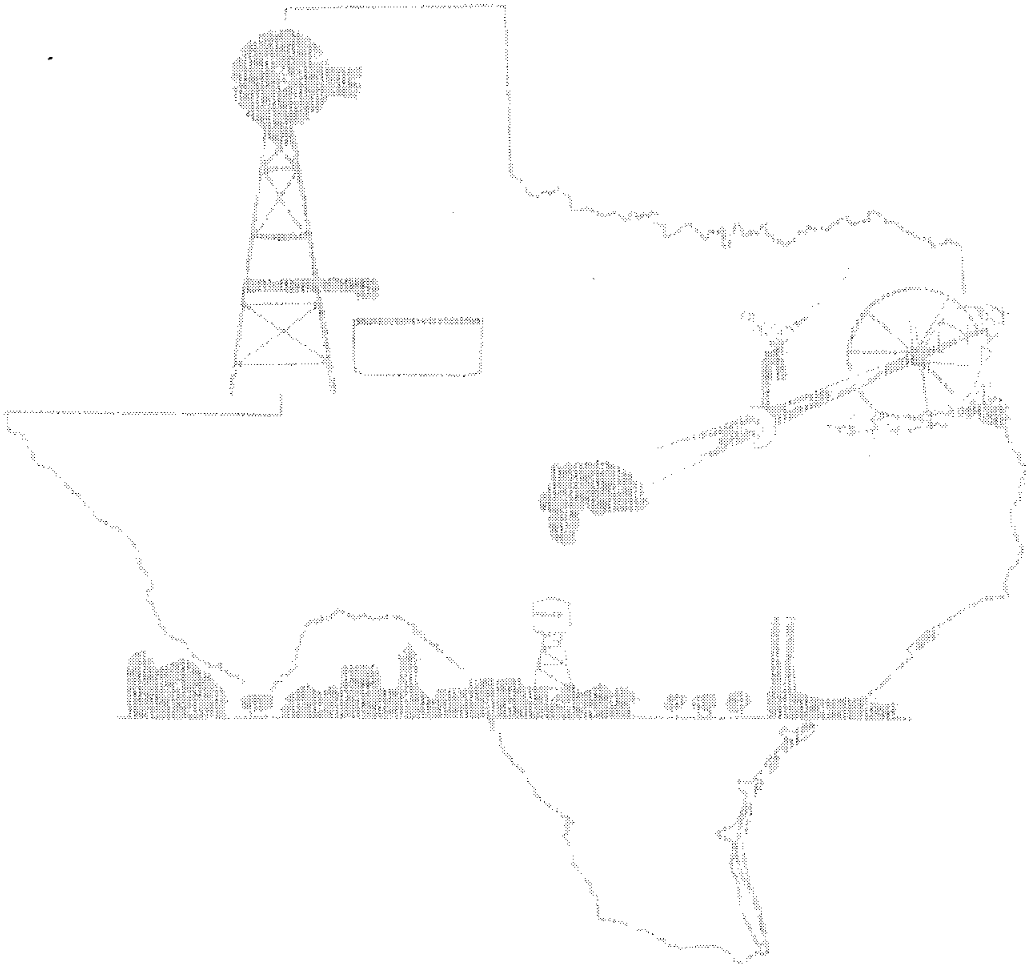
Regional Water Supply Plan to Mitigate Radionuclide Levels in Concho, McCulloch, and San Saba Counties



Hickory Underground Water Conservation District No. 1

September 1991

HDR

HDR Engineering, Inc.

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REGIONAL WATER SUPPLY TO MITIGATE RADIONUCLIDE LEVELS IN CONCHO, McCULLOCH, AND SAN SABA COUNTIES

for

HICKORY UNDERGROUND WATER CONSERVATION DISTRICT NO. 1

by

HDR ENGINEERING, INC.

SEPTEMBER 1991

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50376

9/11/91

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SECTION 1

REGIONAL WATER SUPPLY PLAN TO MITIGATE RADIONUCLIDE LEVELS IN CONCHO, McCULLOCH, AND SAN SABA COUNTIES

1.0 INTRODUCTION

On March 29, 1990, the Texas Department of Health (TDH) informed eight public water systems in Concho, McCulloch, and San Saba Counties that immediate action had to be taken to eliminate the high concentrations of radium in their drinking water supply sources, since radium poses an unreasonable health risk. To that end, TDH and each of these eight public water systems entered into a bilateral agreement that outlined the tasks and associated schedule to reduce the levels of radium to meet the maximum contaminant level(s) (MCLs) of the Drinking Water Standards.

Subsequently, the eight systems, through the Hickory Underground Water Conservation District No. 1, applied to the Texas Water Development Board and received a regional water supply planning grant to fund this study, the first task of the bilateral agreement. As specified in the grant, the purpose of this study is to determine the most practical method to reduce the levels of naturally occurring radium in the public water supply sources so the participants can comply with the rules and regulations of the U.S. Environmental Protection Agency and Texas Department of Health.

The eight public water systems in Concho, McCulloch, and San Saba Counties participating in this study are:

- City of Brady, including the Lake Brady area;
- City of Eden;
- Live Oak Hills Subdivision;
- City of Melvin;
- Millersview-Doole Water Supply Corporation;

- North San Saba Water Supply Corporation;
- Richland Water Supply Corporation (Brady System); and
- Rochelle Water Supply Corporation.

For all eight systems, either combined radium-226 and radium 228, gross alpha, or both constituents exceed the existing MCL in the present or past chemical analyses of their water supply source(s). The MCLs for combined radium-226 and radium-228 and Gross Alpha in the Drinking Water Standards are 5 picoCuries per liter (pCi/l) and 15 pCi/l, respectively.

The first step toward establishing the most practical method to reduce the levels of radium in the drinking water was to determine the immediate as well as future annual water requirements for each of the participants. The second step was to evaluate the quality of each of the water supply sources and identify the major components of each water distribution system, including water supply sources, storage facilities, and pump stations. Once the annual water requirements and the quality of the water supply source was determined, then treatment methods to reduce radionuclides were identified and evaluated, and compared with alternative water supply sources. Then, the most practical method to eliminate radionuclides in the public water supply sources of the study participants was developed and recommendations were prepared.

SECTION 2

2.0 POPULATION AND WATER REQUIREMENT PROJECTIONS

A multi-step process was used to estimate water requirements in the participants' service areas for the 20-year planning period (1990-2010). These requirements were used to size treatment facilities, ascertain deficiencies in existing water supply source(s), calculate the supplemental water supply required, and determine the quantity of water necessary to replace the existing water supply source(s).

The first step in establishing these water requirements was to determine the existing population and per capita water use of each service area. Then the projected service area population and projected per capita water use of each participant was estimated to the end of the planning period based on the historical data available. In the final step, the water requirements for the planning period are calculated by multiplying the projected population by the projected per capita water use.

2.1 Present Population and Water Use

Each of the study participants supplied HDR with the existing number of connections and total annual water use for its service area. This information was compared to similar data reported by the Texas Water Development Board (TWDB) in its Annual Water Use Reports. In general, there were no major differences between the information from the two sources.

While information on numbers of connections and water use was available for all participants, specific population information was available only for the cities of Brady and Eden. Due to the lack of specific population information for the other participants, two different methods of determining the existing population and average daily per capita water use were used, one for the cities of Brady and Eden and another for the other participants.

2.1.1 Cities of Brady and Eden

For the cities of Brady and Eden, the actual 1980 population and estimates of 1981 through 1987 population were obtained from the U.S. Bureau of the Census. For the years in which a population estimate and the reported annual water use were available (1980 through 1987), the average daily per capita water use was calculated by simple division. For the eight-year period, the average annual per capita water use for Brady and Eden was 276 and 158 gallons per capita per day (gpcd), respectively.

2.1.2 Other Participants

For the City of Melvin, the water supply corporations, and Live Oak Hills Subdivision, 1990 population was estimated before making projections of population in the years 2000 and 2010. The 1990 population estimate was based on the per capita water use for the counties in the study area as reported to TWDB, adjusted to reflect only the water use in the unincorporated areas, and the number of connections in 1990 as provided by each participant. Reports for the years 1980 through 1989 were used for the projections in this study (see Table 2.1-1).

TWDB calculates per capita water use under two conditions: average weather conditions and drought conditions. Under drought conditions, water use is usually at the maximum that will occur. The average per capita water use is based on the water use for all the years of the selected period, while the maximum per capita water use is based on the water use for years with little or no precipitation; in other words, a drought condition.

Table 2.1-1
Water Use Information

Member	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989
<u>Number of Connections</u>										
Brady	2,857	2,879	3,127	2,870	2,911	2,888	2,886	2,883	2,934	2,864
Eden	618	625	630	624	624	630	630	630	635	635
Live Oak Hills	--	--	--	--	--	13	8	12	21	--
Melvin	138	135	152	150	150	148	153	--	130	130
Millersville-Doole WSC	1,287	1,287	1,285	1,285	1,285	1,322	1,322	1,340	1,340	1,358
North San Saba WSC	150	167	190	214	235	233	236	234	236	238
Richland WSC (Brady)	--	--	--	--	215	215	216	216	215	215
Rochelle WSC	80	80	80	84	97	97	97	97	97	97
<u>Total Water Use (ac-ft)</u>										
Brady	2,359.7	2,038.5	2,065.0	1,995.6	2,462.1	2,239.9	2,066.1	2,107.6	2,188.0	2,419.8
Eden	256.7	240.8	254.9	209.6	230.5	213.3	227.7	220.9	277.4	279.1
Live Oak Hills	--	--	--	--	--	1.7	2.7	1.3	5.0	--
Melvin	52.4	52.7	55.0	45.8	110.5	98.1	--	54.8	55.2	78.2
Millersville-Doole WSC	440.2	455.5	454.2	464.7	505.5	590.0	559.8	520.1	465.1	526.4
North San Saba WSC	66.9	71.7	77.1	81.8	102.7	88.8	77.1	65.9	84.5	76.6
Richland WSC (Brady)	--	--	--	--	43.5	65.4	70.8	68.8	83.1	94.0
Rochelle WSC	28.8	26.4	33.0	32.6	36.2	33.4	34.5	29.5	38.5	39.1
<u>Water Use (ac-ft/conn.)</u>										
Brady	0.825	0.705	0.660	0.695	0.845	0.775	0.716	0.731	0.745	0.845
Eden	0.415	0.385	0.404	0.335	0.369	0.338	0.361	0.351	0.436	0.439
Live Oak Hills	--	--	--	--	--	0.130	0.330	0.110	0.230	--
Melvin	0.379	0.390	0.362	0.305	0.736	0.663	--	--	0.424	0.601
Millersville-Doole WSC	0.342	0.354	0.353	0.362	0.393	0.446	0.423	0.388	0.347	0.387
North San Saba WSC	0.446	0.429	0.405	0.382	0.437	0.381	0.326	0.282	0.358	0.322
Richland WSC (Brady)	--	--	--	--	0.202	0.304	0.327	0.318	0.386	0.437
Rochelle WSC	0.360	0.330	0.410	0.388	0.373	0.344	0.355	0.304	0.396	0.403

During the 1976 through 1987 time period, a drought existed in the years 1980, 1981, 1982, and 1984. The average and maximum per capita water use for the counties are presented in Table 2.1-2, along with the statewide values for comparison. The per capita water use for Concho County was less than the statewide averages, while McCulloch and San Saba Counties were higher.

Table 2.1-2
County-Wide Per Capita Water Use
For the Period 1976-1987

County	**Per Capita Water Use (gpcd)**	
	Average	Maximum
Concho	148	171
McCulloch	243	286
San Saba	195	214
Statewide	165	194

To determine the per capita water use for the unincorporated areas of the counties where the water supply corporations and Live Oak Hills Subdivision are located, the county-wide per capita water use was adjusted by eliminating the population and water use of the incorporated cities in each county from the county-wide values. The average per capita water use for the unincorporated areas of each county is shown in Table 2.1-3. It should be noted that some water supply corporation customers water livestock and some also irrigate small pecan orchards and gardens. Therefore, the computed per capita water use estimates include these quantities, since the reported total water use does not separate the quantities used by people, livestock, or irrigation.

Table 2.1-3 Per Capita Water Use for the Unincorporated Areas of the Counties	
County	**Per Capita Water Use (gpcd)**
Concho	122
McCulloch	154
San Saba	110

From Table 2.1-1, the average annual water use per connection for the period of record was determined and is shown in Table 2.1-4. This table presents water use in units of acre-feet, which equates to 325,851 gallons.

Table 2.1-4 Average Annual Water Use Per Connection	
Participants	**Acre-Feet Per Connection**
Brady	0.7542
Eden	0.3833
Live Oak Hills	0.2000
Melvin	0.4825
Millersview-Doole WSC	0.3795
North San Saba WSC	0.3768
Richland WSC (Brady)	0.2840
Rochelle WSC	0.3663

The number of persons (capita) per connection was estimated by dividing the average annual water use per connection (shown in Table 2.1-4) by the average daily per capita water use for the unincorporated portions of the counties. The results of these computations are shown in Table 2.1-5.

Table 2.1-5
Water Use Per Connection, Estimated Number of Persons Per Connection and Estimated Population for Service Area

Participant	Number of Connections in 1990	Per Capita Water Use (gpcd)[1]	Number of Persons Per Connection	Estimated Population 1990
Brady	2,915	276[2]	2.04	5,946[4]
Eden	613	158[2]	2.55	1,567[4]
Live Oak Hills Sub.	17	154	N/A	N/A
Melvin	140	154	1.3	184[4]
Millersview-Doole WSC	1,396	138[3]	2.45	3,420
North San Saba WSC	241	138[3]	2.43	586
Richland WSC (Brady)	218	154	1.91	416
Rochelle WSC	97	154	2.12	205

[1] Unincorporated parts of counties, except for Brady and Eden, gallons per capita per day.
[2] Based on reported annual water and U.S. Bureau of Census population estimates.
[3] Average of McCulloch and Concho Counties. North San Saba WSC per capita water use was equated to Millersview-Doole rate, since it appears that the two service areas have similar characteristics.
[4] Census in part for 1990.
N/A means not "applicable"; data series too short.

2.2 Per Capita Water Use Projections

Based on the average per capita water use for 1990 (Table 2.1-5), projections of per capita water use during average and maximum drought conditions were made for each entity for the years 2000 and 2010 (Table 2.2-1), taking into account the potential for water conservation. The water conservation potential assumed in this study is a reduction in the per capita water use by five percent (5.0%) per decade from 1990 through 2010. The means to accomplish this goal is provided in the Water Conservation and Drought Contingency Plan, which is provided in Appendix A.

In order to insure that there is an adequate water supply to meet anticipated maximum demand, it is necessary to determine the maximum per capita water use and maximum total annual water use. Based on the annual water use reported by each entity, the maximum per capita water use was 16 percent higher than the average use during the 1980s. Therefore, for this study, it is assumed that the maximum per capita use for 2000 and 2010 will be 16 percent higher than the average per capita water use. (See Table 2.2-1).

Table 2.2-1
Projected Per Capita Water Use
During Average and Maximum Drought Conditions

Participant	Estimated 1990	2000		2010	
		Average	Maximum	Average	Maximum
	-------------------------------------gallons/capita/day----------------------				
Brady[1]	276	262	305	249	288
Eden[1]	158	150	174	143	165
Live Oak Hills Sub.	154	146	169	139	161
Melvin	154	146	169	139	161
Millersview-Doole WSC	138	131	152	124	144
North San Saba WSC	138	131	152	124	144
Richland WSC (Brady)	154	146	169	139	161
Rochelle WSC	154	146	169	139	161

[1] Texas Water Development Board Water Plan rates.

2.3 Population Projections

Utilizing the TWDB county population projections (Table 2.3-1), the population for 2000 and 2010 of each of the project participants can be estimated. The counties' annual growth rates, both high and low, are also shown. It is noted that the population projections shown in this table do not include the new criminal justice facilities in the area; one is in operation at Eden in Concho County, and another is being constructed at San Saba in San Saba County.

Table 2.3-1
County Population Projections

County	1980 Population	1990		2000		2010	
		Low	High	Low	High	Low	High
Concho	2,915	2,586	2,587	2,487	2,612	2,343	2,487
McCulloch	8,735	9,065	9,084	9,299	9,334	9,555	9,617
San Saba	6,204	5,510	5,605	5,733	5,979	6,047	6,363

County	Compound Annual Growth Rates					
	1980-1990		1990-2000		2000-2010	
	Low	High	Low	High	Low	High
Concho	-1.19	-1.19	-0.38	0.09	-0.59	-0.48
McCulloch	0.37	0.39	0.25	0.27	0.27	0.29
San Saba	-1.18	-1.01	0.39	0.64	0.53	0.62

Source: Texas Water Development Board.

It is anticipated that the new criminal justice facilities located in the area will help the growth potential of the local areas where facilities have been placed (Eden and San Saba). Therefore, the growth rates of these local areas will be significantly higher than the projected growth rates for the counties of the areas shown in Table 2.3-1. In the case of North San Saba Water Supply Corporation, plans have been developed and financing is nearly completed for upgrading and enlarging distribution lines to a large portion of the service area. Expressions of interest have been made for additional connections. These factors have been included in the projected growth rates. In general, the growth rates for the other participants of this project should follow the projected growth rates for the counties. Table 2.3-2 provides the estimated compound annual growth rates, both low and high, for the project participants.

Table 2.3-2
Estimated Annual Growth Rates

Participant	**County**	**Compound Annual Growth Rates (%)**	
		Low	**High**
Brady	McCulloch	0.35	0.40
Eden	Concho	1.1	1.20
Live Oak Hills Subd.	McCulloch	1.0	1.50
Melvin	McCulloch	0.1	0.30
Millersview-Doole WSC	Concho and McCulloch	0.3	0.40
North San Saba WSC	San Saba	1.1	1.60
Richland WSC (Brady)	McCulloch	0.2	0.30
Rochelle WSC	McCulloch	0.2	0.30

Based on the growth rates, both low and high (Table 2.3-2), and the 1990 population (Table 2.1-5), the population for 2000 and 2010 of each of the project participants can be estimated (Table 2.3-3).

Table 2.3-3
Population Projections

Participant	Estimated Population 1990[1]	2000		2010	
		Low	High	Low	High
Brady	5,946	6,157	6,188	6,376	6,440
Eden	1,567	1,748	1,765	1,950	1,988
Live Oak Hills Sub.[2]	51	56	59	62	68
Melvin	184	184	188	185	194
Millersview-Doole WSC	3,420	3,523	3,559	3,630	3,703
North San Saba WSC	586	654	686	729	804
Richland WSC (Brady)	416	424	428	432	441
Rochelle WSC	205	209	211	213	217

[1] From Table 2.1-5
[2] Assumes three persons per connection, since available water use reports cover only four years and computed capita per connection does not appear reasonable.

