussion

Results of Stability Study

The objective of the stability study was to evaluate each compound for recovery variability excluding well-casing material sorption effects. Analyses of duplicate samples determined precision; and analyses at day one, day seven and day nine determined the potential storage effects. Analysis of water blanks also determined background contamination due to the glass jars, liners and storage. The stability study also established a reference for evaluating analytical precision and quality control of

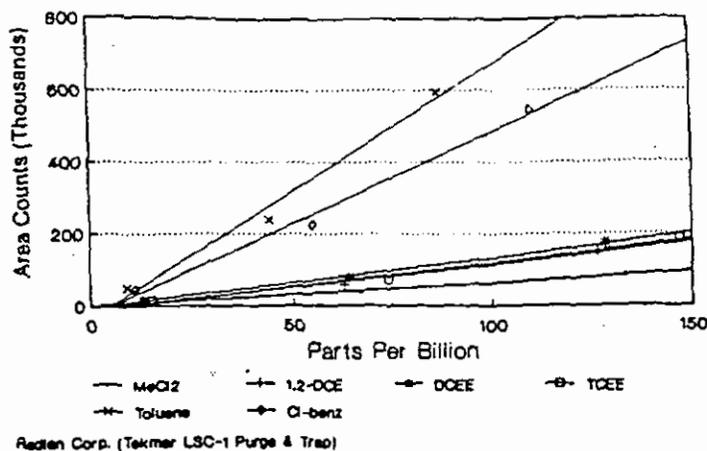


Figure 1. Volatile organic analysis calibration curves of coupon exposures

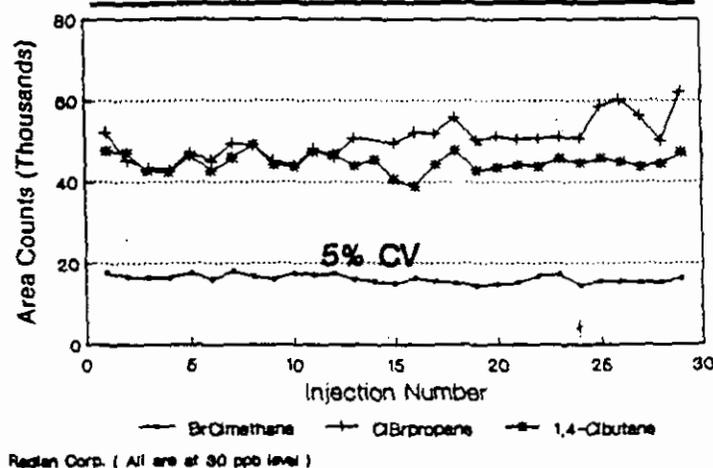


Figure 2. Percent coefficients of variation of three internal standards for coupon exposures

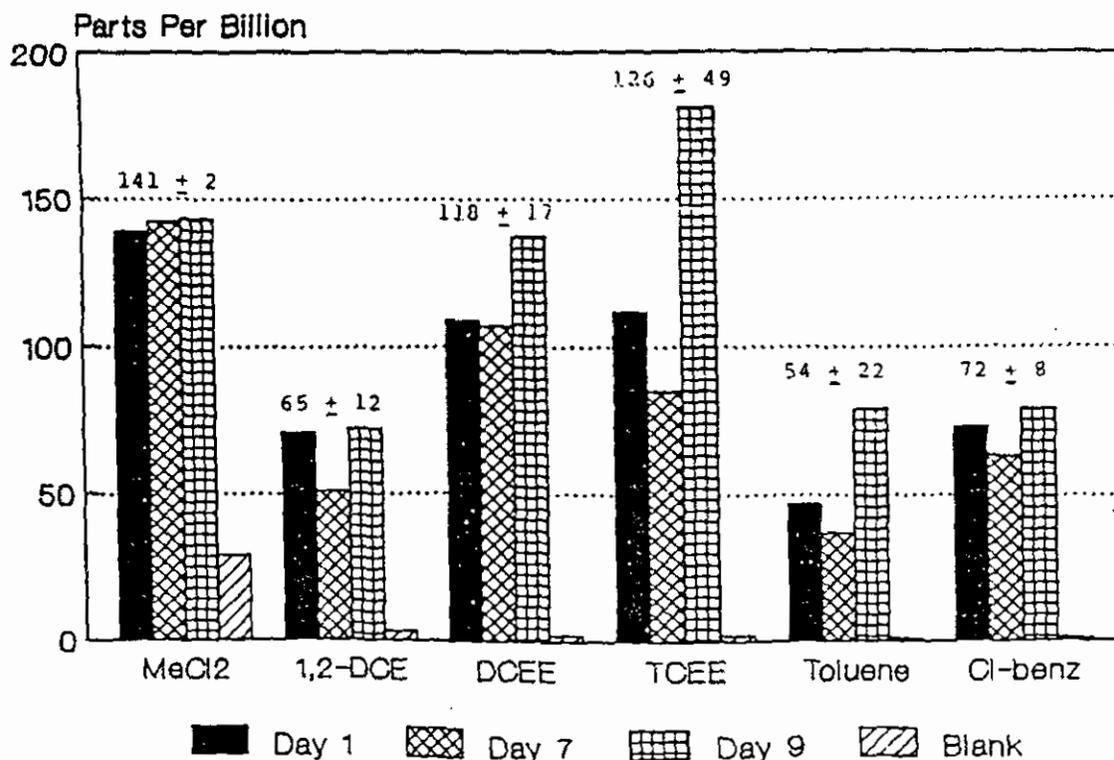


Figure 3. Stability of volatile organics in 5 degree C. water for nine days

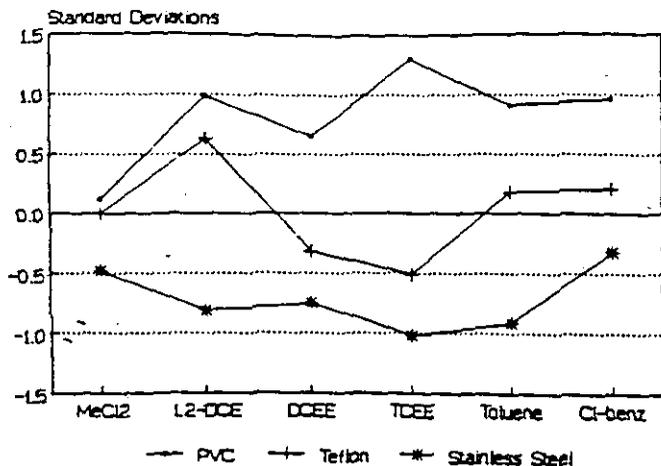


Figure 4. Standard deviation of one hour coupon exposures vs. controls

the method.

The results of the stability study show that for the compounds studied, there is some variability. This variability seems to be associated with the solubility of each compound in water. Figure 3 graphically displays the results for each compound at day one, day seven, and day nine, and a blank water sample.

Results of Material Exposure Study

The objectives of the material exposure study were to determine if there were significant differences in compound sorption between PVC, polytetrafluoroethylene, and stainless steel well-casing materials when exposed to volatile organic hydrocarbons in a simulated "well" environment. The laboratory experiments were designed to simulate actual conditions of sampling ground water containing approximately 100 ppb of hydrocarbons normally found in contaminated waters. In addition, the experiments were designed to determine adsorption or desorption effects of these materials when exposed to these compounds over time. It is necessary to determine if false positive or false negative results bias the actual concentrations of the samples. The study did not, however, determine if the materials released compounds into non-contaminated water.

The results of the material exposure study are shown in Figures 4 and 5. Nine control samples were analyzed for each compound studied. These nine controls were averaged and a standard deviation obtained. The three replicate values for each compound for each type of material studied were also averaged. Figures 4 and 5 represent the effect of sorption at one hour and 24 hours, respectively, for the six compounds on the three casing materials. All results are approximately one standard deviation of the mean for all compounds and all casing materials. A more rigorous analysis of these data will be performed at a later date.

The results of these experiments show that statistically, there is no significant difference between PVC, polytetrafluoroethylene, and 316 stainless steel well casing

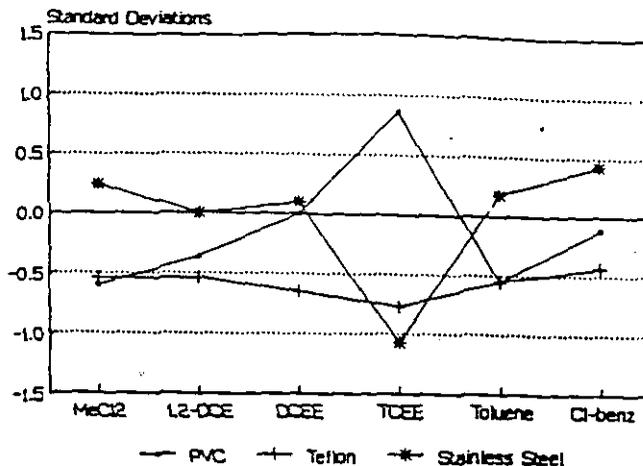


Figure 5. Standard deviation of 24-hour coupon exposures vs. controls

materials when exposed to approximately 100 ppb contaminated water for seven days and 5 C, then exposed for one hour, and then for 24 hours.

Acknowledgment

This project was funded by Waste Management Inc. The authors wish to acknowledge C. Blackley, J. Lummis, N. Cole and T. Buedel for their assistance in conducting these experiments.

References

- EPA Method 602. 1984. Purgeable Aromatics in 40 CFR Part 136. Federal Register, v. 49, no. 209, pp. 40-48.
- Jarke, F.H. 1986. A Review of Materials Used in Monitoring and Monitoring Well Construction. Draft Report. Waste Management Inc., Oak Brook, Illinois.

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APPENDIX D

WELL CONSTRUCTION AND PURGING EFFECTS ON
GROUND-WATER SAMPLES

lowest rankings are for some benzoic acid, aniline, and phenol derivatives; of course, if some other criteria were used, the ranking might have been somehow different, but probably not too much. For example, the ranking obtained by using Freitag and co-worker's chlorobenzene data and the ranking using Ribo and Kaiser's data are very similar. When a new chemical is developed and its properties are known, it can be easily ranked and compared with other known chemicals or any arbitrary standards. The availability of the program in microcomputer form make the routine applicability easy.

A final comment: the development of a suitable index for environmental risk has been widely discussed in the literature (3, 4, 12). An index is a suitable scalar function of the vector distance components with the best chemicals having the lowest index. Since an index is a scalar quantity, problems concerned with the noncomparability of chemicals cannot arise since the chemicals can always be ranked and represented as a chain in a Hasse diagram. Unfortunately, the choice of a particular index affects the results (7).

A simple example can clarify the previous arguments: Let C' and C'' be the vector distance components of a chemical C and let $F = C' + 2C''$ be the chosen index. The chemical C2 previously considered has the components $C'_2 = 2$ and $C''_2 = 3$; thus, $F_2 = 2 + 2 \times 3 = 8$. Analogously, for the chemical C3, $C'_3 = 3$, $C''_3 = 2$, and $F_3 = 3 + 2 \times 2 = 7$. As $F_3 < F_2$, the chemical C3 would be considered safer than C2. Conversely, if the index were $G = 2C' + C''$, then $G_2 < G_3$ and C2 would have to be considered safer than C3. The conclusions are opposite to each other. They depend only on the index chosen. As a consequence, every time the definition of an index cannot be firmly grounded on a theoretical basis, the results can be completely biased, and the index becomes biased toward a subjective meaning.

Registry No. 1, 108-88-3; 2, 108-86-1; 3, 1817-47-6; 4, 611-06-3; 5, 82-68-8; 6, 65-85-0; 7, 586-76-5; 8, 50-30-6; 9, 62-53-3; 10, 106-47-8; 11, 25265-76-3; 12, 92-87-5; 13, 634-93-5; 14, 91-94-1; 15, 108-95-2; 16, 108-39-4; 17, 123-31-9; 18, 100-02-7; 19, 87-86-5; 20, 128-39-2;

21, 120-81-2; 22, 100-00-2; 23, 92-52-1; 24, 13029-08-8; 25, 34883-43-7; 26, 16606-02-3; 27, 62796-65-0; 28, 39485-83-1; 29, 71-43-2; 30, 108-90-7; 31, 100-16-7; 32, 120-82-1; 33, 608-93-5; 34, 118-74-1; 1,2-dichlorobenzene, 95-50-1; 1,3-dichlorobenzene, 541-73-1; 1,2,3-trichlorobenzene, 87-61-6; 1,3,5-trichlorobenzene, 108-70-3; 1,2,3,4-tetrachlorobenzene, 634-66-2; 1,2,3,5-tetrachlorobenzene, 634-90-2; 1,2,4,5-tetrachlorobenzene, 95-94-3.

Literature Cited

- (1) Freitag, D.; Lay, J. P.; Korte, F. In *QSAR in Environmental Toxicology*; Kaiser, K. L. E., Ed.; D. Reidel: Dordrecht, 1984; pp 111-136.
- (2) Kaiser, K. L. E.; Dixon, D. G.; Hodson, P. V. In *QSAR in Environmental Toxicology*; Kaiser, K. L. E., Ed.; D. Reidel: Dordrecht and Boston, 1984; pp 189-206.
- (3) Klein, W.; Geyer, H.; Freitag, D.; Rohleder, H. *Chemosphere* 1984, 13, 203-211.
- (4) Serai, W. F.; Budinger, F. E.; Mueller, P. K. *J. Bacteriol.* 1965, 90, 832-833.
- (5) Preparata, F. P.; Yeh, R. T. *Introduction to Discrete Structures*; Addison-Wesley: Reading, MA, 1973.
- (6) Warfield, J. N. *IEEE Trans. Syst. Man Cybernetics* 1973, SMC-3, 121-132.
- (7) Keeley, R. L.; Raiffa, H. *Decisions with Multiple Objectives: Preferences and Value Tradeoffs*; Wiley: New York and Toronto, 1976; pp 1-400.
- (8) Reggiani, M. G.; Marchetti, F. E. *IEEE Trans. Syst. Man Cybernetics* 1975, SMC-5, 322-330.
- (9) Harary, F. *Graph Theory*; Addison-Wesley, Reading, MA, 1969.
- (10) Ribo, J. M.; Kaiser, K. L. E. *Chemosphere* 1983, 12, 1421-1442.
- (11) Ribo, J. M. "Chlorinated Aromatic Compounds: Microtox Test Results and Structure Toxicity Relationships"; Unpublished Report 84-3, 1984; Environment Canada, National Water Research Institute Publication, Burlington; pp 1-67.
- (12) Koch, R. In *QSAR in Environmental Toxicology*; Kaiser, K. L. E., Ed.; D. Reidel: Dordrecht and Boston, 1984; pp 207-222.

Received for review December 3, 1984. Revised manuscript received February 20, 1986. Accepted June 6, 1986.

Well Construction and Purging Effects on Ground-Water Samples

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Multiple well installations of selected casing materials (i.e., poly(tetrafluoroethylene) (PTFE), 304 stainless steel (SS), and poly(vinyl chloride) (PVC)) were constructed and sampled to determine if well purging and construction procedures would significantly bias chemical constituent determinations in ground water. Water quality results from six monthly sampling dates indicate that proper purging of stagnant water from monitoring wells and isolation of cement seals are essential for the collection of representative chemical data. Significant differences in purged samples taken from PTFE, SS, or PVC wells were observed for total organic carbon and volatile halocarbons, which may be linked to the materials' interaction with ground water. The well casing interferences were not predictably high or low for any of the materials.

Introduction

The effects of well construction procedures and sampling protocols on the reliability of ground-water chemistry investigations have been the subject of a number of research efforts in the past 10 years. The published literature has

dealt mainly with the potential error introduced by water sampling or analysis (1-3) and by the selection of materials that are appropriate for specific monitoring applications (4, 5). A recent publication details procedures by which ground-water sampling protocols may be developed to control systematic sources of sample collection (6). These errors include artifacts of well siting or construction, well purging, and sample retrieval from the well. In general, sample collection errors cannot be accounted for by traditional laboratory quality control measures. Also, large sampling errors coupled with analytical errors of similar severity can result in the collection of grossly-biased data.

Recent work has shown that potential sampling bias due to both sampling mechanisms and flexible tubing materials is of the same order of magnitude (i.e., $\pm 5-20\%$) as analytical errors for volatile organic compounds (7, 8). These results support the need for very careful consideration of volatilization, sorption, and desorption effects in the selection of sampling pumps or tubing for ground-water investigations.

The effects of drilling fluids, grouts, or well casing interactions with the geologic formation or ground water are

more difficult to evaluate for a number of reasons. Foremost among these reasons is that natural variability in ground-water quality has not been studied in detail. Also, there have been no systematic field studies reported on the effects of well purging or casing materials on organic compound levels in ground water where inert materials have been used as controls. Houghton and Berger (9) have reported significant differences in ground-water dissolved organic carbon and trace metal levels in samples from stainless-steel and thermoplastic well casing materials. The observed differences were of the order of analytical bias (i.e., ± 10 –50%). Laboratory investigations may help in evaluating sources of sampling imprecision or bias; however, systematic field studies are needed to evaluate the actual severity of such errors. The unique details of well construction, completion, development, and the in situ geochemical conditions for an actual ground-water monitoring installation may enhance or limit the potential effects of materials or mechanisms that have been observed in controlled laboratory testing (10).

This study was undertaken to compare the effects of well purging and well construction materials on the reliability of determinations of inorganic and organic chemical constituents in ground water at two sanitary landfill sites in east-central Illinois. Due to the fact that the materials' related errors have been documented for ferrous metal well casing materials other than stainless steel, these materials were not considered in the study (5). Similarly, no solvent cements, nonthreaded joints, or uncommon materials were employed in well construction.

Procedures

Site Descriptions and Well Construction. The two 85–100-acre sites had been operated as municipal landfills/dumps for at least 15 years. Household trash, some light industrial wastes, and other refuse made up the 5–10 million tons of waste emplaced at each site prior to closure. Sampling installations were constructed in May 1983 with hollow-stem auger (10-in. o.d., 6-in. i.d.) drilling techniques. All drilling equipment, well casings or screens, split-spoon samplers and steel tapes, etc. were steam-cleaned before they were used to minimize the introduction of foreign materials into the subsurface. Geologic conditions at both sites were characterized as till deposits (i.e., clayey silt with traces of fine to coarse sand) from below the surface topsoil to depths of 6–8 m (20–25 ft). Sand or sand and gravel lenses were observed in split-spoon samples of the till deposits at depths between 8 and 14 m (25–45 ft). All wells were 2 in. in o.d. and were completed at the depth where substantial (>0.5 ft) sand or sand/gravel lenses were observed (see Table 1). Wells were placed within 2 m of each other at each location to minimize the potential effects of horizontal inhomogeneities in the formations at the completion depths. At site 1, upgradient and downgradient wells of poly(tetrafluoroethylene) (PTFE, Teflon), stainless steel (SS), and poly(vinyl chloride) (PVC) were constructed. At site 2, stainless steel and PVC wells were completed at upgradient and downgradient locations, and a Teflon/Al oxide, gas-drive dedicated sampler (Arcad) was installed in place of a PTFE (Teflon) well. All wells were completed with 2-ft screens and packed with Ottawa silica sand from the bottom of the screened interval to at least 1 ft above the top of the screen at site 1. Star slurry seals of 2–5 ft were placed in the bore holes above the sand pack of the site 1 wells. The slurry mix consisted of 1.5:1.0 mixtures by weight of the silica sand with Chem-Comp II shrinkage-compensating cement. The wells were finished by backfilling the bore holes to 4–6 ft from the surface with silica sand, and a cement surface

Table 1. Field Installations Data

location	material ^a	depth of screened interval (below land surface), ft
site 1 (upgradient)	PVC	36.5–38.5
	stainless steel (304)	34.5–36.5
	PTFE	35.5–37.5
site 1 (downgradient)	PVC	25.5–27.5
	stainless steel (304)	26.0–28.0
	PTFE	26.0–28.0
site 2 (upgradient)	PVC	32.5–34.5
	stainless steel (304)	32.0–34.0
	gas-drive sampler (Teflon/Al oxide)	30.75–32
site 2 (downgradient)	PVC	40.5–42.5
	stainless steel (304)	40.5–42.5
	gas-drive sample (Teflon/Al oxide)	41.25–42.5

^a Materials used in the study were as follows: PVC, rigid poly(vinyl chloride) schedule 40 NSF approved for potable water uses; SS, 304 stainless steel; PTFE, poly(tetrafluoroethylene) (PTFE) (Teflon, Du Pont).

then placed to seal the bore hole from surface water influx. At site 2, the native sand or sand/gravel heaved up above the screened interval from 6 to 15 ft. These installations were completed by placing a cement seal from the top of the heaving material to the land surface.

The wells at both locations were developed within a week of the construction date by use of filtered, compressed air (170 cfm) and procedures described previously (3). The wells at site 2 did not require further development work. Although the wells at site 1 were redeveloped at intervals for a period of 12–15 months, turbid water samples were encountered on all sampling dates due to the presence of cement fines.

Well Purging and Sampling. Hydraulic conductivity testing was done on all of the wells at each location to establish their hydraulic performance. The well purging requirements necessary to isolate stagnant water in the well bore were determined by a previously published method (6). Positive-displacement bladder pumps constructed of stainless steel and PTFE with PTFE delivery tubing were utilized for well purging and sampling operations (QED Well Wizard). The pumps were driven by a single controller through a gas manifold, which permitted a simultaneous supply of drive gas (air or N₂) to each pump. Pump intakes were set at the top of the screened interval for both purging and sampling operations.

The experimental design included the measurement of the well purging parameters [pH, Eh, temperature, and conductivity (Ω^{-1})] by use of an in-line flow-through electrode cell prior to well purging. Stagnant water was then collected for NO₂⁻, S²⁻, Fe²⁺, total organic carbon (TOC), alkalinity, and selected volatile organic compounds. After collection of these samples, the outputs of the pumps were reconnected to the manifold of a flow-through electrode cell to monitor the progress of purging the wells simultaneously. Pumping rates during well purging were controlled between 500 and 1000 mL·min⁻¹, and the purging was continued until the levels of the well purging parameter stabilized to within approximately $\pm 10\%$ over a minimum of 4–5 L pumped ($\sim 25\%$ of 1 well volume). All adjacent wells were purged of an equal volume of stagnant water before sampling. The volume depended on site hydrologic conditions and measured hydraulic conductivities. Once well purging was completed, pumping rates were reduced to 100–500 mL·min⁻¹, and samples were taken for the following parameters in order: alkalinity,

Table II. Well-Purging Effects on Ground-Water Quality

	stagnant	purged	difference (+, -)	change (factor of)
Dissolved Ferrous Iron, mg-L ⁻¹ , Site 2 (8/6/85)				
upgradient				
PVC	0.03	0.18	0.15	+5
SS	0.02 ^a	0.25	0.23	+12
downgradient				
PVC	0.04	4.57	4.53	+113
SS	1.23	4.37	3.14	+2.6
Dissolved Sulfide, mg-L ⁻¹ , Site 1 (7/9/85)				
upgradient				
PVC	0.048	0.242	0.194	+4
SS	0.010 ^b	0.012	0.002	+1.2
PTFE	0.031	0.172	0.141	+4.5
downgradient				
PVC	0.308	0.047	0.261	-0.85
SS	0.024	0.384	0.360	+15
PTFE	0.166	0.230	0.064	+0.39

^aMethod detection limit = 0.02 mg-L⁻¹ [accuracy 90% (-10% bias), precision ±15% rad]. ^bMethod detection limit = 0.010 mg-L⁻¹ [accuracy 80% (-20% bias), precision ±20% rad].

TOC, volatile organic compounds, dissolved inorganic constituents, and acid and base/neutral organic compounds. Pressure filtration of samples for dissolved constituent determinations was performed in the field. TOC and volatile organic compound samples were collected in 40-mL glass vials, sealed without head space by PTFE-lined septum caps.

Analytical Methodologies and Quality Control. Organic and inorganic chemical constituents were determined by USEPA-recommended methods (11, 12). Total organic carbon (TOC) determinations were performed by a methodology described previously (13). Field determinations of alkalinity were performed by potentiometric titration, and the results were analyzed by Gran end-point analysis procedures. On each sampling date, a series of field standards and blanks were used to account for transport and storage errors, which supplemented the daily analytical procedure control standards for both laboratory and field methods. Replicate samples were analyzed at intervals during analytical sessions to establish the precision and accuracy of the water chemistry data.

Results and Discussion

Well Purging. Stagnant water samples analyzed for pH, alkalinity, reduced inorganic constituents [S²⁻, Fe(II)], and organic constituents showed higher temporal (i.e., month to month) variability than did the samples obtained after proper well purging. In most cases, well purging resulted in the stabilization of the gross solution chemistry parameters (i.e., pH, E_h, Ω⁻¹, alkalinity) although the concentrations shifted in varying degrees. For example, at site 2, alkalinity levels in stagnant water samples were generally ~25% lower than those measured in water samples after purging. The average magnitude of the alkalinity differences between samples from SS or PVC wells was not significantly different over the course of the study.

In general, the levels of reduced chemical constituents [i.e., Fe(II), S²⁻] shown in Table II would be expected to be lower in the stagnant water in monitoring wells than in ground water obtained from the formation after purging. The actual concentration differences observed between stagnant and purged samples reflected this trend. However, the magnitude of the differences was quite variable and may have been influenced by well casing materials and small-scale heterogeneities in water quality, even in wells

that were finished in the same formation less than 5-ft apart. In downgradient wells at both sites, the stagnant water in the SS well frequently (i.e., four of six sampling dates) showed higher levels of ferrous iron and lower levels of dissolved sulfide than those from the adjacent PVC or PTFE wells. This would be consistent with leaching of iron from the stainless steel and precipitation of sulfide by the excess iron during stagnant periods. Obviously, PVC and PTFE would not be expected to leach iron in this fashion. Example results for ferrous iron given in Table II show stagnant water values from PVC and SS wells that were significantly different at least at the reducing downgradient location. Upon purging, however, ferrous iron values in samples from both upgradient and downgradient locations increased substantially, and the formation water values from both sets of PVC and SS wells were equal within experimental error. It would seem from these observations that either PVC or SS well casing would be appropriate for ferrous iron determinations if purging is complete. Typical results for dissolved sulfide, an example of which is shown in the lower portion of Table II, reflect somewhat random changes on purging. These changes do not consistently correspond to potential well casing effects or the introduction of more reducing formation water. Rather, it seems that variability well above the detection limit and analytical precision can result from natural heterogeneities in water chemistry for some chemical constituents. These average chemical differences in reduced species between stagnant water samples and those obtained after purging were frequently a factor of 5 greater than the errors involved in either the sampling or analytical procedures. Also, purging-related variations in TOC were observed to increase or decrease by ±50% in all wells over the range of 1.20-30.0 mg of C-L⁻¹. No consistent trends in TOC as a function of purging could be associated with the different casing materials. Purging a particular well at either site frequently resulted in greater differences in water quality than was observed between upgradient and downgradient ground locations. Improper well purging can obviously cause gross bias in ground-water monitoring results that far exceeds that due to materials' effects or sampling mechanisms.

Cement Grout Contamination. Despite the fact that all the drilling and well completion operations were held constant, the wells at site 1 exhibited significant water quality differences in both stagnant and formation water samples that could not be attributed to well purging or casing materials. For example, upgradient and downgradient wells at site 1, with the exception of the downgradient PVC well, exhibited pH values in excess of 12 pH units, and the alkalinity was primarily associated with hydroxyl ion. The downgradient PVC well yielded formation samples with pH values between 7 and 8 pH units (mostly bicarbonate alkalinity), and the stagnant water values ranged from 8 to 12 pH units.

Anecdotal references to monitoring well contamination by cement grouts have been reported (14). The usual symptom that has been observed is very high solution pH (i.e., 10-12 pH units), even after exhaustive well development. In this study a total of 12 sampling installations was constructed, and the bore holes were sealed above the screen and gravel pack, as well as at the surface with a sand/shrinkage-compensating cement mixture. The grout seals were introduced as thick slurries into standing water in the bore hole after the well had been constructed.

At site 1, either native geochemical conditions or differential settling of the cement mixture prevented the grout seals from setting up properly. The wells at this site

Table III. Comparison of Ground-Water Quality and Grout/Water Mixture Analytical Results

	contaminated ground water (field site 1, upgradient) ^a	grout-contaminated water (laboratory) ^a
major species		
pH, pH units	12.7	>13
alkalinity (OH ⁻), milliequiv·L ⁻¹	31.6	167
sulfate	35.9	1076
calcium	190	1069
potassium	8.21	578
sodium	26.6	62
silicate	5.84	0.7
conductivity, μ S	1975	11 500
other species		
magnesium	0.05	0.02
aluminum	<0.10	0.17
chloride	11.0	1.04
iron (total)	0.09	0.10
manganese (total)	0.02	<0.02

^a Unless otherwise specified, units are mg·L⁻¹.

consistently showed high pH, alkalinity (>90% hydroxyl or caustic), Ω^{-1} , and Ca^{2+} with the exception of the downgradient PVC well, which showed only slightly elevated levels of caustic alkalinity. The apparent cement contamination problem persisted more than 18 months after construction, despite at least 10 redevelopment attempts. The installations at site 2 did not show any evidence of cement contamination, although several early pH measurements of formation water exceeded the long-term mean of ~7.4 pH units.

The cause of this persistent contamination was investigated by careful analysis of a number of cement/sand/water mixtures prepared in the laboratory from the same materials used in well construction. The data in Table III show the comparison of average solution chemistry results from an upgradient well and the aqueous phase of grout/water mixtures prepared in the laboratory. Clearly, the major ionic species identified in the grout-contaminated ground water are represented in the laboratory grout/water mixture aqueous phase. The alkalinity, conductance, and the calcium values in the ground-water samples were all approximately one-fifth their respective levels in the laboratory sample. Attempts to calculate an ion balance on the basis of OH⁻ as the major anion resulted in ion balances that seldom agreed to better than 40%. There was always an apparent excess negative charge in solution. The possibility of a polymeric or gel phase (e.g., polysilicates) that might interfere with the analytical data or confute the dissolved ion charge balance was examined by NaHCO₃ digestion of the aqueous samples prior to analysis (15). It was verified that no unusual forms of silica or common alkali or alkaline earth cations were present that might cause gross errors in the alkalinity titrations or wet-chemical analytical methods. Therefore, the grout-contamination problem remains an enigma, although it should be obvious that grouting materials can drastically effect the reliability of ground-water chemistry data for long periods after well construction.

The cement grouts should have been placed either after water had been removed from the well bore or by tremie pipe. Also, the sand pack could have been isolated from the cement grout by a bentonite seal.