2.4 Annual Water Requirements

From the projected per capita water use (Table 2.2-1), and the projected population (Table 2.3-3), the projected annual water requirements for each study participant can be calculated. Table 2.4-1 presents the projected annual water requirements for each entity for the planning period in both acre-feet per year and million gallons per day.

Table 2.4-1
Projected Annual Water Requirements

Participant	1990	2000		2010	
		Low	High	Low	High
	------acre-feet------				
Brady	1,838	1,806	2,114	1,778	2,077
Eden	277	293	344	312	367
Live Oak Hills Sub.	9	9	11	10	12
Melvin	32	30	35	29	35
Millersview-Doole WSC	529	517	606	504	597
North San Saba WSC	90	96	117	101	130
Richland WSC (Brady)	72	69	81	67	80
Rochelle WSC	35	34	40	33	39
	------million gallons per day------				
Brady	1.641	1.613	1.887	1.587	1.854
Eden	0.247	0.262	0.307	0.278	0.328
Live Oak Hills Sub.	0.008	0.008	0.009	0.008	0.011
Melvin	0.028	0.026	0.032	0.025	0.031
Millersview-Doole WSC	0.472	0.462	0.541	0.450	0.533
North San Saba WSC	0.081	0.086	0.104	0.094	0.116
Richland WSC (Brady)	0.064	0.062	0.072	0.060	0.071
Rochelle WSC	0.032	0.031	0.035	0.030	0.035

As a general check for accuracy, the annual water requirement for each county was compared to the TWDB's Annual Water Projection under the municipal category for each county. The projections given in Table 2.4-1 compare very closely with the county-wide municipal annual water requirements given in Table 2.4-2.

Table 2.4-2
County-Wide Municipal Annual Water Requirement

County	Base 1980	1990		2000		2010	
		Low	High	Low	High	Low	High
	-------------------------(Acre-Feet)-------------------------						
County (Municipal)[1]							
Concho	477	408	473	373	453	330	407
McCulloch	2,637	2,351	2,770	2,298	2,711	2,244	2,654
San Saba	1,260	1,147	1,276	1,132	1,291	1,129	1,299
County (Total)[2]							
Concho	2,432	5,386	7,143	5,487	7,034	5,444	6,932
McCulloch	8,295	7,710	9,375	8,139	9,864	8,340	10,097
San Saba	12,931	10,166	13,298	10,488	13,309	10,570	13,317

Source: Texas Water Development Board. All projections take into account water conservation.

[1] One acre-foot equals 325,851 gallons or 0.000892 mgd.

[2] Includes irrigation.

SECTION 3

3.0 EXISTING SYSTEM AND WATER QUALITY

In this section, the water supply source(s) and major system components for each of the participants are described. In addition, the latest water quality information for each participant's water supply, as available from the Texas Department of Health files, is presented.

3.1 City of Brady

The City of Brady operates two (2) district water systems. The first system, which will be described in this section, serves the City of Brady. The second system, which is described in the next section, serves the Brady Lake area.

3.1.1 Water Supply

The water supply for the City of Brady is provided by six wells that extend into the Hickory aquifer. The well capacities and locations are presented in Table 3.1-1. Of Brady's six wells, two are used for standby purposes only.

The total water supply capacity which the City of Brady has available to meet the present and future water supply requirements is 3,500 gpm (5.04 MGD). As was shown in Table 2.4-1, by 2010, the City is expected to have water requirements totalling 1.854 MGD. Assuming there is no loss over time in well production capacities, the existing water supply system is more than adequate to meet the future water requirements of the City of Brady.

Table 3.1-1
City of Brady Water Supply

Well No.	Capacity (gpm)	Location	Well Depth (ft)
1	*	Pump Station #1 - at power plant	2114
2	*	Pump Station #1 - at power plant	2112
3	950	Pump Station #2 - Oak Street	2082
4	550	Pump Station #3 - White Street	2127
5	1,000	Pump Station #4 - South China	2060
7	1,000	Pump Station #5 - South Avenue	2249

* Not in normal use at this time (standby use only).

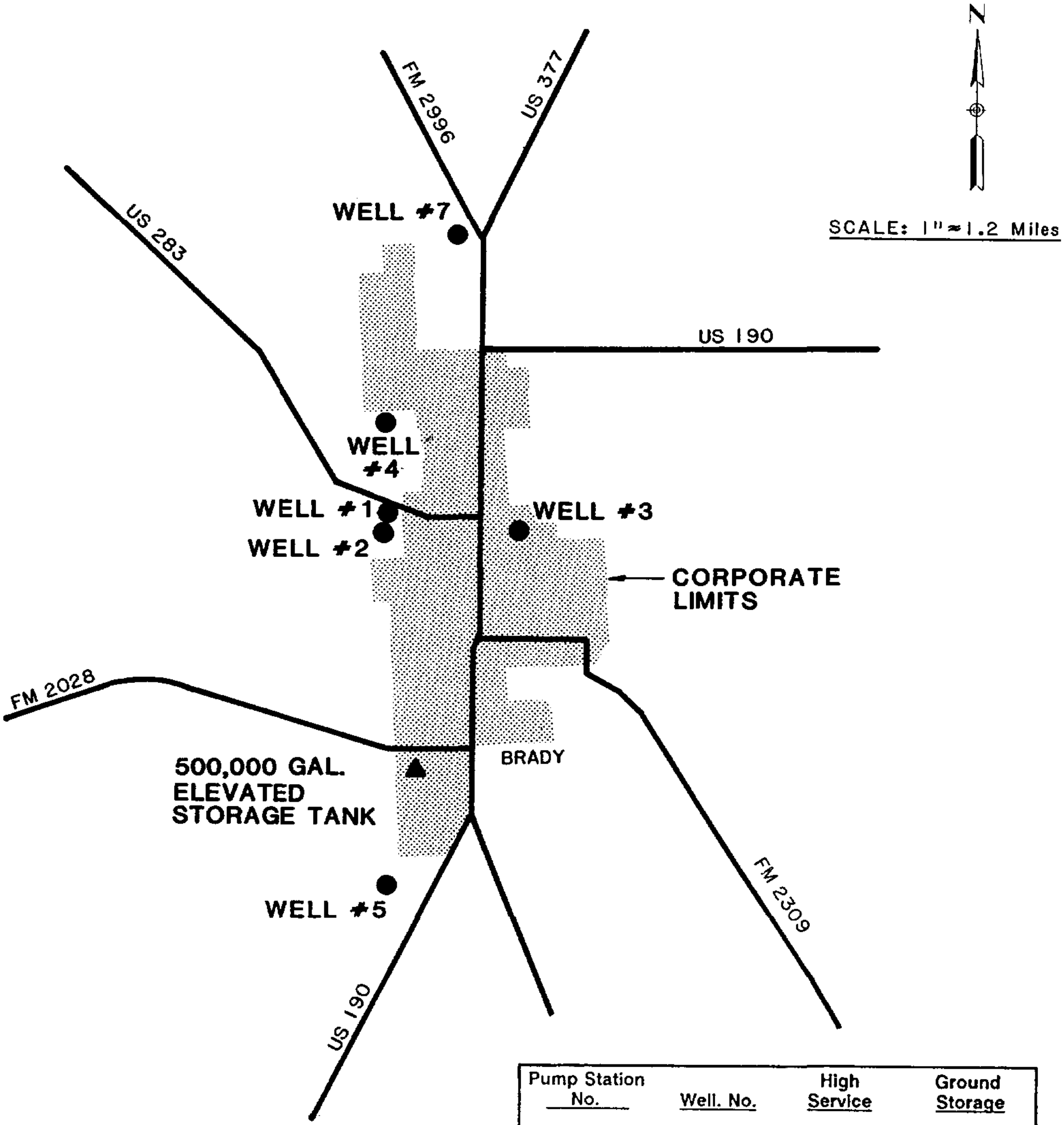
3.1.2 Major Distribution System Components

In addition to wells, other major system components of Brady's water distribution system include ground storage, high service pumping, and elevated storage which provides pressure maintenance. Figure 3.1-1 indicates the general location of the major system components. Note that the standby wells and the wells producing the smallest amount of water are located near the center of town.

Table 3.1-2 summarizes Brady's system components. These components were reviewed for adequacy to meet the minimum system capacity requirements for a public water system in accordance with the Rules and Regulations of the Texas Department of Health. All components meet or exceed these minimum requirements.

3.1.3 Water Quality

Table 3.1-3 presents the latest complete chemical analysis of the water supply for the City of Brady. Shaded areas on the table indicate constituent levels that exceed TDH limits. A quick review of the analysis shows that the City of Brady's drinking water is hard water



Pump Station No.	Well. No.	High Service	Ground Storage
1	1 & 2	*	*
2	3	1-800 gpm	30,000
3	4	1-1,000	25,000
4	5	1-1,250 1-1,000 1-750	1,000,000
5	7	2-500 gpm	

* - Not in use.

NOTE: DISTRIBUTION LINES ARE NOT SHOWN.

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REGIONAL WATER SUPPLY PLAN

CITY OF BRADY - MAJOR DISTRIBUTION SYSTEM COMPONENTS

FIGURE 3.1-1

Table 3.1-2
City of Brady-Major Distribution Supply Components

Storage

Type	Capacity (gal.)	Location
Ground	20,000	Pump Station #1 - at power plant
Ground	30,000	Pump Station #2 - Oak Street
Ground	25,000	Pump Station #3 - White Street
Ground	1,000,000	Pump Station #4 - South China
Elevated	500,000	Corner of Burns and Pine Streets
Pressure	4,000	Pump Station #5 - South Avenue

High Service

Number	Capacity (gpm)	Location
1	700	Pump Station #1 - at power plant
2	1,000	Pump Station #1 - at power plant
3	800	Pump Station #2 - Oak Street
4	1,000	Pump Station #3 - White Street
5	500	Pump Station #5 - South Avenue
6	500	Pump Station #5 - South Avenue
7	1,250	Pump Station #4 - South China
8	1,000	Pump Station #4 - South China
9	750	Pump Station #4 - South China

* Not in normal use at this time (standby use only).

with a high iron concentration level that could cause "red" water problems in the distribution system. A sequestering agent is added to minimize the potential "red" water problems.

In addition, the radionuclide levels, in particular gross alpha and combined radium, exceed the present MCL given in the Drinking Water Standards of the Texas Department of Health and radon exceeds the MCL in the Radionuclide Rules, proposed on June 17, 1991.

Table 3.1-3
City of Brady
Water Analysis

Constituent	Well #3	Well #4	Well #5	Well #7	TDH Limit
Calcium	52	54	52	56	
Chloride	12	19	12	20	300
Fluoride	0.8	0.7	0.8	0.7	2.0
Magnesium	40	44	40	46	
Nitrate (as N)	0.01	0.02	0.01	< 0.01	10
Sodium	21	27	20	28	
Sulfate	43	62	41	67	300
Total Hardness (as CaCO3)	296	317	296	330	
pH (units)	7.9	7.9	7.8	8.0	7.0
Conductivity (umhos/cm)	689	770	675	803	
Tot. Alka. (as CaCO3)	296	302	291	309	
Bicarbonate	361	368	355	377	
Carbonate	0	0	0	0	
Dissolved Solids	353	396	347	412	1,000
P. Alkalinity (as CaCO3)	0	0	0	0	
Arsenic	< 0.010	< 0.010	< 0.010	< 0.010	0.05
Barium	0.050	0.060	0.030	0.050	1.
Cadmium	< 0.005	< 0.005	< 0.005	< 0.005	0.010
Chromium	< 0.02	< 0.02	< 0.02	< 0.02	0.05
Copper	< 0.02	< 0.02	< 0.02	< 0.02	1.0
Iron	0.31	0.35	0.31	0.37	0.3
Lead	< 0.0200	< 0.0200	< 0.0200	<0.0200	0.05
Manganese	< 0.02	< 0.02	< 0.02	< 0.02	0.05
Mercury	0.0004	0.0003	0.0004	0.0004	0.002
Selenium	< 0.002	< 0.002	< 0.002	< 0.002	0.01
Silver	< 0.01	< 0.010	< 0.010	< 0.010	0.05
Zinc	0.07	< 0.02	< 0.02	< 0.02	5.0
Gross Alpha (pCi/l)				37.00 [2]	15
Combined Radium (pCi/l) [1]				12.40 [2]	5
Radium 226 (pCi/l)				4.70 [2]	
Radium 228 (pCi/l)				7.70 [2]	
Gross Beta (pCi/l)				36.00 [2]	
Radon (pCi/l) [3]	668	550	619		

NOTE: Unless otherwise noted, all constituent concentrations are in mg/l.

[1] Combined Radium is the sum of the concentration of Radium-226 an Radium-228.

[2] Composite sample from the distribution system.

[3] The results of the Radon analysis of each supply source are provided in Appendix B.

3.2 City of Brady - Brady Lake Water System

3.2.1 Water Supply

The water supply for the City of Brady - Brady Lake Water System is provided by two wells that extend into the Hickory aquifer. The well capacities and locations are presented in Table 3.2-1.

Table 3.2-1
City of Brady
Brady Lake Water Supply

Well No.	Capacity (gpm)	Location	Well Depth (ft)
6	70	Pump Station near Brady Lake	2225
8	390	Pump Station near Brady Lake	2460

The total water supply capacity which the Brady Lake Water System has available to meet present and future water supply requirements is 460 gpm (0.66 MGD). The water requirement for the Brady Lake Water System is included with the City of Brady's projections (Paragraph 3.1.1), and, as already established, the existing total well capacity exceeds the projected requirements.

Based on TDH minimum system capacity requirements for a well supply (0.6 gpm/connection) and assuming there is no loss over time in well production capacities, the existing well capacity for the Brady Lake water system is capable of serving, at the most, 765 connections, which greatly exceeds the present number of connections (109) served by the system.

3.2.2 Major Distribution System Components

In addition to wells, other major system components of the Brady Lake Water System include ground storage, high service pumping, and pressure maintenance facilities. Figure

3.2-1 indicates the general location of these components.

Table 3.2-2 summarizes Brady's system components. These system components were reviewed for adequacy to meet the minimum system capacity requirements for a public water system in accordance with the Rules and Regulations of the Texas Department of Health. All components meet or exceed these minimum requirements.

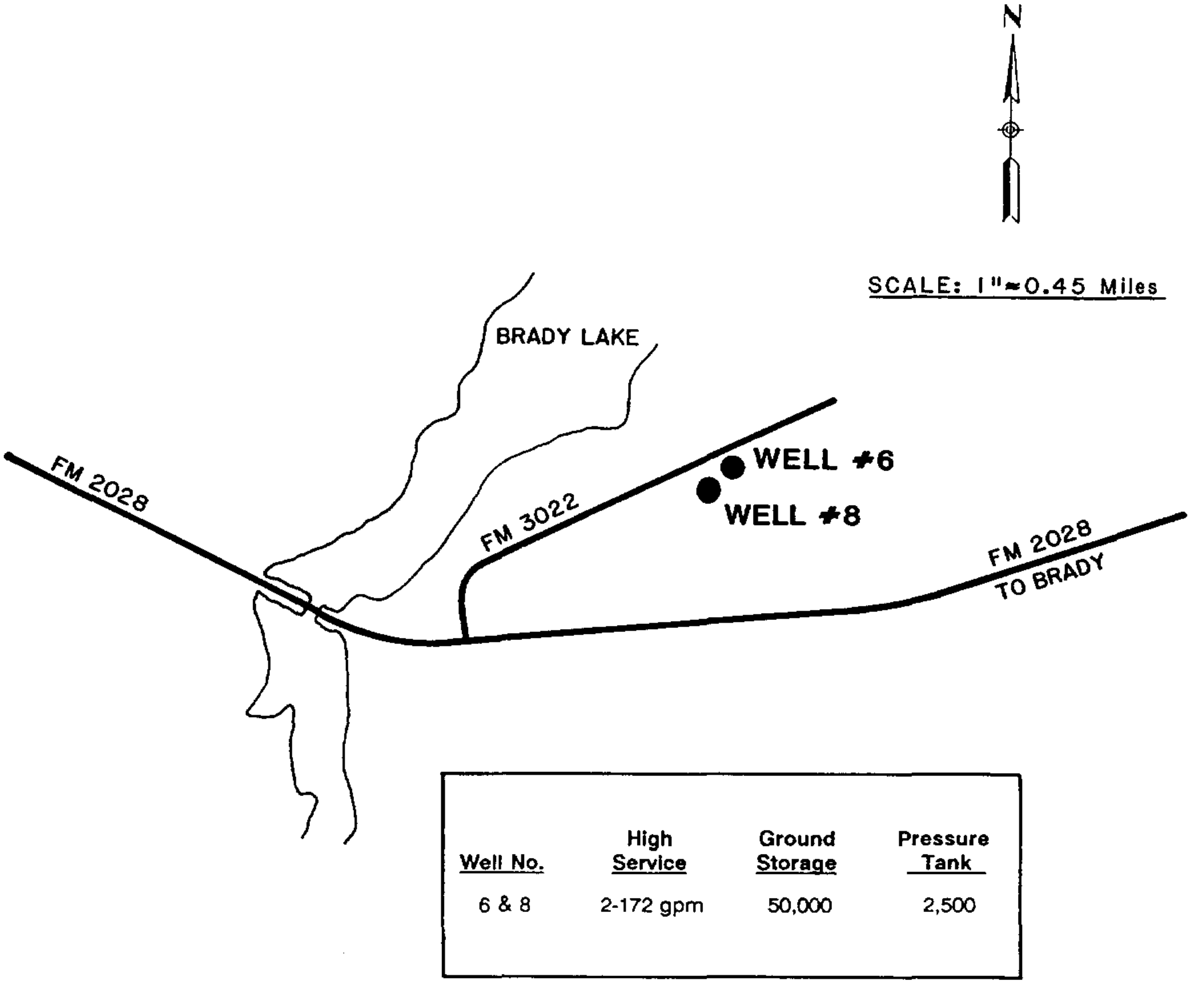
Table 3.2-2
City of Brady
Brady Lake Water System
Major Distribution Supply Components

Storage		
Type	Capacity (gal.)	Location
Ground	50,000	Pump Station near Brady Lake
Pressure	2,500	Pump Station near Brady Lake
High Service		
No.	Capacity (gpm)	Location
1	172	Pump Station near Brady Lake
2	172	Pump Station near Brady Lake

3.2.3 Water Quality

Table 3.2-3 presents the latest complete chemical analysis of the water supply wells for the City of Brady - Brady Lake Water System. Shaded areas on the table indicate constituent levels that exceed TDH limits. A quick review of the analysis shows that the drinking water for the Brady Lake Water System is hard water with a high iron concentration level that could cause "red" water problems in the distribution system. A sequestering agent is added to the water to minimize the potential "red" water problems.