Well Casing Material Effects. Apart from the differences noted above (Tables II and III) in the stagnant water samples, no consistent effects on the inorganic chemical constituent data from the two sites were observed

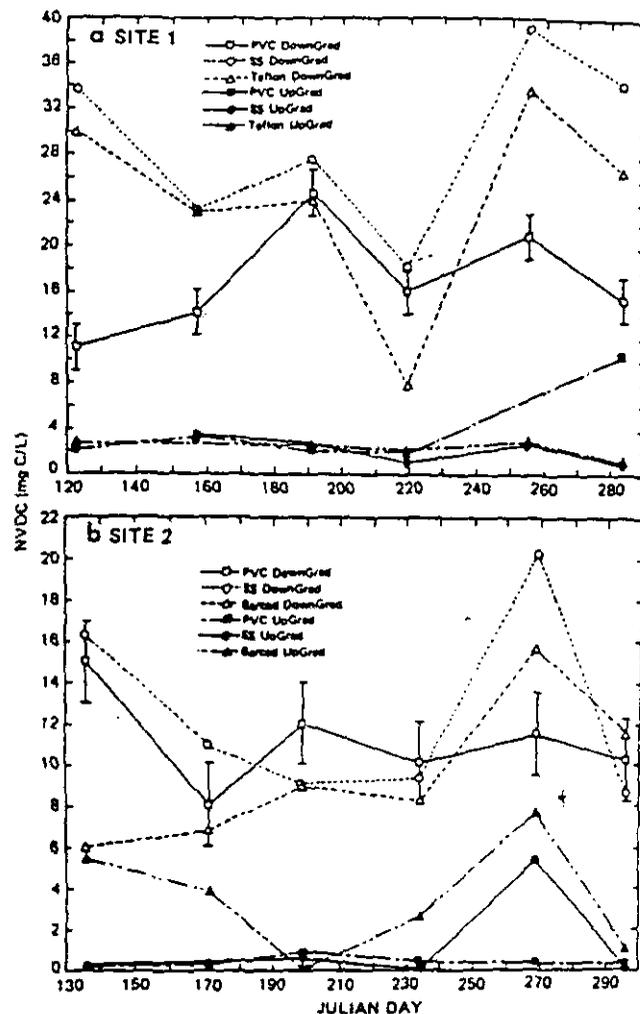


Figure 1. Total organic carbon (mg of C·L⁻¹) in ground water at site 1 and 2. The error bars indicate two standard deviations from the mean of triplicate determinations.

that might be attributed to well casing material exposure alone. Total Fe and Mn concentrations were not significantly different in samples from the SS wells as compared to those from the PVC or PTFE wells. These observations contrast with the elevated metal levels reported by Houghton and Berger on results from a single sampling event (9). The divergent results from this study reflect the influence of actual site conditions, temporal variability, and well purging practices on the type and relative severity of sampling and well construction related errors.

At the downgradient locations, all of the installations showed the effects of apparent landfill leachate contamination (i.e., elevated Cl⁻, Na⁺, K⁺, specific conductance, and TOC). The observed levels of TOC or specific organic compounds, however, did not provide evidence of major organic (i.e., volatiles, base-neutrals, and phenols) contamination of these locations.

The observed levels of the nonvolatile fraction of TOC (i.e., NVOC) in ground water from the upgradient and downgradient wells at site 1 after purging are plotted in Figure 1a as a function of time during the summer and fall of 1984. The upgradient samples from each type of well showed negligible differences from the mean TOC of ~3 mg of C·L⁻¹. The downgradient location, however, generally showed higher levels of TOC in samples from the stainless steel and Teflon wells than did samples from the PVC well. On four of the six sampling dates, the positive differences were significant at the 0.05 confidence level.

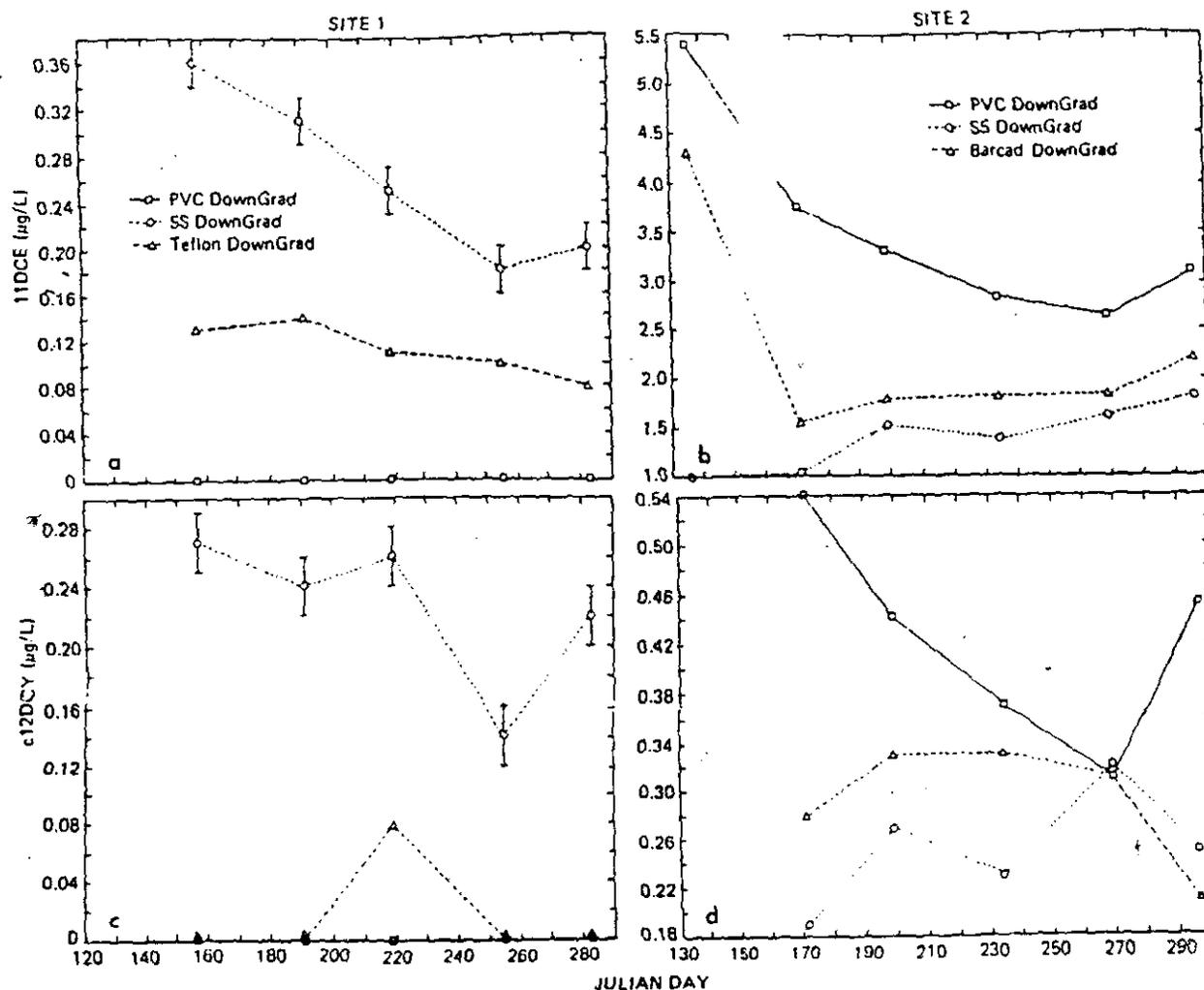


Figure 2. Levels of 1,1-dichloroethane (11DCE) and *cis*-1,2-dichloroethylene (C12DCY) ($\mu\text{g}\cdot\text{L}^{-1}$) in ground water at sites 1 and 2. The error bars indicate two standard deviations from the mean of triplicate determinations. Detection limit = 0.08 and 0.10 $\mu\text{g}\cdot\text{L}^{-1}$, respectively.

The TOC results at site 2 showed a similar pattern; however, the differences in samples from the individual well casing materials were not statistically significant when compared to the analytical and sampling errors.

At site 1, levels of 1,1-dichloroethane (11DCE) and *cis*-1,2-dichloroethylene (c12DCY) after purging also were higher generally in the samples from the downgradient Teflon and stainless steel wells than in those from the PVC well (Figure 2a,c). The error bars on the figure represent two standard deviations from the mean determined by analysis of replicate samples and standards. In these cases, the "stainless steel" results were significantly higher (at the 0.05 level) than either the Teflon or the PVC data. These concentration levels are all quite low but in the range of quantitation of the purge and trap analytical methodology. Therefore, systematic differences in observed trace organic compound distributions may arise from the sorptive effects of polymeric well casings as compared to stainless steel. The samples from the differently cased wells at site 1 showed more net difference in purgeable organic compounds than any single well showed over the study period.

Overall, the levels of 1,1-dichloroethane at site 2 (Figure 2b,d) were 10 times above the levels measured at site 1. In contrast with the other results, levels of purgeable organics in samples from the PVC well were consistently higher than those in samples from either the stainless steel well or the gas-drive sampler. It is unlikely that the paired wells actually intercepted ground water of different mi-

croconstituent quality, as they were finished only ~1 m (4 ft) apart. Yet, the PVC and stainless steel results for 11DCE differ by a factor of 2. This is more than 10 times greater than the precision established for analytical determinations of these compounds. This order of materials-related effect could result in systematic under- or overestimates of the extent of ground-water contamination, under some conditions. This type of error cannot be accounted for in statistical comparisons of data from up-gradient and down-gradient locations unless an "inert" artifact-free well casing material can be identified.

The observations noted for site 2 above are also supported by the levels of total volatile halocarbons in the PVC and stainless steel wells before and after purging. With the exception of the September 25 sampling date, the stagnant levels of volatile halocarbons from the PVC well are nearly a factor of 2 or more greater than those from the stainless steel installation (see Table IV). On the average, the PVC well yielded samples with higher total levels of volatile halocarbons than did the stainless steel well. It may be that under these conditions the sorptive and leaching properties of PVC tend to maintain a higher background level of organic compounds in ground water exposed to this material relative to stainless steel.

Conclusions

The purging of stagnant water from monitoring wells is essential to the collection of reliable ground-water quality data. In this study, the variations in water chemistry on

Table IV. Total Volatile Halocarbon* Levels in Site 2 Wells, $\mu\text{g} \cdot \text{L}^{-1}$

date	PVC		SS	
	stagnant	purged	stagnant	purged
5/15	7.4	6.2	3.7	1.1
6/19	8.7	7.8	1.4	7.5
7/17	8.8	8.3	3.0	4.8
8/21	7.0	6.8	1.9	4.2
9/25	12.9	6.1	12.2	4.6
10/23	11.2	6.5	6.2	3.9
average	9.3	7.0	4.7	4.4

* Included total methylene chloride, dichloroethylene, dichloroethane, trichloroethane, trichloroethylene, tetrachloroethane, and tetrachloroethylene after individual separation and quantitation by gas chromatography with HECD detection.

well purging were generally greater than errors associated with either sampling mechanism, tubing, or apparent well casing material effects.

Cement grout seals, which for one reason or another do not properly set up in the bore hole, can cause severe, persistent contamination of both stagnant and formation water from monitoring wells. Determinations of dissolved inorganic chemical constituents are affected by cement contamination, which significantly changes the background solution composition.

Well casing materials exerted significant, though unpredictable, effects on the results of total organic carbon and specific volatile organic compound determinations. Systematic differences ranging from a factor of 2 to 5 in concentration were observed between samples taken after purging from wells cased with different materials. Though the differences were not consistently higher or lower from site to site, materials' performance may limit the conclusions that may be drawn from ground-water quality results in the low ppb ($\mu\text{g} \cdot \text{L}^{-1}$) range.

Acknowledgments

We thank Mark Sievers, Ed Garske, Mike O'Hearn, and Jim Gibb for their help with various aspects of the project.

Registry No. PTFE, 9002-84-0; PVC, 9002-86-2; SS, 12597-68-1; H_2O , 7732-18-5; Fe, 7439-89-6; S^2 , 18496-25-8; C, 7440-44-0; 11DCE, 75-34-3; C12DCY, 156-59-2.

Literature Cited

- (1) Grisak, G. E.; Jackson, R. E.; Pickens, J. F. *Water Resour. Bull.* 1978, June 12-14, 210-232.
- (2) Scalf, M. R.; McNabb, J. F.; Dunlap, W. J.; Cosby, R. L.; Fryberger, J. *Manual of Ground-Water Quality Sampling*

Procedures; National Water Well Association: Worthington, OH, 1981; 93 pp.

- (3) Gillham, R. W.; Robin, M. J. L.; Barker, J. F.; Cherry, J. A. "Ground Water Monitoring and Sample Bias"; Prepared for the American Petroleum Institute as API Publication 4367; Department of Earth Sciences, University of Waterloo: Waterloo, Ontario, Canada, June 1983; 206 pp.
- (4) Sosebee, J. B.; Geisler, P. C.; Winegardner, D. L.; Fisher, C. R. "Symposium on Hazardous Wastes"; Proceedings of the American Society for Testing and Materials, Orlando, FL, Jan 1982; American Society for Testing and Materials: Philadelphia, PA, 1982; pp 38-50.
- (5) Barcelona, M. J.; Gibb, J. P.; Miller, R. A. "A Guide to the Selection of Materials for Monitoring Well Construction and Ground-Water Sampling"; Contract Report 327; Illinois State Water Survey: Champaign, IL, 1983; EPA/600/S2-84/024, 78 pp.
- (6) Barcelona, M. J.; Gibb, J. P.; Helfrich, J. A.; Garske, E. E. "A Practical Guide for Ground-Water Sampling"; Contract Report 374; Illinois State Water Survey: Champaign, IL, 1985; EPA/600/S2-85/104, 94 pp.
- (7) Barcelona, M. J.; Helfrich, J. A.; Garske, E. E.; Gibb, J. P. *Ground Water Monitor. Rev.* 1984, Spring, 31-42.
- (8) Barcelona, M. J.; Helfrich, J. A.; Garske, E. E. *Anal. Chem.* 1985, 57(2), 460-464.
- (9) Houghton, R. L.; Berger, M. E. "Effects of Well Casing Composition and Sampling Method on Apparent Quality of Ground Water"; Proceedings of the Fourth National Symposium on Aquifer Restoration and Ground Water Monitoring, May 23-25; National Water Well Association: Columbus, OH, 1984; pp 487-490.
- (10) Miller, G. D. "Uptake and Release of Lead, Chromium and Trace Level Volatile Organics Exposed to Synthetic Well Casings"; Proceedings of the Second Annual Symposium on Aquifer Restoration and Ground Water Monitoring; National Water Well Association: Worthington, OH, 1982; pp 236-245.
- (11) USEPA "Methods for Chemical Analysis of Water and Wastes"; USEPA-EMSL: Cincinnati, OH, March 1979; EPA-600/4-79-020.
- (12) USEPA "Methods for Organic Chemical Analysis of Municipal and Industrial Wastewater"; USEPA-EMSL: Cincinnati, OH, 1982; EPA-600/4-82-057, Method 601—Purgeable Halocarbons, Method 604—Phenols, and Method 610—Polycyclic Aromatic Hydrocarbons.
- (13) Barcelona, M. J. *Ground Water* 1984, 22(1), 18-24.
- (14) Dunbar, D.; Tuchfeld, H.; Siegel, R.; Sterbentz, R. *Ground Water Monitor. Rev.* 1985, 5(2), 70-74.
- (15) APHA "Standard Methods for the Examination of Water and Wastewater", 14th ed.; APHA, AWWA, and WPCF: Baltimore, MD, 1975.

Received for review October 11, 1985. Revised manuscript received April 1, 1986. Accepted July 22, 1986. We appreciate the support of the USEPA-RSKERL, Ada, OK, and USEPA-EMSL, Las Vegas, NV (CR80996601), and the Campus Research Board of the University of Illinois.

APPENDIX E

LEACHING OF METAL POLLUTANTS FROM FOUR WELL CASINGS
USED FOR GROUND-WATER MONITORING

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 were 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., in press) 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-dichloroethane, *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., in press) 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 cm². 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 x 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 HNO₃ (G. Fred-

rick 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 HNO₃ 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 cm²/cm³. 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 HNO₃ (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

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

Hg immediately after the ground-water-filled jar was acidified.

In a preliminary experiment, ground water stored in the polypropylene jars was spiked with 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 HNO₃ (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 HNO₃.

	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(NO₃)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(ClO₄)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

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 ($\mu\text{g/L}$). Materials with common underlining are not different at the 95% confidence level as determined by the LSD.

Days	Well casing					Days	Well casing				
<i>a. Barium</i>						<i>c. Lead</i>					
1	Control	PTFE	PVC	SS304	SS316	1	Control	PTFE	SS316	SS304	PVC
(LSD = 1.4)	4.5	6.0	6.6	7.1	7.7	(LSD = 1.45)	0.16	0.35	0.90	1.14	2.46
5	PTFE	Control	PVC	SS304	SS316	5	Control	PTFE	SS316	SS304	PVC
(LSD = 2.1)	5.3	5.8	6.6	7.8	9.9	(LSD = 1.53)	0.21	0.27	1.27	1.56	2.23
20	PTFE	Control	PVC	SS304	SS316	20	Control	PTFE	SS316	PVC	SS304
(LSD = 2.2)	5.5	5.9	6.1	7.4	11.3	(LSD = 2.80)	0.14	0.35	1.00	1.04	2.86
40	PTFE	PVC	Control	SS304	SS316	40	PTFE	Control	PVC	SS316	SS304
(LSD = 2.0)	5.2	5.6	5.9	7.0	10.1	(LSD = 1.52)	0.21	0.33	0.77	1.26	2.06
<i>b. Chromium</i>						<i>d. Copper</i>					
1	Control	PTFE	PVC	SS304	SS316	1	PVC	SS304	Control	PTFE	SS316
(LSD = 9.76)	0.20	0.22	1.23	1.60	6.06	(LSD = 12.0)	9.4	10.7	11.9	12.1	35.8
5	Control	PTFE	PVC	SS316	SS304	5	PTFE	PVC	Control	SS304	SS316
(LSD = 0.31)	0.20	0.22	1.12	1.79	3.34	(LSD = 12.4)	7.8	9.9	10.1	11.0	42.6
20	PTFE	Control	PVC	SS316	SS304	20	PVC	PTFE	Control	SS304	SS316
(LSD = 1.17)	0.19	0.22	1.20	3.30	4.61	(LSD = 38.5)	6.8	8.3	10.1	26.1	81.2
40	Control	PTFE	PVC	SS316	SS304	40	PVC	PTFE	SS304	Control	SS316
(LSD = 1.04)	0.21	0.21	1.11	2.53	5.13	(LSD = 17.2)	4.4	5.2	11.5	14.0	82.3

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. 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(NO₃)₂ (Baker, Reagent Grade) in deionized water. Working standards were prepared in deionized water acidified to 2% v/v with HNO₃ (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

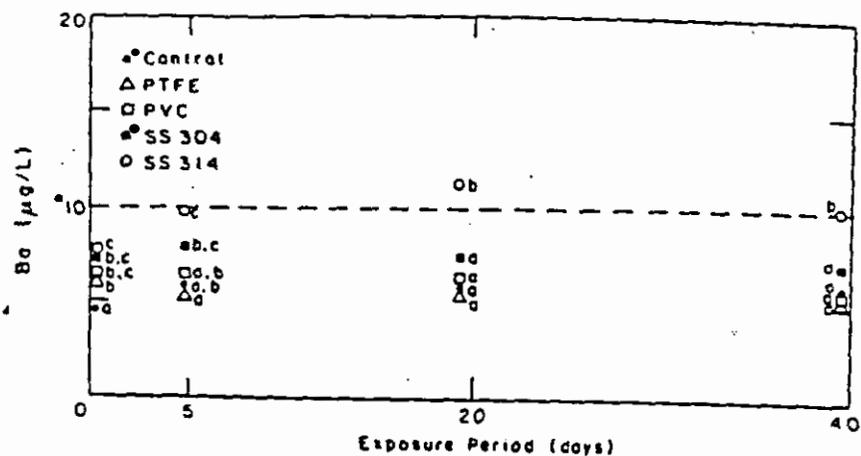
Beyond the 1-day exposure, the analysis consistently demonstrated that both stainless steel well casings contributed significantly greater quantities of Cr to the ground water than the control or the other materials tested (Table 3b). These metal well casings, along with PVC, increased Cr concentrations 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 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.

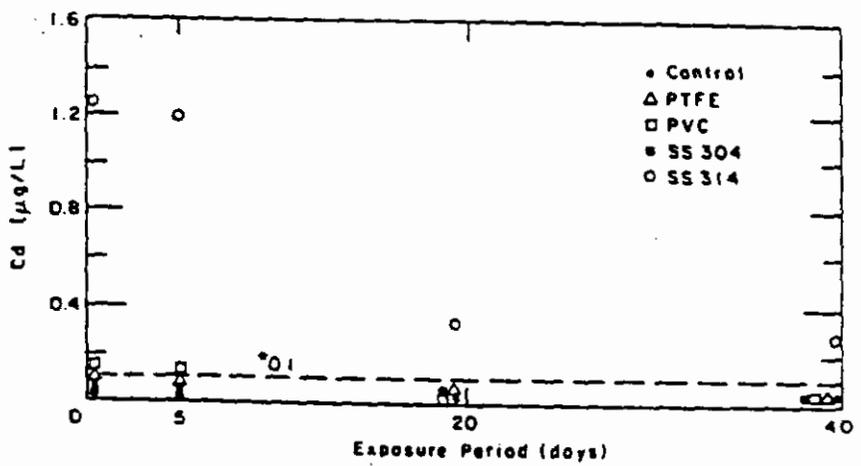
Lead

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).

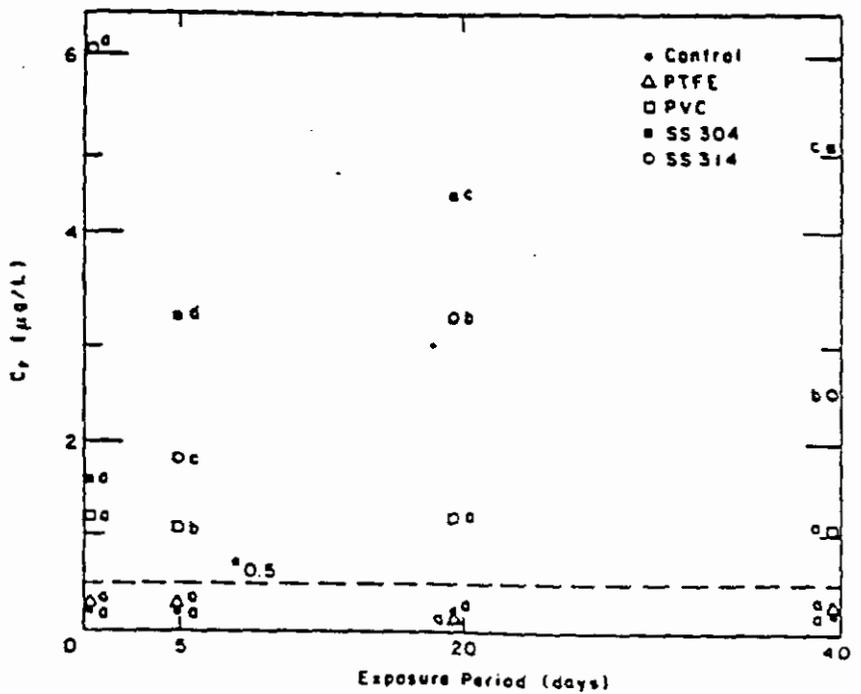
The most distinctive trend was the decrease in Pb with increasing time of exposure for PVC (Fig. 1d). Both stainless steel well casings showed slight



a. Ba.

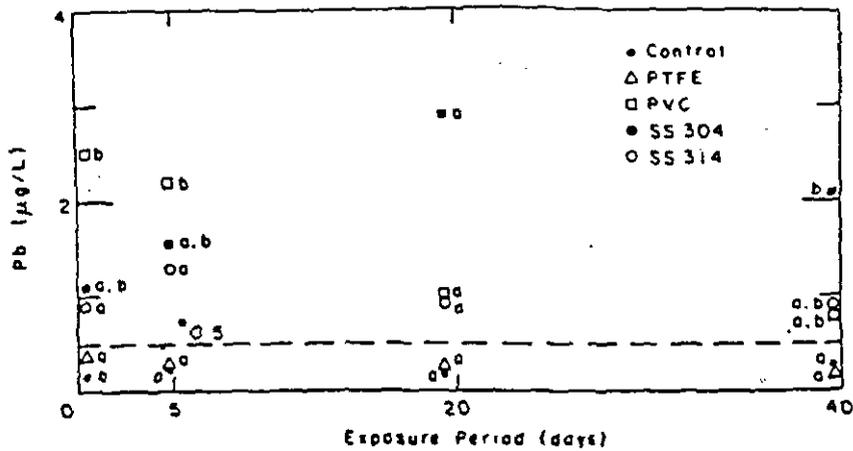


b. Cd.

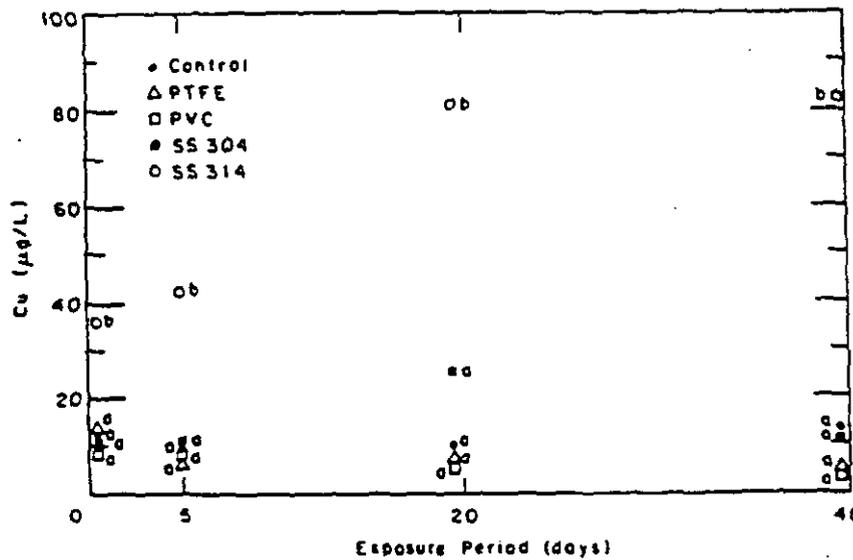


c. Cr.

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).

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.

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 was 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 ($X_1 = 9.8 \pm 0.0$, $n = 4$; $X_2 = 10.3 \pm 0.22$, $n = 5$; $X_3 = 16.0 \pm 0.0$, $n = 3$). Differences between adjacent mean concentrations were established by testing against the maximum variance determined for all of populations (i.e., $X_1 X_2$ and $X_2 X_3$).

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

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 ($\mu\text{g/L}$).

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.	0.19	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	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.

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 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 con-

tinually 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.

LITERATURE CITED

- Barcelona, M.J. and J.A. Helfrich (1986) Well construction and purging effects on ground-water samples. *Environmental Science and Technology*, 20: 1179-1184
- Chao, T.T., E.A. Jenne and L.M. Heppting (1968) Adsorption of traces of silver on sample containers. U.S. Geological Survey Professional Paper, 600-D, D13-D15.
- Coyne, R.V. and J.A. Collins (1972) Loss of mercury from water during storage. *Analytical Chemistry*, 44: 1093-1096.
- Cragin, J. (1979) Increased mercury contamination of distilled and natural waters samples caused by oxidizing preservatives. *Analytical Chimica Acta*, 110: 313-319.
- Eichholz, G.G., A.E. Nagel and R.B. Hughes (1965) Adsorption of ions in dilute aqueous solutions on

glass and plastic surfaces. *Analytical Chemistry*, 37: 867-867.

Federal Register (1984) Definition and procedure for the determination of the method detection limit. Code of Federal Regulations, Part 136, Appendix B, October 26.

Hatch, W.R. and W.L. Ott (1968) Determination of sub-microgram quantities of mercury by atomic absorption spectrophotometry. *Analytical Chemistry*, 40: 2085-2087.

Hewitt, A.D. (1989) Influence of well casing composition on trace metals in ground water. USA Cold Regions Research and Engineering Laboratory, Special Report 89-9.

Houghton, R.L. and M.E. Berger (1984) Effects of well-casing composition and sampling method on apparent quality of ground water. In: *The Fourth National Symposium and Exposition on Aquifer Restoration and Ground Water Monitoring*, Worthington, Ohio: National Water Well Association, p. 203-213.

Lo, J.M. and C.M. Wai (1975) Mercury loss from water during storage: mechanism and prevention. *Analytical Chemistry* 47: 1869-1870.

Miller, G.D. (1982) Uptake and release of lead, chromium and trace level volatile organics exposed to synthetic well casing. *Proceedings, Second Annual Symposium on Aquifer Restoration and Ground Water Monitoring*, Worthington, Ohio: National Water Well Association, p. 236-245.

Parker, L.V., A.D. Hewitt and T.F. Jenkins (1988) Influence of well-casing materials on chemical species in ground water. *Proceedings of the 13th Annual Environmental Quality R+D Symposium*. U.S. Army Toxic and Hazardous Material Agency, p. 450-461.

Parker, L.V., T.F. Jenkins and P.B. Black (in press) Suitability of four well-casing materials for monitoring trace level organics in ground water. USA Cold Regions Research and Engineering Laboratory, Special Report.

Perkin-Elmer (1981) Analytical methods for furnace atomic absorption spectroscopy. The Perkin-Elmer Corporation, Norwalk Ct, Part No. B010-0108.

Reynolds, G.W. and R.W. Gillham (1985) Absorption of halogenated organic compounds by polymer materials commonly used in ground-water monitors. *Proceedings of the Second Canadian/American Conference on Hydrogeology*, p. 125-132.

Subramanian, K.S., C.L. Chakrabarti, J.E. Sueiras, and I.S. Maines (1978) Preservation of some trace metals in samples of natural water. *Analytical Chemistry*, 50: 444-449.

Stumm, W. and J.J. Morgan (1970) *Aquatic Chemistry*. New York: Wiley-Interscience.

Tuncel, G. and O.Y. Atoman (1980) Design and evaluation of a new absorption cell for cold vapor mercury determination by atomic absorption spectrometry. *Atomic Spectroscopy*, 4: 126-128.

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.