In addition, the radionuclide levels, in particular gross alpha and combined radium, have in the past exceeded the present MCL given in the Drinking Water Standards of the Texas Department of Health and radon exceeds the MCL in the Radionuclide Rule proposed on June 17, 1991.



Well No.	High Service	Ground Storage	Pressure Tank
6 & 8	2-172 gpm	50,000	2,500

NOTE: DISTRIBUTION LINES ARE NOT SHOWN.

HDR Engineering, Inc.

REGIONAL WATER SUPPLY PLAN

CITY OF BRADY - BRADY LAKE - MAJOR DISTRIBUTION SYSTEM COMPONENTS

FIGURE 3.2-1

Table 3.2-3
Brady Lake Water System
Water Analysis

Constituent	Well #6	Well #8	TDH Limit
Calcium	53	58	
Chloride	31	13	300
Fluoride	0.7	0.7	2.0
Magnesium	50	45	
Nitrate (as N)	0.02	0.02	10
Sodium	35	23	
Sulfate	82	59	300
Total Hardness (as $CaCO_3$)	337	329	
pH (units)	8.0	8.0	7.0
Conductivity (umhos/cm)	858	785	
Tot. Alka. (as $CaCO_3$)	294	316	
Bicarbonate	359	386	
Carbonate	0	0	
Dissolved Solids	439	395	1,000
P. Alkalinity (as $CaCO_3$)	0	0	
Arsenic	< 0.010	< 0.010	0.05
Barium	0.030	0.060	1.
Cadmium	< 0.005	< 0.005	0.010
Chromium	< 0.02	< 0.02	0.05
Copper	< 0.02	< 0.02	1.0
Iron	0.99	0.95	0.3
Lead	< 0.0200	< 0.0200	0.05
Manganese	< 0.02	< 0.02	0.05
Mercury	0.0003	0.0004	0.002
Selenium	< 0.002	< 0.002	0.01
Silver	< 0.010	< 0.010	0.05
Zinc	< 0.02	< 0.02	5.0
Gross Alpha (pCi/l)		13.30[2]	15
Combined Radium (pCi/l)[1]		7.6[2]	5
Radium 226 (pCi/l)		3.20[2]	
Radium 228 (pCi/l)		4.40[2]	
Gross Beta (pCi/l)		20.30[2]	
Radon (pCi/l)[3]	486	610	

NOTE: Unless otherwise noted, all constituent concentrations are in mg/l.

[1] Combined Radium is the sum of the concentration of Radium-226 and Radium-228.

[2] Composite sample from the distribution system.

[3] The results of the Radon analysis of each supply source are provided in Appendix B.

3.3 City of Eden

3.3.1 Water Supply

The water supply for the City of Eden is provided by six wells. The four shallow wells are producing their waters from Edwards aquifer and associated limestones of the Edwards-Trinity Plateau aquifer. The two deep wells are producing their waters from the "sandy" portions of the San Saba member and Point Peak member of the Wilberns Formation located just below the Ellenburger Group. The well capacities and locations are presented in Table 3.3-1.

Table 3.3-1
City of Eden Water Supply

Well No.	Capacity (gpm)	Location	Well Depth (ft)
1	200	At main pump station	42
2	400	Northeast of hospital	4040
3	150	Ellis Ranch	35
4	150	Canning Ranch	35
5	75	Canning Ranch	35
6	350	East of elevated storage tank	4060

The total water supply capacity which the City of Eden has available to meet present and future water supply requirements is 1,325 gpm (1.91 MGD). As was shown in Table 2.4-1, by 2010, the City is expected to have water requirements totalling 0.328 MGD. Assuming there is no loss over time in well production capacities, the existing water supply system is more than adequate to meet the future water requirements of the City of Eden.

3.3.2 Major Distribution System Components

In addition to wells, other major system components of the City of Eden's water distribution system include ground storage, high service pumping, and elevated storage which provides pressure maintenance. Figure 3.3-1 indicates the general location of the major system components.

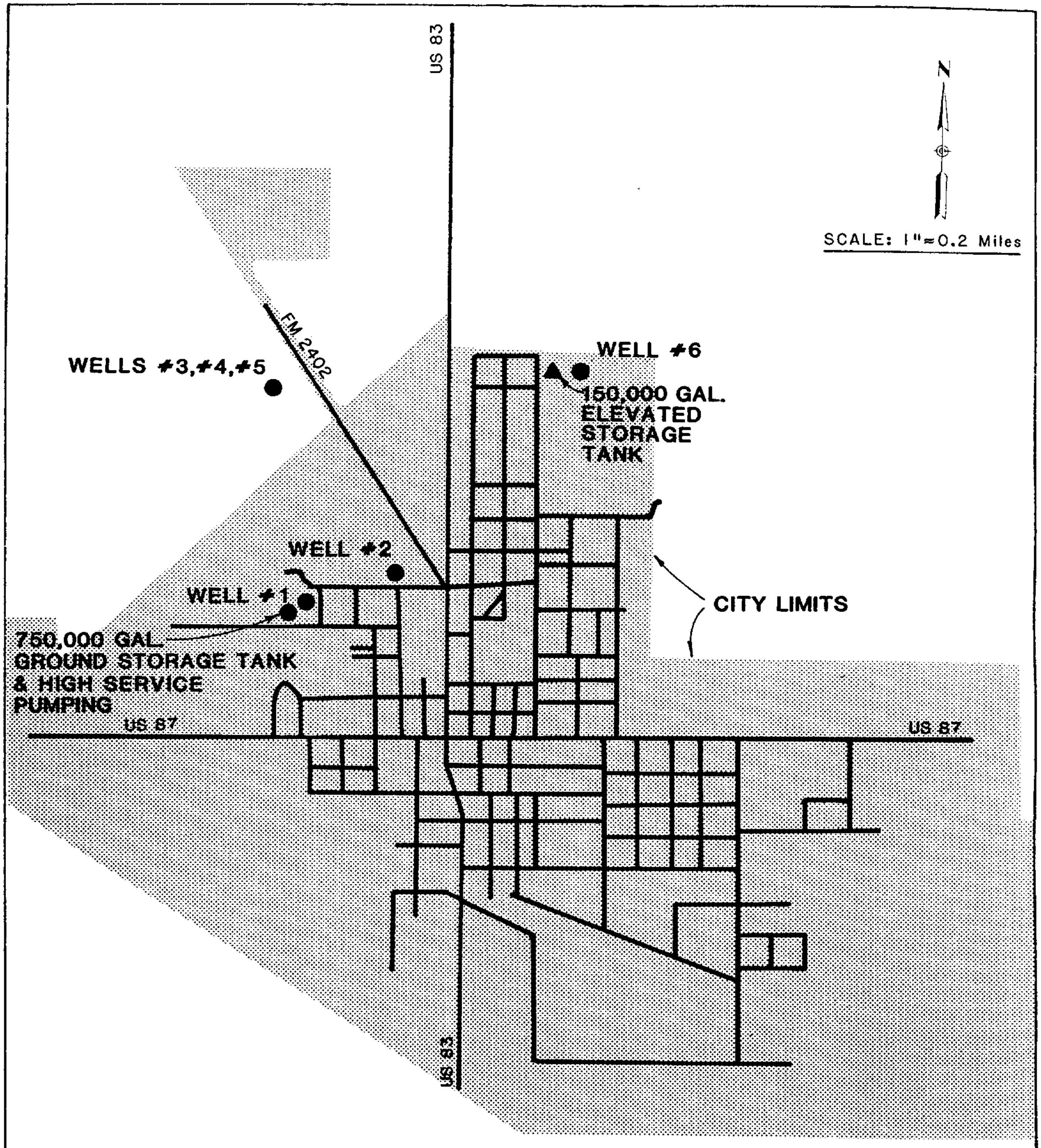
Table 3.3-2 summarizes Eden's system components. These components were reviewed for adequacy to meet the minimum system capacity requirements for a public water system in accordance with the Rules and Regulations of the Texas Department of Health. All components meet or exceed these minimum requirements.

Table 3.3-2
City of Eden
Major Distribution Supply Components

Storage		
Type	Capacity (gal.)	Location
Ground	750,000	At pump station
Elevated	150,000	Northeast part of town
High Service		
No.	Capacity (gpm)	Location
1	500	Southeast of ground storage tank
2	1,000	Southeast of ground storage tank
3	500	Southeast of ground storage tank

3.3.3 Water Quality

Table 3.3-3 presents the latest complete chemical analysis and Table 3.3-4 presents the latest Radon Analysis of the water supply wells for the City of Eden. Shaded areas on the tables indicate constituent levels that exceed TDH limits. A quick review of the analysis shows that the drinking water for the City of Eden is hard water.



NOTE: DISTRIBUTION LINES ARE NOT SHOWN.

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REGIONAL WATER SUPPLY PLAN

CITY OF EDEN - MAJOR DISTRIBUTION SYSTEM COMPONENTS

FIGURE 3.3-1

Table 3.3-3
City of Eden
Water Analysis

Constituent	Distribution[2]	TDH Limit
Calcium	76	
Chloride	28	300
Fluoride	0.5	2.0
Magnesium	20	
Nitrate (as N)	5.52	10
Sodium	24	
Sulfate	21	300
Total Hardness (as $CaCO_3$)	274	
pH (units)	7.0	7.0
Conductivity (umhos/cm)	671	
Tot. Alka. (as $CaCO_3$)	247	
Bicarbonate	301	
Carbonate	0	
Dissolved Solids	343	1,000
P. Alkalinity (as $CaCO_3$)	0	
Arsenic	< 0.010	0.05
Barium	0.20	1.
Cadmium	< 0.005	0.010
Chromium	< 0.02	0.05
Copper	< 0.02	1.0
Iron	0.03	0.3
Lead	< 0.0200	0.05
Manganese	< 0.02	0.05
Mercury	0.0011	0.002
Selenium	< 0.002	0.01
Silver	< 0.01	0.05
Zinc	0.28	5.0
Gross Alpha (pCi/l)	4.20	15
Combined Radium (pCi/l)[1]	7.40	5
Radium 226 (pCi/l)	2.60	
Radium 228 (pCi/l)	4.80	
Gross Beta (pCi/l)	14.00	

NOTE: Unless otherwise noted, all constituent concentrations are in mg/l.

[1] Combined Radium is the sum of the concentration of Radium-226 and Radium-228.

[2] Composite sample from the distribution system.

Table 3.3-4
City of Eden
Radon Analysis

Location	Well #1	Well #2	Well #3	Well #4	Well #5	Well #6	Tower Effluent
Concentration (pCi/l)	672	400	774	666	653	171	< 100

NOTE: The results of the Radon analysis of each supply source are provided in Appendix B.

In addition, the radionuclide levels, in particular gross alpha and combined radium, have in the past exceeded the present MCL given in the Drinking Water Standards of the Texas Department of Health and radon exceed the MCL in the Radionuclide Rule, proposed on June 17, 1991.

3.4 Live Oak Hills Subdivision

3.4.1 Water Supply

The water supply for the Live Oak Hills Subdivision is provided by one well that extends into the Cap Mountain Member of the Riley Formation, of which the Hickory aquifer is a lower member. The well capacity and location is presented in Table 3.4-1.

Table 3.4-1
Live Oak Hills Subdivision Water Supply

Well No.	Capacity (gpm)	Location	Well Depth (ft)
1	45	Next to pump station and ground storage tank	1230

The total water supply capacity which Live Oak Hills Subdivision has available to meet the present and future water supply requirements is 45 gpm (0.065 MGD). As was shown in Table 2.4-1, by 2010, the Subdivision is expected to have water requirements totalling 0.011 MGD. Assuming there is no loss over time in well production capacities, the existing water supply system is more than adequate to meet the future water requirements of the Live Oak Hills Subdivision.

3.4.2 Major Distribution System Components

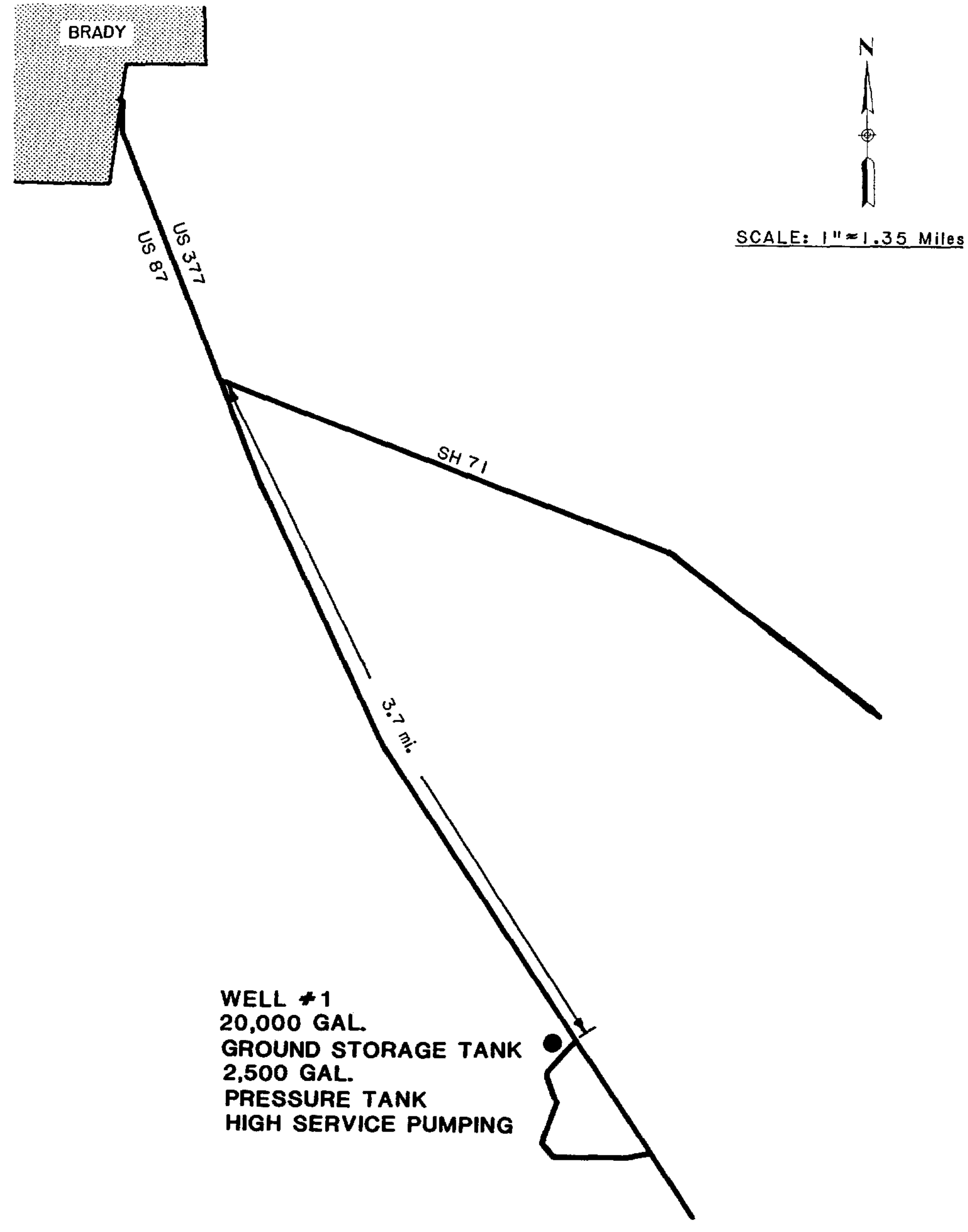
In addition to the well, other major system components of the Subdivision's water distribution system include ground storage, high service pumping, and pressure maintenance facilities. Figure 3.4-1 indicates the general location of the major system components. Table 3.4-2 summarizes the Subdivision's system components. These components were reviewed for adequacy to meet the minimum system capacity requirements for a public water system in accordance with the Rules and Regulations of the Texas Department of Health. All components meet or exceed these minimum requirements.

Table 3.4-2
Live Oak Hills Subdivision
Major Distribution Supply Components

Storage		
Type	Capacity (gal.)	Location
Ground	20,000	At pump station
Pressure	2,500	At pump station
High Service		
No.	Capacity (gpm)	Location
1	35	At pump station
2	35	At pump station

3.4.3 Water Quality

Table 3.4-3 presents the latest complete chemical analysis of the water supply for the Live Oak Hills Subdivision. Shaded areas on the table indicate constituent levels that



NOTE: DISTRIBUTION LINES ARE NOT SHOWN.

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REGIONAL WATER SUPPLY PLAN

LIVE OAK HILLS SUBDIVISION - MAJOR DISTRIBUTION SYSTEM COMPONENTS

FIGURE 3.4-1

Table 3.4-3
Live Oak Hills Subdivision
Water Analysis

Constituent	Well #1	TDH Limit
Calcium	59	
Chloride	19	300
Fluoride	0.9	2.0
Magnesium	36	
Nitrate (as N)	0.10	10
Sodium	22	
Sulfate	44	300
Total Hardness (as $CaCO_3$)	297	
pH (units)	8.1	7.0
Conductivity (umhos/cm)	720	
Tot. Alka. (as $CaCO_3$)	287	
Bicarbonate	350	
Carbonate	0	
Dissolved Solids	361	1,000
P. Alkalinity (as $CaCO_3$)	0	
Arsenic	< 0.010	0.05
Barium	0.09	1.
Cadmium	< 0.005	0.010
Chromium	< 0.02	0.05
Copper	0.02	1.0
Iron	0.17	0.3
Lead	< 0.0200	0.05
Manganese	< 0.02	0.05
Mercury	< 0.0002	0.002
Selenium	< 0.002	0.01
Silver	< 0.01	0.05
Zinc	0.027	5.0
Gross Alpha (pCi/l)	21.00	15
Combined Radium (pCi/l) [1]	10.40	5
Radium 226 (pCi/l)	4.90	
Radium 228 (pCi/l)	5.50	
Gross Beta (pCi/l)	23.00	
Radon (pCi/l) [2]	535	

NOTE: Unless otherwise noted, all constituent concentrations are in mg/l.