REPORT DOCUMENTATION PAGE

Form Approved
OMB NO. 0704-0188
Exp. Date: Jun 30, 198

1a. REPORT SECURITY CLASSIFICATION Unclassified		18. RESTRICTIVE MARKINGS	
2a. SECURITY CLASSIFICATION AUTHORITY		3. DISTRIBUTION/AVAILABILITY OF REPORT	
2b. DECLASSIFICATION/DOWNGRADING SCHEDULE		Approved for public release; distribution is unlimited.	
4. PERFORMING ORGANIZATION REPORT NUMBER(S) Special Report 89- 32		5. MONITORING ORGANIZATION REPORT NUMBER(S) CETHA-TE-CR-89186	
6a. NAME OF PERFORMING ORGANIZATION U.S. Army Cold Regions Research and Engineering Laboratory	6b. OFFICE SYMBOL (If applicable) CECRL	7a. NAME OF MONITORING ORGANIZATION U.S. Army Toxic and Hazardous Materials Agency	
6c. ADDRESS (City, State, and ZIP Code) 72 Lyme Road Hanover, N.H. 03755-1290		7b. ADDRESS (City, State, and ZIP Code) Aberdeen Proving Ground Maryland	
8a. NAME OF FUNDING/SPONSORING ORGANIZATION	8b. OFFICE SYMBOL (If applicable)	9. PROCUREMENT INSTRUMENT IDENTIFICATION NUMBER	
8c. ADDRESS (City, State, and ZIP Code)		10. SOURCE OF FUNDING NUMBERS	
		PROGRAM ELEMENT NO.	PROJECT NO.
		TASK NO.	WORK UNIT ACCESSION NO.
11. TITLE (Include Security Classification) Leaching of Metal Pollutants From Four Well Casings Used for Ground-Water Monitoring			
12. PERSONAL AUTHOR(S) Hewitt, Alan D.			
13a. TYPE OF REPORT	13b. TIME COVERED FROM _____ TO _____	14. DATE OF REPORT (Year, Month, Day) September 1989	15. PAGE COUNT 15
16. SUPPLEMENTARY NOTATION			
17. COSATI CODES			18. SUBJECT TERMS (Continue on reverse if necessary and identify by block number)
FIELD	GROUP	SUB-GROUP	
			Ground-water monitoring PTFE Well casings Ground-water pollution PVC Metal pollutants Stainless steel
19. ABSTRACT (Continue on reverse if necessary and identify by block number) 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			
20. DISTRIBUTION/AVAILABILITY OF ABSTRACT <input checked="" type="checkbox"/> UNCLASSIFIED/UNLIMITED <input type="checkbox"/> SAME AS RPT. <input type="checkbox"/> DTIC USERS		21. ABSTRACT SECURITY CLASSIFICATION Unclassified	
22a. NAME OF RESPONSIBLE INDIVIDUAL Alan D. Hewitt		22b. TELEPHONE (Include Area Code) 603-646-4100	22c. OFFICE SYMBOL CECRL-RS

APPENDIX F

INFLUENCE OF WELL CASING COMPOSITION ON
TRACE METALS IN GROUND WATER

Influence of Well Casing Composition on Trace Metals in Ground Water

ALAN D. HEWITT

INTRODUCTION

Representative ground water sampling requires apparatus made from materials that are chemically inert with respect to the analytes of interest under a variety of environmental conditions. Several materials are being considered by the Environmental Protection Agency (EPA) for use as well casings for ground water monitoring. Four well casing materials now employed were examined in this study: polyvinylchloride (PVC), polytetrafluoroethylene (PTFE), stainless steel 304 (SS304) and stainless steel 316 (SS316).

A review of the literature showed that all of the well casing materials being tested sorb appreciable quantities of certain ion species (Eichholz et al. 1965, Miller 1982). Both natural and contaminated ground water vary widely in composition, so testing a single ground water solution is insufficient (Barcelona et al. 1983). In this study, well casings were exposed to varying concentrations of metals, pH and total organic carbon (TOC) in ground water. The metals selected (arsenic, cadmium, chromium and lead) have been cited by the EPA as priority pollutants.

In previous studies of the effects of well casing composition on inorganic constituents in ground water, it was observed that, in general, steel and stainless steel release metals (Houghton and Berger 1984, Barcelona and Helfrich 1986). The corrosion of stainless steel was also cited as causing a hydrous iron precipitate that could remove species from solution by sorption and coprecipitation (Barcelona and Helfrich 1986). The only laboratory study cited in the literature used deionized water as an aqueous solution, and samples were taken weekly (Miller 1982). Determinations of Cr and Pb showed that the latter was more susceptible to sorption and that PVC was a more active exchange surface than either polyethylene or polypropylene (Miller 1982).

MATERIALS AND METHODS

This study monitored metal concentrations in a variety of ground water solutions exposed to the four well casing materials. The results were analyzed for trends in sorption and leaching, the effects caused by the variables introduced into the solution, and the randomness of sample pair variability for the analytes tested. The degree to which the analytes were affected was used to rank the casing materials. The test procedure was to submerge samples of well casings in ground water solutions with different metal concentrations, TOC and pH. After 0.5, 4, 8, 24 and 72 hours, aliquots of the solutions were collected for analysis. An increase in the metal concentration would indicate that the well casing material was releasing metals into the solution, while a decrease in concentration would indicate that metals were being sorbed by the casing. Both situations are undesirable.

Test design

A full 2³ factorial screening experiment was used to test each of the four well casing materials. The variables selected were aqueous metal concentration, pH and TOC. The ground water was obtained from a domestic well system in Weathersfield, Vt. The two added concentrations of metals were the maximum level cited by the EPA for primary drinking water quality and one fifth that concentration (Table 1). Experiments were run both at the natural pH (7.8) and TOC (not determined) of the ground water and at modified levels (pH 5.8, natural TOC plus 5 mg/L humic acid, Aldrich). Acidity was raised by the addition of HCl (reagent grade, Baker), creating an initial pH of 5.8. The natural buffering capacity of the ground water allowed the pH to drift to 6.2 by the end of the 72-hour experiment.

Duplicate samples of each pipe material, along

Table 1. Aqueous metal concentrations added to ground water for testing well casing materials.

Metal	High spike* (mg/L)	Low spike (mg/L)
Arsenic	50	10
Cadmium	10	2
Chromium	50	10
Lead	50	10

* EPA interim primary drinking water standard (1983)

with two controls, were exposed to each set of conditions in the factorial matrix (Fig. 1). Controls consisted of a container and the sample solution without a pipe sample. Replication allows for the measurement of random error (consisting of sample pair variability and analytical precision) and thus confidence levels can be assigned to observed effects. A three-variable factorial has eight sets of variable combinations, creating an experiment with 80 samples (4 materials plus 1 control x 2 replicates x 8 conditions). To collect aliquots within 30 minutes, the factorial was blocked, that is, the solutions with high metal concentrations were sampled 2 hours before the samples with low concentrations.

The 80 aqueous solutions were prepared by first adding 97 mL of ground water to each sample container. The condition of high TOC was created by adding 1 mL of 500 mg/L humic acid; no humic acid was added for the low condition. The condition of high acidity (low pH) was created by adding 1 mL of 0.183 M HCl; no acid was added for the low condition. The condition of high metal concentration was created by adding 1 mL of 5 mg/L As, Cr

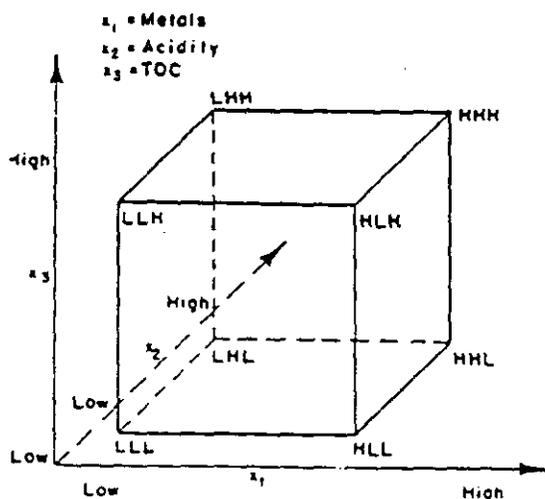


Figure 1. Diagram of factorial matrix.

and Pb and 1 mg/L Cd of mixed metal spike; the condition of low metal concentration was created by adding 1 mL of 1 mg/L As, Cr and Pb and 0.2 mg/L of Cd mixed metal spike. The volume in each sample container was increased to 100 mL with the addition of 1 or 2 mL of reagent-grade water (Milli Q, Millipore Corp.). Pipe sections were the last constituent added to the vessels containing the ground water solutions. All vessels were sealed in a room with no natural light at approximately 24°C. Five aliquots of 2.5 mL each were removed at < 0.5, 4, 8, 24 and 72 hours to produce 400 samples for analysis.

Materials

Polypropylene 125-mL jars (Model 6185-E37, Thomas Scientific) served as the sample containers. The jars were rinsed and soaked for 24 hours in reagent-grade water prior to use. The PVC and stainless steel well casings were manufactured by Johnson Well Screen, and the PTFE pipe by MIP Inc. All of these pipes were specifically made for ground water monitoring. Approximately 2-cm lengths of 5-cm inner-diameter well casings were cut for the experiment. The exact length of the rings varied with the wall thickness and diameter of the pipe lot tested, maintaining a constant surface area of 80 cm². Cut surfaces made up approximately 17% of the area for the PTFE and PVC well casings but less than 10% for the steels. Care was taken during the milling of the pipe sections to prevent contact with any foreign materials (i.e. grease, dirt, oil, solvents and excessive handling). Individual 2-cm lengths of pipe were rinsed with reagent-grade water and dried before use. No attempt was made to remove surface discoloration or ink present on the pipes due to the manufacturing process. Cleaning of ground monitoring pipes for field applications often consists only of rinsing with the cleanest water available before installation. The pipe sections were completely submerged in the 100 mL of ground water in each container, creating a pipe-surface-area/aqueous-volume ratio of 0.8 cm². This experimental design has the expected surface/aqueous-solution ratio for well casing at the bottom of the well; however, the ratio is lower than would be encountered at the well screen.

Sample vials (7.5 mL CPE, Nalgene) were cleaned by rinsing with reagent-grade water, soaked for 24 hours in 10% v/v G. Fredrick Smith (GFS) distilled nitric acid, rinsed with reagent-grade water, and air dried. Aliquots (2.5 mL) were transferred to these small vials with an Eppendorf syringe and then immediately acidified to below pH 1 with 50 µL of concentrated GFS distilled nitric acid. Studies have

shown that acidification below pH 1.5 is an effective method for preventing the loss of trace metal species from natural waters (Subramanian et al. 1978).

The experiment and all cleaning operations for the pipe materials and sample containers were performed in class 100 cleanrooms. Ground water, and ground water treated with the spiking solutions of humic acid and HCl, showed no detectable amounts of As, Cd, Cr and Pb at the sensitivity level used for the analysis.

Instrumentation

Metal concentrations were determined using a Perkin-Elmer (PE) model 703 atomic absorption spectrophotometer coupled with a PE model 2200 heated graphite atomizer. Sample injections of 10 and 20 mL were made with a PE AS-1 autosampler. Each sample aliquot was analyzed at least twice. Analyte concentrations were based on average peak height from strip-chart recordings. Calibration used standards of the same acid composition. Furnace programs and other instrument parameters followed recommended settings for aqueous metal analysis (Perkin-Elmer 1981). Arsenic determinations required deuterium arc background correction to eliminate interferences from the acid in the samples and standards. All metal determinations were completed within three weeks after collecting the sample aliquots. In addition to running aqueous standards with acid concentrations matched to the sample aliquots, EPA trace metal reference standards were analyzed independently to check day-to-day standard preparation.

Conductivity and pH measurements were made on all of the ground water solutions. Acidity was checked at the beginning and end of the 72-hour exposure period (Table 2). All pH measurements

were obtained with a semi-micro glass combination Ross model 81-03 electrode (Orion). The electrode was calibrated with both high and low ionic strength buffers prior to analysis of the ground water solutions. A Leeds and Northrup electrolytic conductivity bridge was used to measure conductivity.

Aqueous metal spikes

Aqueous metal solutions (As, Cd, Cr and Pb) for standards and sample spikes were made by diluting 1000-mg/L certified atomic absorption reference solutions (Fisher Scientific Corp.). Lead and cadmium were introduced into solution as metals, chromium as potassium dichromate, and arsenic as the trioxide for these reference standards. Mixed metal sample and control spiking solutions without acidification were prepared just prior to doping the ground water. Standards were prepared daily from a separate mixed metal stock. The standard stock solution and working standards were prepared in reagent-grade water (Milli Q) acidified with 2% v/v GFS HNO₃.

RESULTS AND DISCUSSION

Ground water was collected and stored in sealed glass bottles for 24 hours prior to making the factorial matrix solutions. From the time of collection to the end of the experiment, shifts in chemical equilibria undoubtedly occurred due to the new environment and added constituents. Once removed from its anoxic environment, ground water may undergo redox and precipitation reactions, such as oxidation of organics, conversion of sulfide to sulfate, and conversion of ferrous iron to ferric with subsequent precipitation of hydrous iron oxides (Stumm and Morgan 1970). Lowering the pH would shift the carbonate equilibrium from predominantly bicarbonate species toward carbon dioxide (Manahan 1972). Clearly these changes could alter the trace metal species distribution. None of these possible changes to the ground water composition were monitored quantitatively.

Qualitatively it was apparent iron was oxidizing on the metal pipes since surface rust developed on the stainless steel in 14 of 32 vessels containing this material (Table 3). In four of the vessels, sufficient oxidation occurred to form a hydrous iron oxide precipitate. Stainless steel 316 was more susceptible to surface oxidation (11 of 16) than SS304 (3 of 16). The hydrous iron oxide precipitate only developed with SS316 in low pH (high acidity) solutions. Rust formed on or near the cut surface of the SS304 pipe, but in the SS316 sections it was predominantly located near the weld on the exterior wall.

Table 2. Conductivity of ground water solutions and pH measurements at the beginning and end of the 72-hour exposure.

Matrix no.	Key*	Initial pH	Final pH	Conductance (mho)
1	HHH	5.6	6.1	2.46x10 ⁻¹
2	HHL	5.6	6.2	2.46x10 ⁻¹
3	LHH	5.6	6.3	2.42x10 ⁻¹
4	LHL	5.7	6.2	2.44x10 ⁻¹
5	HLH	7.7	7.7	1.94x10 ⁻¹
6	HLL	7.6	7.8	1.92x10 ⁻¹
7	LLH	7.9	7.9	1.98x10 ⁻¹
8	LLL	7.7	7.9	1.96x10 ⁻¹

*Key

Order: Metal, acidity, TOC

H = High

L = Low

Table 3. Physical state of stainless steel pipes after 72 hours of exposure to ground water solutions.

	HHH	HHL	LHH	LHL	HLH	HLL	LLH	LLL
SS304	O/E	O/E	—	—	—	—	—	—
SS304	—	—	O/E	—	—	—	—	—
SS316	F/E	—	—	F/W	O/W	O/W	O/W	O/E
SS316	F/W	F/E	—	—	O/W	O/W	—	O/W

* See Table 2

KEY: O - oxidation on surface

F - hydrous iron oxide precipitate in solution

E - oxidation near edge

W - oxidation on wall

Table 4. Design matrix for 2³ factorial run in duplicate (-1 represents the low level and +1 the high level of each variable).

	X ₁	X ₂	X ₃	X ₁ X ₂	X ₁ X ₃	X ₂ X ₃	X ₁ X ₂ X ₃	Y*
LLL	-1	-1	-1	+1	+1	+1	-1	24.3
LLL	-1	-1	-1	+1	+1	+1	-1	20.9
HLL	+1	-1	-1	-1	-1	+1	+1	27.8
HLL	+1	-1	-1	-1	-1	+1	+1	27.8
LHL	-1	+1	-1	-1	+1	-1	+1	56.1
LHL	-1	+1	-1	-1	+1	-1	+1	56.1
HHL	+1	+1	-1	+1	-1	-1	-1	61.6
HHL	+1	+1	-1	+1	-1	-1	-1	49.6
LLH	-1	-1	+1	+1	-1	-1	+1	54.9
LLH	-1	-1	+1	+1	-1	-1	+1	53.1
HLH	+1	-1	+1	-1	+1	-1	-1	50.2
HLH	+1	-1	+1	-1	+1	-1	-1	50.2
LHH	-1	+1	+1	-1	-1	+1	-1	96.3
LHH	-1	+1	+1	-1	-1	+1	-1	85.2
HHH	+1	+1	+1	+1	+1	+1	+1	73.7
HHH	+1	+1	+1	+1	+1	+1	+1	73.7

* Example of normalized responses, Pb - SS304 - 24 hours.

KEY: X₁ = Metal concentration

X₂ = Acidity

X₃ = TOC

Table 5. ANOVA for Pb at 24 hours with SS304 well casing. The average normalized response was 53.8 and the standard deviation was 4.2.

Factor	Effects	df	MS	F	Sig.*
X ₁ (Metal conc.)	-4.04	1		3.70	NS
X ₂ (Acidity)	30.4	1		209	VS
X ₃ (TOC)	26.6	1		161	VS
X ₁ X ₂	-4.74	1		5.09	NS
X ₁ X ₃	-6.39	1		9.26	S
X ₂ X ₃	-0.26	1		0.02	NS
X ₁ X ₂ X ₃	-1.89	1		0.81	NS
ERROR	39.7	8	17.6		

*S = significant effect at a significance level of 5%

VS = very significant, order of magnitude greater than F value [F_{0.99}(df 1,8) = 5.32] at a significance level of 5%

NS = not a significant effect at 5% level.

Aqueous metal concentrations for the vessels containing well casing pipe sections were compared to the average amount determined in the two controls within each matrix set and multiplied by 100. Handling the data in this fashion normalized the metal concentrations observed for the two levels tested, allowing an analysis of variance (ANOVA) on the entire data set. Consequently the expected value in every case would be 100 if the pipe casing exerted no influence.

An ANOVA was performed on a matrix of eight duplicates for each time interval (Table 4). A representative ANOVA table containing all computed effects, mean squares and F-ratios appears in Table 5. The complete array of ANOVA tables is given in Appendix A. Typically interaction effects are small compared to main effects. Consequently the data summaries that follow will be based exclusively on main effects and trends in those effects as a function of time. The average response is based on all 16 measurements in a set. Each effect is the average of the eight responses when a variable is at the high setting minus the average of the eight responses at the low settings. The MS error is simply the sum of squares for error between the replicates divided by eight degrees of freedom (df) (one df from each pair of values). The standard deviation is the square root of the MS and represents the random error associated with an individual experiment.

Table 6 summarizes the effects noted in ANOVA for the aqueous metal concentrations in contact with the four pipe materials. Since sample aliquots were taken at five separate time intervals, effects that followed a pattern provided support for the analysis of variance conclusions. Effects were established at the 95% confidence level. The effects shown in Table 6 represent the influence that the matrix variables have on the analyte concentration when a single constituent changes from the low to the high level. The last two columns in this table show the random error variance associated with the duplicate determinations. The standard deviation was obtained by taking the square root of the mean sums of squares for the error. The larger this term becomes, the less likely that the analysis will be able to distinguish real effects from random variations. Standard deviations greater than 10% depict large random discrepancies between sample pairs.

Arsenic

No consistent pattern of effects was observed by the ANOVA on the aqueous arsenic concentrations. In general PTFE and PVC showed no change in concentration throughout the 72-hour exposure period. The lack of influence may be due to the

Table 6. Summary of significant ($\alpha = 0.05$) main effects and the random error of measurement for the four metals. The standard deviation given here is the square root of the MS error and represents the random error associated with an individual measurement.

Time (hr)	Pipe	Average response	Metal X_1	Acidity X_2	TOC X_3	MS error	Standard deviation
Arsenic							
0.5	PVC	99.1		-2.6		14.8	3.8
	PTFE	99.9				24.9	5.0
	SS304	99.7				33.4	5.7
	SS316	99.4				16.0	4.0
4.0	PVC	102	+2.8			20.4	4.5
	PTFE	99.3				27.4	5.2
	SS304	97.8				40.4	6.3
	SS316	94.5				36.6	6.0
8.0	PVC	100				45.1	4.5
	PTFE	101		-4.3		96.3	9.8
	SS304	96.2				32.3	5.7
	SS316	94.5			-4.5	46.0	6.8
24.0	PVC	99.4				40.8	6.4
	PTFE	99.2			-3.8	28.9	5.4
	SS304	89.4				26.3	5.1
	SS316	85.3				63.7	8.0
72.0	PVC	103				21.4	4.6
	PTFE	102				20.2	4.5
	SS304	89.1		-5.9		70.9	8.4
	SS316	87.4				68.4	8.3
Cadmium							
0.5	PVC	101			-2.6	6.06	2.5
	PTFE	101		-1.2	-1.2	1.28	1.1
	SS304	106	-3.3		-2.5	12.9	3.6
	SS316	104	-2.8		-2.7	4.47	2.1
4.0	PVC	113	-7.1	+9.2		13.9	3.7
	PTFE	103				29.3	5.4
	SS304	117		+14.5		217	15
	SS316	124				2428	49
8.0	PVC	115	-8.5	+14.2	-2.4	14.0	3.7
	PTFE	103	+1.6	+1.8		2.42	1.6
	SS304	116	-9.4	+21.6		207	14
	SS316	130	-29.4	+34.8		2207	47
24.0	PVC	116	-12.5	+14.5	-4.4	31.4	5.6
	PTFE	103	-1.6			2.90	1.7
	SS304	112	-12.3	+24.6		147	12
	SS316	136				4614	68
72.0	PVC	114	-14.5	+14.9		24.5	4.9
	PTFE	102	-1.9			4.86	2.2
	SS304	103	-14.7	+22.4		186	14
	SS316	1254	-38.4	+44.2		4419	66
Chromium							
0.5	PVC	101				3.34	1.8
	PTFE	101				0.51	0.7
	SS304	101				2.61	1.6
	SS316	102				2.12	1.5

Table 6 (cont'd). Summary of significant ($\alpha=0.05$) main effects and the random error of measurement for the four metals. The standard deviation given here is the square root of the MS error and represents the random error associated with an individual measurement.

Time (hr)	Pipe	Average response	Metal X_1	Acidity X_2	TOC X_3	MS error	Standard deviation
4.0	PVC	99.9				1.71	1.3
	PTFE	101	+0.75		+0.75	1.16	1.1
	SS304	95.7		-3.55		13.8	3.7
	SS316	92.1	-3.2	-5.1	+5.0	27.6	5.2
8.0	PVC	100				1.96	1.4
	PTFE	98.9	-1.6			3.64	1.9
	SS304	97.2				251	16
	SS316	87.2	-7.0	-7.8	+5.8	102	10
24.0	PVC	100		-1.4		2.57	1.6
	PTFE	101	+1.9			5.67	2.4
	SS304	103				1381	37
	SS316	85.5		-11.2	+8.7	118	11
72.0	PVC	101	+1.3			3.35	1.8
	PTFE	100	+1.8			1.62	1.3
	SS304	103				1795	42
	SS316	83.6		-11.3	+9.4	98.6	9.9
0.5	PVC	99.9				0.82	0.9
	PTFE	100		Lead -0.55		6.53	2.6
	SS304	102				0.64	0.8
	SS316	101				6.49	2.5
4.0	PVC	88.9		-2.2	+2.9	9.19	3.0
	PTFE	97.4	+1.6			3.69	1.9
	SS304	78.4	-2.2	+6.8	+6.4	12.1	3.5
	SS316	80.3	-5.5	+7.2	+7.4	59.7	7.7
8.0	PVC	89.3			+3.7	12.4	3.5
	PTFE	98.5		+1.9		9.98	3.2
	SS304	69.9	-3.4	+12.1	+9.0	9.61	3.1
	SS316	80.4	-7.7	+10.8	+9.7	100	10
24.0	PVC	80.8		-5.4	+6.9	25.7	5.1
	PTFE	95.1			+2.4	16.2	4.0
	SS304	53.8		+15.2	+13.3	17.6	4.2
	SS316	79.3	-10.7		+14.8	351	19
72.0	PVC	74.3		-5.9	+9.6	41.4	6.4
	PTFE	89.9	+3.0	-2.2	+4.1	11.5	3.4
	SS304	45.2		+15.7	+12.4	37.3	6.1
	SS316	72.0		+16.3	+18.0	2936	17

preferred state of this metal in aqueous solution. Arsenic exists as arsenates ($H_2AsO_4^-$, $HAsO_4^{2-}$ and $HAAsO_4^{2-}$) under oxidizing conditions and as arsenites ($H_2AsO_3^-$, $HAsO_3^{2-}$ and $HAAsO_3^{2-}$) under moderate reducing conditions in natural waters (Fowler et al. 1979). These partly dissociated species are negatively charged and are not likely to interact with

surfaces such as plastics, which are better known for their cation exchange capabilities (Masse and Maessen 1981). The vessels containing both types of stainless steel showed, on the average, about a 10% decrease in aqueous arsenic relative to the controls for the two longest exposures.

Cadmium

On the average, aqueous cadmium concentration increased relative to the controls for all pipe sections except PTFE, which showed no consistent effects and no change in aqueous cadmium concentration. PVC in general showed a constant 15% average enrichment in cadmium after 4 hours of exposure. This enrichment was affected negatively by the concentration of the initial metal spike (the percentage of cadmium enrichment was less at 10 mg/L than at 2 mg/L; however, the absolute amount of Cd contributed by the pipe was approximately 0.5 mg/L in both solutions) and positively by hydrogen ion (cadmium concentration increased as hydrogen ion concentration increased). Increasing TOC showed a very small and inconsistent negative effect. Stainless steel 304 showed an average 15% increase in cadmium for the 4-, 8- and 24-hour aliquots; however, it returned to the same level as the control for the last collection. This material was also affected by the concentration of the metal spike (negative) and hydrogen ion (positive).

A similar pattern was observed for SS316 except that it did not return all the way to the control level after 72 hours and the effects were less consistent due to large random errors. For all the pipe materials that showed enrichment of cadmium, a maximum was reached before 72 hours of exposure, and the effect was most prominent at low metal concentration and high hydrogen ion concentration (low pH). This suggests that the release of cadmium to solution from PVC, SS304 and SS316 was small and that some sorption occurred with time. Cadmium may have been employed as a UV stabilizer during the manufacturing of PVC (Wilson et al. 1982), and it may exist as an impurity in stainless steel.

The standard deviation for SS304 was greater than 10%, showing large random discrepancies between sample duplicates. Random variation was a dominant feature with SS316, where after 4 hours of exposure the standard deviation was greater than 47%. In sample duplicates the presence of a single SS316 pipe section with surface oxidation was the major source of variance. In contrast the standard deviation for both PVC and PTFE were consistently below 6%.

Chromium

PTFE, PVC and SS304 showed no consistent effects and on the average no change in aqueous chromium relative to the controls. The lack of interaction with plastic may be due to the salt of the metal employed to make the aqueous solutions. Potassium dichromate in solution predominantly exists as dichromate and chromate ($\text{Cr}_2\text{O}_7^{2-}$, CrO_4^{2-}). Negatively charged species are not as likely to exchange with plastic surfaces,

Stainless steel 316 showed a 16% average reduction in chromium after 72 hours of exposure. This response was affected by the concentration of hydrogen ion (negative) and TOC (positive). The standard deviation reached 10% after 8 hours for SS316. At the lower pH there was increased surface oxidation with regard to SS316, which may have created sorption sites and consequently increased random variation. Humic species increased the stability of aqueous chromium, perhaps by acting as a complexing agent. The standard deviation for SS304 steadily increased from 3.7% at 4 hours to 42% by 72 hours. Again the pairs with the greatest discrepancies had one member with surface oxidation. In general, surface oxidation appeared to be a chromium source with SS304 and a sink with SS316.

Lead

All four pipe materials showed a continuous loss of aqueous lead relative to the controls with time (Fig. 2). This metal was by far the most active species in terms of sorbing onto the well casing surfaces. PTFE had the least-active surface, followed by PVC, SS316 and SS304. The average losses ranged from 10% for PTFE to 55% for SS304 after 72 hours of exposure. No effects were either large or consistent with respect to solutions in contact with PTFE. For SS316, there were large effects that were significant when they were consistent but that were sometimes not statistically significant due to large random error. The trend was toward negative effects for metal concentration and positive effects for hydrogen ion and TOC concentration. Aqueous concentrations of lead in contact with both PVC and SS304 were consistently affected by matrix variables. TOC

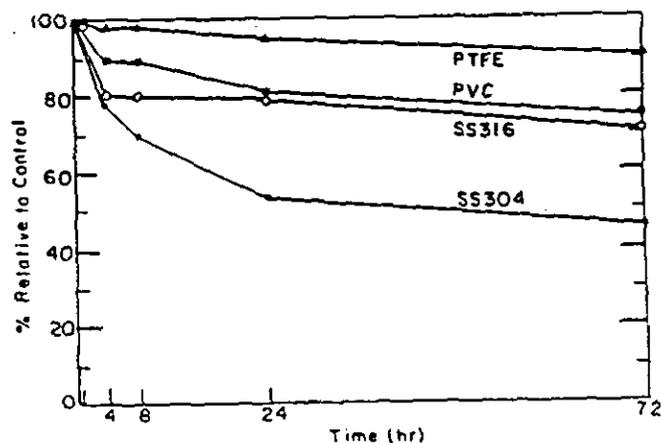


Figure 2. Relative loss of lead in solutions exposed to well casings.

concentration showed a positive influence on aqueous lead concentrations in the presence of PVC and SS304 pipe sections. Stainless steel 304 was also affected positively by the concentration of hydrogen ion. Humic material apparently again acted as a complexing agent, making lead more stable in solution. Lower pH also increased the ability of lead to remain in solution for the vessels containing SS304. Acidity was increased in the ground water solutions by the addition of HCl; with this species in solution, hydrogen ions may compete for sorption sites.

CONCLUSION

In laboratory testing, SS316 and SS304 were found susceptible to oxidation at locations near cuts and welds in ground water solutions. Surface oxidation, presumably by galvanic action, provides active sites for sorption and also releases impurities and major constituents. This random source of error most likely depends on the specific production batch and supplier of the well casing pipe. Installation is also an important variable since any abrasions would readily act as oxidation sites. Stainless steel sorbed both anions and cations faster than PVC or PTFE. Therefore, stainless steels are prone to imposing specific signatures on ground water and are not suitable where trace metal determinations are planned.

PVC was a source for low levels of cadmium and it acts as a moderately active surface for the sorption of lead. Both of these processes were affected by the ground water composition. Lead was affected positively by increasing the TOC concentration, and the effect on cadmium depended on the analyte concentration. Regardless, PVC should be considered as a well casing candidate based on economics (PVC is one sixth the price of PTFE) and the strong possibility that the effects cited previously are of less concern at well recharge rates of less than 24 hours. PTFE showed no marked interactions with any of the metals tested. This material is superior to the others because it did not influence trace inorganics in ground water of various compositions.

LITERATURE CITED

Barcelona, M.J. and J.A. Helfrich (1986) Well construction and purging effects on ground-water samples. *Environmental Science and Technology*, 20: 1179-1184.

Barcelona, M.J., J.P. Gibb, R.A. Miller (1983) A guide to the selection on materials for monitoring well construction and ground-water sampling. Illinois State Water Survey, Champaign, Illinois, Contract Report 374, EPA/600/S2-84/024.

Fowler, B.A., R.S. Braman, K.Y. Chen, P.A. Gilderhus, P.V. Hodson, M. Katz, J.C. Kent, L.L. King, J.M. Mckim, L.W. Nicholson, D.R.M. Passino and W.R. Penrose (1979) Arsenic. In: A review of the EPA Red Book: Quality criteria for water (R.V. Thurston, R.C. Rosso, C.M. Fetteroff, Jr., T.A. Edsall and Y.M. Baker, Jr., Ed.). Water Quality Section, American Fisheries Society, Bethesda, Maryland, p. 19-33.

Eichholz, G.G., A.E. Nagel and R.B. Hughes (1965) Adsorption of ions in dilute aqueous solutions on glass and plastic surfaces. *Analytical Chemistry*, 37: 863-867.

Houghton, R.L. and M.E. Berger (1984) Effects of well-casing composition and sampling method on apparent quality of ground water. *The Fourth National Symposium and Exposition on Aquifer Restoration and Ground Water Monitoring*, National Water Well Association, Worthington, Ohio, p. 203-213.

Manahan, S.E. (1972) *Environmental Chemistry*. Boston: Willard Grant Press.

Massee, R. and F.J.M.J. Maessen (1981) Losses of silver, arsenic, cadmium, selenium and zinc traces from distilled water and artificial sea-water by sorption of various container surfaces. *Analytica Chimica Acta*, 127: 187-193.

Miller, G.D. (1982) Uptake and release of lead, chromium and trace level volatile organics exposed to synthetic well casing. *Proceedings, Second Annual Symposium on Aquifer Restoration and Ground Water Monitoring*, National Water Well Association, Worthington, Ohio, p. 236-245.

Perkin-Elmer (1981) Analytical methods for furnace atomic absorption spectroscopy. The Perkin-Elmer Corporation, Norwalk Ct, Part No. B010-0108.

Stumm, W. and J.J. Morgan (1970) *Aquatic Chemistry*. New York: Wiley-Interscience.

Subramanian, K.S., C.L. Chakrabarti, J.E. Sueiras, and I.S. Maines (1978) Preservation of some trace metals in samples of natural water. *Analytical Chemistry*, 50: 444-449.

Wilson, D.C., P.J. Young, B.C. Hudson and G. Baldwin (1982) Leaching of cadmium from pigmented plastics in a landfill site. *Environmental Science and Technology*, 16: 560-566

APPENDIX A: ANOVA TABLES FOR ALL RESULTS
The significance levels are the same as in Table 5

Table A1. ANOVA for As at 0.5 hours with PVC well casing. The average normalized response was 99.1 and the standard deviation was 3.8.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-3.26	1		2.87	NS
X ₁ (Acidity)	-5.14	1		7.11	S
X ₂ (TOC)	3.49	1		3.27	NS
X ₁ X ₂	-3.01	1		2.45	NS
X ₁ X ₃	1.26	1		0.43	NS
X ₂ X ₃	-3.21	1		2.78	NS
X ₁ X ₂ X ₃	-1.74	1		0.81	NS
Error		8	14.8		

Table A2. ANOVA for As at 4 hours with PVC well casing. The average normalized response was 102 and the standard deviation was 4.5.

Factor	Effects	df	MS	F	Sig
X1 (Metal conc)	5.70	1		6.35	S
X2 (Acidity)	-3.85	1		2.90	NS
X ₃ (TOC)	-0.82	1		0.13	NS
X ₁ X ₂	0.90	1		0.16	NS
X ₁ X ₃	5.78	1		6.52	S
X ₂ X ₃	-7.22	1		10.2	S
X ₁ X ₂ X ₃	-2.87	1		1.62	NS
Error		8	20.5		

Table A3. ANOVA for As at 8 hours with PVC well casing. The average normalized response was 100 and the standard deviation was 6.7.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	2.18	1		0.42	NS
X ₂ (Acidity)	-1.32	1		0.16	NS
X ₃ (TOC)	-4.02	1		1.44	NS
X ₁ X ₂	-0.22	1		0.00	NS
X ₁ X ₃	-1.02	1		0.09	NS
X ₂ X ₃	-0.08	1		0.00	NS
X ₁ X ₂ X ₃	-5.22	1		2.42	NS
Error		8	45.1		

Table A4. ANOVA for As at 24 hours with PVC well casing. The average normalized response was 99.4 and the standard deviation was 6.4.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-1.75	1		0.30	NS
X ₂ (Acidity)	0.00	1		0.00	NS
X ₃ (TOC)	0.55	1		0.03	NS
X ₁ X ₂	1.88	1		0.34	NS
X ₁ X ₃	-4.42	1		1.92	NS
X ₂ X ₃	1.88	1		0.34	NS
X ₁ X ₂ X ₃	-0.85	1		0.07	NS
Error		8	40.8		

Table A5. ANOVA for As at 72 hours with PVC well casing. The average normalized response was 103 and the standard deviation was 4.6.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-4.84	1		4.37	NS
X ₁ (Acidity)	-3.96	1		2.93	NS
X ₂ (TOC)	-2.94	1		1.61	NS
X ₁ X ₂	-1.09	1		0.22	NS
X ₁ X ₃	2.74	1		1.40	NS
X ₂ X ₃	0.46	1		0.04	NS
X ₁ X ₂ X ₃	0.54	1		0.05	NS
Error		8	21.4		

Table A6. ANOVA for As at 0.5 hours with PTFE well casing. The average normalized response was 99.9 and the standard deviation was 5.0.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-2.30	1		0.85	NS
X ₂ (Acidity)	0.70	1		0.08	NS
X ₃ (TOC)	-0.65	1		0.07	NS
X ₁ X ₂	-2.32	1		0.87	NS
X ₁ X ₃	2.28	1		0.83	NS
X ₂ X ₃	2.22	1		0.80	NS
X ₁ X ₂ X ₃	-1.40	1		3.32	NS
Error		8	24.9		

Table A7. ANOVA for As at 4 hours with PTFE well casing. The average normalized response was 99.3 and the standard deviation was 5.2.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	5.20	1		3.90	NS
X ₂ (Acidity)	4.50	1		2.92	NS
X ₃ (TOC)	-2.40	1		0.83	NS
X ₁ X ₂	3.38	1		1.64	NS
X ₁ X ₃	1.88	1		0.51	NS
X ₂ X ₃	-0.67	1		0.07	NS
X ₁ X ₂ X ₃	0.45	1		0.03	NS
Error		8	27.7		

Table A8. ANOVA for As at 8 hours with PTFE well casing. The average normalized response was 101 and the standard deviation was 9.8.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	2.45	1		1.99	NS
X ₂ (Acidity)	0.45	1		0.07	NS
X ₃ (TOC)	-2.62	1		2.29	NS
X ₁ X ₂	0.08	1		0.00	NS
X ₁ X ₃	-7.70	1		19.7	S
X ₂ X ₃	-5.26	1		9.16	S
X ₁ X ₂ X ₃	-5.32	1		9.42	S
Error		8	12.0		

Table A9. ANOVA for As at 24 hours with PTFE well casing. The average normalized response was 99.2 and the standard deviation was 5.4.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	2.26	1		0.71	NS
X ₂ (Acidity)	-1.29	1		0.23	NS
X ₃ (TOC)	-7.71	1		8.21	S
X ₁ X ₂	2.73	1		1.04	NS
X ₁ X ₃	-3.79	1		1.98	NS
X ₂ X ₃	0.81	1		0.09	NS
X ₁ X ₂ X ₃	-1.36	1		0.26	NS
Error		8	29.0		

Table A10. ANOVA for As at 72 hours with PTFE well casing. The average normalized response was 102 and the standard deviation was 4.5.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-4.56	1		4.11	NS
X ₂ (Acidity)	-1.56	1		0.48	NS
X ₃ (TOC)	-1.56	1		0.48	NS
X ₁ X ₂	6.26	1		7.75	S
X ₁ X ₃	3.11	1		1.92	NS
X ₂ X ₃	5.71	1		6.45	S
X ₁ X ₂ X ₃	-2.11	1		0.88	NS
Error		8	20.2		

Table A11. ANOVA for As at 0.5 hours with SS304 well casing. The average normalized response was 99.7 and the standard deviation was 5.7.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	2.62	1		0.82	NS
X ₂ (Acidity)	-2.40	1		0.69	NS
X ₃ (TOC)	1.98	1		0.47	NS
X ₁ X ₂	-2.05	1		0.50	NS
X ₁ X ₃	1.62	1		0.32	NS
X ₂ X ₃	-1.85	1		0.41	NS
X ₁ X ₂ X ₃	0.70	1		0.06	NS
Error		8	33.4		

Table A12. ANOVA for As at 4 hours with SS304 well casing. The average normalized response was 97.8 and the standard deviation was 6.3.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-2.12	1		0.44	NS
X ₂ (Acidity)	-1.95	1		0.38	NS
X ₃ (TOC)	-2.70	1		0.72	NS
X ₁ X ₂	-1.25	1		0.15	NS
X ₁ X ₃	2.65	1		0.70	NS
X ₂ X ₃	-4.72	1		2.21	NS
X ₁ X ₂ X ₃	-0.88	1		0.08	NS
Error		8	40.4		

Table A13. ANOVA for As at 8 hours with SS304 well casing. The average normalized response was 96.2 and the standard deviation was 5.7.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-5.09	1		3.20	NS
X ₂ (Acidity)	-8.59	1		9.12	S
X ₃ (TOC)	-3.74	1		1.73	NS
X ₁ X ₂	-4.46	1		2.46	NS
X ₁ X ₃	2.79	1		0.96	NS
X ₂ X ₃	2.54	1		0.80	NS
X ₁ X ₂ X ₃	-10.5	1		13.7	S
Error		8	32.3		

Table A14. ANOVA for As at 24 hours with SS304 well casing. The average normalized response was 89.4 and the standard deviation was 5.1.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-0.56	1		0.05	NS
X ₂ (Acidity)	-1.96	1		0.59	NS
X ₃ (TOC)	-2.89	1		1.27	NS
X ₁ X ₂	-3.09	1		1.45	NS
X ₁ X ₃	4.04	1		2.48	NS
X ₂ X ₃	7.29	1		8.09	S
X ₁ X ₂ X ₃	-1.39	1		0.29	NS
Error		8	26.3		

Table A15. ANOVA for As at 72 hours with SS304 well casing. The average normalized response was 89.1 and the standard deviation was 8.4.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-8.80	1		4.37	NS
X ₂ (Acidity)	-11.7	1		7.75	S
X ₃ (TOC)	-4.15	1		0.97	NS
X ₁ X ₂	-13.6	1		10.5	S
X ₁ X ₃	23.0	1		30.0	S
X ₂ X ₃	8.02	1		3.63	NS
X ₁ X ₂ X ₃	1.62	1		0.14	NS
Error		8	70.9		

Table A16. ANOVA for As at 0.5 hours with SS316 well casing. The average normalized response was 99.4 and the standard deviation was 4.0.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-4.12	1		4.25	NS
X ₂ (Acidity)	-2.75	1		1.89	NS
X ₃ (TOC)	2.20	1		1.21	NS
X ₁ X ₂	2.30	1		1.32	NS
X ₁ X ₃	1.45	1		0.52	NS
X ₂ X ₃	-2.18	1		1.18	NS
X ₁ X ₂ X ₃	-0.68	1		0.11	NS
Error		8	16.0		

Table A17. ANOVA for As at 4 hours with SS316 well casing. The average normalized response was 94.5 and the standard deviation was 6.0.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	3.52	1		1.34	NS
X ₂ (Acidity)	3.80	1		1.58	NS
X ₃ (TOC)	-2.35	1		0.62	NS
X ₁ X ₂	-0.75	1		0.06	NS
X ₁ X ₃	7.60	1		6.32	S
X ₂ X ₃	-2.62	1		0.75	NS
X ₁ X ₂ X ₃	-2.62	1		0.75	NS
Error		8	36.6		

Table A18. ANOVA for As at 8 hours with SS316 well casing. The average normalized response was 94.5 and the standard deviation was 6.8.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	2.55	1		0.57	NS
X ₂ (Acidity)	-1.35	1		0.16	NS
X ₃ (TOC)	-9.10	1		7.20	S
X ₁ X ₂	-0.82	1		0.06	NS
X ₁ X ₃	8.28	1		5.95	S
X ₂ X ₃	-7.02	1		4.29	NS
X ₁ X ₂ X ₃	-7.55	1		4.95	NS
Error		8	46.0		

Table A19. ANOVA for As at 24 hours with SS316 well casing. The average normalized response was 85.3 and the standard deviation was 8.0.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	0.05	1		0.00	NS
X ₂ (Acidity)	1.58	1		0.16	NS
X ₃ (TOC)	2.55	1		0.41	NS
X ₁ X ₂	-0.60	1		0.02	NS
X ₁ X ₃	5.32	1		1.78	NS
X ₂ X ₃	1.39	1		0.11	S
X ₁ X ₂ X ₃	1.52	1		0.15	NS
Error		8	63.7		

Table A20. ANOVA for As at 72 hours with SS316 well casing. The average normalized response was 87.4 and the standard deviation was 8.3.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	3.69	1		0.79	NS
X ₂ (Acidity)	7.74	1		3.50	NS
X ₃ (TOC)	-7.63	1		3.41	NS
X ₁ X ₂	-8.51	1		4.23	NS
X ₁ X ₃	14.6	1		12.4	S
X ₂ X ₃	-0.94	1		0.05	NS
X ₁ X ₂ X ₃	5.66	1		1.87	NS
Error		8	68.4		

Table A21. ANOVA for Cd at 0.5 hours with PVC well casing. The average normalized response was 101 and the standard deviation was 2.5.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-2.49	1		4.08	NS
X ₂ (Acidity)	-2.71	1		4.85	NS
X ₃ (TOC)	-5.13	1		17.4	S
X ₁ X ₂	-1.19	1		0.93	NS
X ₁ X ₃	-0.91	1		0.54	NS
X ₂ X ₃	0.41	1		0.11	NS
X ₁ X ₂ X ₃	-1.11	1		0.82	NS
Error		8	6.06		

Table A22. ANOVA for Cd at 4 hours with PVC well casing. The average normalized response was 113 and the standard deviation was 3.7.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-14.2	1		58.1	VS
X ₂ (Acidity)	18.4	1		97.1	VS
X ₃ (TOC)	-1.90	1		1.04	NS
X ₁ X ₂	-9.70	1		27.1	S
X ₁ X ₃	3.95	1		4.50	NS
X ₂ X ₃	-0.35	1		0.04	NS
X ₁ X ₂ X ₃	-0.10	1		0.00	NS
Error		8	13.9		

Table A23. ANOVA for Cd at 8 hours with PVC well casing. The average normalized response was 115 and the standard deviation was 3.7.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-19.0	1		103	VS
X ₂ (Acidity)	28.4	1		230	VS
X ₃ (TOC)	-4.85	1		6.70	S
X ₁ X ₂	-12.6	1		45.4	S
X ₁ X ₃	6.70	1		12.8	S
X ₂ X ₃	-4.48	1		5.70	S
X ₁ X ₂ X ₃	4.42	1		5.57	S
Error		8	14.1		

Table A24. ANOVA for Cd at 24 hours with PVC well casing. The average normalized response was 116 and the standard deviation was 5.6.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-25.0	1		79.8	VS
X ₂ (Acidity)	29.0	1		107	VS
X ₃ (TOC)	-8.82	1		9.93	S
X ₁ X ₂	-17.0	1		36.9	S
X ₁ X ₃	9.52	1		11.6	S
X ₂ X ₃	-10.6	1		14.4	S
X ₁ X ₂ X ₃	8.02	1		8.21	S
Error		8	31.4		

Table A25. ANOVA for Cd at 72 hours with PVC well casing. The average normalized response was 114 and the standard deviation was 4.9.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-29.1	1	137		VS
X ₂ (Acidity)	29.8	1	144		VS
X ₃ (TOC)	0.39	1	0.02		NS
X ₁ X ₂	-20.0	1	65.2		VS
X ₁ X ₃	12.3	1	24.7		S
X ₂ X ₃	-7.86	1	10.1		S
X ₁ X ₂ X ₃	7.21	1	8.47		S
Error		8	24.6		

Table A26. ANOVA for Cd at 0.5 hours with PTFE well casing. The average normalized response was 101 and the standard deviation was 1.1.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	0.46	1	0.67		NS
X ₂ (Acidity)	-2.43	1	18.5		S
X ₃ (TOC)	-2.44	1	18.5		S
X ₁ X ₂	-1.04	1	3.36		NS
X ₁ X ₃	1.66	1	8.62		S
X ₂ X ₃	0.31	1	0.30		NS
X ₁ X ₂ X ₃	1.51	1	7.14		S
Error		8	1.28		

Table A27. ANOVA for Cd at 4 hours with PTFE well casing. The average normalized response was 103 and the standard deviation was 5.4.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-1.74	1	0.41		NS
X ₂ (Acidity)	0.91	1	0.11		NS
X ₃ (TOC)	-0.04	1	0.00		NS
X ₁ X ₂	0.96	1	0.12		NS
X ₁ X ₃	2.66	1	0.97		NS
X ₂ X ₃	1.56	1	0.33		NS
X ₁ X ₂ X ₃	-4.09	1	2.28		NS
Error		8	29.3		

Table A28. ANOVA for Cd at 8 hours with PTFE well casing. The average normalized response was 103 and the standard deviation was 1.6.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	3.15	1	16.4		S
X ₂ (Acidity)	3.65	1	22.1		S
X ₃ (TOC)	-0.75	1	0.93		NS
X ₁ X ₂	3.70	1	22.7		S
X ₁ X ₃	-0.70	1	0.81		NS
X ₂ X ₃	0.30	1	0.15		NS
X ₁ X ₂ X ₃	-2.45	1	9.94		S
Error		8	2.42		

Table A29. ANOVA for Cd at 24 hours with PTFE well casing. The average normalized response was 103 and the standard deviation was 1.7.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-0.34	1	0.15		NS
X ₂ (Acidity)	1.86	1	4.79		NS
X ₃ (TOC)	-3.16	1	13.8		S
X ₁ X ₂	4.56	1	28.8		S
X ₁ X ₃	1.74	1	4.17		NS
X ₂ X ₃	-5.86	1	47.5		S
X ₁ X ₂ X ₃	2.14	1	6.31		S
Error		8	2.90		

Table A30. ANOVA for Cd at 72 hours with PTFE well casing. The average normalized response was 102 and the standard deviation was 2.2.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-3.92	1	12.7		S
X ₂ (Acidity)	0.90	1	0.67		NS
X ₃ (TOC)	0.78	1	0.49		NS
X ₁ X ₂	-1.65	1	2.24		NS
X ₁ X ₃	-2.18	1	3.89		NS
X ₂ X ₃	1.75	1	2.52		NS
X ₁ X ₂ X ₃	-0.90	1	0.67		NS
Error		8	4.86		

Table A31. ANOVA for Cd at 0.5 hours with SS304 well casing. The average normalized response was 106 and the standard deviation was 3.6.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-6.68	1	13.8		S
X ₂ (Acidity)	-0.02	1	0.00		NS
X ₃ (TOC)	-5.02	1	7.82		S
X ₁ X ₂	-0.82	1	0.21		NS
X ₁ X ₃	2.08	1	1.33		NS
X ₂ X ₃	0.02	1	0.00		NS
X ₁ X ₂ X ₃	1.28	1	0.50		NS
Error		8	12.9		

Table A32. ANOVA for Cd at 4 hours with SS304 well casing. The average normalized response was 117 and the standard deviation was 15.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-15.2	1	4.26		NS
X ₂ (Acidity)	28.9	1	15.4		S
X ₃ (TOC)	-7.16	1	0.95		NS
X ₁ X ₂	-16.7	1	5.11		NS
X ₁ X ₃	5.14	1	0.49		NS
X ₂ X ₃	-6.41	1	0.76		NS
X ₁ X ₂ X ₃	3.09	1	0.18		NS
Error		8	217		

Table A33. ANOVA for Cd at 8 hours with SS304 well casing. The average normalized response was 116 and the standard deviation was 14.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-18.9	1		6.86	S
X ₂ (Acidity)	43.2	1		36.1	S
X ₃ (TOC)	-5.35	1		0.55	NS
X ₁ X ₂	-21.5	1		8.95	S
X ₁ X ₃	2.42	1		0.11	NS
X ₂ X ₃	-5.22	1		0.53	NS
X ₁ X ₂ X ₃	5.00	1		0.48	NS
Error		8	207		

Table A34. ANOVA for Cd at 24 hours with SS304 well casing. The average normalized response was 112 and the standard deviation was 12.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-24.6	1		16.4	S
X ₂ (Acidity)	49.2	1		65.4	VS
X ₃ (TOC)	-10.0	1		2.73	NS
X ₁ X ₂	-24.3	1		16.0	S
X ₁ X ₃	6.68	1		1.21	NS
X ₂ X ₃	-10.7	1		3.07	NS
X ₁ X ₂ X ₃	8.18	1		1.81	NS
Error		8	147		

Table A35. ANOVA for Cd at 72 hours with SS304 well casing. The average normalized response was 103 and the standard deviation was 14.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-29.4	1		18.6	S
X ₂ (Acidity)	44.8	1		43.3	S
X ₃ (TOC)	-6.65	1		0.95	NS
X ₁ X ₂	-31.0	1		20.7	S
X ₁ X ₃	-0.00	1		0.00	NS
X ₂ X ₃	-2.98	1		0.19	NS
X ₁ X ₂ X ₃	4.22	1		0.38	NS
Error		8	186		

Table A36. ANOVA for Cd at 0.5 hours with SS316 well casing. The average normalized response was 104 and the standard deviation was 2.1.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-5.69	1		29.0	S
X ₂ (Acidity)	0.46	1		0.19	NS
X ₃ (TOC)	-5.46	1		26.7	S
X ₁ X ₂	-1.86	1		3.12	NS
X ₁ X ₃	5.71	1		29.2	S
X ₂ X ₃	0.51	1		0.24	NS
X ₁ X ₂ X ₃	0.24	1		0.05	NS
Error		8	4.47		

Table A37. ANOVA for Cd at 4 hours with SS316 well casing. The average normalized response was 124 and the standard deviation was 49.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-52.4	1		4.53	NS
X ₂ (Acidity)	53.4	1		4.70	NS
X ₃ (TOC)	-35.1	1		2.03	NS
X ₁ X ₂	-47.5	1		3.72	NS
X ₁ X ₃	38.0	1		2.38	NS
X ₂ X ₃	-32.6	1		1.76	NS
X ₁ X ₂ X ₃	34.7	1		1.99	NS
Error		8	2430		

Table A38. ANOVA for Cd at 8 hours with SS316 well casing. The average normalized response was 130 and the standard deviation was 47.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-58.7	1		6.25	S
X ₂ (Acidity)	69.7	1		8.80	S
X ₃ (TOC)	-32.8	1		1.94	NS
X ₁ X ₂	-57.5	1		5.99	S
X ₁ X ₃	40.25	1		2.94	NS
X ₂ X ₃	-33.7	1		2.06	NS
X ₁ X ₂ X ₃	41.9	1		3.18	NS
Error		8	2210		

Table A39. ANOVA for Cd at 24 hours with SS316 well casing. The average normalized response was 136 and the standard deviation was 68.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-73.6	1		4.70	NS
X ₂ (Acidity)	78.2	1		5.31	NS
X ₃ (TOC)	-45.9	1		1.83	NS
X ₁ X ₂	-66.4	1		3.82	NS
X ₁ X ₃	57.8	1		2.89	NS
X ₂ X ₃	-57.6	1		2.88	NS
X ₁ X ₂ X ₃	51.8	1		2.33	NS
Error		8	4610		

Table A40. ANOVA for Cd at 72 hours with SS316 well casing. The average normalized response was 125 and the standard deviation was 6.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-76.8	1		5.34	S
X ₂ (Acidity)	88.5	1		7.09	S
X ₃ (TOC)	-47.0	1		2.00	NS
X ₁ X ₂	-65.3	1		3.86	NS
X ₁ X ₃	48.2	1		2.10	NS
X ₂ X ₃	-51.3	1		2.38	NS
X ₁ X ₂ X ₃	47.8	1		2.07	NS
Error		8	4420		

Table A41. ANOVA for Cr at 0.5 hours with PVC well casing. The average normalized response was 101 and the standard deviation was 1.8.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-1.68	1		3.36	NS
X ₂ (Acidity)	-1.45	1		2.52	NS
X ₃ (TOC)	-1.72	1		3.36	NS
X ₁ X ₂	1.88	1		4.21	NS
X ₁ X ₃	0.90	1		0.97	NS
X ₂ X ₃	1.08	1		1.39	NS
X ₁ X ₂ X ₃	-1.50	1		2.70	NS
Error		8	3.34		

Table A42. ANOVA for Cr at 4 hours with PVC well casing. The average normalized response was 99.9 and the standard deviation was 1.3.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	0.96	1		2.15	NS
X ₂ (Acidity)	0.62	1		0.90	NS
X ₃ (TOC)	-1.06	1		2.62	NS
X ₁ X ₂	-1.09	1		2.81	NS
X ₁ X ₃	0.68	1		1.09	NS
X ₂ X ₃	0.09	1		0.02	NS
X ₁ X ₂ X ₃	-0.97	1		2.20	NS
Error		8	1.71		

Table A43. ANOVA for Cr at 8 hours with PVC well casing. The average normalized response was 100 and the standard deviation was 1.4.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-1.44	1		4.21	NS
X ₂ (Acidity)	0.71	1		1.04	NS
X ₃ (TOC)	-1.33	1		3.65	NS
X ₁ X ₂	-0.71	1		1.04	NS
X ₁ X ₃	-1.41	1		4.07	NS
X ₂ X ₃	1.99	1		8.05	S
X ₁ X ₂ X ₃	-0.09	1		0.02	NS
Error		8	1.96		

Table A44. ANOVA for Cr at 24 hours with PVC well casing. The average normalized response was 100 and the standard deviation was 1.6.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	1.75	1		4.77	NS
X ₂ (Acidity)	0.15	1		0.04	NS
X ₃ (TOC)	-2.78	1		12.0	NS
X ₁ X ₂	-0.10	1		0.02	NS
X ₁ X ₃	-1.12	1		1.97	NS
X ₂ X ₃	-0.48	1		0.35	NS
X ₁ X ₂ X ₃	-0.08	1		0.01	NS
Error		8	2.57		

Table A45. ANOVA for Cr at 72 hours with PVC well casing. The average normalized response was 101 and the standard deviation was 1.8.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	2.71	1		8.80	S
X ₂ (Acidity)	-1.39	1		2.30	NS
X ₃ (TOC)	0.69	1		0.57	NS
X ₁ X ₂	-1.11	1		1.48	NS
X ₁ X ₃	1.01	1		1.23	NS
X ₂ X ₃	1.21	1		1.76	NS
X ₁ X ₂ X ₃	0.89	1		0.94	NS
Error		8	3.35		

Table A46. ANOVA for Cr at 0.5 hours with PTFE well casing. The average normalized response was 101 and the standard deviation was 0.7.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	0.30	1		0.70	NS
X ₂ (Acidity)	-0.78	1		4.68	NS
X ₃ (TOC)	-0.12	1		0.12	NS
X ₁ X ₂	0.72	1		4.10	NS
X ₁ X ₃	0.62	1		3.05	NS
X ₂ X ₃	-0.45	1		1.58	NS
X ₁ X ₂ X ₃	-0.45	1		1.58	NS
Error		8	0.51		

Table A47. ANOVA for Cr at 4 hours with PTFE well casing. The average normalized response was 101 and the standard deviation was 1.1.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	1.49	1		7.73	S
X ₂ (Acidity)	0.98	1		3.33	NS
X ₃ (TOC)	1.47	1		7.47	S
X ₁ X ₂	-0.13	1		0.06	NS
X ₁ X ₃	-0.97	1		3.25	NS
X ₂ X ₃	1.09	1		4.14	NS
X ₁ X ₂ X ₃	-1.49	1		7.73	S
Error		8	1.16		

Table A48. ANOVA for Cr at 8 hours with PTFE well casing. The average normalized response was 98.9 and the standard deviation was 1.9.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-3.16	1		11.0	S
X ₂ (Acidity)	-0.79	1		0.68	NS
X ₃ (TOC)	-1.46	1		2.35	NS
X ₁ X ₂	-3.21	1		11.3	S
X ₁ X ₃	2.26	1		5.62	S
X ₂ X ₃	4.49	1		22.1	S
X ₁ X ₂ X ₃	1.41	1		2.19	NS
Error		8	3.64		

Table A49. ANOVA for Cr at 24 hours with PTFE well casing. The average normalized response was 101 and the standard deviation was 2.4.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-3.16	1		11.0	S
X ₂ (Acidity)	0.28	1		0.05	NS
X ₃ (TOC)	-1.22	1		1.06	NS
X ₁ X ₂	0.00	1		0.00	NS
X ₁ X ₃	-0.95	1		0.63	NS
X ₂ X ₃	-1.00	1		0.71	NS
X ₁ X ₂ X ₃	-1.98	1		2.77	NS
Error		8	5.64		

Table A50. ANOVA for Cr at 72 hours with PTFE well casing. The average normalized response was 100 and the standard deviation was 1.3.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	3.59	1		31.8	S
X ₂ (Acidity)	-1.19	1		3.49	NS
X ₃ (TOC)	0.74	1		1.35	NS
X ₁ X ₂	-0.21	1		0.11	NS
X ₁ X ₃	-2.14	1		11.3	S
X ₂ X ₃	0.84	1		1.74	NS
X ₁ X ₂ X ₃	-0.79	1		1.53	NS
Error		8	1.62		

Table A51. ANOVA for Cr at 0.5 hours with SS304 well casing. The average normalized response was 101 and the standard deviation was 1.6.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-0.99	1		1.50	NS
X ₂ (Acidity)	-1.19	1		2.16	NS
X ₃ (TOC)	-0.09	1		0.01	NS
X ₁ X ₂	0.71	1		0.78	NS
X ₁ X ₃	1.01	1		1.57	NS
X ₂ X ₃	-0.64	1		0.62	NS
X ₁ X ₂ X ₃	-0.29	1		0.13	NS
Error		8	2.61		

Table A52. ANOVA for Cr at 4 hours with SS304 well casing. The average normalized response was 95.7 and the standard deviation was 3.7.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-1.39	1		0.56	NS
X ₂ (Acidity)	-7.14	1		14.7	S
X ₃ (TOC)	1.96	1		1.11	NS
X ₁ X ₂	-2.11	1		1.29	NS
X ₁ X ₃	-0.81	1		0.19	NS
X ₂ X ₃	-0.86	1		0.22	NS
X ₁ X ₂ X ₃	-2.94	1		2.50	NS
Error		8	13.8		

Table A53. ANOVA for Cr at 8 hours with SS304 well casing. The average normalized response was 97.2 and the standard deviation was 16.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-10.7	1		1.83	NS
X ₂ (Acidity)	-5.16	1		0.42	NS
X ₃ (TOC)	8.26	1		1.09	NS
X ₁ X ₂	-9.66	1		1.48	NS
X ₁ X ₃	-10.6	1		1.80	NS
X ₂ X ₃	9.76	1		1.52	NS
X ₁ X ₂ X ₃	9.79	1		1.52	NS
Error		8	251		