[1] Combined Radium is the sum of the concentration of Radium-226 and Radium-228.

[2] The results of the Radon analysis for each supply source are provided in Appendix B.

exceed TDH limits. A quick review of the analysis shows that the drinking water for the Subdivision is hard water.

In addition, the radionuclide levels, in particular gross alpha and combined radium, exceed the present MCL given in the Drinking Water Standards of the Texas Department of Health and radon exceeds the MCL in the Radionuclide Rule, proposed on June 17, 1991.

3.5 City of Melvin

3.5.1 Water Supply

The water supply for the City of Melvin is provided by two wells, which produce their waters from the "sandy" portions of the San Saba members of the Wilberns Formation located just below the Ellenburger Group. The well capacities and locations are presented in Table 3.5-1.

Table 3.5-1
City of Melvin Water Supply

Well No.	Capacity (gpm)	Location	Well Depth (ft)
2	195	Northwest corner of town at pump station	2500
3	170	40 yards west of elevated storage tank	2400

The total water supply capacity which the City of Melvin has available to meet the present and future water supply requirements is 365 gpm (0.526 MGD). As was shown in Table 2.4-1, by 2010, the City is expected to have water requirements totalling 0.031 MGD. Assuming there is no loss over time of well production capacities, the existing water supply system is more than adequate to meet the future water requirements of the City of Melvin.

3.5.2 Major Distribution System Components

In addition to wells, other major system components of Melvin's water distribution system include ground storage, high service pumping, and elevated storage which provides pressure maintenance. Figure 3.5-1 indicates the general location of the major system components.

Table 3.5-2 summarizes Melvin's system components. These components were reviewed for adequacy to meet the minimum system capacity requirements for a public water system in accordance with the Rules and Regulations of the Texas Department of Health. All components meet or exceed these minimum requirements.

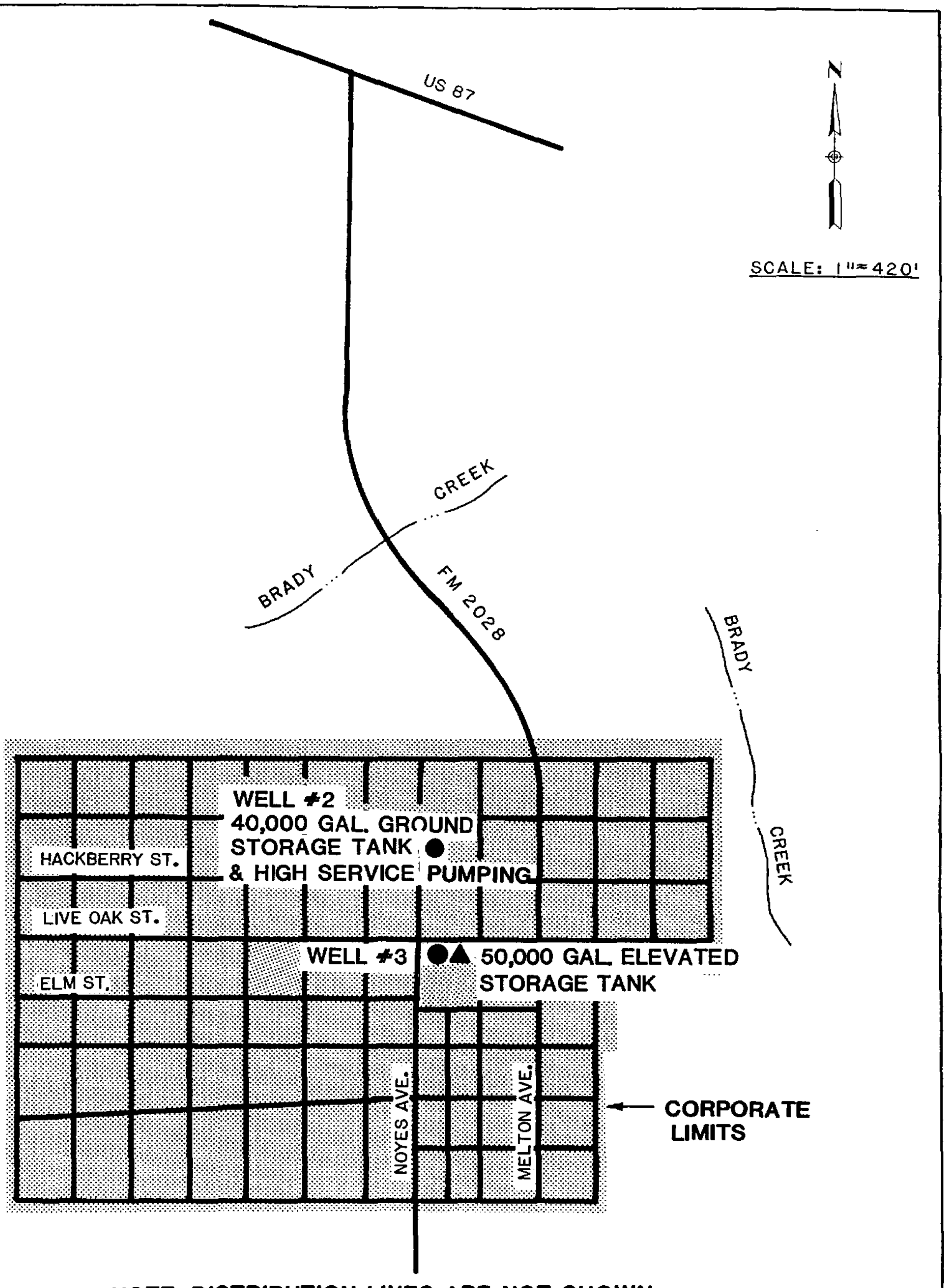
Table 3.5-2
City of Melvin
Major Distribution Supply Components

Storage		
Type	Capacity (gal.)	Location
Ground	40,000	Northwest corner of town at p.s.
Elevated	50,000	West of town square
High Service		
No.	Capacity (gpm)	Location
1	320	At pump station
2	320	At pump station
3	250	At elevated storage tank

*Not in normal use at this time (standby use only).

3.5.3 Water Quality

Table 3.5-3 presents the latest complete chemical analysis of the water supply for the City of Melvin. Shaded areas on the table indicate constituent levels that exceed TDH limits. A quick review of the analysis shows that Melvin's drinking water is hard water with high iron and manganese concentration levels that could cause "red" and "black" water problems in the distribution system.



NOTE: DISTRIBUTION LINES ARE NOT SHOWN.

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REGIONAL WATER SUPPLY PLAN

CITY OF MELVIN - MAJOR DISTRIBUTION SYSTEM COMPONENTS

FIGURE 3.5-1

Table 3.5-3
City of Melvin
Water Analysis

Constituent	Distribution	Well #2	TDH Limit
Chloride	53	44	300
Fluoride	0.8	0.8	2.0
Magnesium	23	22	
Nitrate (as N)	0.02	0.01	10
Sodium	46	43	
Sulfate	37	35	300
Total Hardness (as $CaCO_3$)	245	230	
pH (units)	8.2	8.1	7.0
Conductivity (umhos/cm)	740	698	
Tot. Alka. (as $CaCO_3$)	242	243	
Bicarbonate	295	296	
Carbonate	0	0	
Dissolved Solids	374	356	1,000
P. Alkalinity (as $CaCO_3$)	0	0	
Arsenic		< 0.010	0.05
Barium		< 0.50	1.
Cadmium		< 0.005	0.010
Chromium		< 0.02	0.05
Copper		< 0.02	1.0
Iron	1.04	9.14	0.3
Lead		< 0.02	0.05
Manganese	0.02	0.16	0.05
Mercury		0.0002	0.002
Selenium		< 0.002	0.01
Silver		< 0.01	0.05
Zinc		< 0.02	5
Gross Alpha (pCi/l)	24.00		15
Combined Radium (pCi/l)[1]	7.90		5
Radium 226 (pCi/l)	5.70		
Radium 228 (pCi/l)	2.20		
Gross Beta (pCi/l)	20.00		
Radon (pCi/l)[3]	318[2]	< 100	

NOTE: Unless otherwise noted, all constituent concentrations are in mg/l.
[1] Combined Radium is the sum of the concentration of Radium-226 and Radium-228.
[2] The radon concentration shown under the Distribution heading is for Well #1.
[3] The results of the Radon analysis for each supply source are provided in Appendix B.

In addition, the radionuclide levels, in particular gross alpha and combined radium, exceed the present MCL given in the Drinking Water Standards of the Texas Department of Health and radon just exceeds the MCL in the Radionuclide Rule, proposed June 17, 1991.

3.6 Millersview-Doole Water Supply Corporation

3.6.1 Water Supply

The water supply for Millersview-Doole Water Supply Corporation is provided primarily from two wells (Well Nos. 2 and 3) that extend into the Hickory aquifer. Well No. 1, which has an unknown capacity, is not in normal use at this time and extends into the water-bearing sands of the Point Peak member of the Wilberns Formation. The well capacities and locations are presented in Table 3.6-1.

Table 3.6-1
Millersview-Doole Water Supply Corporation Water Supply

Well No.	Capacity (gpm)	Location	Well Depth (ft)
1	*	About 1 mile northeast of Well No. 3	2485
2	410	About 0.5 mile south of Well No. 3	3209
3	420	About 1.8 miles east of Melvin	3295

* Capacity is unknown and well is not in normal use at this time.

The total water supply capacity, based on the capacity of Well Nos. 2 and 3, which the Millersview-Doole WSC has available to meet present and future water supply requirements is 830 gpm (1.195 MGD). As was shown in Table 2.4-1, by 2010, the Corporation is expected to have water requirements totalling 0.533 MGD. Assuming there is no loss over time in well production capacities, the existing water supply system is more than adequate to meet the future water requirements of the Millersview-Doole Water Supply Corporation.

3.6.2 Major Distribution System Components

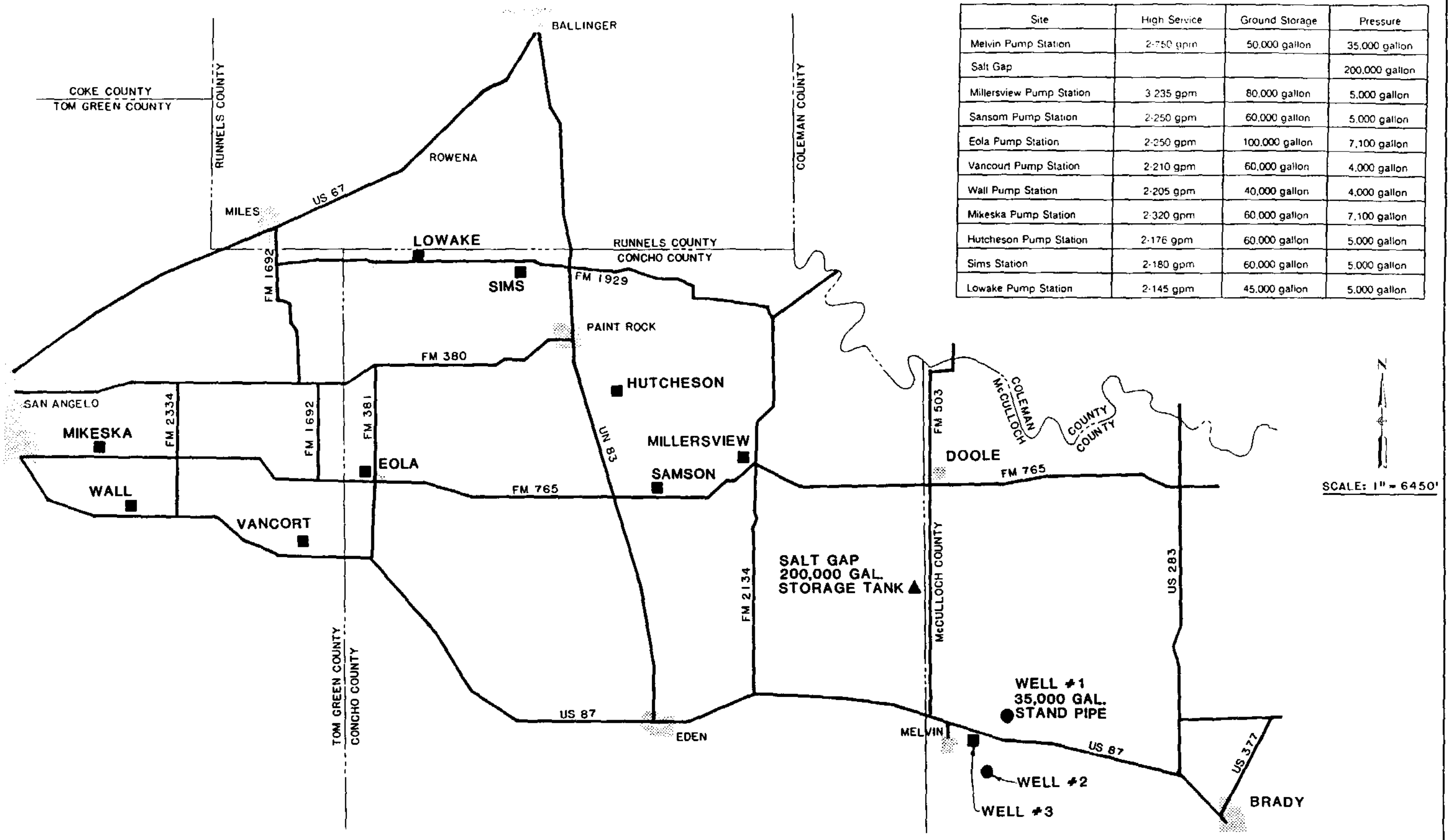
In addition to wells, other major system components of the Corporation's water distribution system include ground storage, high service pumping, and elevated storage which provides pressure maintenance. Figure 3.6-1 indicates the general location of the major system components.

Table 3.6-2 summarizes the Corporation's system components. These components were reviewed for adequacy to meet the minimum system capacity requirements for a public water system in accordance with the Rules and Regulations of the Texas Department of Health. All components meet or exceed these minimum requirements. The only potential problem is the minimum system capacity requirements for well capacity. Based on the TDH requirement of 0.6 gpm/connection, the Corporation will need to develop a new supply source in the near future.

3.6.3 Water Quality

Table 3.6-3 presents the latest complete chemical analysis of the water supply wells for the Millersview-Doole Water Supply Corporation. Shaded areas on the table indicate constituent levels that exceed TDH limits. A quick review of the analysis shows that the drinking water for the Corporation is hard water and the fluoride concentration exceeds the secondary MCL. In addition, a sequestering agent is added to minimize the potential high iron levels causing "red" water in the distribution system.

In addition, the radionuclide levels, in particular gross alpha and combined radium, exceed the present MCL given in the Drinking Water Standards of the Texas Department of Health and radon exceeds the MCL in the Radionuclide Rule proposed on June 17, 1991.



Site	High Service	Ground Storage	Pressure
Melvin Pump Station	2-750 gpm	50,000 gallon	35,000 gallon
Salt Gap			200,000 gallon
Millersview Pump Station	3 235 gpm	80,000 gallon	5,000 gallon
Sansom Pump Station	2-250 gpm	60,000 gallon	5,000 gallon
Eola Pump Station	2-250 gpm	100,000 gallon	7,100 gallon
Vancourt Pump Station	2-210 gpm	60,000 gallon	4,000 gallon
Wall Pump Station	2-205 gpm	40,000 gallon	4,000 gallon
Mikeska Pump Station	2-320 gpm	60,000 gallon	7,100 gallon
Hutcheson Pump Station	2-176 gpm	60,000 gallon	5,000 gallon
Sims Station	2-180 gpm	60,000 gallon	5,000 gallon
Lowake Pump Station	2-145 gpm	45,000 gallon	5,000 gallon

LEGEND FOR SYMBOLS

- ● WELL
- ▲ ELEVATED STORAGE TANK
- ■ PUMP STATION

NOTE: DISTRIBUTION LINES ARE NOT SHOWN.

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REGIONAL WATER SUPPLY PLAN

MILLERSVIEW - DOOLE WATER SERVICE CORP. - MAJOR DISTRIBUTION SYSTEM COMPONENTS

FIGURE 3.6-1

Table 3.6-2
Millersview-Doole Water Supply Corporation
Major Distribution Supply Components

Storage

Type	Capacity (gal.)	Location
Ground	50,000	Melvin Pump Station
Elevated	200,000	Salt Gap Station
Ground	60,000	Sansom Pump Station
Pressure	5,000	Sansom Pump Station
Ground	100,000	Eola Pump Station
Pressure	7,000	Eola Pump Station
Ground	60,000	Mikeska Pump Station
Pressure	7,000	Mikeska Pump Station
Ground	40,000	Vancourt Pump Station
Pressure	4,000	Vancourt Pump Station
Ground	70,000	Wall Pump Station
Pressure	4,000	Wall Pump Station
Ground	60,000	Sims Pump Station
Pressure	5,000	Sims Pump Station
Ground	60,000	Hutcheson Pump Station
Pressure	5,000	Hutcheson Pump Station
Ground	80,000	Millersview Pump Station
Pressure	5,000	Millersview Pump Station
Pressure	38,000	Well #1 (Pear Gap) - Standpipe
Ground	45,000*	Lowake Pump Station
Pressure	5,000*	Lowake Pump Station
Pressure	5,000*	Hwy. 283 Near Lohn

* Not in normal use at this time (standby use only).

Table 3.6-3
Millersview-Doole Water Supply Corporation
Water Analysis

Constituent	Well #3	Well #2	TDH Limit
Calcium		38	
Chloride		99	300
Fluoride		2.2	2.0
Magnesium		17	
Nitrate (as N)		0.17	10
Sodium		137	
Sulfate		107	300
Total Hardness (as $CaCO_3$)		162	
pH (units)		8.3	7.0
Conductivity (umhos/cm)		1085	
Tot. Alka. (as $CaCO_3$)		238	
Bicarbonate		290	
Carbonate		0	
Dissolved Solids		569	1,000
P. Alkalinity (as $CaCO_3$)		0	
Arsenic		< 0.010	0.05
Barium		< 0.50	1.
Cadmium		< 0.005	0.010
Chromium		< 0.02	0.05
Copper		0.30	1.0
Iron		0.25	0.3
Lead		0.02	0.05
Manganese		< 0.02	0.05
Mercury		< 0.0002	0.002
Selenium		< 0.002	0.01
Silver		< 0.01	0.05
Zinc		0.09	5.0
Gross Alpha (pCi/l)		113.00 [2]	15
Combined Radium (pCi/l) [1]		79.00 [2]	5
Radium 226 (pCi/l)		18.00 [2]	
Radium 228 (pCi/l)		61.00 [2]	
Gross Beta (pCi/l)		113.00 [2]	
Radon (pCi/l) [3]	572	605	300 [4]

NOTE: Unless otherwise noted, all constituent concentrations are in mg/l.