Table 54. ANOVA for Cr at 24 hours with SS304 well casing. The average normalized response was 103 and the standard deviation was 37.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-7.74	1		0.17	NS
X ₂ (Acidity)	3.46	1		0.03	NS
X ₃ (TOC)	20.3	1		1.20	NS
X ₁ X ₂	-11.0	1		0.35	NS
X ₁ X ₃	-18.6	1		1.00	NS
X ₂ X ₃	19.2	1		1.07	NS
X ₁ X ₂ X ₃	-20.9	1		1.26	NS
Error		8	1380		

Table A55. ANOVA for Cr at 72 hours with SS304 well casing. The average normalized response was 103 and the standard deviation was 42.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	5.50	1		0.07	NS
X ₂ (Acidity)	5.88	1		0.08	NS
X ₃ (TOC)	32.8	1		2.39	NS
X ₁ X ₂	1.25	1		0.00	NS
X ₁ X ₃	-18.9	1		0.79	NS
X ₂ X ₃	29.6	1		1.94	NS
X ₁ X ₂ X ₃	-15.8	1		0.56	NS
Error		8	1800		

Table A56. ANOVA for Cr at 0.5 hours with SS316 well casing. The average normalized response was 102 and the standard deviation was 1.5.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-1.51	1		4.43	NS
X ₂ (Acidity)	-1.49	1		4.17	NS
X ₃ (TOC)	0.59	1		0.65	NS
X ₁ X ₂	-0.34	1		0.21	NS
X ₁ X ₃	1.69	1		5.37	S
X ₂ X ₃	-0.49	1		0.45	NS
X ₁ X ₂ X ₃	-0.89	1		1.48	NS
Error		8	2.12		

Table A57. ANOVA for Cr at 4 hours with SS316 well casing. The average normalized response was 92.1 and the standard deviation was 5.2.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-6.54	1		6.20	S
X ₂ (Acidity)	-10.3	1		15.4	S
X ₃ (TOC)	10.0	1		14.6	S
X ₁ X ₂	-5.99	1		5.20	NS
X ₁ X ₃	0.11	1		0.00	NS
X ₂ X ₃	6.54	1		6.20	S
X ₁ X ₂ X ₃	0.11	1		0.00	NS
Error		8	27.6		

Table A58. ANOVA for Cr at 8 hours with SS316 well casing. The average normalized response was 87.2 and the standard deviation was 10.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-14.1	1		7.77	S
X ₂ (Acidity)	-15.6	1		9.56	S
X ₃ (TOC)	11.8	1		5.46	S
X ₁ X ₂	-9.19	1		3.32	NS
X ₁ X ₃	-1.59	1		0.10	NS
X ₂ X ₃	10.6	1		4.42	NS
X ₁ X ₂ X ₃	1.99	1		0.15	NS
Error		8	102		

Table A59. ANOVA for Cr at 24 hours with SS316 well casing. The average normalized response was 85.5 and the standard deviation was 11.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-9.91	1		3.32	NS
X ₂ (Acidity)	-22.4	1		17.0	S
X ₃ (TOC)	17.4	1		10.2	S
X ₁ X ₂	-11.9	1		4.77	NS
X ₁ X ₃	-1.71	1		0.10	NS
X ₂ X ₃	11.4	1		4.42	NS
X ₁ X ₂ X ₃	2.26	1		0.17	NS
Error		8	118		

Table A60. ANOVA for Cr at 72 hours with SS316 well casing. The average normalized response was 83.6 and the standard deviation was 9.9.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-5.86	1		1.39	NS
X ₂ (Acidity)	-22.6	1		20.7	S
X ₃ (TOC)	18.8	1		14.3	S
X ₁ X ₂	-7.66	1		2.38	NS
X ₁ X ₃	-2.36	1		0.23	NS
X ₂ X ₃	12.9	1		6.77	S
X ₁ X ₂ X ₃	3.64	1		0.54	NS
Error		8	98.6		

Table A61. ANOVA for Pb at 0.5 hours with PVC well casing. The average normalized response was 99.9 and the standard deviation was 0.9.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-0.09	1		0.04	NS
X ₂ (Acidity)	-1.06	1		5.52	S
X ₃ (TOC)	-0.61	1		1.83	NS
X ₁ X ₂	0.74	1		2.66	NS
X ₁ X ₃	2.09	1		21.3	S
X ₂ X ₃	-0.04	1		0.01	NS
X ₁ X ₂ X ₃	-0.84	1		3.43	NS
Error		8	0.82		

Table A62. ANOVA for Pb at 4 hours with PVC well casing. The average normalized response was 88.9 and the standard deviation was 3.0.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-0.71	1		0.22	NS
X ₂ (Acidity)	-4.16	1		7.54	S
X ₃ (TOC)	5.89	1		15.1	S
X ₁ X ₂	2.34	1		2.37	NS
X ₁ X ₃	-3.21	1		4.49	NS
X ₂ X ₃	4.84	1		10.2	S
X ₁ X ₂ X ₃	-1.71	1		1.28	NS
Error		8	9.19		

Table A63. ANOVA for Pb at 8 hours with PVC well casing. The average normalized response was 89.3 and the standard deviation was 3.5.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-2.96	1		2.84	NS
X ₂ (Acidity)	-0.06	1		0.00	NS
X ₃ (TOC)	7.46	1		18.0	S
X ₁ X ₂	-0.79	1		0.20	NS
X ₁ X ₃	-0.41	1		0.06	NS
X ₂ X ₃	3.64	1		4.27	NS
X ₁ X ₂ X ₃	-2.54	1		2.08	NS
Error		8	12.4		

Table A64. ANOVA for Pb at 24 hours with PVC well casing. The average normalized response was 80.8 and the standard deviation was 5.1.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-2.78	1		1.20	NS
X ₂ (Acidity)	-10.7	1		17.9	S
X ₃ (TOC)	13.9	1		30.1	S
X ₁ X ₂	0.30	1		0.01	NS
X ₁ X ₃	-3.55	1		1.96	NS
X ₂ X ₃	7.95	1		9.81	S
X ₁ X ₂ X ₃	-2.68	1		1.11	NS
Error		8	25.7		

Table A65. ANOVA for Pb at 72 hours with PVC well casing. The average normalized response was 74.3 and the standard deviation was 6.4.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	0.20	1		0.00	NS
X ₂ (Acidity)	-11.7	1		13.3	S
X ₃ (TOC)	19.2	1		35.7	S
X ₁ X ₂	1.67	1		0.27	NS
X ₁ X ₃	-2.75	1		0.73	NS
X ₂ X ₃	8.02	1		6.23	S
X ₁ X ₂ X ₃	-2.98	1		0.86	NS
Error		8	41.4		

Table A66. ANOVA for Pb at 0.5 hours with PTFE well casing. The average normalized response was 100 and the standard deviation was 2.6.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	0.12	1		0.01	NS
X ₂ (Acidity)	1.25	1		0.96	NS
X ₃ (TOC)	0.92	1		0.52	NS
X ₁ X ₂	-0.08	1		0.00	NS
X ₁ X ₃	1.40	1		1.20	NS
X ₂ X ₃	0.28	1		0.05	NS
X ₁ X ₂ X ₃	-1.95	1		2.32	NS
Error		8	6.53		

Table A67. ANOVA for Pb at 4 hours with PTFE well casing. The average normalized response was 97.4 and the standard deviation was 1.9.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	3.30	1		11.8	S
X ₂ (Acidity)	0.88	1		0.83	NS
X ₃ (TOC)	-0.42	1		0.20	NS
X ₁ X ₂	3.28	1		11.6	S
X ₁ X ₃	-4.52	1		22.3	S
X ₂ X ₃	-2.65	1		7.61	S
X ₁ X ₂ X ₃	-2.30	1		5.73	S
Error		8	3.69		

Table A68. ANOVA for Pb at 8 hours with PTFE well casing. The average normalized response was 98.5 and the standard deviation was 3.2.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	1.24	1		0.61	NS
X ₂ (Acidity)	3.84	1		5.90	S
X ₃ (TOC)	0.66	1		0.18	NS
X ₁ X ₂	-3.21	1		4.13	NS
X ₁ X ₃	-0.09	1		0.00	NS
X ₂ X ₃	-1.49	1		0.89	NS
X ₁ X ₂ X ₃	2.11	1		1.79	NS
Error		8	9.98		

Table A69. ANOVA for Pb at 24 hours with PTFE well casing. The average normalized response was 95.1 and the standard deviation was 4.0.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	0.08	1		0.00	NS
X ₂ (Acidity)	-2.75	1		1.86	NS
X ₃ (TOC)	4.75	1		5.56	S
X ₁ X ₂	-3.38	1		2.81	NS
X ₁ X ₃	-0.68	1		0.11	NS
X ₂ X ₃	-0.25	1		0.02	NS
X ₁ X ₂ X ₃	-0.48	1		0.06	NS
Error		8	16.2		

Table A70. ANOVA for Pb at 72 hours with PTFE well casing. The average normalized response was 89.9 and the standard deviation was 3.4.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	5.96	1		12.3	NS
X ₂ (Acidity)	-4.49	1		6.99	S
X ₃ (TOC)	8.16	1		23.1	S
X ₁ X ₂	-1.21	1		0.51	NS
X ₁ X ₃	-1.51	1		0.79	NS
X ₂ X ₃	0.94	1		0.31	NS
X ₁ X ₂ X ₃	-2.89	1		2.90	NS
Error		8	11.5		

Table A71. ANOVA for Pb at 0.5 hours with SS304 well casing. The average normalized response was 102 and the standard deviation was 0.8.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	0.05	1		0.02	NS
X ₂ (Acidity)	0.50	1		1.57	NS
X ₃ (TOC)	0.08	1		0.04	NS
X ₁ X ₂	-0.82	1		4.26	NS
X ₁ X ₃	-0.35	1		0.77	NS
X ₂ X ₃	-0.80	1		4.01	NS
X ₁ X ₂ X ₃	0.52	1		1.73	NS
Error		8	0.64		

Table A72. ANOVA for Pb at 4 hours with SS304 well casing. The average normalized response was 78.4 and the standard deviation was 3.5.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-4.48	1		6.61	S
X ₂ (Acidity)	13.5	1		60.6	VS
X ₃ (TOC)	12.9	1		55.1	VS
X ₁ X ₂	1.90	1		1.19	NS
X ₁ X ₃	-2.58	1		2.19	NS
X ₂ X ₃	-5.15	1		8.75	S
X ₁ X ₂ X ₃	-0.85	1		0.24	NS
Error		8	12.1		

Table A73. ANOVA for Pb at 8 hours with SS304 well casing. The average normalized response was 69.9 and the standard deviation was 3.1.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-6.76	1		19.0	S
X ₂ (Acidity)	24.2	1		243	VS
X ₃ (TOC)	118.0	1		134	VS
X ₁ X ₂	-5.01	1		10.5	S
X ₁ X ₃	-2.09	1		1.81	NS
X ₂ X ₃	-6.61	1		18.2	S
X ₁ X ₂ X ₃	0.26	1		0.03	NS
Error		8	9.61		

Table A74. ANOVA for Pb at 24 hours with SS304 well casing. The average normalized response was 53.8 and the standard deviation was 4.2.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-4.04	1		3.67	NS
X ₂ (Acidity)	30.4	1		209	VS
X ₃ (TOC)	26.6	1		161	VS
X ₁ X ₂	-4.74	1		5.09	NS
X ₁ X ₃	-6.39	1		9.26	S
X ₂ X ₃	-0.26	1		0.02	NS
X ₁ X ₂ X ₃	-1.89	1		0.81	NS
Error		8	17.6		

Table A75. ANOVA for Pb at 72 hours with SS304 well casing. The average normalized response was 45.2 and the standard deviation was 6.1.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-0.78	1		0.06	NS
X ₂ (Acidity)	31.4	1		106	VS
X ₃ (TOC)	24.8	1		65.8	VS
X ₁ X ₂	-0.18	1		0.00	NS
X ₁ X ₃	-3.65	1		1.43	NS
X ₂ X ₃	3.72	1		1.49	NS
X ₁ X ₂ X ₃	-2.30	1		0.57	NS
Error		8	37.3		

Table A76. ANOVA for Pb at 0.5 hours with SS316 well casing. The average normalized response was 101 and the standard deviation was 2.5.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-2.51	1		3.89	NS
X ₂ (Acidity)	0.31	1		0.06	NS
X ₃ (TOC)	1.06	1		0.69	NS
X ₁ X ₂	-0.64	1		0.25	NS
X ₁ X ₃	1.01	1		0.63	NS
X ₂ X ₃	-0.01	1		0.00	NS
X ₁ X ₂ X ₃	0.86	1		0.46	NS
Error		8	6.49		

Table A77. ANOVA for Pb at 4 hours with SS316 well casing. The average normalized response was 80.3 and the standard deviation was 7.7.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-11.0	1		8.14	S
X ₂ (Acidity)	14.4	1		13.8	S
X ₃ (TOC)	14.8	1		14.7	S
X ₁ X ₂	-2.25	1		0.34	NS
X ₁ X ₃	0.80	1		0.04	NS
X ₂ X ₃	-5.05	1		1.71	NS
X ₁ X ₂ X ₃	6.12	1		2.51	NS
Error		8	59.7		

Table A78. ANOVA for Pb at 8 hours with SS316 well casing. The average normalized response was 80.4 and the standard deviation was 10.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-15.4	1		9.47	S
X ₂ (Acidity)	21.5	1		18.4	S
X ₃ (TOC)	19.4	1		15.0	S
X ₁ X ₂	-7.75	1		2.39	NS
X ₁ X ₃	4.02	1		0.64	NS
X ₂ X ₃	-4.36	1		0.76	NS
X ₁ X ₂ X ₃	2.00	1		0.16	NS
Error		8	101		

Table A79. ANOVA for Pb at 24 hours with SS316 well casing. The average normalized response was 79.3 and the standard deviation was 19.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-21.4	1		5.23	NS
X ₂ (Acidity)	17.7	1		3.58	NS
X ₃ (TOC)	29.6	1		10.0	S
X ₁ X ₂	-5.02	1		0.29	NS
X ₁ X ₃	2.60	1		0.08	NS
X ₂ X ₃	3.20	1		0.11	NS
X ₁ X ₂ X ₃	-0.40	1		0.00	NS
Error		8	351		

Table A80. ANOVA for Pb at 72 hours with SS316 well casing. The average normalized response was 72.0 and the standard deviation was 17.

Factor	Effects	df	MS	F	Sig
X ₁ (Metal conc)	-2.65	1		0.10	NS
X ₂ (Acidity)	32.5	1		14.4	S
X ₃ (TOC)	35.0	1		16.7	S
X ₁ X ₂	0.80	1		0.01	NS
X ₁ X ₃	6.40	1		0.56	NS
X ₂ X ₃	-6.82	1		0.64	NS
X ₁ X ₂ X ₃	-1.70	1		0.04	NS
Error		8	293		

APPENDIX G

EVALUATION OF FOUR WELL CASING MATERIALS FOR
MONITORING SELECTED TRACE LEVEL ORGANICS IN GROUND WATER

Evaluation of Four Well Casing Materials for Monitoring Selected Trace Level Organics in Ground Water

LOUISE V. PARKER, THOMAS F. JENKINS AND PATRICK B. BLACK

INTRODUCTION

Background

Several different materials have been used in the manufacture of well casings and screens for monitoring ground water. These materials include virgin fluorocarbon resins (i.e., fluorinated ethylene propylene (FEP), polytetrafluoroethylene (PTFE and Teflon (R)), and stainless steel (304, 316, or 2205), cast iron, galvanized steel, polyvinyl chloride (PVC), polyethylene (PE), epoxy bisphenol and polypropylene (PP) (U.S. EPA 1986). Until relatively recently (1985) PVC was the preferred casing material. However, in 1985 the initial draft of the U.S. EPA's "Resource Conservation and Recovery Act (RCRA) Ground-Water Monitoring Technical Enforcement Guidance Document" was published. This document stated, "in constructing wells, the owner/operator should use Teflon, Stainless Steel 316, or other proven chemically and physically stable materials."

The EPA's concern was that many of the materials commonly used in ground water monitoring affected the quality of the samples or did not have the long-term structural characteristics required of RCRA monitoring wells. The EPA document stated that "steel casings deteriorated in corrosive environments; PVC deteriorated in contact with ketones, esters and aromatic hydrocarbons; polyethylene deteriorated in contact with aromatic and halogenated hydrocarbons; and polypropylene deteriorated in contact with oxidizing acids, aliphatic hydrocarbons and aromatic hydrocarbons." The EPA was also concerned that steel, PVC, polyethylene and polypropylene might adsorb or leach constituents, thereby affecting the composition of the ground water samples.

Because of the furor that followed publication of the initial draft of this document, the requirement was reduced slightly in the final version.

This version stated that "fluorocarbon resins or stainless steel should be specified for use in the saturated zone when volatile organics are to be determined, or may be tested, during a 30-year period" (U.S. EPA 1986). The RCRA document further stated that "National Sanitation Foundation (NSF) or ASTM-approved polyvinyl chloride (PVC) well casing and screens may be appropriate if only trace metals or nonvolatile organics are the contaminants anticipated."

It is generally recognized that metal pipes can corrode, that polymeric materials such as PVC, PE and PP can soften and swell in the presence of either pure or highly concentrated solutions of some organic solvents, and that fluoropolymers are resistant to attack by almost all chemical species. However, in ground water monitoring situations very high concentrations or neat solvents are usually not encountered. Therefore, PVC casings may be suitable for monitoring organics in the concentration range most commonly found. This report focuses on the interactions between well casings (PVC, PTFE and stainless steel) and trace-level organic constituents.

Literature review

Ideally the long-term interaction between a well casing and the ground water being monitored should not result in gain or loss of analyte or interference with the analytical method used for determination. Analyte loss can result from sorption of analyte by the casing material, from chemical or microbiological destruction of the analyte as a result of interaction with the surface, or leaching of a substance from the casing material.

Masse et al. (1981) outlined the factors involved in sorptive losses of metals on containers:

1. The chemical nature of the analyte and its concentration.
2. The characteristics of the solution—the pres-

ence of acids, dissolved material, complexing agents, dissolved gases (especially oxygen), suspended matter, and microorganisms.

3. The properties of the container—chemical composition, surface roughness, surface cleanliness, relative surface area, history (i.e., age, prior cleaning, and previous exposure).

4. External factors—temperature, contact time, access of light, and agitation.

These factors are generally applicable to well casing materials, and many of them are also applicable when considering the sorption of organics from solution.

There have been several studies examining the sorption of organics by rigid PVC. Lawrence and Tosine (1976) found that PVC chips were quite efficient in adsorbing PCBs from water and waste water. However, PVC appeared to be effective only at sorbing PCBs when their concentrations were close to their solubility limits.*

Pettyjohn et al. (1981) claimed that metal surfaces can also strongly adsorb organic compounds. Although they did not present any supporting data, they claimed that, for example, DDT is strongly adsorbed by stainless steel.

Miller (1982) conducted a six-week laboratory study that tested three types of well casing materials, including schedule 40 PVC, for sorption of trace levels (2–14 $\mu\text{g}/\text{L}$) of six volatile organics. The substances tested were bromoform, trichlorofluoromethane, trichloroethylene, 1,1,1-trichloroethane, 1,1,2-trichloroethane, and tetrachloroethylene. While the data he presented were only semiquantitative, tetrachloroethylene appeared to be sorbed by the PVC casing material (25 to 50% loss after six weeks). It is not clear why this compound was preferentially sorbed. However, it could be that its planar geometry allowed it to more easily penetrate the pores of the polymer (Berens and Hopfenberg 1982, Parker and Jenkins 1986).

Curran and Tomson (1983) tested five plastics for adsorption of trace levels (0.5 $\mu\text{g}/\text{L}$) of naphthalene and *p*-dichlorobenzene. The plastics tested included Teflon, and glued or nonglued rigid PVC. This study was performed by pumping a set volume (20 L) of the aqueous organic solution through the tubings. The data they reported were only semiquantitative, but they estimated that 80–100% of both of these organics were recovered.

Houghton and Berger (1984) compared adjacent wells, made of PVC and steel, to assess the effect that composition of the well casing material

had on the composition of sampled water. These wells had been in the ground for two years. Each well was only sampled twice. Although they found that concentrations of dissolved organic carbon and total organic carbon were 10% higher in the PVC well than in the steel well, the PVC casing was joined using organic solvents, which may explain the elevated organic carbon content. Perhaps their most significant finding was that sampling methods had a greater effect on the ground water composition than the type of casing.

Our laboratory studied the suitability of PVC well casings for monitoring low levels of military munitions and their breakdown products (Parker and Jenkins 1986). Specifically, the substances studied were 2,4,6-trinitrotoluene (TNT), hexahydro-1,3,5-trinitro-1,3,5-triazine (RDX), octahydro-1,3,5,7-tetranitro-1,3,5,7-tetrazocine (HMX) and 2,4-dinitrotoluene (DNT). Our initial study, conducted for 60 days under non-sterile conditions, indicated significant loss of TNT and to a lesser extent HMX in the presence of PVC well casing. However, a 21-day follow-up study, conducted using both sterile and nonsterile conditions, indicated that TNT loss seemed to be associated with increased microbial degradation in the presence of PVC rather than sorption by PVC. In the nonsterile samples, loss was only 4% greater after 21 days than when compared with the controls.

Reynolds and Gillham (1986) have conducted perhaps the most definitive study on PVC and PTFE materials to date. In a laboratory study they determined the sorption of low levels of five halogenated compounds by six polymeric materials including rigid PVC rod and PTFE tubing. The organic compounds tested, in concentrations ranging from 20–45 $\mu\text{g}/\text{L}$, were 1,1,1-trichloroethane, 1,1,2,2-tetrachloroethane, bromoform, tetrachloroethylene and hexachloroethane.

For each polymer, several pieces were placed into sixty 160-mL hypovials, which were then filled with the aqueous organic solution without headspace and sealed. Thirty control samples were prepared identically except that they did not contain any added polymer. The results were expressed by taking the final concentration C and dividing it by the initial concentration C_0 . A relative concentration of 1.0 represented no sorption.

Both PVC and PTFE sorbed four of the five compounds tested. Sorption was generally slow; decreases in solution concentrations were generally less than 50% after 5 weeks, except for tetrachloroethylene, which was reduced by 50% in approximately 8 hours by PTFE. Reynolds and Gillham

*D.C. Leggett, CRREL, personal communication 1986.

(1986) determined the time at which the relative concentration (C/C_0) was reduced to 0.9 for each polymer and then ranked them in order. The order the compounds were sorbed varied between polymers. Reynolds and Gillham (1986) compared this order of loss with the compounds' octanol/water partition coefficient ($\log K_{ow}$), undecane/water partition coefficient, and solubility in water, but they did not find any relationship (Table A1).

They attributed the loss by the polymer materials to absorption. They developed a model where uptake of an organic compound first proceeds by sorption/dissolution into the polymer surface, followed by diffusion into the polymer matrix. Their analytical model is given in eq 1:

$$\frac{C}{C_0} = \exp\left[\frac{KDt}{A^2}\right] \operatorname{erfc}\left[\frac{(KD)^{1/2} t^{1/2}}{A}\right] \quad (1)$$

where C = concentration in solution ($\mu\text{g/L}$) at time t (sec)

C_0 = initial solution concentration ($\mu\text{g/L}$)

C/C_0 = relative concentration (dimensionless)

K = partition coefficient between the organic compound in solution and the polymer (dimensionless)

D = diffusion coefficient in the polymer (cm^2/sec).

The product of K and D is defined as the permeability coefficient (P).

Using this model, Reynolds and Gillham (1986) fitted the curves through the data and found reasonable agreement between eq 1 and most of their experimental results. They were unable to fit a curve through the data for absorption of bromoform by PTFE or 1,1,1-trichloroethane by PVC, because they did not measure any absorption of these compounds after five weeks. They also found that after three weeks hexachloroethane and bromoform were more rapidly absorbed by PVC than eq 1 predicted. However, enhanced biodegradation in the presence of PVC (similar to what Parker and Jenkins [1986] observed with nitroaromatics) could also explain this additional loss. They noted additional peaks in the chromatograms of these samples; these peaks were similar to ones they had observed in degraded stock solutions of bromoform and hexachloroethane. No precautions were taken in their study to prevent biodegradation of the analytes.

Reynolds and Gillham (1986) felt that their results for PVC compared well with those from Miller (1982), except for the results for bromoform

with PVC. Miller noted no loss after six weeks, whereas they found 43% loss after five weeks.

Reynolds and Gillham (1986) concluded that PVC absorption was sufficiently slow so that any resulting bias would most likely not be significant for these compounds, provided the well is developed and sampled on the same day. They also concluded that the same was true for PTFE except for tetrachloroethylene. However, they did not feel they had sufficient data to recommend the use of PVC over PTFE, and they also could not predict which organic chemicals were most susceptible to absorption. Moreover, we feel that some caution should be used when extrapolating their data since they did not use actual well casings.

Sykes et al. (1986) evaluated sorptive losses of organics by well casing materials in a laboratory study that may more closely parallel a real ground water monitoring situation. Control samples, which contained only the aqueous organic solution, were compared with samples that also contained either pieces of PVC, stainless steel or PTFE well casing. The organics tested were methylene chloride, 1,2-dichloroethane, trans-1,2-dichloroethylene, trichloroethylene, toluene, and chlorobenzene. Concentrations ranged from 87 to 150 $\mu\text{g/L}$. After seven days at 5°C, solutions were decanted and replaced with fresh solution (at the initial concentrations). Samples were then taken after 1 hour, the sample solutions were again refreshed, and final samples were taken after 24 hours. They reported that for both exposure times and all organics tested, the mean values for the solutions exposed to casing materials (three replicate samples) were usually within 1 standard deviation of the mean control values (nine replicate samples). They concluded that there were no statistically significant differences between the control samples and those containing well casings.

Barcelona and Helfrich (1986) conducted an in-situ study to determine the effect of well construction material on the reliability of determinations of organic chemical constituents in ground water. They constructed adjacent wells at upgradient and downgradient locations at two sanitary landfill sites. Casing materials were PTFE, 304 stainless steel, and PVC. No solvent cements, threaded joints, or uncommon materials were employed in well construction. Their findings are based on samples taken once a month for 6 months.

At site 1 Barcelona and Helfrich (1986) generally found, at the downgradient location, higher levels of total organic carbon content (TOC) in samples from the stainless steel and Teflon wells

than from the PVC well. The levels of 1,1-dichloroethane (DCE) were generally higher in samples taken from the downgradient Teflon and stainless steel wells than those taken from the PVC well. The values for *cis*-1,2-dichloroethylene (CDCE) were considerably higher in the samples taken from the stainless steel well than in those from either plastic-lined well.

At site 2, the levels of DCE were 10 times higher than at site 1. In contrast to site 1, the levels of purgeable organics were consistently higher for the PVC well samples than for the stainless steel or Teflon samples. The concentration of DCE was two times greater for the samples from the PVC well than for those from the stainless steel well. Because these wells were only about 1 m apart, Barcelona and Helrich felt it unlikely that they had intercepted ground water of different microconstituent quality. However, while they concluded that well casing materials exerted significant, though unpredictable, effects on the determination of total organic carbon and specific volatile organic compounds, we feel a much larger statistical base than two data sets is needed before any conclusions of this type can be drawn. Also, other differences in the construction of the wells may be responsible for these differences.

Gossett and Hegg (1987) compared three sampling devices, including a handmade Teflon bailer and a PVC bailer, to determine their effects on the recovery of three volatile organic compounds in ground water. The three organics used were chloroform, benzene and 1,2-dichloroethane; the initial concentrations were 749, 439 and 628 mg/L, respectively. They used two experimental wells: one constructed with PVC casing and the other with stainless steel. Based on analysis of variance, they claimed that neither sampler type nor well casing material had a significant effect. However, with only one sampler of each type of material and no report of the number of replicate samples, we caution against extrapolating these results to a larger population of samplers or casings.

In a laboratory study, Jones and Miller (1988) examined several different well casing materials for sorption of several trace level (parts per billion) organic constituents. The materials included PVC, ABS, Teflon, stainless steel 304, and Kynar (polyvinylidene fluoride or PVDF). Although they found losses for most of the compounds tested, there were no control samples that could be used for comparison. Therefore, losses could result from sorption by the glass containers or Teflon caps, or from chemical degradation or biodegradation since

no precautions were taken to prevent biodegradation.

Aside from possible losses due to sorption, casings may leach substances that could interfere with analyses or could cause, or aid in, alteration of the analyte in question.

Several components of rigid PVC may possibly leach. These components include vinyl chloride monomer (VCM), thermal stabilizers, pigments, lubricants, fillers, impurities, and transformation products. While older studies (Banzer 1977 and Dressman and McFarren 1978) found that significant concentrations of VCM leached from PVC pipe into water, this problem has been greatly moderated by reducing the residual VCM levels in the resin and finished products (Barcelona et al. 1984). While we were not able to find much specific information on the substances used as thermal stabilizers in PVC well casings, in the United States organo-tin compounds have been widely used in PVC potable water pipes (Boettner et al. 1981). Lead compounds are more widely used in other parts of the world such as Great Britain. Specific organo-tin compounds used in the U.S. include methyl-, butyl-, and octyl-tin esters of lauric, maleic, and thioglycolic acids (Boettner et al. 1981). Other stabilizers that have received approval for use in potable water pipes include compounds containing antimony, antimony-tin, calcium-zinc, and zinc (McClelland 1981). While the inorganic components of stabilizers have been found to leach from PVC pipe at measurable levels (Packham 1971a,b,c; Gross et al. 1974; Dietz et al. 1977; Boettner et al. 1981 and McClelland 1981), there is little information regarding leaching of the organic components. Presumably organic species are less soluble and therefore would not leach as readily. Metal leaching is greatest initially (mostly occurring within the first few days) and can be reduced by either precleaning the pipe with detergent, prerinsing it, or by treating it with dilute mineral acid (Packham 1971 a and c). This may also be true with respect to leaching organic constituents. Plasticizers (phthalate esters) are also components of flexible PVC products; but we would not expect to see them leaching from well casings since rigid PVC products do not contain them (plasticizers are added to give flexibility).

In addition to the actual components of well casing materials that may leach substances into ground water, well casings that have been joined by solvent bonds can significantly leach the solvents used to join the pipe (Boettner et al. 1981. Sosebee et al. 1982). Commonly used bonding

solvents are tetrahydrofuran, cyclohexanone, methylethylketone, and methylisobutylketone. Because these solvents have been detected leaching into ground water several months after installation of monitoring wells (Sosebee et al. 1982 and Miller 1982), it is generally recommended that only casings and bailers with threaded joints be used for ground water monitoring. These solvents may also dissolve some of the PVC polymer, thereby releasing chloroform and carbon tetrachloride (Desrosiers and Dunnigan 1983).