[1] Combined Radium is the sum of the concentration of Radium-226 and Radium-228.

[2] Composite sample from the distribution system.

[3] The results of the Radon analysis are provided in Appendix B.

[4] The sample location for this Radon Analysis is the high service pumps after storage.

3.7 North San Saba Water Supply Corporation

3.7.1 Water Supply

The water supply for the North San Saba Water Supply Corporation is provided by one well that extends into the Hickory aquifer. The well capacity and location is presented in Table 3.7-1.

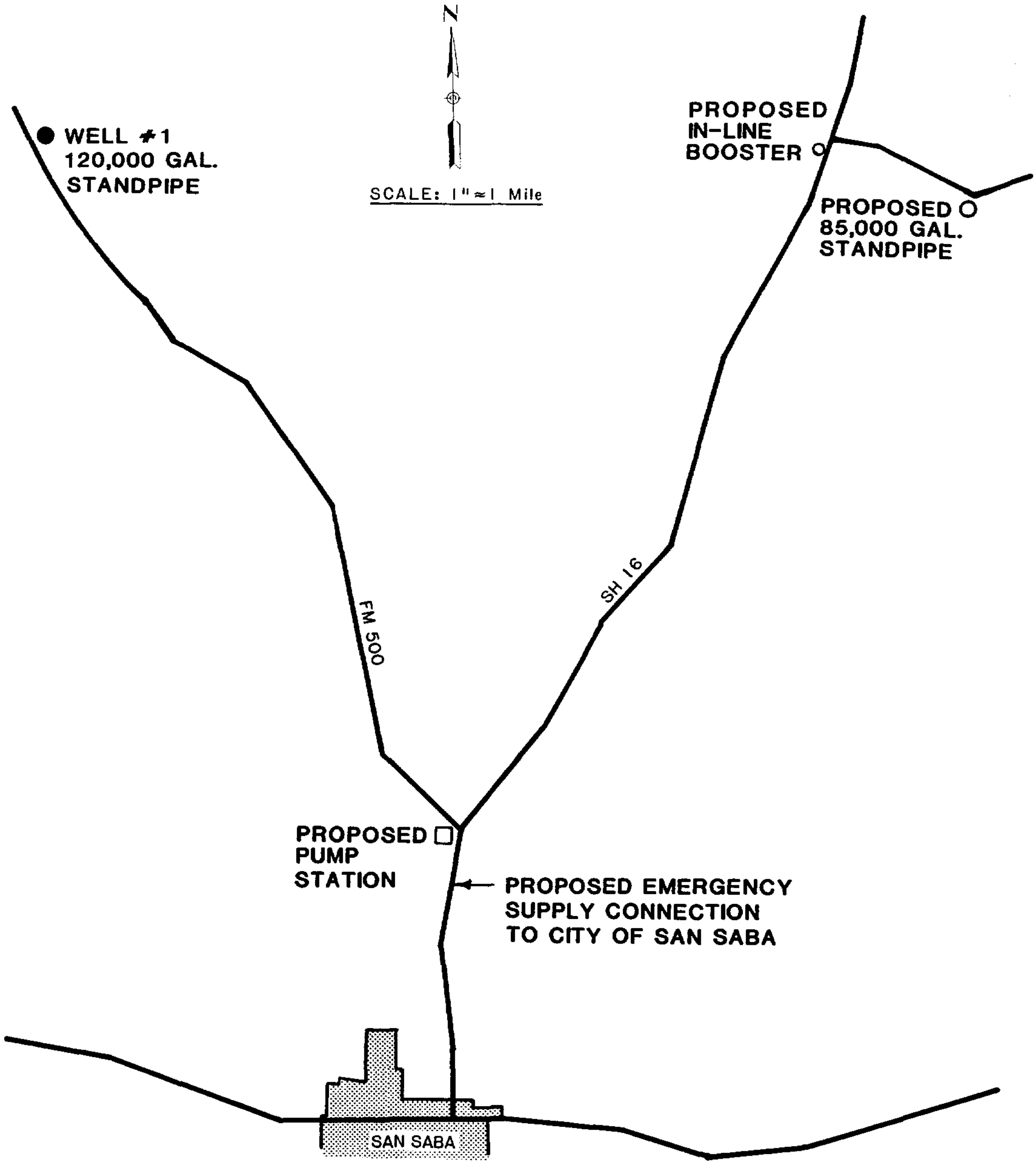
Table 3.7-1
North San Saba Water Supply Corporation
Water Supply

Well No.	Capacity (gpm)	Location	Well Depth (ft)
1	237	FM 500	3488

The total water supply capacity which the North San Saba Water Supply Corporation has available to meet present and future water supply requirements is 237 gpm (0.34 MGD). As was shown in Table 2.4-1, by 2010, the Corporation is expected to have water requirements totalling 0.116 MGD. Assuming no loss over time in well production capacity, the existing water supply system is more than adequate to meet the future water requirements of the North San Saba Water Supply Corporation. However, the Corporation is about to begin construction on an enhancement project that will provide a supplemental or emergency supply from the City of San Saba.

3.7.2 Major Distribution System Components

In addition to the one well, other major system components of the Corporation's water distribution system include ground storage, high service pumping, and pressure maintenance facilities. Figure 3.7-1 indicates the general location of the major system



NOTE: DISTRIBUTION LINES ARE NOT SHOWN.

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REGIONAL WATER SUPPLY PLAN

NORTH SAN SABA WATER SERVICE CORP. - MAJOR DISTRIBUTION SYSTEM COMPONENTS

FIGURE 3.7-1

components.

As part of the construction project that is about to begin, the Corporation will be adding two (2) booster pump stations and standpipes to improve the service to existing and future customers.

Table 3.7-2 summarizes the Corporation's existing system components. These components were reviewed for adequacy to meet the minimum system capacity requirements for a public water system in accordance with the Rules and Regulations of the Texas Department of Health. All existing components meet or exceed these minimum requirements based on the present member of customers.

Table 3.7-2
North San Saba Water Supply Corporation
Major Distribution Supply Components

Storage		
Type	Capacity (gal.)	Location
Pressure	110,000	FM 500 - standpipe
High Service		
No.	Capacity (gpm)	Location
None		

3.7.3 Water Quality

Table 3.7-3 presents the latest complete chemical analysis of the water supply for the North San Saba Water Supply Corporation. Shaded areas on the table indicate constituent levels that exceed TDH limits. A quick review of the analysis shows that the drinking water for the Corporation is very soft water (< 50 ppm as $CaCO_3$ of Total Hardness) with a total dissolved solids (TDS) concentration that is just below the Drinking Water Standards MCL.

In addition, the radionuclide levels, in particular gross alpha and combined radium, exceed the present maximum contaminant level given in the Drinking Water Standards of the Texas Department of Health and radon exceeds the MCL in the Radionuclide Rule proposed on June 17, 1991.

3.8 Richland Water Supply Corporation - Brady System

3.8.1 Water Supply

The water supply for the Richland Water Supply Corporation - Brady System is provided by one well that extends into the Hickory aquifer. The well capacity and location is presented in Table 3.8-1.

Table 3.8-1
Richland Water Supply Corporation - Brady System
Water Supply

Well No.	Capacity (gpm)	Location	Well Depth (ft)
1	310	Farm Road 1121 and U.S Highway 377	2640

The total water supply capacity which the Richland Water Supply Corporation System has available to meet present and future water supply requirements is 310 gpm (0.446 MGD). As was shown in Table 2.4-1, by 2010, the Corporation is expected to have water requirements totalling 0.071 MGD. Assuming there is no loss over time in well production capacity, the existing water supply system is more than adequate to meet the future water requirements of the Richland Water Supply Corporation - Brady System.

Table 3.7-3
North San Saba Water Supply Corporation
Water Analysis

Constituent	Well #1	TDH Limit
Calcium	5	
Chloride	288	300
Fluoride	1.6	2.0
Magnesium	1	
Nitrate (as N)	0.04	10
Sodium	354	
Sulfate	25	300
Total Hardness (as $CaCO_3$)	17	
pH (units)	8.5	7.0
Conductivity (umhos/cm)	1796	
Tot. Alka. (as $CaCO_3$)	364	
Bicarbonate	427	
Carbonate	8	
Dissolved Solids	901	1,000
P. Alkalinity (as $CaCO_3$)	7	
Arsenic	< 0.010	0.05
Barium	< 0.50	1.
Cadmium	< 0.005	0.010
Chromium	< 0.02	0.05
Copper	< 0.02	1.0
Iron	0.02	0.3
Lead	< 0.02	0.05
Manganese	< 0.02	0.05
Mercury	< 0.0002	0.002
Selenium	< 0.002	0.01
Silver	< 0.01	0.05
Zinc	< 0.02	5.0
Gross Alpha (pCi/l)	28.00	15
Combined Radium (pCi/l)[1]	29.40	5
Radium 226 (pCi/l)	8.10	
Radium 228 (pCi/l)	21.30	
Gross Beta (pCi/l)	18.00	
Radon (pCi/l)[2]	748	

NOTE: Unless otherwise noted, all constituent concentrations are in mg/l.

[1] Combined Radium is the sum of the concentration of Radium-226 and Radium-228.

[2] The results of the Radon analysis are provided in Appendix B.

3.8.2 Major Distribution System Components

In addition to the one well, other major system components of the Corporation's water distribution system include ground storage, high service pumping, and elevated storage which provides pressure maintenance. Figure 3.8-1 indicates the general location of the major system components.

Table 3.8-2 summarizes the Corporation's system components. These major distribution system components were reviewed for adequacy to meet the minimum system capacity requirements for a public water system in accordance with the Rules and Regulations of the Texas Department of Health. All components meet or exceed these minimum requirements.

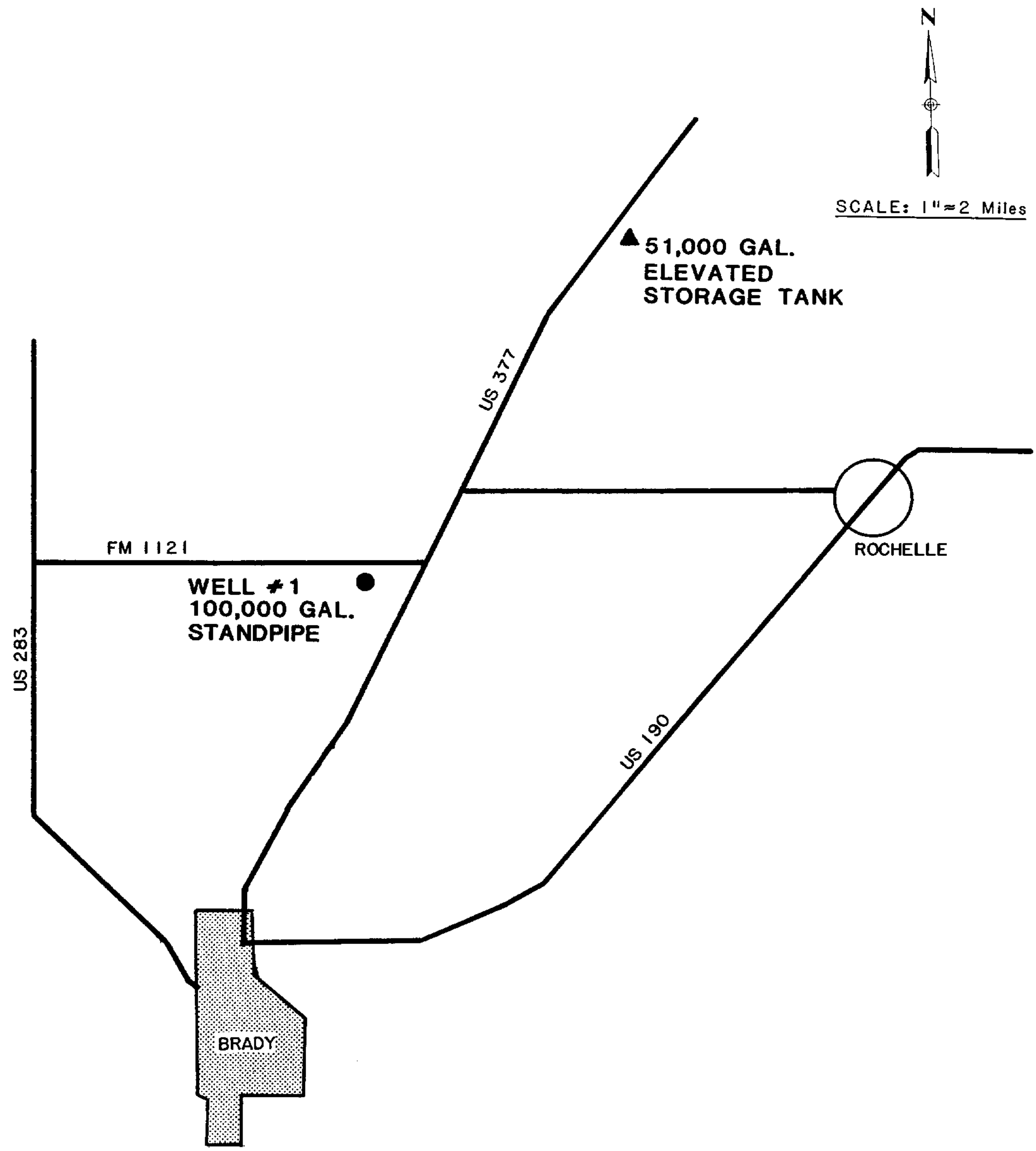
Table 3.8-2
Richland Water Supply Corporation - Brady System
Major Distribution Supply Components

Storage

Type	Capacity (gal.)	Location
Elevated	51,000	East of U.S. 377 - Placid Mountain
Pressure	105,000	Farm Road 1121 & U.S. 377 - standpipe

3.8.3 Water Quality

Table 3.8-3 presents the latest complete chemical analysis of the water supply for the Richland Water Supply Corporation - Brady System. Shaded areas on the table indicate constituent levels that exceed TDH limits. A quick review of the analysis shows that the drinking water for the Corporation is hard water with a high iron concentration level that could cause "red" water problems in the distribution system. A sequestering agent is added to minimize the potential "red" water problems.



NOTE: DISTRIBUTION LINES ARE NOT SHOWN.

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REGIONAL WATER SUPPLY PLAN

RICHLAND WATER SERVICE CORP. - BRADY SYSTEM - MAJOR DISTRIBUTION SYSTEM COMPONENTS

FIGURE 3.8-1

Table 3.8-3
Richland Water Supply Corporation - Brady System Water Analysis

Constituent	Well #1	TDH Limit
Calcium	58	
Chloride	31	300
Fluoride	0.7	2.0
Magnesium	39	
Nitrate (as N)	0.02	10
Sodium	38	
Sulfate	61	300
Total Hardness (as $CaCO_3$)	305	
pH (units)	8.2	7.0
Conductivity (umhos/cm)	840	
Tot. Alka. (as $CaCO_3$)	302	
Bicarbonate	368	
Carbonate	0	
Dissolved Solids	421	1,000
P. Alkalinity (as $CaCO_3$)	0	
Arsenic	< 0.010	0.05
Barium	0.06	1.
Cadmium	< 0.005	0.010
Chromium	< 0.02	0.05
Copper	< 0.02	1.0
Iron	0.34	0.3
Lead	< 0.0200	0.05
Manganese	< 0.02	0.05
Mercury	< 0.0002	0.002
Selenium	< 0.002	0.01
Silver	< 0.01	0.05
Zinc	< 0.02	5.0
Gross Alpha (pCi/l)	86.10	15
Combined Radium (pCi/l)[1]	39.00	5
Radium 226 (pCi/l)	11.00	
Radium 228 (pCi/l)	28.00	
Gross Beta (pCi/l)	28.00	
Radon (pCi/l)[2]	655	

NOTE: Unless otherwise noted, all constituent concentrations are in mg/l.

[1] Combined Radium is the sum of the concentration of Radium-226 and Radium-228.

[2] The results of the Radon analysis are provided in Appendix B.

In addition, the radionuclide levels, in particular gross alpha and combined radium, exceed the present maximum contaminant level given in the Drinking Water Standards of the Texas Department of Health and radon exceeds the MCL in the Radionuclide Rule, proposed on June 17, 1991.

3.9 Rochelle Water Supply Corporation

3.9.1 Water Supply

The water supply for the Rochelle Water Supply Corporation is provided by three wells - one major well that extends into the Hickory aquifer and two (2) older wells that extend into the Marble Falls aquifer. The well capacities and locations are presented in Table 3.9-1.

Table 3.9-1
Rochelle Water Supply Corporation
Water Supply

Well No.	Capacity (gpm)	Location	Well Depth (ft)
1	45	At pump station on U.S. Highway 150	2350
2	20*	One block south of the pump station	285
3	20*	Six blocks south east of the pump station	300

* Not in normal use at this time (standby use only).

The total water supply capacity which the Rochelle Water Supply Corporation has available to meet present and future water supply requirements is 85 gpm (0.122 MGD). As was shown in Table 2.4-1, by 2010, the Corporation is expected to have water requirements totalling 0.035 MGD. Assuming there is no loss over time in well production capacities, the existing water supply system is more than adequate to meet the future water requirements of the Rochelle Water Supply Corporation.

3.9.2 Major Distribution System Components

In addition to the wells, other major system components of the Corporation's water distribution system include ground storage, high service pumping, and pressure maintenance facilities. Figure 3.9-1 indicates the general location of the major system components.