Miller (1982) looked for leaching of solvent extractable substances, such as plasticizers and other additives, from PVC well casings that had been exposed for 3 to 6 weeks to solutions containing trace levels of several metal and organic substances. The samples were extracted with solvent, concentrated by a factor of 1000, and analyzed using flame ionization gas chromatography (GC-FID). Although Miller did not find any identifiable substances in these leachates, he cautions that leaching may be greater in an actual monitoring situation where ground water is flowing and may contain other more aggressive pollutants.

Curran and Tomson (1983) also tested PVC and Teflon for leaching of contaminants; in their test, water was actually pumped through the tubings. The samples were processed and analyzed using methods very similar to those used by Miller (1982). Curran and Tomson (1983) did not find any analytical interferences in the samples that had been exposed to either Teflon or PVC that had been previously washed with detergent. They concluded that rigid PVC was acceptable for ground water monitoring if the casing is thoroughly washed and rinsed prior to installation.

We also tested several samples of PVC well casing for the leaching of substances that could interfere with analytical determination of these munitions (Parker and Jenkins 1986). We did not find any detectable interferences using reversed-phase HPLC analysis (Jenkins et al. 1986).

Organic lubricants such as inks or lubricants used during manufacture could possibly leach from stainless steel casings.

In addition to possible analytical problems arising from substances that can be leached from well casing materials, desorption of substances that have been previously sorbed by casing materials could raise the concentration of analytes if the concentrations in the well were to decrease substantially.

Only two studies have addressed desorption of organic constituents (Miller 1982, Jones and Miller

1988). Because the study of Jones and Miller (1988) did not include any control samples, we will not discuss those results. However, Miller (1982) found slight (25%) desorption of tetrachloroethylene from PVC during the first two weeks.

Digest of the literature and proposed study

Generally, we feel that the literature on the interactions of trace level organics with SS, PVC, and PTFE casings is incomplete. Many of the studies we cited only examined one or two of these casing materials; this makes it difficult to compare all four casings. Also, there were problems in the experimental design of a number of these studies; often there was no replication or controls, the data were not quantitative, or effects such as biodegradation could not be ruled out. In addition, many of the authors failed to report the actual data, thereby precluding an independent assessment of the authors' conclusions.

In spite of these problems, some conclusions can be drawn from the literature. First, at least some of the smaller halogenated alkanes and alkenes were slowly sorbed by both PVC and PTFE, and in one instance tetrachloroethylene was rapidly sorbed by PTFE (50% loss within 8 hours). However, based on the data so far, we cannot predict which compounds are most susceptible to loss or the rate of loss. While few studies have examined whether this loss is reversible, there is evidence in at least one study that tetrachloroethylene that has been sorbed by PVC is also slowly desorbed. There does not appear to be any problem with organic substances leaching from PTFE. While there are a number of compounds that possibly could leach from PVC casings and several metal species have been found to leach, there does not appear to be a serious problem with organic substances leaching, especially if the casing is washed with detergent and water prior to use. Also, while one would not expect to find organic substances leaching from stainless steel casings, again the casings should be washed to eliminate any surface contaminants.

The purpose of this study was to compare the performance of these four casing materials when subjected to trace levels of a variety of organic species including several volatile species. This study included control samples and sufficient replication to allow objective statistical analysis of the results. Biocide (mercuric chloride) was added to all the samples to eliminate losses due to biodegradation.

MATERIALS AND METHODS

Five-centimeter (2-in.)-diameter threaded well casings designed specifically for ground water monitoring were used in this study. The casings tested were schedule 40 PVC, Teflon, and 304 and 316 stainless steel. Sections 11 mm in length were cut from the PVC and Teflon casings and those 14 mm in length from the two stainless steel casings. Because the thickness of the walls of the well casings varied, the length was varied so that the final surface area would be the same for all the casings. These ring-shaped sections were then cut into quarters. Special care was taken to eliminate contamination from grease or oil in the cutting process. For each casing material, the pieces were then placed in a large beaker containing deionized water plus detergent and sonicated for 10 minutes. The pieces were then rinsed with deionized water until no suds remained, placed in fresh deionized water, and sonicated for 20 minutes. The water was then poured off, and the pieces were left to air dry on lint-free paper towel. Two pieces of casing were placed in each 40-mL vial. The vials were filled with the aqueous test solution so that there was no headspace and then capped with Teflon-lined plastic caps. Similar vials with no well casing material served as controls. The ratio of the surface area of the casing to solution volume was $0.79 \text{ cm}^2/\text{mL}$; this ratio was determined by dividing the surface area inside a 5-cm-diameter pipe by the volume that the pipe would hold, or $SA/V = 2/r$ where $r = 2.54 \text{ cm}$. The ratio of solution volume to volume of casing material was approximately 10.

In the first experiment, the test solution was prepared by adding each of the organics directly to 2.2 L of well water (taken from a deep water well in Weathersfield, Vermont) in a stoppered glass bottle. The organics used were RDX, trinitrobenzene (TNB), cis-1,2-dichloroethylene (CDCE), trans-1,2-dichloroethylene (TDCE), m-nitrotoluene (MNT), trichloroethylene (TCE), chlorobenzene (CLB), o-dichlorobenzene (ODCB), p-dichlorobenzene (PDCB), and m-dichlorobenzene (MDCB). The criteria used for selecting these substances included whether they were an EPA priority pollutant, molecular structure, solubility in water, K_{ow} value, and retention time (using reversed-phase HPLC analysis). The final concentration was approximately 2 mg/L for each organic constituent. The solution also contained 40 mg/L HgCl_2 to prevent biodegradation of the organics. The bottle was filled to capacity to eliminate any headspace, capped with a ground glass

stopper, and then stirred with a magnetic stirrer for 24 hours. The solution was then poured into scintillation vials and capped; separate vials were prepared for each sampling period so that the test solution could be discarded after sampling. For each material and time there were three replicate samples. Sample times were 0 hours, 1 hour, 8 hours, 24 hours (1 day), 72 hours (3 days), 168 hours (1 week) and approximately 1000 hours (6 weeks).

After removing an aliquot for analysis from each of the 1000-hour samples, the vials were emptied, and the pieces of well casing were rinsed with fresh, uncontaminated well water to remove any residual solution adhering to the surfaces. The casing pieces were then placed in clean vials with fresh unspiked well water, capped, and allowed to sit for 3 days. Aliquots taken from these samples were analyzed to determine if desorption had occurred.

In the second experiment 2.0 g of NaCl was also added per liter of solution to test the effect high chloride concentrations had on sorption/desorption. Sampling times were the same except that the last sample was taken after approximately 1200 hours (7 weeks).

All analytical determinations were performed using reversed-phase high performance liquid chromatography (RP-HPLC). A modular system was employed consisting of a Spectra Physics SP 8810 isocratic pump, a Dynatech LC-241 autosampler with a 100- μL loop injector, a Spectra-Physics

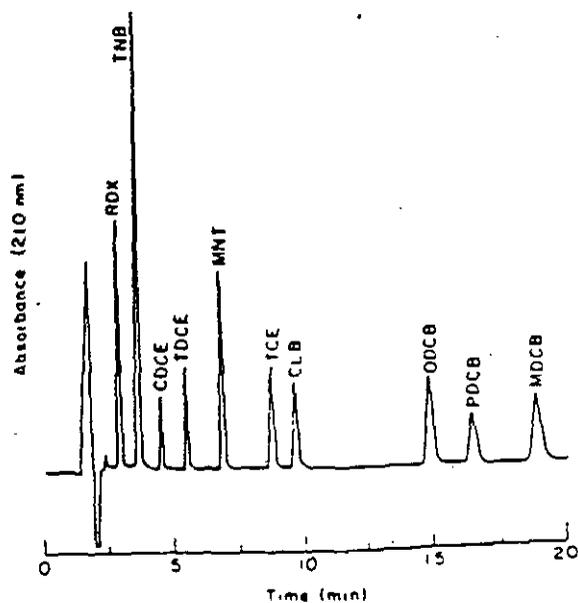


Figure 1. Chromatogram for 10 analytes.

SP8490 variable wavelength UV detector set at 210 nm, a Hewlett-Packard 3393A digital integrator and a Linear model 555 strip chart recorder. Separations were obtained on a 25-cm \times 4.6-mm (5 μ m) LC-18 column (Supelco) eluted with 1.5 mL/min of 62/38 (V/V) methanol-water. Retention times varied from 3.0 to 18.8 minutes (Table A2). Baseline separation was achieved for all analytes (Fig. 1). Detector response was obtained from the digital integrator operating in the peak height mode. Analytical precision (% RSD) ranged from 0.4 to 3.9% (mean = 1.6%) as determined by the pooled standard deviation of triplicate initial measurements from both studies (Table A2).

Prior to conducting the two experiments described above, a preliminary leaching study was conducted. This study was conducted to determine if any substances leached from the (four) casing materials that could interfere with our analytical method. For this study, two pieces of each type of well casing were placed in each of two vials. The vials were then filled with fresh well water so that there was no headspace, capped and allowed to sit for one week. An aliquot was taken from each vial and analyzed. No detectable peaks were observed in any of the samples.

RESULTS AND DISCUSSION

In the first experiment we compared the four well casing materials with control samples to determine whether there were any losses of the 10 analytes from solution. The complete data from this study are presented in Appendix Tables A3-A12. These data are summarized in Table 1, where the normalized concentrations for the well casings

are given with time. For each analyte and time, a one-way analysis of variance test (ANOVA) was performed to determine if the well casing material had any significant effect (at the 95% confidence level). When significant differences were found, a multiple range test was also performed to determine which materials were significantly different from each other. Those values that were significantly different from the control samples were marked with an asterisk in Table 1.

Examining these data reveals that 1) the stainless steel well casings did not affect the concentration of any of the analytes in solution, while PVC and Teflon casings did affect the concentration of some of the analytes, 2) the effect of PVC was considerably less than that of Teflon, and 3) the amount of analyte lost varied with the substance. As an example, Figure 2 shows the concentration of MDCB as a function of time for the four well casing materials. There was no loss of analyte in the samples that contained either stainless steel casing. Loss of MDCB was slow in those samples that contained PVC casing; after 1000 hours the loss was 20%. However, for the samples containing Teflon casing, loss was much more rapid; 20% of the MDCB was lost within the first 24 hours and over 70% was lost after 1000 hours.

There were no statistically significant losses of RDX or TNB in solutions containing any of the well casing materials, even after 1000 hours (Table 1). Loss of MNT was only statistically significant after 1000 hours, when 10% was lost in the samples containing Teflon casings. However, there was significant loss of the remainder of the substances in samples containing Teflon casings and for many of those containing PVC casings.

Loss of CDCE in samples containing Teflon

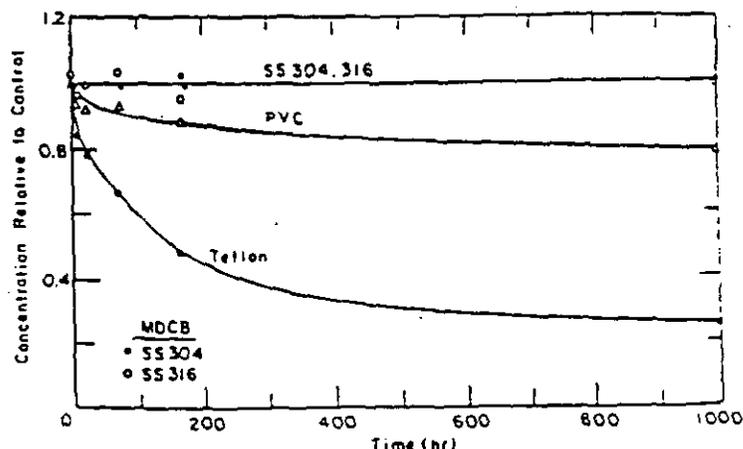


Figure 2. Sorption of MDCB by the four well casing materials.

Table 1. Normalized* concentrations of analytes for the four well casings with time.

Analyte	Treatment	1 hour	8 hours	24 hours	72 hours	168 hours	1000 hours
RDX	PTFE	1.03	1.00	1.00	1.02	0.91	0.99
	PVC	1.01	1.00	0.98	1.00	1.02	1.00
	SS304	0.99	0.99	1.01	1.02	1.10	0.98
	SS316	1.01	0.99	1.01	1.02	1.11	1.00
TNB	PTFE	1.01	1.00	1.00	0.98	0.95	1.01
	PVC	1.01	1.00	0.98	1.02	1.01	1.02
	SS304	0.99	1.00	1.00	1.05	1.07	1.00
	SS316	1.02	0.99	1.01	1.07	1.06	1.02
C12DCE	PTFE	1.01	0.96 [†]	0.96 [†]	0.94	0.91 [†]	0.79 [†]
	PVC	1.00	0.99	0.95 [†]	0.96	0.95	0.90
	SS304	0.97	1.00	1.00	0.96	1.04	0.98
	SS316	0.95	0.99	1.00	1.01	0.98	0.99
T12DCE	PTFE	1.00	0.92 [†]	0.88 [†]	0.83	0.66	0.56 [†]
	PVC	1.00	0.98	0.93 [†]	1.06	0.83	0.83
	SS304	0.95 [†]	1.00	1.00	0.96	1.11	1.00
	SS316	1.00	0.99	1.00	1.12	1.03	1.00
MNT	PTFE	1.03	1.00	0.99	0.99	0.90	0.90 [†]
	PVC	1.02	1.00	0.98	1.05	0.99	0.94
	SS304	1.00	1.00	1.01	1.00	1.08	1.07
	SS316	1.02	1.00	1.02	1.08	1.10	0.99
TCE	PTFE	1.00	0.90 [†]	0.85 [†]	0.78 [†]	0.64 [†]	0.40 [†]
	PVC	1.01	0.98	0.94 [†]	0.99	0.94 [†]	0.88 [†]
	SS304	0.96	1.00	1.01	0.96	1.04	0.99
	SS316	1.00	0.99	1.00	1.04	0.98	1.00
CLB	PTFE	1.01	0.93 [†]	0.90 [†]	0.85 [†]	0.74 [†]	0.51 [†]
	PVC	1.01	0.98	0.95 [†]	0.98	0.94 [†]	0.86 [†]
	SS304	0.98	1.00	1.00	0.97	1.05	0.99
	SS316	0.99	0.99	1.01	1.04	0.98	0.99
ODCB	PTFE	1.01	0.91 [†]	0.88 [†]	0.81 [†]	0.68 [†]	0.43 [†]
	PVC	1.02	0.97 [†]	0.94 [†]	0.98	0.93	0.86 [†]
	SS304	0.98	0.99	1.00	0.99	1.04	1.00
	SS316	1.01	0.98 [†]	1.01	1.03	0.98	1.00
PDCB	PTFE	0.92 [†]	0.84 [†]	0.77 [†]	0.64 [†]	0.47 [†]	0.26 [†]
	PVC	0.95	0.95 [†]	0.92 [†]	0.97	0.88 [†]	0.80 [†]
	SS304	0.91 [†]	0.98	1.00	0.98	1.02	1.02
	SS316	0.94	0.97 [†]	1.00	1.04	0.97	1.02
MDCB	PTFE	1.00	0.84 [†]	0.78 [†]	0.66 [†]	0.48 [†]	0.26 [†]
	PVC	1.02	0.95 [†]	0.92 [†]	0.97	0.88 [†]	0.80 [†]
	SS304	0.99	0.96 [†]	1.00	0.99	1.02	1.02
	SS316	1.03	0.96 [†]	1.00	1.04	0.96	1.01

* The values given here are determined by dividing the mean concentration of a given analyte at a given time and for a particular well casing by the mean concentration (for the same analyte) of the control samples taken at the same time.

[†] Values significantly different from control values.

well casings was relatively slow; losses did not exceed 10% until after 72 hours (Fig. 3). Loss of this compound never exceeded 6% for the samples containing PVC casings.

The trans-isomer of 1,2DCE (TDCE) was lost more rapidly than the cis-isomer from solutions containing Teflon casings (Fig. 4). Generally, loss was significantly greater in the samples with the Teflon casings than in the samples with the PVC

casings (Fig. 4). Because significant loss occurred after only 8 hours in samples containing Teflon casings (8% loss), this could impact the water quality of samples taken from wells with longer recharge times (8 to 24 hours). However, this seems less likely for PVC cased wells since loss was only 7% after 24 hours. After 1000 hours, loss was 44% in samples containing Teflon casings and 17% for those containing PVC casings.

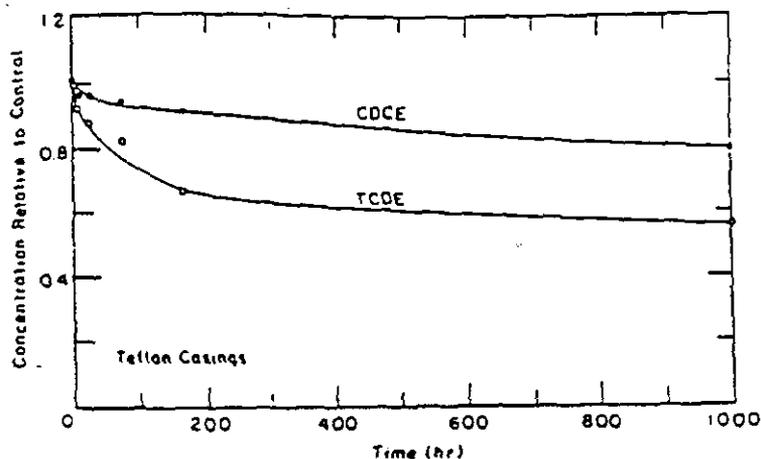


Figure 3. Sorption of CDCE and TDCE by Teflon well casings.

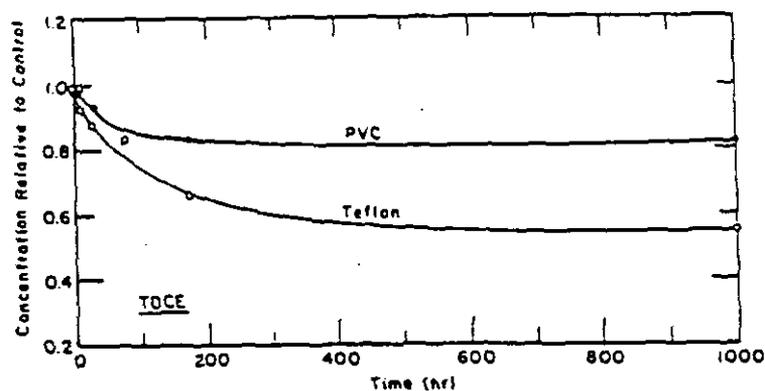


Figure 4. Sorption of TDCE by plastic casings.

The results for TCE were very similar to those of TDCE, except that the final loss was more in samples containing Teflon casings; loss was 15% after 24 hours and 60% after 1000 hours (Fig. 5). For the samples that contained PVC casings, loss was 5% after 24 hours and only 12% after 1000 hours.

A similar pattern of loss was seen with CLB, ODCB, MDCB, and PDCB. Figure 6 shows the rate of loss of these compounds for samples that contained Teflon casings. The order of loss was MDCB and PDCB > ODCB > CLB. After eight hours, losses were significant in the samples containing Teflon casings; loss was 7% for CLB, 9% for ODCB, and 16% for PDCB and MDCB. For PDCB loss was significant after only 1 hour (8% loss). Although loss of CLB isomers was significant in the samples that contained PVC well casings after only 8 hours, loss was less than 5%. Even after 24 hours losses were less than 10% for CLB and the three DCB compounds.

We also tested the 1000-hour samples to detect if there was any desorption of the sorbed organics

from the well casings. After 3 days no analytes were detected in the samples containing either type of stainless steel casing. These results were as expected since no organic had a measurable loss in the samples containing the stainless steel casings. However, for samples containing plastic casings, we did recover measurable quantities of all the organics where significant losses had been observed in the sorption experiment. The results are given in Table 2. While this experiment did not give us any of the kinetics of desorption, generally the amount of analyte desorbed closely paralleled the amount sorbed. No RDX or TNB was recovered from either casing. For those substances that were sorbed, the amount of MNT recovered was the lowest for both casings, and the amount of CDCE recovered was next lowest. However, it is interesting that, for the samples containing Teflon casings, the compounds that were sorbed to the greatest extent (PDCB and MDCB) were not necessarily the substances that were desorbed to the greatest extent (TCE and TDCE were). Diffusion

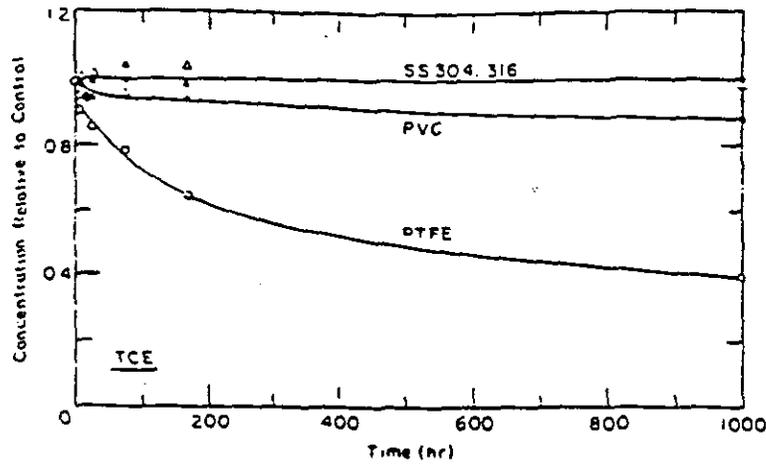


Figure 5. Sorption of TCE by the four well casing materials.

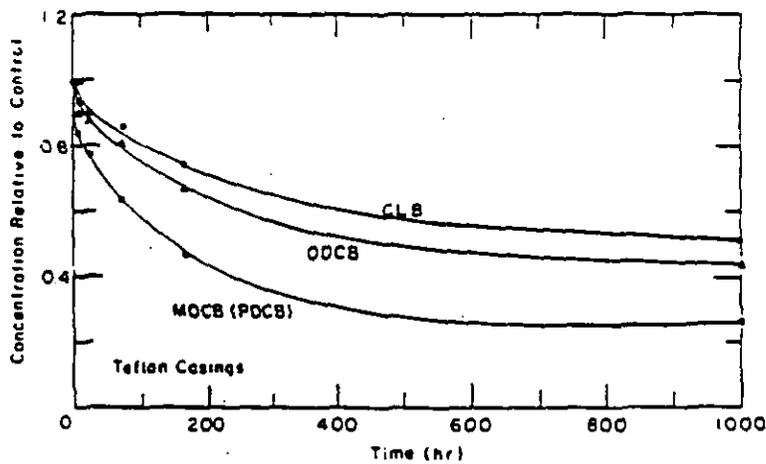


Figure 6. Sorption of CLB, ODCB, MDCB and PDCB by Teflon well casings.

Table 2. Results of desorption study.

Casing material	Concentration in mg/L after 3 days equilibration									
	RDX	TNB	CDCE	TDCE	MNT	TCE	CLB	ODCB	PDCB	MDCB
Teflon	ND	ND	0.20	0.43	0.075	0.47	0.28	0.38	0.30	0.35
	ND	ND	0.21	0.45	0.076	0.48	0.28	0.35	0.34	0.36
	ND	ND	0.01*	0.06*	0.074	0.10*	0.06*	0.09*	0.10*	0.12*
PVC	ND	ND	0.079	0.15	0.046	0.14	0.10	0.15	0.17	0.18
	ND	ND	0.080	0.14	0.046	0.12	0.10	0.15	0.16	0.21
	ND	ND	0.080	0.15	0.043	0.13	0.11	0.16	0.16	0.20

* Sample probably had a loose cap.
 ND—Not detected.

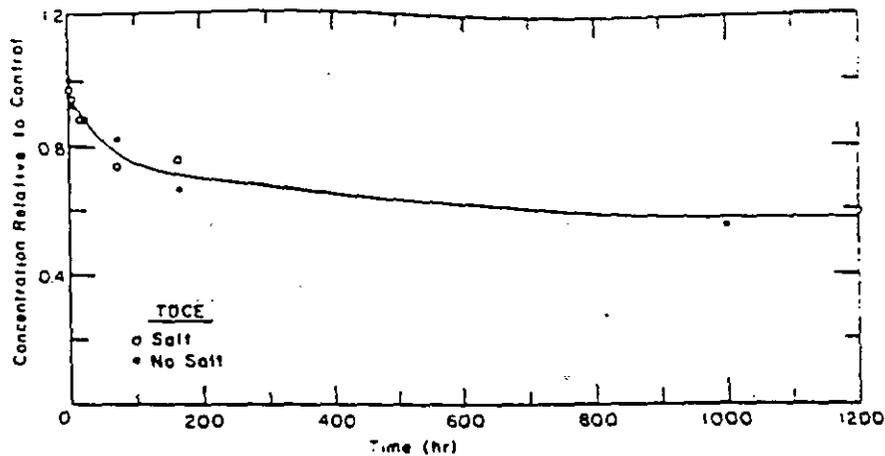


Figure 7. Sorption of TDCE by Teflon well casings in the presence and absence of salt.

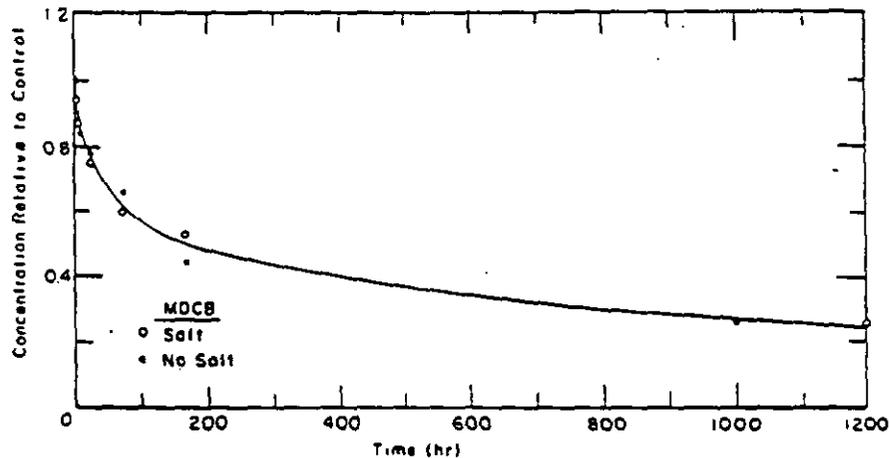


Figure 8. Sorption of MDCB by Teflon well casings in the presence and absence of salt.

out of the polymer may be more rapid for the smaller, more planar molecules.

Experiment with NaCl-amended ground water

In the second experiment we added NaCl to raise the chloride concentration above 1000 mg/L. High chloride concentrations are known to be corrosive to 304 stainless steel. Specifically, we wondered if over the long term, rusting would have any effect on the performance of the stainless casings. It is also possible that sorption on plastic materials would increase with increasing ionic strength.

Rusting of the stainless casings was visible after only 8 hours for SS 304 and, after 24 hours for SS 316. However, the addition of sodium chloride did not seem to affect the rate of loss of any of the analytes studied for either the stainless steel or

plastic casings. Tables A13-A22 give the data for the 10 analytes. The data were analyzed using standard analysis of variance to determine any significant effects, and multiple range tests were performed to determine which materials were significantly different from each other. Table A23 summarizes the data by giving the normalized values for the well casings; values that were significantly different from the control values are marked with an asterisk.

Figures 7 and 8 are plots of the concentrations of TDCE and MDCB, respectively, as a function of time for sample solutions, with and without added chloride, containing Teflon casings. Clearly the addition of salt did not markedly affect the rate or amount of sorption of these analytes. This was also found to be true when similar plots were drawn for the TCE, CLB, and ODCB.

Table 3. Several physical constants of the analytes.

Analyte	Molecular weight ¹	Solubility (mg/L)	Log K_{ow}
RDX	222.15	43 ⁴	0.87 ² 0.88 ⁶
TNB	213.11	—	1.18 ⁶
CDCE	96.95	3500 ⁵	1.63 ⁷
TDCE	96.95	6300 ⁵	1.93 ⁷
MNT	137.13	498 ¹	2.45 ³
TCE	131.40	1100 ⁵	2.29 ² 2.42 ³
CLB	112.56	488 ¹	2.18 ³ 2.46 ³ 2.84 ³
ODCB	147.01	145 ¹	3.38 ³ 3.40 ²
PDCB	147.01	79 ¹	3.38 ³ 3.39 ³ 3.37 ²
MDCB	147.01	123 ¹	3.39 ³ 3.44 ²

1. Hodgman (1955).
2. Banerjee et al. (1980).
3. Hansch and Leo (1979).
4. Leggett (CRREL, pers. comm, 1986).
5. Mackay and Shiu (1981).
6. Jenkins (1989).
7. Estimated for HPLC capacity factor (McDuffie 1981).

Relationship of sorption to analyte properties

We performed regression analyses on the concentration of analyte in the 1000-hour samples containing either the PVC or Teflon casings vs the corresponding aqueous solubilities, molecular weights, or log octanol/water partition coefficients (K_{ow}) of the analytes (see Table 3 for constants). We found a statistically significant (95% confidence level) inverse relationship between the concentration of analyte relative to the control samples and the log K_{ow} values for both the PVC and Teflon casings. Tables A24 and A25 summarize the regression analyses for the Teflon and PVC data respectively, and Figures 9 and 10 show the corresponding plots of (normalized) concentration of each analyte vs its log K_{ow} . The relationship with K_{ow} was the most highly significant and the only one that was significant when a stepwise multiple regres-

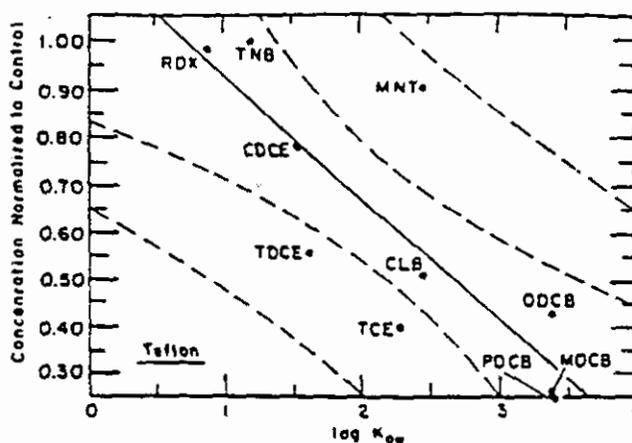


Figure 9. Regression analysis for concentration vs log K_{ow} for samples containing Teflon casings.