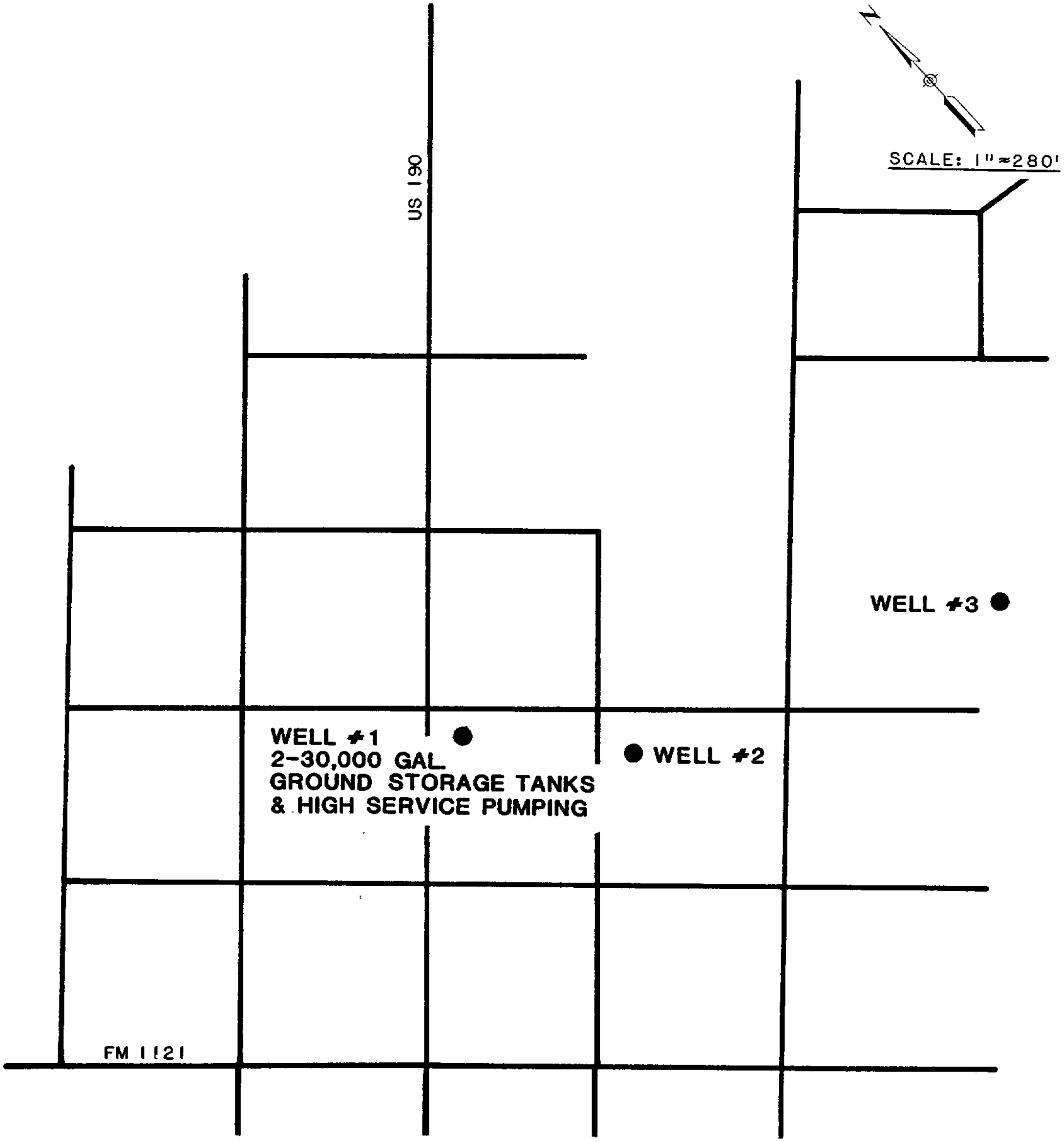
Table 3.9-2 summarizes the Corporation's system components. These components were reviewed for adequacy to meet the minimum system capacity requirements for a public water system in accordance with the Rules and Regulations of the Texas Department of Health. All components meet or exceed these minimum requirements.

Table 3.9-2
Rochelle Water Supply Corporation
Major Distribution Supply Components

Storage		
Type	Capacity (gal.)	Location
Ground	30,000	At pump station on U.S. Highway 190
Ground	30,000	At pump station on U.S. Highway 190
Pressure	2,570	At pump station on U.S. Highway 190
High Service		
No.	Capacity (gpm)	Location
1	180	At pump station on U.S. Highway 190
2	180	At pump station on U.S. Highway 190

3.9.3 Water Quality

Table 3.9-3 presents the latest complete chemical analysis of the water supply wells for the Rochelle Water Supply Corporation. Shaded areas on the table indicate constituent levels that exceed TDH limits. A quick review of the analysis shows that the drinking water for the Corporation is hard water with a high iron concentration level that could cause "red"



NOTE: DISTRIBUTION LINES ARE NOT SHOWN.

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REGIONAL WATER SUPPLY PLAN

ROCHELLE WATER SERVICE CORP. - MAJOR DISTRIBUTION SYSTEM COMPONENTS

FIGURE 3.9-1

Table 3.9-3
Rochelle Water Supply Corporation
Water Analysis

Constituent	Distribution	TDH Limit
Calcium	42	
Chloride	52	300
Fluoride	0.8	2.0
Magnesium	31	
Nitrate (as N)	0.02	10
Sodium	59	
Sulfate	80	300
Total Hardness (as $CaCO_3$)	233	
pH (units)	8.3	7.0
Conductivity (umhos/cm)	840	
Tot. Alka. (as $CaCO_3$)	230	
Bicarbonate	281	
Carbonate	0	
Dissolved Solids	428	1,000
P. Alkalinity (as $CaCO_3$)	0	
Arsenic	< 0.010	0.05
Barium	0.505	1.
Cadmium	< 0.005	0.010
Chromium	< 0.02	0.05
Copper	< 0.02	1.0
Iron	0.35	0.3
Lead	< 0.0200	0.05
Manganese	< 0.02	0.05
Mercury	< 0.0002	0.002
Selenium	< 0.002	0.01
Silver	< 0.01	0.05
Zinc	< 0.02	5.0
Gross Alpha (pCi/l)	22.00	15
Combined Radium (pCi/l) [1]	9.40	5
Radium 226 (pCi/l)	4.40	
Radium 228 (pCi/l)	5.00	
Gross Beta (pCi/l)	40.00	
Radon (pCi/l) [2]	530 [3]	187 [3]

NOTE: Unless otherwise noted, all constituent concentrations are in mg/l.

[1] Combined Radium is the sum of the concentration of Radium-226 and Radium-228.

[2] The results of the Radon analysis are provided in Appendix B.

[3] For the Radon Analysis, the value shown under the Distribution Heading is for Well #1 and the value shown under the TDH Limit Heading is for Well #2.

water problems in the distribution system. A sequestering agent is added to minimize the potential "red" water problem.

In addition, the radionuclide levels, in particular gross alpha and combined radium, exceed the present maximum contaminant level given in the Drinking Water Standards of the Texas Department of Health and radon exceeds the MCL in the Radionuclide Rule, proposed on June 17, 1991.

SECTION 4

4.0 TREATMENT OPTIONS

Naturally occurring radionuclides, in particular radium-226, radium-228, and radon, have been found in the drinking water supply sources of the participants of this study. The treatment options for the removal of these radionuclides should have the following general characteristics:

- Low first costs for the complete treatment system;
- Easy operation and maintenance;
- Low annual operation and maintenance costs;
- Minimal wastes produced by the treatment process; and
- Fully automatic operation due to remote locations of potential treatment sites.

In this section, the treatment options available to remove the naturally occurring radioactive materials (NORM) from the participant's water supply sources will be reviewed. First, the levels of NORM will be presented for each system. Then, the existing as well as anticipated maximum contaminant levels (MCL) of the radionuclides will be given. Based on these levels, the treatment options identified must be able to reduce the concentrations of NORM found in the water to meet the MCLs for the radionuclides. The percentage of reduction in the levels of NORM is called the "removal efficiency" of the treatment technology. For each treatment option identified, a waste stream will be generated, and the disposal options of this waste stream will be given in the last section.

4.1 Present Levels of NORM

Analyses of the water supply sources for each of the participants have been performed to determine the concentrations of NORM in the water. In Table 4.1-1, the maximum levels (or concentrations) of NORM that have been found to date are given.

Table 4.1-1
Maximum Reported Levels of NORM

Participants	Radium-226	Radium-228	Gross Alpha	Radon
Brady	4.7	7.7	40.2	668
Brady Lake	3.2	4.4	13.3	610
Eden	2.6	4.8	27.0	774
Live Oak Hills Sub.	4.9	5.5	17.0	535
Melvin	5.7	2.2	49.0	318
Millersview-Doole WSC	18.0	61.0	93.0	605
North San Saba WSC	8.1	21.3	22.0	748
Richland WSC (Brady)	11.0	28.0	86.1	655
Rochelle WSC	4.4	5.0	22.0	530

Note: All constituent concentrations are in pCi/l.

4.2 MCL For NORM

In July, 1976, the U.S. Environmental Protection Agency (EPA) published interim regulations that limit the concentrations of radionuclides in public water systems. Subsequent to this, the Texas Department of Health also adopted these same published limits into its Drinking Water Standards. At the present time, there is a maximum contaminant level for only two constituents of NORM. Those constituents and the appropriate MCLs are:

Combined Radium-226 & Radium-228	5 pCi/l
Gross Alpha	15 pCi/l

In general, one or more of the levels of NORM given in Table 4.1-1 for each of the participants exceeds the present MCL for that constituent.

On June 17, 1991, EPA published rules proposing new MCLs for radionuclides found in public water systems. The proposed constituents of NORM to be regulated and subsequent MCLs are:

Radium-226	20 pCi/l
Radium-228	20 pCi/l
Uranium	20 mg/l
Radon	300 pCi/l
Gross Alpha (adjusted)	15 pCi/l
Gross Beta	4 millirem/ede/year

Based on the proposed constituents and their MCLs, most of the water supply sources for the participants will meet the individual radium and uranium concentration levels. However, none of the water supply sources will meet the proposed MCL for radon, except for one well (Well No. 6) supplying the City of Eden.

The constituents and MCLs shown above are based on the proposed rule and not on the final rule. Therefore, caution should be exercised because the constituent(s) to be regulated as well as the MCLs could (and probably will) change in the final rule. To provide the participants with the flexibility to meet the changing regulatory environment, treatment options will be identified to provide removal efficiencies that will meet the existing as well as any future regulations of NORM.

4.3 Treatment Alternatives for the Removal of Radium

There are numerous treatment alternatives for the removal of naturally occurring radium from public water supply sources. Table 4.3-1 presents a summary of the most common radium removal treatment alternatives. On the following pages, the treatment process for each of the alternatives will be described in general, as well as the predicted

Table 4.3-1
Radium Removal Treatment Alternatives

Treatment Process	Efficiency (percent)	Full-Scale Plant Operation	Type(s) of Waste
Sodium Cation Exchange	95	Yes	Backwash, brine and rinse wastewater
Lime-Soda Ash Softening	80 - 90	Yes	Lime sludge and filter backwash water
Reverse Osmosis	95	Yes	Continuous brine wastewater stream
Low-Pressure Membrane Filters	> 93	Yes	Continuous brine wastewater stream
Manganese Greensand with Potassium Permanganate Feed	50	Limited	Backwash and permanganate feed wastewater
Manganese Dioxide Impregnated Filter	90	Limited	No backwashing or regeneration
Iron Oxidation-Filtration Process	0 - 25	Yes	Filter backwash water
Radium Selective Removal Resins	98	Yes	Media may require periodic backwashing

removal efficiency, and the type of waste generated. Cost information will also be presented for two different scenarios based on the present TDH Drinking Standards and the proposed Radionuclide Rule.

To provide a general basis for comparing the costs of all treatment and supplemental supply alternatives, all costs will be reduced to a cost per thousand gallons based on 1990 and 2010 water use.

4.3.1 Sodium Cation Exchange

The sodium cation exchange (or zeolite) process depends upon the ability of certain substances to exchange cations with other cations dissolved in water. For instance, when hard water is passed through a sodium cation exchanger, the calcium and magnesium ions in the hard water replace the sodium ions on the exchange medium and thus the water is softened. Since the reaction is reversible, after all of the sodium ions have been removed from the medium and exchanged, the cation exchange medium is regenerated by reintroducing the sodium ions to the medium with a sodium chloride solution. In regenerating the medium, the calcium and magnesium ions on the cation exchange medium are replaced with a new supply of sodium ions from the brine solution used for regeneration. Then, after backwashing the exchange medium to clean it of the calcium and magnesium cations and any excess salt, the regenerated exchanger is ready to begin the exchange process again and soften a new supply of hard water.

Radium is an alkaline earth metal or a cation with a valence state like that of calcium and magnesium. Therefore, a sodium cation exchange process will remove radium in the same way as it does calcium and magnesium. However, radium is the preferred ion over calcium or magnesium for removal by the synthetic exchange resin; therefore, a hardness (calcium or magnesium) breakthrough, which means that the resin can no longer remove these ions, will indicate that the exchange medium needs regeneration before radium will break through.

The removal efficiency of the cation exchanger varies somewhat during the normal operation cycle between regenerations. In general, the sodium cation exchange process will remove 95 percent of the radium found in the raw water.

The sodium cation exchange process has typically been used for turning "hard" water into "soft" water and there are numerous operating facilities, both large and small, located throughout Texas as well as the United States. However, only recently has the sodium cation exchange process been utilized for the removal of radium from drinking water.

The typical waste generated by the sodium cation exchange process is a liquid stream of brine waste, backwash (or backflush), and rinse of the medium. Typically, this waste stream is approximately five to eight percent of the raw water flow. However, technology advances in the regeneration of the medium have decreased the waste stream to approximately two percent of the raw water flow.

One of the problems with the sodium cation exchange process is a high level of sodium in the treated water. To minimize and possibly eliminate this problem, studies indicate that potassium chloride can be substituted for sodium chloride without any problem. However, potassium chloride is approximately 50 to 70 percent more expensive than sodium chloride in the quantities that would be typically required for most of the treatment facilities for this area.

Table 4.3.1-1 presents the costs of constructing, operating, and maintaining a sodium cation exchange process for the removal of radium for each of the study participants to meet the existing TDH Drinking Water Standards. The general design parameters for the design of the facilities are as follows:

Treatment Capacity -	Two times the 2010 high projected annual water requirements (from Table 2.4-1) so that capacity is available to meet the potential peak day demands of the system.
Radium Removal Efficiency -	95%.

Finish Water Radionuclide
Concentration - 4 pCi/l Combined Radium-226 & Radium-228 or 13 pCi/l Gross Alpha to meet the existing TDH Drinking Water Standards.

Table 4.3.1-1
Construction, Operation and Maintenance Cost
For Sodium Cation Exchange Process
to Produce a Finish Water that
Complies with the Existing TDH Standards

Participant	Total Project Cost	Initial O&M Cost ($/1,000 gallons)	Number of Connections	Monthly Debt Service Cost/ Connection*
Brady	$2,700,000	$0.098	2,915	$7.23
Eden	213,000	0.083	613	2.71
Live Oak Hills Sub.	45,300	0.049	17	20.80
Melvin	73,750	0.055	140	4.11
Millersview-Doole WSC	381,250	0.088	1,396	2.13
North San Saba WSC	128,250	0.075	241	4.16
Richland WSC	132,000	0.070	218	4.73
Rochelle WSC	73,750	0.056	97	5.94

*Assuming 8% interest for 25 years.

The cost of the sodium cation exchange treatment option for removing the radium to levels meeting the TDH Drinking Water Standards, assuming the total project cost is financed for eight percent interest for 25 years, without capitalizing interest, varies for each participant, but the average cost per thousand gallons for all the participants is $0.57 based on 1991 water use. With O&M, the total 1990 average cost would be $0.65 per thousand gallons. By 2010, the total average cost per thousand gallons would reduce to $0.55.

Table 4.3.1-2 presents the costs of constructing, operating, and maintaining a sodium cation exchange process for the removal of radium for Millersview-Doole WSC, North San

Saba WSC, and Richland WSC to meet the proposed Radionuclide Rule. The water supplies for the other participants will meet the proposed MCLs for Radium-226 and Radium-228, so no treatment is required. The general design parameters for the design of the faclities are as follows:

Finish Water Capacity -	Two times the 2010 high projected annual water requirements (from Table 2.4-1) so that capacity is available to meet the potential peak day demands of the system through treatment and blending.
Radium Removal Efficiency -	95%.
Finish Water Radionuclide Concentration -	15 pCi/l Radium-226 or Radium-228.

Table 4.3.1-2
Construction, Operation and Maintenance Cost For Sodium Cation Exchange Process to Produce a Finish Water that Complies with the Proposed Radionuclide Rule

Participant	Total Project Cost	Initial O&M Cost ($/1,000 gallons)	Number of Connections	Monthly Debt Service Cost/ Connection*
Millersview-Doole WSC	305,000	0.086	1,396	1.71
North San Saba WSC	73,750	0.056	241	2.39
Richland WSC	88,500	0.059	218	3.16

*Assuming 8% interest for 25 years.

The total project cost includes the following items:

A. Equipment, piping, and valving;
B. Building for equipment;
C. Electrical and controls;
D. Land for and construction of evaporation ponds for disposal of the concentrate (no cost for disposal of the residue has been included);
E. Engineering; and
F. Legal and administration.

The cost of the sodium cation exchange treatment option for removing the radium levels meeting the Radionuclide Rule, assuming the total project cost is financed for eight percent interest for 25 years, without capitalizing interest, varies for the three participants, but the average cost per thousand gallons for the three participants is $0.24 based on 1991 water use. With O&M, the total 1990 average cost would be $0.31 per thousand gallons. By 2010, the total average cost per thousand gallons would reduce to $0.28.

4.3.2 Lime - Soda Ash Softening

The lime - soda ash process uses lime and soda ash to change soluble calcium, magnesium, and radium compounds into nearly insoluble compounds that are removed by flocculation, settling, and filtration in a typical water treatment plant. Lime is added to raise the pH of the raw water to a level where the maximum precipitation of magnesium and radium occurs. After the precipitation of magnesium and radium, the water is recarbonated with carbon dioxide to lower the pH precipitation of calcium. Soda ash is normally added to precipitate the remaining noncarbonate hardness. The water is then filtered to remove any remaining solids before it is pumped into the water distribution system.

The removal efficiency of the lime - soda ash process varies somewhat, depending on the maximum pH of the groundwater. In general, however, the lime - soda ash softening process removes 80 to 90 percent of the radium found in raw water.

The lime - soda ash softening process has typically been used for turning "hard" water into "soft" water, and there are numerous operating facilities, both large and small, located throughout Texas as well as the United States. However, only recently has lime - soda ash softening been used for the removal of radium from drinking water.

Typically, the waste generated in the lime - soda ash softening process is a semi-liquid stream containing sludge from the settling basins and filter backwash. Usually, the volume of this waste stream is approximately eight to 10 percent of the raw water flow.