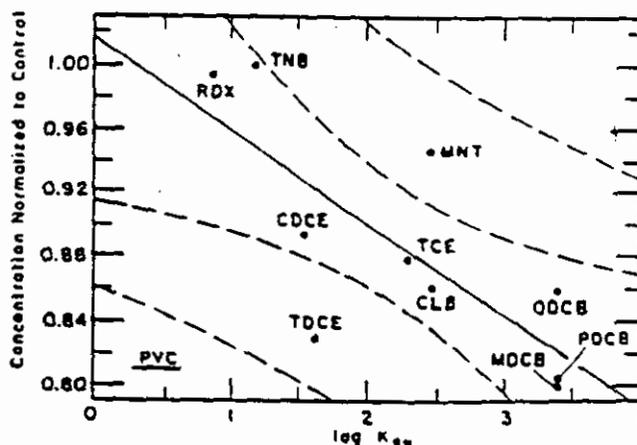


Figure 10. Regression analysis for concentration vs log K_{ow} for samples containing PVC casings.

sion was performed to test for the addition of the other variables (using "Statgraphics" software by STSC Inc., Rockville, Md.).

Modeling the sorption process

While these experiments clearly demonstrated that the loss of organic chemicals from solutions exposed to plastic casing materials is a sorption process, it is not clear whether this is a surface phenomenon or whether penetration into the polymer matrix occurred. During the desorption studies the sorbed analytes were released back into solution, thereby demonstrating that the process is at least partially reversible. While surface adsorption cannot be ruled out, the evidence suggests that diffusion into the polymer matrix occurred. Zhang et al. (1988) showed that organic molecules penetrate plasticized PVC membranes. In our ex-

periments sorption appears to be slow (taking hundreds of hours to attain equilibrium), which suggests that partitioning into the bulk of the material occurs. Desorption of some analytes from Teflon also appeared to be slow. If we assume this to be the case, the process can be modeled using classical partitioning by treating the plastic casing as an immiscible liquid phase in contact with water and relating partitioning of individual analytes to their K_{ow} values. While immiscible liquids other than octanol may be better structural models of Teflon or PVC, the most extensive collection of partition coefficients is available for octanol. This is because K_{ow} values have been used successfully to predict the behavior of drugs in the human body and the sorption of environmental pollutants on sediments and soils.

Because it appears that we can predict the behavior of the various analytes exposed to plastic casings on the basis of their K_{ow} values, we modeled the partitioning process as follows. First, if we assume that the sorption process is a simple, reversible first order process (eq 2), we can write the rate equation as shown in eq 3 (Gould 1959):

$$X_{aq} \frac{k_1}{k_2} X_{sorb} \quad (2)$$

$$\frac{d[X_{aq}]}{dt} = -k_1[X_{aq}] + k_2[X_{sorb}] \quad (3)$$

where $[X_{aq}]$ = the concentration of the analyte X in aqueous solution

$[X_{sorb}]$ = the concentration of analyte X sorbed in the plastic material

k_1 = the rate constant for sorption

k_2 = the rate constant for desorption

t = time in hours.

Since in our experiments the volume of the solution was 10 times the volume of the plastic casing, or

$$V_{aq} = 10 \cdot V_{sorb} \quad (4)$$

then

$$[X_{aq}] = [X_0] - \left[\frac{X_{sorb}}{10} \right] \quad (5)$$

where $[X_0]$ is the initial concentration of X in solution. Solving for $[X_{sorb}]$ we have

$$[X_{sorb}] = 10([X_0] - [X_{aq}]) \quad (6)$$

Substituting back into eq 2 we have

$$\frac{d[X_{aq}]}{dt} = -k_1[X_{aq}] + k_2(10[X_0] - [X_{aq}]) \quad (7)$$

Regrouping terms we have

$$\frac{d[X_{aq}]}{dt} = -(k_1 + k_2)[X_{aq}] + 10k_2[X_0] \quad (8)$$

Since k_1 , k_2 and $[X_0]$ are constants, we can rewrite this as

$$\frac{d[X_{aq}]}{dt} = -A[X_{aq}] + B \quad (9)$$

where

$$A = k_1 + k_2 \quad (10)$$

and

$$B = 10k_2[X_0] \quad (11)$$

If we then integrate the rate equation we have a nonlinear relation for $[X_{aq}]$ as a function of t and

$$\frac{\ln[A[X_{aq}] + B]}{A} = t \quad (12)$$

two constants A and B.

We obtained the optimal values for A and B for each analyte, where sorption loss was observed, by application of the Gauss-Newton method of nonlinear curve fitting using the measured aqueous concentrations at 1, 8, 24, 72, 128 and 1000 hours. Then using these values for A and B, we simultaneously solved eq 10 and 11 for each analyte to obtain values for k_1 and k_2 (the rate constants for the forward and reverse processes). The values for Teflon are shown in Table 4. Since the process we describe is assumed to be reversible and of first order, the ratio of the rate constants, k_1/k_2 , is the equilibrium constant, K_{eq} . The K_{eq} and the $\log K_{ow}$ values for each analyte are also given in Table 4.

When we plotted the eight values of K_{eq} given in Table 4 vs $\log K_{ow}$ six of the eight points appeared to fall on a straight line, while the points for MNT and ODCB did not (Fig. 11). The poor fit for MNT and the lack of significant sorption for TNB and RDX can be explained by the tendency of nitro-containing organic molecules to form strong hydrogen bonds, which keeps them in solution. While octanol can be a donor in hydrogen bonding, Teflon cannot. Thus, if we predict partitioning into Teflon for these molecules based on their octanol/water coefficients, we will overestimate the amount of sorption. For example, based on our

Table 4. Sorption (k_1) and desorption (k_2) rate constants and equilibrium constant (K_{eq}) for exposure to Teflon.

Analyte	$k_1 \times 10^2$ (cm^{-1})	$k_2 \times 10^4$ (cm^{-1})	K_{eq}	Log K_{eq}
RDX	.	.	-	0.88 ²
TNB	.	.	-	1.18 ¹
MNT	0.699	3.100	22.3	2.40 ¹
C12DCE	1.590	6.253	25.4	1.63 ³
T12DCE	1.935	6.116	31.6	1.93 ¹
ODCB	1.100	3.064	35.9	3.38 ¹
CLB	0.827	2.300	36.0	2.46 ¹
TCE	1.543	4.067	37.9	2.29 ¹
MDCB	1.408	2.779	50.7	3.38 ¹
PDCB	1.558	3.005	51.9	3.39 ¹

1. Hansch and Leo (1979).

2. Jenkins (1989).

3. Estimated for RP-HPLC capacity factor using method of McDuffie (1981).

* Loss not statistically significant so no estimate possible.

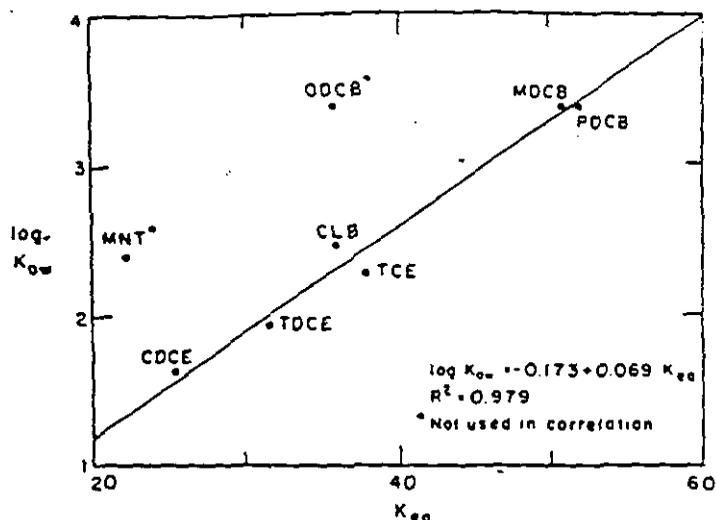


Figure 11. Correlation between log octanol-water partition coefficient (K_{ow}) and equilibrium constant (K_{eq}) for solutes exposed to Teflon well casing.

regression equation we predict a K_{eq} of 38 for MNT; however, the observed K_{eq} was only 22.3.

The poor prediction for ODCB can be explained by the well known "ortho effect." This effect is a complex combination of electronic and steric influences, which often results in ortho-di-substituted aromatic molecules behaving much differently than the meta- and para-isomers.

We did not create a similar model predicting the loss of analyte for PVC because the percent sorbed was small when compared with the experimental error and this would create an unacceptable degree of uncertainty in the calculated rate constants.

Therefore, we conclude that, for hydrophobic organic molecules that are not subject to hydrogen bonding, the relationship presented in Figure 11 can be used to estimate the equilibrium partitioning of an analyte between the aqueous phase and Teflon. Obviously, in a well, the ground water is refreshed and one would not observe the levels of depletion we observed in our study. However, eventually the plastic casing should reach equilibrium with the aqueous phase if the concentration of the analyte in ground water is relatively constant with time.

While K_{eq} will determine the equilibrium concentrations of each analyte in the water and plastic phases, it is the magnitude of k_1 that will determine how quickly various analytes are depleted. For

small, planar molecules like TCE, the k_1 values are quite high compared to those of the other analytes. This may explain the rapid loss of tetrachloroethylene from solutions containing Teflon casings observed by Miller (1982) and Reynolds and Gillham (1986).

CONCLUSIONS AND RECOMMENDATIONS

These studies indicate that Teflon was clearly the poorest choice of the four well casing materials tested when samples are to be analyzed for trace level organics. Significant losses of all the chlorinated compounds occurred within 1-8 hours, and one nitroaromatic compound was also lost after prolonged exposure (1000 hours). While losses were also seen for several compounds exposed to PVC, the rate of loss was always much slower than for the Teflon casings; usually 24 hours lapsed before significant losses occurred. There was no loss of any organic tested in the presence of either SS casing. However, rusting of both types of stainless casings occurred relatively quickly, in some instances overnight.

The desorption study showed that loss of organics from aqueous solution is due to a sorption process, and that the sorption process is partially reversible. Desorption from well casing material

could result in falsely high concentrations of analytes if their concentrations were to decrease in the well water.

We were able to correlate the loss of hydrophobic organic constituents in the well water containing Teflon casings with the substance's K_{ow} values. However, for hydrophilic organic substances this correlation overestimates losses.

Our results indicate that in a monitoring situation, where the well is purged and then sampled within 8–24 hours, PVC cased wells are probably suited for sampling most organics while Teflon cased wells are probably not. However, there are two conflicting effects that must be considered when extrapolating our test data to a real monitoring situation: 1) we tested casings, not well screens; the greater surface area of well screens could substantially increase the rate of sorptive losses in the screened portion of the well, and 2) this experiment was conducted under static conditions. If there is a long time between purging the well and sampling, it is possible that the water being sampled would be at least partially replenished, and this would tend to mitigate losses due to sorption by the casing material.

The larger question is what is the best casing for ground water monitoring? Our study attempts to answer only part of the question—how suitable are these four well casing materials for monitoring organic constituents? Inorganic constituents must also be considered and for that we refer the reader to Hewitt (1989). Hewitt's results for inorganics show nearly opposite behavior. He found that Teflon casings were the best for monitoring four species of metals (Cd, Cr, As and Pb) while stainless steel casings were the worst; rusting by the stainless steel casings presented serious problems with several of the analytes. Clearly, selecting a single casing material, from those tested, for monitoring both inorganic and organic constituents in ground water will necessarily have to involve compromise.

LITERATURE CITED

Banerjee, S., S. H. Yalkowsky and S.C. Valvani (1980) Water solubility and octanol/water partition coefficients of organics: Limitations of the solubility-partition coefficient correlation. *Environmental Science and Technology*, 14(10): 1227–1229.

Banzer, J. D. (1977) The migration of vinyl chloride monomer from PVC pipe into water. In *Proceedings of the Safety and Health with Plastics National*

Technical Conference, Denver, Colorado. Society of Plastic Engineering, p 30–34.

Barcelona, M.J. and J.A. Helfrich (1986) Well construction and purging effects on groundwater samples. *Environmental Science and Technology*, 20: 1179–1184.

Barcelona, M.J., J.P. Gibb and R.A. Miller (1984) A guide to the selection of materials for monitoring well construction and ground water sampling. U.S. Environmental Protection Agency, Report no. EPA-600/2-84-024.

Barcelona, M.J., J.A. Helfrich and E.E. Garske (1985) Sampling tubing effects on ground water samples. *Analytical Chemistry*, 57: 460–464.

Berens, A.R. and H.B. Hopfenberg (1982) Diffusion of organic vapors at low concentrations in glassy PVC, polystyrene, and PMMA. *Journal of Membrane Sciences*, 10: 283.

Boettner, E.A., G.L. Ball, Z. Hollingsworth and R. Aquino (1981) Organic and organotin compounds leached from PVC and CPVC pipe. U.S. Environmental Protection Agency, Report Number EPA-600/1-81-062.

Curran, C.M. and M.B. Tomson (1983) Leaching of trace organics into water from five common plastics. *Groundwater Monitoring Review*, 3: 68–71.

Desrosiers, D.G. and P.C. Dunnigan (1983) The diffusion of chloroform and carbon tetrachloride from rigid PVC pipe and rigid CPVC pipe into water. *Journal of Vinyl Technology*, 5(4): 187–191.

Dietz, G.R., J.D. Banzer and E.M. Miller (1977) Water extraction of additives from PVC pipe. In *Proceeding of the Safety and Health With Plastics National Technical Conference, Denver, Colorado*. Society of Plastic Engineering, p. 25–29.

Dressman, R.C. and E.F. McFarren (1978) Determination of vinyl chloride migration from polyvinyl chloride pipe into water. *Journal of the American Water Works Association*, January, p. 29–30.

Gossett, R.E. and R.O. Hegg (1987) A comparison of three sampling devices for measuring volatile organics in ground water. *Transactions of the American Society of Agricultural Engineers*, 30(2), p. 387–390.

Gould, E. S. (1959) *Mechanism and Structure in Organic Chemistry*. New York: Holt, Rinehart and Winston, p. 166–167.

Gross, R.C., B. Engelbart and S. Walter (1974) Aqueous extraction of lead stabilizers from PVC compounds. Society of Plastic Engineering, Technical Paper 20, p. 529–531.

Hansch C. S and A. Leo (1979) *Substitution Constants for Correlation Analysis in Chemistry and Biology*. New York: Wiley.

APPENDIX A: TEST DATA

Table A1. Time at which absorption reduced the relative concentration (C/C₀) in solution to 0.9.¹

Polymer	Least Absorption ----- Most Absorption				
	TRI	TET	BRO	HEX	TEY
PVC	> 5 weeks	- 2 weeks	- 3 days	- 1 day	- 1 day
PTFE	BRO > 5 weeks	TET - 2 weeks	TRI - 1 day	HEX - 1 day	TEY < 5 min.
Log (undecane/water partition coefficient)	TET 2.04	BRO 2.10	TRI 2.62	HEX Not Reported	TEY 3.43
Water Solubility (mg/L)	BRO 3100	TET 2962	TRI 1495	TEY 150	HEX 50
Log (Octanol/Water partition coefficient)	BRO 2.30	TRI 2.49	TET 2.56	TEY 2.60	HEX 3.34

TRI ~ 1,1,1-trichloroethane
 TET ~ 1,1,2,2-tetrachloroethane
 BRO ~ bromoform
 HEX ~ hexachloroethane
 TEX ~ tetrachloroethylene

¹ Reynolds and Gillham, 1986.

Table A2. Retention times and analytical precision.

Substance	Abbreviation	Precision RSD (%)
1,3,5-trinitro-1,3,5-triazine	RDX	1.0
1,3,5-trinitrobenzene	TNB	0.9
cis-1,2-dichloroethylene	CDCE	3.9
trans-1,2-dichloroethylene	TDCE	1.9
m-nitrotoluene	MNT	0.4
trichloroethylene	TCE	2.2
chlorobenzene	CLB	1.6
o-dichlorobenzene	ODCB	1.4
p-dichlorobenzene	PDCE	1.6
m-dichlorobenzene	MDCB	1.5

Table A3. Concentration of RDX with time.

treatment	Concentration mg/L						
	0hr	1hr	8hr	24hr	72hr	168hr	1000hr
SS304	1.79	1.74	1.72	1.82	1.75	2.19	1.63
SS304	1.79	1.75	1.70	1.77	1.76	2.20	1.70
SS304	1.78	1.69	1.71	1.72	1.76	1.90	1.69
SS316	1.79	1.76	1.70	1.77	1.75	2.20	1.71
SS316	1.79	1.76	1.71	1.77	1.76	2.19	1.72
SS316	1.78	1.78	1.70	1.75	1.75	1.97	1.70
PVC	1.79	1.76	1.70	1.74	1.73	2.19	1.70
PVC	1.79	1.78	1.72	1.71	1.76	1.74	1.70
PVC	1.78	1.73	1.72	1.70	1.69		1.71
TEFLON	1.79	1.77	1.72	1.77	1.73	1.73	1.67
TEFLON	1.79	1.90	1.72	1.76	1.77	1.73	1.69
TEFLON	1.78	1.73	1.72	1.70	1.74	1.73	1.71
CONTROL	1.79	1.74	1.74	1.76	1.77	2.14	1.67
CONTROL	1.79	1.75	1.70	1.74	1.70	1.68	1.68
CONTROL	1.78	1.74	1.72	1.74	1.69		1.77

Table A4. Concentration of TNB with time.

treatment	Concentration mg/L						
	0hr	1hr	8hr	24hr	72hr	168hr	1000hr
SS304	2.37	2.29	2.28	2.33	2.26	2.52	2.19
SS304	2.37	2.28	2.26	2.35	2.33	2.53	2.30
SS304	2.36	2.25	2.27	2.28	2.34	2.26	2.32
SS316	2.37	2.35	2.25	2.34	2.45	2.50	2.33
SS316	2.37	2.32	2.27	2.32	2.28	2.45	2.31
SS316	2.36	2.31	2.26	2.34	2.35	2.31	2.30
PVC	2.37	2.32	2.25	2.25	2.09	2.53	2.24
PVC	2.37	2.36	2.28	2.25	2.20	2.18	2.34
PVC	2.36	2.28	2.28	2.29	2.45	2.19	2.34
TEFLON	2.37	2.34	2.28	2.33	2.29	2.24	2.28
TEFLON	2.37	2.36	2.28	2.35	2.12	2.06	2.29
TEFLON	2.36	2.27	2.27	2.24	2.09	2.18	2.32
CONTROL	2.37	2.29	2.29	2.29	2.16	2.48	2.23
CONTROL	2.37	2.30	2.26	2.33	2.17	2.27	2.28
CONTROL	2.36	2.29	2.28	2.31	2.27	2.08	2.30

Table A5. Concentration of MNT with time.

treatment	Concentration mg/L						
	0hr	1hr	8hr	24hr	72hr	168hr	1000hr
SS304	2.32	2.20	2.23	2.30	2.20	2.63	
SS304	2.32	2.23	2.23	2.33	2.23	2.70	2.02
SS304	2.30	2.25	2.23	2.25	2.23	2.29	
SS316	2.32	2.33	2.23	2.33	2.57	2.75	2.00
SS316	2.32	2.23	2.22	2.32	2.25	2.61	2.02
SS316	2.30	2.24	2.23	2.32	2.24	2.41	2.08
PVC	2.32	2.30	2.21	2.21	2.21	2.60	2.02
PVC	2.32	2.32	2.22	2.21	2.25	2.24	1.87
PVC	2.30	2.20	2.23	2.25	2.53	2.15	1.95
TEFLON	2.32	2.31	2.22	2.28	2.17	2.13	
TEFLON	2.32	2.32	2.22	2.30	2.20	2.11	1.87
TEFLON	2.30	2.20	2.22	2.19	2.17	2.09	1.84
CONTROL	2.32	2.21	2.22	2.26	2.22	2.65	2.11
CONTROL	2.32	2.23	2.22	2.30	2.28	2.28	
CONTROL	2.30	2.22	2.24	2.28	2.13	2.12	2.01

Table A6. Concentration of CDCE with time.

treatment	Concentration mg/L						
	0hr	1hr	8hr	24hr	72hr	168hr	1000hr
SS304	2.79	2.55	2.78	2.57	2.51	2.37	2.21
SS304	2.84	2.66	2.76	2.63	2.35	2.42	
SS304	2.73	2.60	2.75	2.62	2.10	2.41	
SS316	2.79	2.59	2.76	2.64	2.52	2.37	2.09
SS316	2.84	2.68	2.72	2.59	2.46	2.16	2.30
SS316	2.73	2.39	2.73	2.56	2.37		2.26
PVC	2.79	2.71	2.75	2.44	2.33	2.24	1.94
PVC	2.84	2.73	2.76	2.47	2.31	2.24	1.95
PVC	2.73	2.61	2.68	2.48	2.34	2.12	2.15
TEFLON	2.79	2.74	2.68	2.48	2.28	2.19	1.77
TEFLON	2.84	2.74	2.64	2.45	2.33	2.10	1.76
TEFLON	2.73	2.64	2.66	2.52	2.21	2.02	
CONTROL	2.79	2.66	2.79	2.57	2.43	2.38	2.13
CONTROL	2.84	2.71	2.79	2.63	2.56	2.38	2.36
CONTROL	2.73	2.68	2.72	2.60	2.26	2.19	

Table A7. Concentration of TDCE with time.

treatment	Concentration mg/L						
	0hr	1hr	8hr	24hr	72hr	168hr	1000hr
SS304	2.71	2.43	2.70	2.43	2.29	2.86	
SS304	2.77	2.53	2.68	2.44	2.22	2.93	2.03
SS304	2.63	2.38	2.67	2.39	1.87	2.21	
SS316	2.71	2.66	2.66	2.46	2.87		1.89
SS316	2.77	2.47	2.65	2.41	2.37	2.84	2.13
SS316	2.63	2.58	2.69	2.35	2.21	2.07	2.11
PVC	2.71	2.62	2.58	2.22	2.11	1.99	1.82
PVC	2.77	2.61	2.66	2.25	2.11	1.99	1.64
PVC	2.63	2.48	2.64	2.26	2.83	1.97	1.63
TEFLON	2.71	2.62	2.48	2.08	1.82	1.64	
TEFLON	2.77	2.62	2.45	2.16	1.88	1.49	1.13
TEFLON	2.63	2.50	2.46	2.13	1.83	1.58	1.14
CONTROL	2.71	2.55	2.70	2.38	2.22	2.88	2.14
CONTROL	2.77	2.61	2.72	2.44	2.34	2.15	
CONTROL	2.63	2.58	2.63	2.41	2.12	2.14	1.95

Table A8. Concentration of TCE with time.

treatment	Concentration mg/L						
	0hr	1hr	8hr	24hr	72hr	168hr	1000hr
SS304	2.80	2.48	2.77	2.55	2.38	2.29	
SS304	2.85	2.59	2.75	2.60	2.29	2.36	2.21
SS304	2.71	2.52	2.74	2.52	1.96	2.40	
SS316	2.80	2.76	2.74	2.61	2.40	2.31	2.08
SS316	2.85	2.53	2.72	2.56	2.47		2.30
SS316	2.71	2.64	2.76	2.48	2.34	2.15	2.29
PVC	2.80	2.70	2.67	2.37	2.31	2.17	2.07
PVC	2.85	2.72	2.74	2.40	2.33	2.15	1.92
PVC	2.71	2.55	2.72	2.42	2.25	2.11	1.89
TEFLON	2.80	2.69	2.50	2.13	1.72	1.53	
TEFLON	2.85	2.68	2.47	2.20	1.86	1.41	0.89
TEFLON	2.71	2.53	2.48	2.15	1.80	1.43	0.90
CONTROL	2.80	2.61	2.78	2.51	2.31	2.32	2.31
CONTROL	2.85	2.67	2.80	2.57	2.46	2.31	
CONTROL	2.71	2.64	2.70	2.55	2.16	2.18	2.14

Table A9. Concentration of CLB with time.

treatment	Concentration $\mu\text{g/L}$						
	0hr	1hr	8hr	24hr	72hr	168hr	1000hr
SS304	2.18	1.95	2.13	2.04	1.95	1.90	
SS304	2.12	2.03	2.13	2.08	1.91	1.97	1.56
SS304	2.13	2.05	2.14	2.12	1.73	1.98	
SS316	2.18	1.98	2.14	2.05	1.98	1.91	1.47
SS316	2.12	2.05	2.12	2.09	2.03		1.59
SS316	2.13	2.06	2.11	2.12	1.97	1.80	1.62
PVC	2.18	2.12	2.08	1.93	1.88	1.78	1.32
PVC	2.12	2.12	2.11	1.95	1.91	1.78	1.32
PVC	2.13	2.00	2.13	1.97	1.84	1.74	1.43
TEFLON	2.18	2.12	1.98	1.84	1.57	1.45	
TEFLON	2.12	2.12	1.99	1.85	1.69	1.35	0.81
TEFLON	2.13	1.99	2.00	1.90	1.64	1.37	0.81
CONTROL	2.18	2.05	2.11	2.05	1.92	1.93	
CONTROL	2.12	2.04	2.15	2.07	2.03	1.92	1.51
CONTROL	2.13	2.08	2.16	2.10	1.80	1.82	4.62

Table A10. Concentration of ODCB with time.

treatment	Concentration $\mu\text{g/L}$						
	0hr	1hr	8hr	24hr	72hr	168hr	1000hr
SS304	2.31	2.12	2.28	2.22	2.16		
SS304	2.36	2.20	2.29	2.28	2.12	2.16	1.97
SS304	2.40	2.21	2.29	2.32	1.99	2.17	
SS316	2.31	2.14	2.26	2.25	2.16	2.13	1.87
SS316	2.36	2.21	2.26	2.30	2.22		1.97
SS316	2.40	2.35	2.29	2.31	2.16	1.94	2.04
PVC	2.31	2.15	2.22	2.09	2.06	1.95	1.64
PVC	2.36	2.31	2.24	2.11	2.09	1.97	1.65
PVC	2.40	2.31	2.27	2.16	2.03	1.87	1.78
TEFLON	2.31	2.12	2.10	1.94	1.65	1.48	
TEFLON	2.36	2.29	2.10	1.99	1.76	1.31	0.85
TEFLON	2.40	2.30	2.11	2.03	1.72	1.39	0.85
CONTROL	2.31	2.20	2.28	2.25	2.15	2.16	2.02
CONTROL	2.36	2.20	2.31	2.26	2.23	2.13	
CONTROL	2.40	2.24	2.33	2.29	1.95	1.96	1.92

Table A11. Concentration of MDCB with time.

treatment	Concentration mg/L						
	0hr	1hr	8hr	24hr	72hr	168hr	1000hr
SS304	2.27	1.99	2.14	2.21	2.09	1.94	
SS304	2.29	2.01	2.12	2.25	2.03	2.06	1.32
SS304	2.23	2.15	2.20	2.16	1.86	2.06	
SS316	2.27	2.11	2.10	2.15	2.09	2.01	1.70
SS316	2.29	2.05	2.14	2.24	2.15		1.81
SS316	2.23	2.28	2.20	2.25	2.06	1.82	1.89
PVC	2.27	2.23	2.14	2.06	1.95	1.76	1.39
PVC	2.29	1.96	2.18	2.02	1.96	1.80	1.39
PVC	2.23	2.21	2.09	2.00	1.96	1.71	1.51
TEFLON	2.27	2.21	1.86	1.75	1.26	1.02	0.47
TEFLON	2.29	2.13	1.87	1.67	1.38	0.89	0.48
TEFLON	2.23	1.92	1.89	1.72	1.35	0.96	
CONTROL	2.27	2.06	2.19	2.19	2.06	2.06	1.75
CONTROL	2.29	2.06	2.22	2.20	2.15	2.04	1.83
CONTROL	2.23	2.11	2.31	2.23	1.84	1.86	

Table A12. Concentration of PDCB with time.

treatment	Concentration mg/L						
	0hr	1hr	8hr	24hr	72hr	168hr	1000hr
SS304	1.97	1.79	1.94	1.87	1.79	1.68	
SS304	2.02	1.86	1.94	1.90	1.74	1.78	1.55
SS304	2.05	1.85	1.94	1.95	1.60	1.78	
SS316	1.97	2.00	1.92	1.86	1.80	1.75	1.50
SS316	2.02	1.81	1.92	1.93	1.84		1.56
SS316	2.05	1.88	1.93	1.94	1.81	1.58	1.60
PVC	1.97	1.81	1.87	1.73	1.66	1.52	1.21
PVC	2.02	1.97	1.88	1.74	1.68	1.55	1.22
PVC	2.05	1.97	1.90	1.77	1.72	1.47	1.24
TEFLON	1.97	1.76	1.64	1.42	1.06	0.85	0.40
TEFLON	2.02	1.90	1.65	1.47	1.16	0.74	0.38
TEFLON	2.05	1.91	1.67	1.49	1.15	0.82	
CONTROL	1.97	2.02	1.93	1.89	1.77	1.78	1.49
CONTROL	2.02	2.05	1.97	1.90	1.84	1.74	1.55
CONTROL	2.05	1.97	2.03	1.93	1.63	1.61	

Table A13. Concentration of RDX with time--salt study.

treatment	Concentration mg/L						
	0hr	1hr	8hr	24hr	72hr	168hr	1200hr
SS304	1.87	2.06	2.03	2.09	2.02	2.18	2.01
SS304	1.92	2.05	2.04	2.10	2.00	2.18	2.02
SS304	1.99	2.08	2.05	2.07	1.98	2.17	1.97
SS316	1.87	2.78	2.05	2.59	1.96	2.17	2.06
SS316	1.92	2.07	2.07	2.13	2.01	2.18	2.04
SS316	1.99	2.07	2.04	2.14	2.01	2.19	1.92
PVC	1.87	2.07	2.04	2.07	1.91	2.18	1.98
PVC	1.92	2.07	2.05	2.12	2.00	2.18	2.03
PVC	1.99	2.04	2.06	2.07	1.93	2.18	2.06
TEFLON	1.87	2.00	1.99	2.06	1.96	2.18	2.00
TEFLON	1.92	2.07	2.03	2.13	2.02	2.19	2.01
TEFLON	1.99	2.07	1.89	2.60	1.90	2.19	2.04
CONTROL	1.87	2.01	2.04	2.09	1.97	2.18	1.99
CONTROL	1.92	2.02	2.04	2.12	1.96	2.18	2.05
CONTROL	1.99	2.07	2.06	2.61	1.96	2.17	2.06

Table A14. Concentration of TNE with time--salt study.

treatment	Concentration mg/L						
	0hr	1hr	8hr	24hr	72hr	168hr	1200hr
SS304	2.21	2.25	2.18	2.25	2.22	2.27	2.30
SS304	2.26	2.23	2.20	2.26	2.20	2.26	2.19
SS304	2.28	2.26	2.17	2.22	2.15	2.25	2.21
SS316	2.21	2.24	2.13	2.65	2.26	2.27	2.17
SS316	2.26	2.25	2.21	2.30	2.21	2.27	2.19
SS316	2.28	2.76	2.19	2.31	2.06	2.25	2.14
PVC	2.21	2.22	2.22	2.27	2.23	2.29	2.21
PVC	2.26	2.25	2.23	2.30	2.20	2.28	2.21
PVC	2.28	2.25	2.21	2.24	2.19	2.27	2.23
TEFLON	2.21	2.18	2.15	2.66	2.26	2.29	2.31
TEFLON	2.26	2.25	2.18	2.32	2.22	2.29	2.29
TEFLON	2.28	2.27	2.19	2.23	2.17	2.28	2.16
CONTROL	2.21	2.19	2.21	2.68	2.26	2.28	2.30
CONTROL	2.26	2.21	2.20	2.30	2.26	2.27	2.26
CONTROL	2.28	2.26	2.19	2.25	2.27	2.28	2.26