Although the lime - soda ash softening process is a very viable method of removing radium, the costs of constructing, operating and maintaining a lime - soda ash softening treatment plant are not very economical at the capacities required for the participants of this study. For example, the cost of constructing a lime - soda ash softening treatment plant at the City of Brady's Well No. 5 (capacity 1,000 gpm) is over $2,500,000. Brady has three well sites of a similar capacity, so the total project cost for the City would be $7,500,000. The operation and maintenance cost for a lime-soda ash softening plant of this size would be approximately $1.10 per thousand gallons. If the project cost is financed at eight percent for 25 years, without capitalized interest, the annual debt service would be $702,600, which equates to $1.13 per 1,000 gallons based on Brady's 1990 water use. With O&M, the total 1990 cost would be $2.43 per thousand. By 2010, or when water requirements reach 1.934 mgd, the total cost per thousand gallons would reduce to $2.10.

4.3.3 Reverse Osmosis

In the reverse osmosis (RO) demineralizing process, water is pressurized and piped into a reverse osmosis unit where relatively pure water diffuses through a semi-permeable membrane and becomes the product water, leaving a concentrated reject water. The normal operating pressure of a typical reverse osmosis unit is approximately 240 to 440 psi.

The removal efficiency of the reverse osmosis process varies somewhat, but, in general, RO will remove 95 percent of the radium found in the raw water.

The reverse osmosis process has typically been used for the removal of dissolved solids and other contaminants found in water, and there are numerous operating facilities, both large and small, located throughout Texas as well as the United States. However, only recently has the reverse osmosis process been utilized for the removal of radium from drinking water.

The typical waste generated by the reverse osmosis process is a concentrated liquid stream of the reject from the RO units and the wastes from the membrane cleaning system. Typically, this waste stream is approximately 20 to 30 percent of the raw water flow.

The costs of constructing, operating, and maintaining a reverse osmosis treatment plant, like a lime - soda ash treatment plant, are not very economical at the capacities required for the participants of this study. For example, the cost of constructing a reverse-osmosis treatment plant for North San Saba WSC (capacity 130 gpm) is over $500,000 with an operation and maintenance cost of $1.25 per thousand gallons. In addition, the waste stream would be 20 to 30 percent of the raw water flow, which is significantly higher than other treatment alternatives. The greater volume of waste, which must be disposed of, greatly increases the total cost of this alternative.

Based on a total project cost of $500,000 for North San Saba WSC and financing at eight percent interest for 25 years, without capitalizing interest, the annual debt service would be $46,840, which equates to $1.58 per 1,000 gallons of 1990 water use. With O&M, the total 1990 cost would be $2.84 per thousand. By 2010, or when water requirements reach 0.116 mgd, the total cost per thousand gallons would reduce to $2.36.

4.3.4 Low-pressure Membrane Filters

The low-pressure membrane filter process is similar to the reverse osmosis process, except the membranes are designed to selectively remove undesirable constituents at a lower pressure. The normal operating pressure of a low-pressure membrane filter system is approximately 70 psi, which is significantly less than the minimum of 240 psi required by an RO system.

The radium removal efficiency of the low-pressure membrane filter process is generally at least 93 percent, which is slightly less than that of the reverse osmosis process.

The low-pressure membrane filter process has typically been used for the selective removal of divalent cations, such as calcium and magnesium, as well as other contaminants found in water. There are numerous large and small facilities using this process throughout the United States. Only recently, however, has the low-pressure membrane filter process been used to remove radium drinking water.

The typical waste generated by the low-pressure membrane filter process, like that from RO, is a liquid stream containing the concentrate (or reject) from the filter units and the wastes from the membrane cleaning system. Typically, this waste stream is approximately 20 to 30 percent of the raw water flow.

Like RO, the low-pressure membrane filter process is very costly at the capacities required by the study participants. And, like reverse osmosis, the waste stream volume is very high compared to other processes. Initially, this treatment alternative is not a viable method for the participants; however, membrane technology is rapidly changing and, over time, this method may merit further review.

4.3.5 Manganese Greensand

Manganese greensand is typically used to remove excess soluble iron and manganese found in groundwater. Manganese greensand is manufactured by treating glauconite with manganous sulfate and potassium permanganate to provide an active supply of iron and manganese oxides on sand grains. Radium is absorbed by the hydrous oxides of iron and manganese typically found in a manganese greensand filter bed. When the oxidizing capacity of the bed is exhausted, it is regenerated with permanganate, returned to service, and reused. In lieu of batch regeneration, continuous regeneration can be performed by feeding a solution of potassium permanganate continuously into the raw water ahead of the filter to reduce the amount of soluble iron and manganese applied to the filter.

The removal efficiency of the manganese greensand filter process is relatively poor compared to other processes. In general, manganese greensand will remove 45 to 55 percent of the radium in raw water.

There are very few facilities in the United States utilizing manganese greensand for the removal of iron, manganese, or radium. Only recently has the manganese greensand filter process been utilized for the removal of radium from drinking water.

The typical waste generated by the manganese greensand filter process is the liquid stream used to backwash the filter bed. Typically, this waste stream is approximately 20 to 30 percent of the raw water flow.

Manganese greensand is a relatively new process as evidenced by the limited number of installations in the United States. In addition, the waste stream volume is very high and radium removal efficiency is low compared to other treatment alternatives. Initially, this treatment alternative does not appear to be a viable method for the removal of radium in the participants' water supplies.

4.3.6 Manganese Dioxide Impregnated Filter

Manganese dioxide is loaded or coated on a prewoven filter element to form a manganese dioxide impregnated filter. Typically, 10 to 25 percent (by weight) of manganese dioxide is coated on the filter media to provide an adsorption area for the removal of radium as well as the other divalent alkaline earth metals, such as calcium and manganese. In this process, the manganese dioxide filter cannot be regenerated because the radium is adsorbed permanently onto the filter media.

The removal efficiency of the manganese dioxide impregnated filter process is relatively good, generally removing about 90 percent of the radium found in the raw water.

The manganese dioxide impregnated filter process is a relatively new process that has typically been used only in laboratory tests to determine its ability to remove radium. There are no facilities in the United States utilizing manganese dioxide impregnated filters for the removal of iron and manganese or radium.

The typical waste generated by the manganese dioxide impregnated filter process is a solid consisting of the filter itself, since radium is permanently absorbed to the media. As long as the level of radium is low enough, the filter(s) can be disposed of in the nearest sanitary landfill.

Manganese dioxide impregnated filters are a relatively new process, as evidenced by the lack of installations in the United States. In addition, the waste stream volume is very high compared to other treatment alternatives. Since it is experimental, this alternative is currently not a viable treatment option for the participants; however, after additional testing, including use in full scale plants and implementation of mass production of the filters, this process may prove to be cost-effective. If the filters can be produced at a reasonable cost, their use could become popular since waste disposal would not require dewatering or other means of drying the waste prior to disposal.

4.3.7 Iron Oxidation-Filtration Process

Potassium permanganate, chlorine, chlorine dioxide, or other oxidants can be used to oxidize soluble iron and manganese in water. After oxidation, the iron and manganese forms a hydroxide that easily precipitates in a settling basin or can be removed by filtration. Radium can also be removed from water by this process since the precipitate sometimes adsorbs radium. Then, the radium is subsequently removed in the settling basin or filters, along with the precipitate.

The removal efficiency of the iron oxidation-filtration process is relatively poor since the process usually removes no more than 25 percent of the radium found in the raw water. In some cases, very little or no radium may be removed by the precipitate because the ionic charge of the precipitate is the same as radium. Therefore, radium will not be absorbed to and removed by the precipitate.

The iron oxidation-filtration process has typically been used for the removal of iron, and it is sometimes used to remove manganese. There are numerous large and small facilities in Texas and the United States utilizing this process for iron removal. Only recently has the iron oxidation-filtration process been utilized for the removal of radium from drinking water.

The typical waste generated by the iron oxidation-filtration process is a semi-liquid consisting of the iron oxide particles with the radium permanently adsorbed into them. As long as the level of radium in the particles is low enough, it can be disposed of in the nearest sanitary landfill.

Iron oxidation-filtration process is not a viable treatment alternative for the study participants due to its very low radium removal efficiency. This process does not merit further consideration.

4.3.8 Radium Selective Complexer

The radium selective complexer is comprised of resins that will selectively remove only radium from the water passing through it. The radium is permanently absorbed onto the complexer resin and cannot be removed.

The removal efficiency of the radium selective complexer is exceptionally good. In general, the complexer will remove 98 percent of the radium found in the raw water.

The radium selective complexer is a new treatment process for the removal of radium. There are only a few operating facilities in the United States.

The typical waste generated by the radium selective complexer is a solid consisting of the complexer resin with the radium permanently adsorbed into it. As long as the level of radium is low enough, the resin can be removed and disposed of in the nearest sanitary landfill.

At first glance, the radium selective complexer would appear to be a viable method for the removal of radium. However, the manufacturer of the complexer has stated that it is an experimental process and not available for normal use. Therefore, this treatment alternative is not a viable method. If the manufacturer makes the complexer available for normal use, this option may be worth consideration in the future.

4.3.9 Point of Use Treatment

The typical point of use (POU) (i.e., at the location of the connection) treatment would be sodium cation exchange and reverse osmosis processes. The system would be installed under the sink, if possible, or in a small out-building. The system would provide water on an as-needed basis.

The removal efficiency of the point of use treatment processes would be the same as that given above for sodium cation exchange or reverse osmosis. In general, POU treatment will remove 90 to 99 percent of the radium found in drinking water.

The typical waste generated by the POU treatment systems is a liquid consisting of the brine from the RO unit or the ion exchange unit, which can be disposed of at an on-site septic system or a publicly owned treatment works.

The Texas Department of Health has issued a policy statement that states that POU treatment is not considered a viable alternative to centralized treatemnt. Another major issue to be addressed with POU treatment is who is responsible for the maintenance and operation of the POU devices. The water purveyor is responsible for supplying potable water to the customers that meets all rules and regulations of the Texas Department of Health. Therefore, the water purveyor is technically responsible for the operation and maintenance of the POU treatment units. Depending on the locaton of and access to the POU treatment units, this may or may not be a problem.

The typical cost of a POU treatment varies widely. For example, the typical sodium cation exchange system for treating all the water entering the house costs $1,800 to $2,500 without a building. The normal operation and maintenance costs for the sodium cation exchange system is $5 to $8 per month. The typical life of this POU treatment devices is 15-20 years.

For cost comparison purposes, we will assume that a POU treatment system is the selected treatment option to be installed by the participants in each home in lieu of centralized treatment. The cost of the project will be financed at eight percent interest for 15 years (the minimum life of the units), without capitalized interest, the average cost per thousand gallons for all the participants is $2.43 based on 1990 water use. With O&M, the total 1990 average cost would be $2.83 per thousand gallons. By 2010, the total average cost per thousand gallons would reduce to $2.61.

4.4 Treatment Alternatives For the Removal Of Radon

Radon is one of the naturally occurring radioactive materials that will be regulated by the Radionuclide Rule to be promulgated by EPA in the near future. Radon is the only gas of the radionuclides and is a member of the noble gas group, which includes helium, argon, and other chemically inert monatomic gases. Therefore, the treatment alternatives for the removal of radon include stripping the gaseous radon from the water, providing detention time in a reservoir to allow the decay and dissipation of the radon, and absorption onto another media such as granular activated carbon.

4.4.1 Stripping

Because radon is a gas, it can easily be removed or stripped from the water by aeration. The methods for stripping radon can be simple, such as providing splash plates in ground storage tanks to aerate the water, or complex, such as trickling water over a media in a packed tower to form water droplets which are then aerated by air which is forced through the media.

The removal efficiency of each method varies according to the amount of energy expended in aeration. The simple stripping method of splash plates in a ground storage tank, removes approximately 50 percent of the radon in the raw water. The more complex packed tower aeration method, removes approximately 98 percent of the radon in the raw water.

It was not within the scope of this study to perform detailed review of each of the participants' ground storage tank installations to determine an estimate of the cost to modify each reservoir to air strip radon. However, the only modifications required would be to change the tank inlet from the bottom to the top and add splash plates in the tank. It is estimated that the costs of these modifications would be in range of $5,000 to $10,000, depending on the size of the tank and piping.

4.4.2 Detention

Due to its short half-life and the fact that it is a gas, radon can be removed from water by detaining the water in a storage tank until the gas has dissipated or the radon has decayed. Research has shown that if water containing high levels of radon is allowed to stand in a tank for five to six days, radon will be reduced approximately 70 to 80 percent through decay and dissipation.

By using supplemental aeration along with extending detention time in the reservoir, removal efficiencies can be increased significantly. Supplemental aeration can be added by including a small blower and air diffuser in the ground storage tank or an external blower and a venturi that injects air into the water in the tank. Depending on the amount of supplemental aeration added, the removal efficiency can be increased to the point that 90 to 99 percent of the radon in the raw water is removed.

Based on discussions with the study participants, it appears that there is excess storage tank capacity for at least most of the year, particularly for the smaller systems. However, during times of heavy usage, detention times on the order of five to six days are not feasible. Therefore, to provide year round reduction of radon, all participants would have to construct additional storage facilities to provide the required detention time. Typically, the cost of constructing a new above ground steel reservoir ranges from $60,000 for a 10,000 gallon tank to $300,000 for a 500,000 gallon tank. For the smaller systems, extending detention time may be a viable alternative to reduce radon, but for the larger systems, such as Brady, it is not.

4.4.3 Adsorption by Carbon

Granular activated carbon (GAC) is a very strong adsorbent media, and it is the best available treatment technology for the removal of most volatile contaminants. Since radon is a very volatile gas, GAC will easily remove radon from water.

To remove radon using GAC, the treatment system must include a tower or filter bed filled with GAC over which water is distributed and allowed to trickle down. As water trickles through the GAC, radon is adsorbed (or attached to) the GAC. Then, due to its short half-life, the adsorbed radon is removed from the media over time by decay, or it is dissipated into the air. GAC removes 50 to 80 percent of the radon in raw water.

One of the main problems with GAC is its high adsorption capacity. In addition to radon, GAC will remove other constituents in the water, including iron, manganese, particulates, and microorganisms. The accumulation of these constituents will necessitate more frequent periodic backwashing or pretreatment to prevent significant increase in headloss across the filter which reduces the capacity of the filter. In addition, due to accumulation of radioactive materials, radioactive emissions from the GAC filter could be higher than normal background levels, requiring the filter to be shielded to protect personnel in the area.

4.5 Waste Disposal Alternatives

As discussed in previous sections describing treatment alternatives, the wastes to be disposed of are either liquid, such as the concentrate or brine, or solid, such as a sludge or filter media. The disposal alternatives for each are very different.

To complicate matters, the regulations covering the disposal of such wastes are changing. Presently, the only regulations covering the disposal of waste from naturally occuring radioactive materials (NORM) are federal regulations covering the disposal of low level radioactive wastes. Under these federal regulations, the only alternative available for the disposal of such wastes is deposition in a federally regulated low level radioactive waste disposal site. Currently, there are only four low level radioactive disposal sites in the United States at which NORM wastes can be disposed, and the closest site to this area is located in Nevada. However, NORM wastes typically contain levels of radioactivity that are significantly below the levels of radiation of the wastes normally deposited in federally regulated disposal sites.

Within the last few months, the Texas Department of Health has begun the process of developing rules and regulations for the disposal of NORM wastes. It is anticipated that it will take six months to one year to complete the rule-making process so state rules and regulations presently cover the disposal of NORM wastes. It is expected that the state rules will allow NORM wastes to be discharged into a wastewater treatment plant or deposited in a sanitary landfill. The disposal alternatives discussed in the following sections are based on the proposed state level rules and regulations covering the disposal of NORM wastes as we understand them at this time. The study participants should monitor the development of these rules and regulations and select disposal alternatives after the final rules have been implemented.

4.5.1 Disposal at POTW

The best alternative for the disposal of NORM wastes mixed with water is to discharge it in the closest available publicly owned treatment works (POTW) for the treatment and disposal of sewage. Of all the participants of this study, only the City of Brady owns and operates a POTW. The wastes can be discharged to the sanitary sewer or otherwise transported to the sewage treatment plant for treatment and disposal with the normal sewage wastes and sludges. Disposing of NORM wastes in this manner would eliminate or minimize the need for additional treatment.

The controlling factor in the disposal of NORM wastes in a POTW is the concentration of radioactivity in the wastewater sludge. However, the volume of NORM wastes and the amount of radioactivity in the water treatment sludge should be minor compared to the volume of sanitary sewage being treated at the POTW, and the resultant levels of radioactivity in the wastewater sludge should be below the proposed maximum level(s) in the new state regulations in order for the solids to be deposited in a sanitary landfill.

4.5.2 Disposal at a Landfill

After the Texas Department of Health rules are implemented, another alternative for the disposal of NORM wastes will be the nearest sanitary landfill. The wastes would typically be required to be in a solid (or near solid) state before they can be disposed of in the landfill. For instance, if the chosen treatment alternative produces a liquid or semi-liquid waste stream, the wastes will have to be consolidated into a transportable solid before disposal.

The consolidation process could involve treatment ranging from removing the radium from the liquid waste stream by another treatment process such as the radium selective complexer that produces a disposable solid. Removing the water from the sludges produced in lime-soda ash softening by devices such as centrifuges results in a semi-solid sludge cake. Also, evaporating the water in evaporation ponds will produce a substance that can be disposed of in a sanitary landfill.

Regardless of the water treatment process, if the NORM waste stream consists primarily of water, further treatment will be required to produce a substance that can be disposed of in a sanitary landfill. And, as discussed for disposal via wastewater sludge, the concentration of radioactivity in the substance deposited in the sanitary landfill must meet the proposed maximum level required by the new state regulations.

4.5.3 Radon Disposal

As discussed earlier, aeration (or stripping) is the best treatment alternative for the removal of radon. After stripping, the stripped radon is typically disposed of by dispersing it into the air. In this process, radon is mixed with air so that the concentration of radon in the air does not exceed the normal background level.

If granular activated carbon (GAC) is used to remove radon, the spent GAC must be reactivated. The radon removed in this process is also disposed of by dispersing it into the air. If necessary, the spent GAC can be disposed of in a sanitary landfill.

4.6 Summary

For the participants of this study, the potential treatment options for removing the radium and radon have been identified. Table 4.6-1 lists the costs for the potential treatment options for the removal of radium.

Table 4.6-1
Project Cost of Potential Treatment Options for the Removal of Radium
(Cost per 1,000 Gallons Based on 2010 Water Use)

Sodium Cation Exchange	$0.55
Lime-Soda Ash Softening	2.10
Reverse Osmosis	2.36
Point of Use	2.61

SECTION 5

5.0 ALTERNATIVE WATER SUPPLY SOURCES

5.1 Introduction

As discussed in the previous section, there are a number of water treatment alternatives available to treat the study participants' water supplies to remove radionuclides. In addition to treating their water, the participants could develop alternative water supply sources to either replace or supplement their existing supply sources. In this chapter, various alternative supply sources will be identified.