Table A15. Concentration of MNT with time--salt study.

treatment	Concentration mg/L						
	0hr	1hr	8hr	24hr	72hr	168hr	1200hr
SS304	2.16	2.20	2.10	2.12	2.19	2.12	2.28
SS304	2.19	2.16	2.11	2.13	2.17	2.13	2.13
SS304	2.16	2.16	2.09	2.10	2.12	2.10	2.14
SS316	2.16	2.16	2.11	2.21	2.19	2.12	2.17
SS316	2.19	2.16	2.11	2.16	2.18	2.10	2.15
SS316	2.16	2.14	2.09	2.17	2.11	2.09	2.09
PVC	2.16	2.17	2.13	2.13	2.16	2.10	2.07
PVC	2.19	2.17	2.13	2.14	2.14	2.08	2.05
PVC	2.16	2.15	2.11	2.08	2.10	2.06	2.06
TEFLON	2.16	2.10	2.05	2.16	2.16	2.05	2.06
TEFLON	2.19	2.16	2.08	2.15	2.12	2.04	2.03
TEFLON	2.16	2.16	2.09	2.06	2.08	2.05	1.85
CONTROL	2.16	2.15	2.12	2.21	2.17	2.12	2.29
CONTROL	2.19	2.15	2.10	2.15	2.21	2.09	2.20
CONTROL	2.16	2.16	2.10	2.12	2.22	2.11	2.21

Table A16. Concentration of CDCE with time--salt study.

treatment	Concentration mg/L						
	0hr	1hr	8hr	24hr	72hr	168hr	1200hr
SS304	2.58	2.58	2.46	2.45	2.40	2.28	2.22
SS304	2.58	2.54	2.47	2.47	2.46	2.35	1.86
SS304	2.61	2.58	2.38	2.49	2.16	2.21	2.05
SS316	2.58	2.51	2.42	2.51	2.36	2.29	1.85
SS316	2.58	2.47	2.50	2.54	2.36	2.23	2.21
SS316	2.61	2.51	2.42	2.46	2.41	2.24	2.05
PVC	2.58	2.58	2.46	2.39	2.36	2.19	1.59
PVC	2.58	2.57	2.51	2.41	2.31	2.24	1.73
PVC	2.61	2.52	2.43	2.40	2.31	2.18	1.88
TEFLON	2.58	2.45	2.40	2.43	1.86	2.01	1.53
TEFLON	2.58	2.50	2.42	2.45	2.24	2.09	1.65
TEFLON	2.61	2.57	2.37	2.36	2.25	2.14	1.74
CONTROL	2.58	2.57	2.44	2.54	2.39	2.25	2.17
CONTROL	2.58	2.52	2.41	2.53	2.46	2.24	1.88
CONTROL	2.61	2.53	2.50	2.49	2.41	2.32	1.88

Table A17. Concentration of TDCE with time--salt study.

treatment	Concentration mg/L						
	0hr	1hr	8hr	24hr	72hr	168hr	1200hr
SS304	1.93	1.91	1.82	1.79	1.70	1.58	1.39
SS304	1.91	1.88	1.81	1.81	1.75	1.65	1.01
SS304	1.95	1.91	1.74	1.84	1.47	1.52	1.21
SS316	1.93	1.85	1.78	1.84	1.71	1.59	1.00
SS316	1.91	1.83	1.85	1.87	1.67	1.54	1.40
SS316	1.95	1.85	1.77	1.79	1.67	1.55	1.27
PVC	1.93	1.92	1.80	1.73	1.65	1.46	0.76
PVC	1.91	1.90	1.85	1.74	1.59	1.50	0.87
PVC	1.95	1.86	1.77	1.74	1.58	1.45	1.03
TEFLON	1.93	1.78	1.70	1.65	1.39	1.12	0.66
TEFLON	1.91	1.82	1.71	1.67	1.36	1.19	0.63
TEFLON	1.95	1.88	1.66	1.60	1.07	1.23	0.76
CONTROL	1.93	1.91	1.80	1.87	1.69	1.55	1.37
CONTROL	1.91	1.87	1.77	1.86	1.75	1.56	1.03
CONTROL	1.95	1.87	1.85	1.83	1.70	1.61	1.00

Table A18. Concentration of TCE with time--salt study.

treatment	Concentration mg/L						
	0hr	1hr	8hr	24hr	72hr	168hr	1200hr
SS304	2.82	2.80	2.66	2.60	2.51	2.34	2.10
SS304	2.79	2.75	2.64	2.61	2.58	2.43	1.48
SS304	2.86	2.80	2.54	2.65	2.17	2.24	1.82
SS316	2.82	2.73	2.61	2.69	2.55	2.34	1.46
SS316	2.79	2.70	2.70	2.72	2.46	2.27	2.03
SS316	2.86	2.71	2.59	2.61	2.44	2.27	1.86
PVC	2.82	2.81	2.71	2.55	2.49	2.21	1.21
PVC	2.79	2.80	2.60	2.55	2.39	2.27	1.36
PVC	2.86	2.72	2.65	2.54	2.32	2.19	1.63
TEFLON	2.82	2.59	2.44	2.30	1.90	1.47	0.80
TEFLON	2.79	2.64	2.44	2.33	1.84	1.55	0.68
TEFLON	2.86	2.73	2.38	2.24	1.50	1.62	0.66
CONTROL	2.82	2.79	2.63	2.72	2.49	2.29	2.04
CONTROL	2.79	2.74	2.59	2.70	2.60	2.30	1.54
CONTROL	2.86	2.74	2.70	2.65	2.51	2.38	1.50

Table A19. Concentration of CLB with time--salt study.

treatment	Concentration mg/L						
	0hr	1hr	8hr	24hr	72hr	168hr	1200hr
SS304	1.78	1.75	1.63	1.66	1.49	1.55	1.31
SS304	1.78	1.77	1.68	1.67	1.67	1.49	1.40
SS304	1.81	1.78	1.69	1.68	1.69	1.60	1.59
SS316	1.78	1.72	1.65	1.44	1.62	1.50	1.31
SS316	1.78	1.73	1.67	1.73	1.64	1.51	1.41
SS316	1.81	1.73	1.70	1.74	1.67	1.55	1.51
PVC	1.78	1.74	1.65	1.41	1.54	1.45	1.10
PVC	1.78	1.77	1.69	1.64	1.58	1.47	1.17
PVC	1.81	1.78	1.71	1.61	1.62	1.49	1.24
TEFLON	1.78	1.67	1.56	1.50	1.19	1.14	0.65
TEFLON	1.78	1.70	1.58	1.56	1.35	1.18	0.72
TEFLON	1.81	1.75	1.59	1.56	1.39	1.22	0.79
CONTROL	1.78	1.74	1.65	1.69	1.64	1.52	1.34
CONTROL	1.78	1.75	1.67	1.71	1.67	1.53	1.34
CONTROL	1.81	1.78	1.70	0.34	1.71	1.57	1.57

Table A20. Concentration of ODCB with time--salt study.

treatment	Concentration mg/L						
	0hr	1hr	8hr	24hr	72hr	168hr	1200hr
SS304	2.48	2.42	2.32	2.32	2.34	2.25	1.86
SS304	2.46	2.43	2.32	2.33	2.37	2.28	1.90
SS304	2.49	2.46	2.25	2.34	2.15	2.15	2.20
SS316	2.48	2.36	2.28	2.11	2.26	2.16	1.92
SS316	2.46	2.38	2.32	2.40	2.31	2.16	2.03
SS316	2.49	2.40	2.34	2.42	2.36	2.23	1.88
PVC	2.48	2.44	2.28	2.05	2.21	2.04	1.64
PVC	2.46	2.43	2.33	2.31	2.28	2.09	1.57
PVC	2.49	2.39	2.35	2.29	2.15	2.10	1.61
TEFLON	2.48	2.29	2.14	2.03	1.78	1.54	0.73
TEFLON	2.46	2.27	2.14	2.12	1.69	1.57	0.87
TEFLON	2.49	2.33	2.11	2.13	1.88	1.62	0.90
CONTROL	2.48	2.40	2.32	2.35	2.31	2.24	2.18
CONTROL	2.46	2.42	2.28	2.39	2.37	2.20	1.94
CONTROL	2.49	2.45	2.33	2.45	2.42	2.21	1.94

Table A21. Concentration of MDCB with time--salt study.

treatment	Concentration mg/L						
	0hr	1hr	8hr	24hr	72hr	168hr	1200hr
SS304	2.49	2.40	2.29	2.28	2.28	2.18	2.05
SS304	2.48	2.42	2.20	2.28	2.31	2.21	1.70
SS304	2.49	2.46	2.32	2.31	2.04	2.06	1.64
SS316	2.49	2.38	2.29	2.30	2.23	2.15	1.77
SS316	2.48	2.33	2.32	2.37	2.16	2.07	1.88
SS316	2.49	2.36	2.23	1.98	2.33	2.06	1.67
PVC	2.49	2.40	2.25	1.90	2.20	1.88	1.28
PVC	2.48	2.38	2.31	2.18	2.10	1.96	1.35
PVC	2.49	2.44	2.29	2.21	2.05	1.97	1.39
TEFLON	2.49	2.21	1.95	1.79	1.48	1.09	0.39
TEFLON	2.48	2.25	2.00	1.85	1.38	1.12	0.55
TEFLON	2.49	2.31	1.98	1.86	1.30	1.17	0.50
CONTROL	2.49	2.37	2.25	2.31	2.31	2.12	2.01
CONTROL	2.48	2.40	2.32	2.36	2.36	2.12	1.73
CONTROL	2.49	2.44	2.28	2.43	2.29	2.16	1.73

Table A22. Concentration of PDCB with time--salt study.

treatment	Concentration mg/L						
	0hr	1hr	8hr	24hr	72hr	168hr	1200hr
SS304	2.09	2.07	2.00	1.96	1.95	1.86	1.79
SS304	2.09	2.09	1.92	1.96	1.97	1.75	1.48
SS304	2.11	2.09	1.98	1.97	1.74	1.87	1.45
SS316	2.09	2.05	2.00	1.98	2.01	1.82	1.54
SS316	2.09	2.04	1.94	2.03	1.92	1.77	1.64
SS316	2.11	2.02	2.01	2.08	1.85	1.76	1.47
PVC	2.09	2.05	1.94	1.86	1.88	1.60	1.14
PVC	2.09	2.07	1.99	1.91	1.79	1.66	1.18
PVC	2.11	2.06	2.00	2.03	1.73	1.66	1.19
TEFLON	2.09	1.91	1.72	1.51	1.09	0.93	0.34
TEFLON	2.09	1.95	1.70	1.58	1.16	0.97	0.43
TEFLON	2.11	1.99	1.68	1.58	1.25	0.89	0.47
CONTROL	2.09	2.03	2.01	2.08	1.96	1.80	1.76
CONTROL	2.09	2.08	1.96	1.86	1.97	1.80	1.53
CONTROL	2.11	2.09	1.98	1.96	2.03	1.84	1.52

Table A23. Normalized¹ concentrations of analytes taken from samples containing salt.

Analyte	Treatment	1 Hour	6 Hour	24 Hour	72 Hour	168 Hour	1000 Hour
ROX	Teflon	1.01	0.96	1.00	1.00	1.00	0.99
	PVC	1.01	1.00	0.92	0.99	1.00	1.00
	SS304	1.02	1.00	0.92	1.02	1.00	0.98
	SS316	1.13	1.00	1.01	1.02	1.00	0.99
YNE	Teflon	1.01	0.99	1.00	1.04	1.00	0.99
	PVC	1.01	1.01	0.94	1.01	1.00	0.98
	SS304	1.01	0.99	0.93	1.01	0.99	0.98
	SS316	1.09	0.99	1.00	1.02	0.99	0.95*
COCE	Teflon	0.99	0.98	0.96*	0.88*	0.92*	0.83*
	PVC	1.01	1.01	0.95*	0.96	0.97	0.88
	SS304	1.01	1.00	0.98	0.87	1.00	1.03
	SS316	0.98	1.00	0.98	0.98	0.99	1.03
IDCE	Teflon	0.97*	0.94*	0.88*	0.74*	0.75*	0.60*
	PVC	1.00	1.00	0.94*	0.94	0.94	0.78
	SS304	1.01	0.99	0.98	0.86	1.01	1.06
	SS316	0.98	1.00	0.98	0.98	0.99	1.08
MNT	Teflon	0.98	0.98*	0.98	0.96*	0.97	0.86*
	PVC	1.00	1.01	0.98	0.97*	0.99	0.92*
	SS304	1.01	1.00	0.98	0.98	1.00	0.98
	SS316	1.00	1.00	1.01	0.98	1.00	0.98
TCE	Teflon	0.86*	0.82*	0.83**	0.69*	0.67*	0.42*
	PVC	1.01	1.00	0.93*	0.95	0.96	0.83
	SS304	1.02	0.99	0.97	0.96	1.01	1.08
	SS316	0.98	1.00	0.99	0.98	0.99	1.01
CLE	Teflon	0.87*	0.94*	0.91	0.78	0.77	0.51*
	PVC	1.00	1.01	0.91	0.94	0.95	0.83*
	SS304	1.01	1.00	0.98	0.97	1.00	1.02
	SS316	0.98	1.00	0.96	0.96	0.99	1.00
OOCB	Teflon	0.95*	0.82*	0.87*	0.75*	0.71*	0.41*
	PVC	1.00	1.00	0.82	0.94*	0.94*	0.80*
	SS304	1.01	0.99	0.99	0.97	1.00	0.98
	SS316	0.98	1.00	0.96	0.98	0.98	0.96
POCB	Teflon	0.94	0.86	0.79*	0.59*	0.51*	0.26*
	PVC	1.00	1.00	0.98	0.91*	0.91*	0.73*
	SS304	1.01	0.99	1.00	0.85	1.01	0.98
	SS316	0.96	1.00	1.03	0.97	0.98	0.97
MOCB	Teflon	0.94*	0.87*	0.77*	0.56*	0.53*	0.26*
	PVC	1.00	1.00	0.89*	0.91*	0.91*	0.74*
	SS304	1.01	0.99	0.94	0.95	1.01	0.99
	SS316	0.98	1.00	0.97	0.97	0.98	0.97

¹ The values given here are determined by dividing the mean concentration of a given analyte at a given time and for a particular well casing by the mean concentration (for the same analyte) of the control samples taken at the same time.

Table A24. Regression analysis for samples containing Teflon casings (vs K_{ov}).

Regression Analysis -- Linear model: $Y = a+bX$

Dependent variable: Normalized conc. with ref Independent variable: K_{ov}

Parameter	Estimate	Standard Error	T Value	Prob. Level
Intercept	1.1789	0.148297	7.94958	.00005
Slope	-0.252084	0.061157	-4.12191	.00334

Analysis of Variance

Source	Sum of Squares	Df	Mean Square	F-Ratio	Prob. Level
Model	.502285	1	.502285	16.99017	.00334
Error	.2365063	8	.0295633		

Total (Corr.) .7387916 9

Correlation Coefficient = -0.824545 R-squared = 67.99 percent
 Std. Error of Est. = 0.17194

Table A25. Regression analysis for samples containing PVC casings (vs K_{ov}).

Regression Analysis -- Linear model: $Y = a+bX$

Dependent variable: Normalized conc. with pvc Independent variable: K_{ov}

Parameter	Estimate	Standard Error	T Value	Prob. Level
Intercept	1.01749	0.0440471	23.1001	.00000
Slope	-0.0581521	0.0181648	- 3.20136	.01259

Analysis of Variance

Source	Sum of Squares	Df	Mean Square	F-Ratio	Prob. Level
Model	.026729	1	.026729	10.24874	.01259
Error	.0208646	8	.0026081		

Total (Corr.) .0475941 9

Correlation Coefficient = -0.749409 R-squared = 56.16 percent
 Std. Error of Est. = 0.0510693

APPENDIX H
CHEMICAL RESISTANCE GUIDE
PVC TYPE 1



CHEMICAL RESISTANCE GUIDE

PVC TYPE 1

This guide to the proper application of PVC Type 1 pipe and fittings is offered as a service to customers of Silver-Line. Much of the data was collected from outside sources; therefore, we cannot warrant its accuracy, although every effort has been made to verify the information shown. Compounds designated for "Limited" use (L) and substances not listed should be tested prior to installation, as they may not be suitable under some conditions.

A = Acceptable			L = Limited			U = Unsuitable		
	PVC I			PVC I			PVC I	
	73°F.	140°F.		73°F.	140°F.		73°F.	140°F.
Acetaldehyde	U	U	Amyl chloride	U	U	Bromine water	A	A
Acetaldehyde, 40%	L	U	Aniline	U	U	Butadiene	A	A
Acetamide	U	U	Aniline chlorohydrate	U	U	Butane	A	U
Acetate solvents - crude	U	U	Aniline dyes	U	U	Butanediol	A	A
Acetate solvents - pure	U	U	Aniline hydrochloride	U	U	Butantetrol (erythritol)	A	U
Acetic acid 0 - 20%	A	A	Anthraquinone	A	U	Buttermilk	A	A
Acetic acid 20 - 30%	A	A	Anthraquinonesulfonic acid	A	A	Butyl acetate	L	U
Acetic acid 30 - 60%	A	A	Antimony trichloride	A	A	Butyl phenol	U	U
Acetic acid 80%	A	L	Aqua regia	L	U	Butylene	A	L
Acetic acid - glacial	L	U	Aromatic hydrocarbons	U	U	Butyric acid 20%	L	U
Acetic acid - vapors	L	U	Arsenic acid - 80%	A	L	Butyric acid	A	U
Acetic anhydride	U	U	Arylsulfonic acid	A	A			
Acetone	U	U	Asphalt	A	A	Cadmium cyanide	U	U
Acetylene	L	L				Calcium hydroxide	A	A
Adipic acid	A	A	Barium carbonate	A	A	Calcium hypochlorite	A	A
Alcohol, allyl - 96%	L	U	Barium chloride	A	A	Calcium salts	A	A
Alcohol, amyl	L	U	Barium hydroxide 10%	A	A	Cane sugar liquors	A	A
Alcohol, benzyl	U	U	Barium sulfate	A	A	Carbolic acid	A	A
Alcohol, butyl (n-butanol)	A	A	Barium sulfide	A	A	Carbon bisulfide	U	U
Alcohol, butyl (2-butanol)	A	U	Beer	A	A	Carbon dioxide (aqueous)	A	A
Alcohol, ethyl	A	L	Beet sugar liquor	A	A	Carbon dioxide gas (dry)	A	A
Alcohol, hexyl	A	A	Benzaldehyde 10%	U	U	Carbon dioxide gas (wet)	A	A
Alcohol, isopropyl (2-propanol)	A	A	Benzene	U	U	Carbon monoxide	A	A
Alcohol, methyl	A	U	Benzenesulfonic acid - 10%	A	A	Carbon tetrachloride	L	U
Alcohol, propyl (1-propanol)	A	U	Benzenesulfonic acid	U	U	Carbonated water	A	A
Allyl chloride	U	U	Benzoic acid	A	A	Carbonic acid	A	A
Alum	A	A	Benzol	U	U	Casain	A	A
Alum, chrome	A	U	Bismuth carbonate	A	A	Castor oil	A	A
Aluminum hydroxide	A	A	Black liquor (paper industry)	A	A	Caustic potash	A	A
Aluminum oxychloride	A	A	Bleach - 12.5% active Cl ₂	A	L	Caustic soda	A	A
Aluminum salts	A	A	Borax	A	A	Cellosolve	L	U
Ammonia - gas (dry)	A	A	Borax liquors	A	A	Cellosolve acetate	A	U
Ammonia - gas (wet)	U	U	Boric acid	A	A	Chloral hydrate	A	A
Ammonia - liquid	U	U	Boron trifluoride	A	A	Chloramine	A	U
Ammonium fluoride 0 - 25%	L	U	Breeder pellets - (fish deriv)	A	A	Chloric acid 20%	A	A
Ammonium hydroxide 0 - 28%	A	A	Brine	A	A	Chloride water	A	A
Ammonium salts	A	A	Bromic acid	A	A	Chlorinated solvents	U	U
Amyl acetate	U	U	Bromine - liquid	U	U	Chlorine (dry)	U	U
			Bromine gas - 25%	A	A	Chlorine gas (moist)	U	U

THE DATA SHOWN HAS BEEN COMPILED FROM VARIOUS SOURCES. IT IS NOT ALL-INCLUSIVE AND IS INTENDED FOR USE ONLY AS A GENERAL GUIDE. SILVER-LINE CANNOT ASSUME RESPONSIBILITY FOR ANY PARTICULAR APPLICATION AND URGES USERS TO CONDUCT QUALIFICATION TESTING.

	PVC I			PVC I			PVC I	
	73°F.	140°F.		73°F.	140°F.		73°F.	140°F.
Chlorine, liquid	U	U	Ethers	U	U	Hydrofluoric acid 4% - 100%	A	L
Chlorine water	A	A	Ethyl esters	U	U	Hydrogen	A	A
Chloroacetic acid	A	U	Ethyl halides	U	U	Hydrogen peroxide -		
Chlorobenzene	U	U	Ethylene glycol	A	A	30% - 90%	A	A
Chlorobenzyl chloride	U	U	Ethylene halides	U	U	Hydrogen phosphide	A	A
Chloroform	U	U	Ethylene oxide	U	U	Hydrogen sulfide - dry or		
Chlorosulfonic acid (100%)	A	U				solu.	A	A
Chrome alum	A	A	Fatty acids	A	A	Hydroquinone	A	A
Chromic acid 10% - 40%	A	L	Ferric salts	A	A	Hydroxylamine sulfate	A	A
Chromic acid 50%	U	U	Ferrous salts	A	A	Hypochlorous acid	A	A
Citric acid	A	A	Fish solubles	A	A	Hypo-(sodium thiosulfate)	A	A
Coconut oil	A	A	Fluorine gas - dry & wet	L	U			
Coke oven gas	U	U	Fluoboric acid - 25%	A	A	Iodine	U	U
Copper salts	A	A	Fluosilicic acid	A	A	Iodine (in alcohol)	U	U
Core oils	A	A	Food products such as milk,			Iodine solution (10%)	U	U
Corn oil	A	A	buttermilk, molasses, salad					
Corn syrup	A	A	oils, fruit	A	A	Jet fuels, JP-4 & JP-5	A	A
Cottonseed oil	A	A	Formaldehyde	A	A			
Cresol	U	U	Formic acid	A	U	Kerosene	A	A
Cresylic acid 50%	A	A	Freon - F11, F12, F113, F114	A	A	Ketones	U	U
Crotonaldehyde	U	U	Freon - F21, F22	U	U	Kraft liquor (paper industry)	A	A
Crude oil - sour	A	A	Fructose	A	A			
Crude oil - sweet	A	A	Fruit pulps and juices	A	A	Lacquer thinners	U	U
Cupric fluoride	A	A	Fuel oil (containing H ₂ SO ₄)	U	U	Lactic acid 25%	A	A
Cupric sulfate	A	A	Furfural	U	U	Lard oil	A	A
Cuprous chloride	A	A				Lauric acid	A	A
Cyclohexane	U	U	Gallic acid	A	A	Lauryl chloride	A	L
Cyclohexanol	U	U	Gas - coke oven	U	U	Lauryl sulfate	A	A
Cyclohexanone	U	U	Gas - manufactured	U	U	Lead salts	A	A
			Gas - natural (dry & wet)	A	A	Lime sulfur	A	A
Deminerlized water	A	A	Gasolines	A	A	Linoleic acid	A	A
Detergents	A	A	Gelatine	A	A	Linseed oil	A	A
Dextrin	A	A	Glucose	A	A	Liqueurs & liquors	A	A
Dextrose	A	A	Glue	A	A	Lithium salts	A	A
Diazo salts	A	A	Glycerine (glycerol)	A	A	Lubricating oil - ASTM 1-23	A	A
Dibutyl phthalate	U	U	Glycol	A	A			
Dibutyl sebacate	L	U	Glycolic acid 30%	A	A	Machine oil	A	A
Dichlorobenzene	U	U	Grape sugar	A	A	Magnesium hydroxide	A	A
Dichloroethylene	U	U	Green liquor (paper	A	A	Magnesium salts	A	A
Diesel fuels	A	A	industry)			Maleic acid	A	A
Diethylamine	U	U	Heptane	A	A	Malic acid	A	A
Diethyl ether	U	U	Hexane	A	U	Manganese sulfate	A	A
Diglycolic acid	A	L	Hexanol tertiary	A	A	Mercuric salts	A	A
Dimethylamine	L	L	Hydrazine	U	U	Mercurous nitrate	A	A
Dimethyl formamide	U	U	Hydrobromic acid - 20%	A	A	Mercury	A	A
Diocetyl phthalate	U	U	Hydrochloric acid - 0 - 40%	A	A	Mesityl oxide	U	U
Dioxane - 1, 4	U	U	Hydrocyanic acid or			Metallic soaps	A	A
Disodium phosphate	A	A	hydrogen cyanide	A	A	Methane	A	A
						Methyl acetate	U	U

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	PVC I			PVC I			PVC I	
	73°F.	140°F.		73°F.	140°F.		73°F.	140°F.
Methyl bromide	U	U	Palmitic acid 10%	A	A	Silicic acid	A	A
Methyl cellosolve	U	U	Palmitic acid 70%	A	U	Silicone oil	A	U
Methyl chloride	U	U	Paraffin	A	A	Silver salts	A	A
Methyl chloroform	U	U	Pentane	L	L	Soaps	A	A
Methyl cyclohexanone	U	U	Peracetic acid 40%	L	U	Soap solution	A	A
Methyl ethyl ketone (MEK)	U	U	Perchloric acid 10%	A	L	Sodium hydroxide	A	A
Methyl iso-butyl ketone (MIBK)	U	U	Perchloric acid 15% - 70%	L	U	Sodium hypochlorite	A	A
Methyl methacrylate (MMA)	A	U	Perchloroethylene	L	L	Sodium salts	A	A
Methyl salicylate	A	A	Petrolatum	A	A	Sour crude oil	A	A
Methyl sulfate	A	U	Petroleum liquifier	A	A	Stannic chloride	A	A
Methyl sulfonic acid	A	A	Phenol	U	U	Stannous chloride	A	A
Methylene halides	U	U	Phenylcarbinol	U	U	Stenous chloride (50%)	A	A
Milk	A	A	Phenylhydrazine	U	U	Starch	A	A
Mineral oils	A	L	Phenylhydrazine hydrochloride	U	U	Stearic acid	A	A
Mixed acids (H ₂ SO ₄ & HNO ₃)	L	U	Phosgene (gas)	A	U	Stoddard solvent	U	U
Molasses	A	A	Phosgene (liquid)	U	U	Sulfite liquor	A	A
Monochlorobenzene	U	U	Phosphoric acid 0 - 25%	A	A	Sulfur	A	A
Monethanolamine	U	U	Phosphoric acid 25 - 85%	A	A	Sulfur dioxide gas - dry	A	A
Muriatic acid	A	A	Phosphorus (red)	A	A	Sulfur dioxide gas - wet	A	U
Naphtha	A	A	Phosphorus (yellow)	A	L	Sulfur dioxide - liquid	L	U
Naphthalene	U	U	Phosphorus pentoxide	A	L	Sulfur trioxide - dry	A	A
Natural gas, dry & wet	A	A	Phosphorus trichloride	U	U	Sulfur trioxide, wet	A	L
Nickel salts	A	A	Photographic chemicals	A	A	Sulfuric acid 0 - 70%	A	A
Nicotine	A	A	Photographic solutions	A	A	Sulfuric acid 70 - 90%	A	L
Nicotinic acid	A	A	Picric acid	U	U	Sulfuric acid, 90 - 100%	L	U
Nitric acid anhydrous	U	U	Plating solutions: metals	A	L	Sulfurous acid	U	U
Nitric acid 10%	A	A	Potassium acid sulfate	A	L	Tall oil	A	A
Nitric acid 20% - 70%	A	L	Potassium alkyl xanthates	A	U	Tannic acid	A	A
Nitric acid 80%	L	L	Potassium hydroxide	A	A	Tanning liquors	A	A
Nitric acid 90%	L	U	Potassium hypochlorite	A	L	Tartaric acid	A	A
Nitric acid 100%	U	U	Potassium permanganate 10%	A	A	Terpineol	U	U
Nitric acid red fuming	U	U	Potassium permanganate 25%	L	U	Tertrachloroethane	L	L
Nitrobenzene	U	U	Potassium salts	A	A	Tetraethyl lead	L	U
Nitroglycerine	U	U	Propane	A	A	Tetrahydro furan	U	U
Nitroglycol	U	U	Propylene dichloride	U	U	Thionyl chloride	U	U
Nitropropane	L	L	Propylene glycol	A	A	Tin chloride	A	A
Nitrous acid (10%)	A	L	Propylene oxide	U	U	Titanium tetrachloride	U	U
Nitrous oxide	A	L	Pyridine	U	U	Toluol (toluene)	U	U
			Pyrogalllic acid	L	L	Tributyl citrate	A	U
			Rayon coagulating bath	A	A	Tributyl phosphate	U	U
			Rochelle salts	A	A	Trichloroacetic acid	A	A
Dil and fats	A	A				Trichloroethylene	U	U
Oleic acid	A	A	Salicylaldehyde	L	L	Tricresylphosphate	U	U
Oleum	U	U	Salicylic acid	A	A	Triethanolamine	L	U
Olive oil	L	U	Salt water	A	A	Triethylamine	A	L
Oxalic acid	A	A	Sea water	A	A	Trimethyl propane	L	U
Oxygen	A	A	Selenic acid	A	A	Trisodium phosphate	A	A
Ozone	A	L	Sewage, residential	A	A	Turpentine	A	A

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	PVC I			PVC I			PVC I	
	73°F.	140°F.		73°F.	140°F.		73°F.	140°F.
Urea	A	A	Water - acid mine	A	A	Xylene (xylol)	U	U
Urine	A	A	Water - distilled	A	A	Zinc salts	A	A
			Water - fresh	A	A			
Vaseline	U	U	Water - salt	A	A			
Vegetable oil	A	A	Water - sewage	A	A			
Vinegar	A	A	Whiskey	A	A			
Vinyl acetate	U	U	White gasoline	A	A			
			White liquor (paper industry)	A	A			
			Wines	A	A			



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