For this study, it has been assumed that the water distribution systems of the participants will be adequate to receive, store, and distribute water from an alternative water supply source. However, in reality, distribution systems built around wells scattered throughout the system typically require modification to efficiently serve customers from a single source. In addition, it has been assumed that study participants or entities in the surrounding area with an under-utilized supply source would be willing to supply water to any other entity that participates in a regional water supply system.

The costs outlined for each regional water supply system include the capital cost of a water treatment plant, if required, to treat the water to meet the TDH Drinking Water Standards and the transmission system necessary to transport water from the plant to each participating entity. In addition, the cost estimates include the projected annual operation and maintenance cost.

5.2 Surface Water Sources

There are five potential sources of surface water in the area that might be used to either replace or supplement the existing water supplies of the study participants. These potential sources are Lake Brady, O.H. Ivie Reservoir, Lake Brownwood, the City of San Angelo, and the potential San Saba Reservoir. Each of these potential surface water sources is described in greater detail below.

5.2.1 Lake Brady

Lake Brady, completed in 1963, is owned by the City of Brady, which has a permit from the Texas Water Commission to use 3,000 acre-feet of water annually from the lake for municipal purposes and 500 acre-feet annually for industrial purposes. The dependable yield of the lake is estimated to be 3,100 acre-feet or 2.76 million gallons of water per day (mgd)[1]. However, this estimate of the dependable yield of Lake Brady has been questioned by the City of Brady and others in the area. Therefore, a new dependable yield study of Lake Brady should be performed before this alternative supply source is utilized. But, for the purposes of this study, the dependable yield of Lake Brady is assumed to be 3,100 acre feet per year.

5.2.1.1 Lake Brady Regional System I

Lake Brady could supply the City of Brady and its neighboring participants, including Live Oak Hills Subdivision, Richland Water Supply Corporation, and Rochelle Water Supply Corporation. As shown in Table 5.2.1-1, the 2010 projected annual water requirement (from Table 2.4-1) for these four participants is 1.971 mgd, which is less than Lake Brady's

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[1] Unpublished planning information, Texas Water Development Board, Austin, Texas, 1977.

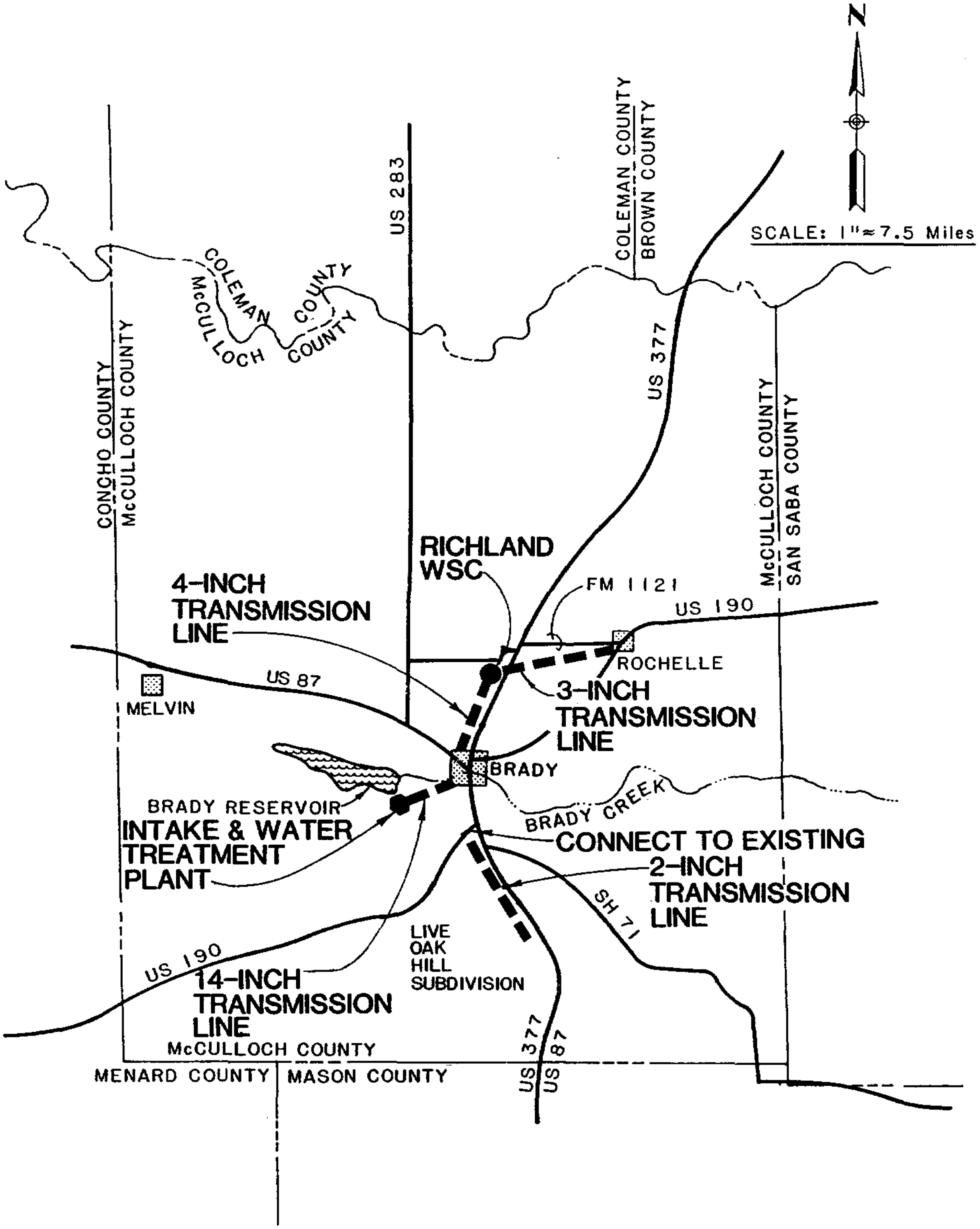
Table 5.2.1-1 Current and Projected Water Use for Lake Brady Regional System I		
Participant	Water Use[1] in 1990 (mgd)	Projected Water Requirements in 2010[1] (mgd)
Brady	1.641	1.854
Live Oak Hills	0.008	0.011
Richland WSC	0.064	0.071
Rochelle WSC	0.032	0.035
TOTAL	1.745	1.971

[1] From Table 2.4-1

estimated yield.

Lake Brady Regional System I (Figure 5.2.1-1) would be comprised of an intake and a 4 mgd water treatment plant to be located at Lake Brady, and a transmission system to transport the treated water to the participating entities. The transmission system would include a 14-inch transmission line from the water treatment plant to the City of Brady's elevated storage tank; a 2-inch line extending from Brady's water distribution system to Live Oak Hills Subdivision; a booster pump at Brady's Pump Station No. 5 and a 4-inch line to Richland WSC's standpipe located along FM 1121; and a 3-inch line from Richland's water distribution system to Rochelle WSC.

The estimated total project cost for Lake Brady Regional System I is $8,200,000, and the estimated operation and maintenance cost is $0.63 per one thousand gallons of treated water. If the project cost is financed at eight percent interest for 25 years, without



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REGIONAL WATER SUPPLY PLAN

LAKE BRADY REGIONAL SYSTEM I

FIGURE 5.2.1-1

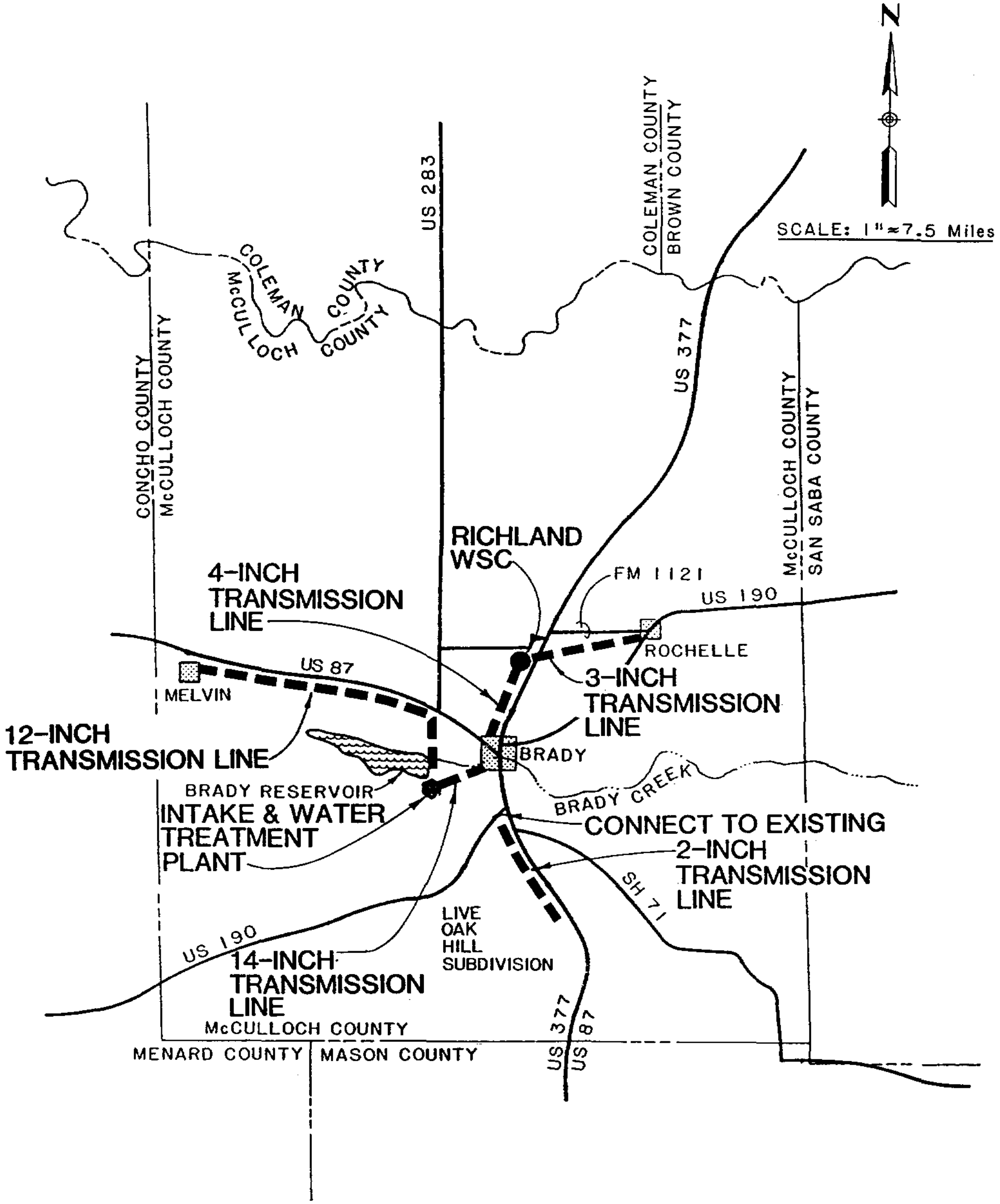
capitalized interest, the annual debt service would be $768,000, which equates to $1.21 per 1,000 gallons of 1990 water use. With O&M, the total 1990 cost would be $1.84 per thousand. By 2010, or when water requirements increase to 1.971 mgd, the total cost per thousand gallons would reduce to $1.65.

5.2.1.2 Lake Brady Regional System II

Lake Brady could also be an alternative water supply source for the City of Melvin and Millersview-Doole WSC. By adding these two entities with the entities in System I, the total projected annual water requirements (from Table 2.4-1) would be 2.535 mgd (see Table 5.2.1-2). The 2.76 mgd yield of Lake Brady is adequate to meet the projected water requirements of these six entities.

The Lake Brady Regional System II (shown in Figure 5.2.1-2) would be comprised of an intake and a 5 mgd water treatment plant at Lake Brady and a transmission system to transport the treated water to each of the participating entities. The transmission system would include the transmission lines from System I as well as 13 miles of a 12-inch transmission line from the water treatment plant to Millersview-Doole's Melvin Pump Station and one mile of 4-inch line extending to the City of Melvin.

The estimated total project cost for Lake Brady Regional System II is $12,700,000, and the estimated operation and maintenance cost is $0.60 per one thousand gallons of treated water. If the project cost is financed at eight percent interest for 25 years, without capitalized interest, the annual debt service would be $1,190,000, which equates to $1.45 per 1,000 gallons of 1990 water use. With O&M, the total 1990 cost would be $2.05 per thousand. By 2010, or when water requirements reach 2.535 mgd, the totals cost per



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REGIONAL WATER SUPPLY PLAN

LAKE BRADY REGIONAL SYSTEM II

FIGURE 5.2.1-2

Table 5.2.1-2
Current and Projected Water Use for Lake Brady Regional System II

Participant	Water Use[1] in 1990 (mgd)	Projected Water Requirements in 2010[1] (mgd)
Brady	1.641	1.854
Live Oak Hills	0.008	0.011
Richland WSC	0.064	0.071
Rochelle WSC	0.032	0.035
Millersview-Doole WSC	0.472	0.533
Melvin	0.028	0.031
TOTAL	2.245	2.535

[1] From Table 2.4-1

thousand gallons would reduce to $1.89.

5.2.2 O.H. Ivie

O.H. Ivie Reservoir, owned by the Colorado River Municipal Water District (CRMWD), was completed in 1990. The reservoir is permitted for municipal and industrial water use by customers of CRMWD. This potential source of water is included because of its proximity to the area. However, water for the participants of this study would have to be obtained from CRMWD, and at the present time, arrangements have not been made to discuss obtaining water from CRMWD for the study area. Based on discussions with CRMWD, the price for raw water from the O.H. Ivie Resevoir would be $120/acre-foot (or $0.37/1,000 gallons) on a take or pay basis.

5.2.2.1 O.H. Ivie System I

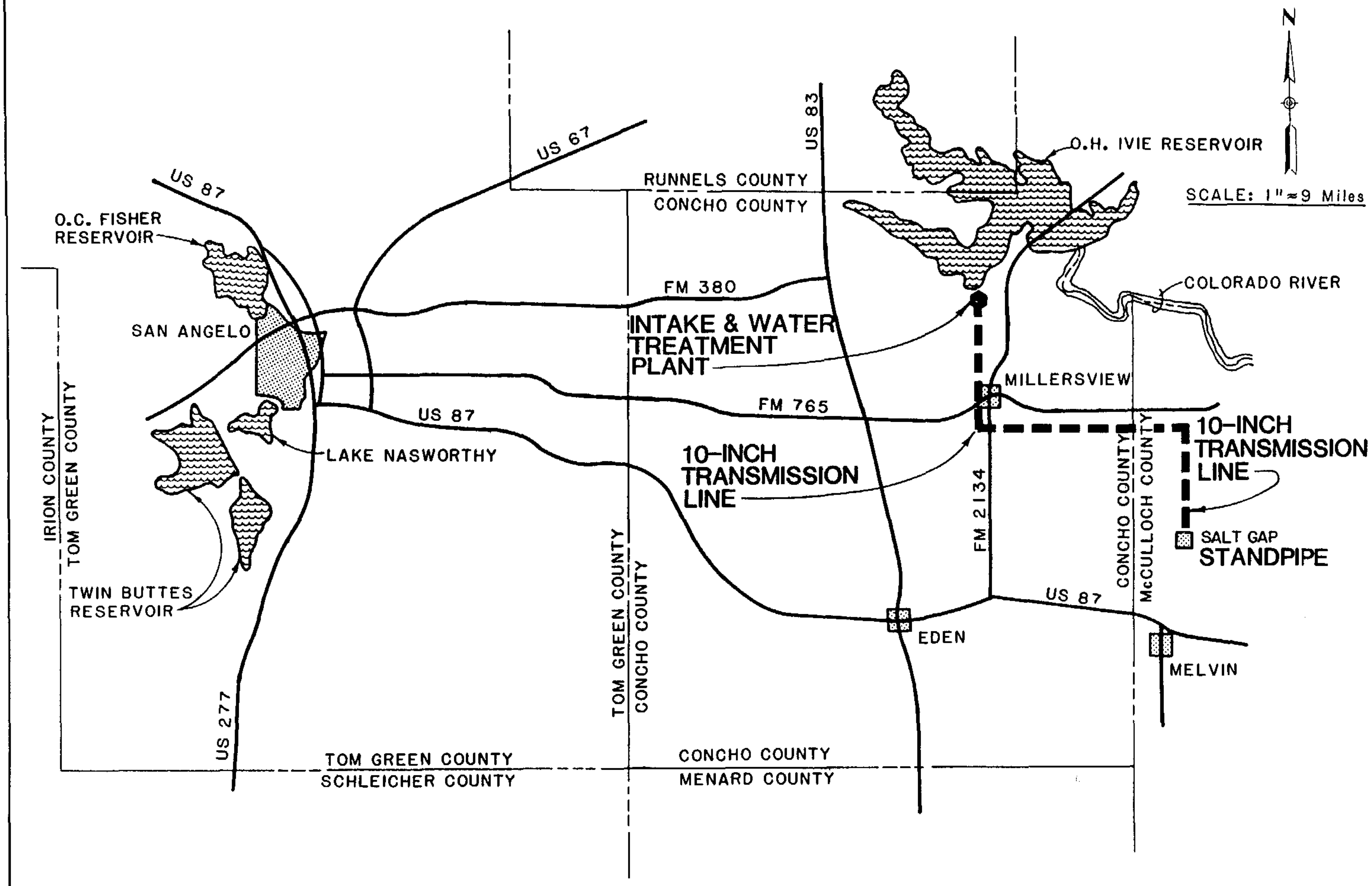
Ivie Reservoir could be an alternative supply source for Millersview-Doole WSC. A 1.0 mgd water treatment plant would be constructed at O.H. Ivie Reservoir, and treated water would be pumped via a 10-inch transmission line to Millersview-Doole WSC's Salt Gap storage tank and then distributed to its customers. The projected water requirements for Millersview-Doole WSC are 0.533 mgd in 2010, and the system's water use in 1990 was estimated to be 0.472 mgd.

The estimated total project cost for Ivie System I (shown in Figure 5.2.2-1) is $3,500,000, and the estimated operation and maintenance cost is $1.72 per thousand gallons of treated water including the cost of raw water. If the project cost is financed at eight percent interest for 25 years, without capitalized interest, the annual debt service would be $328,000, which equates to $1.90 per 1,000 gallons of 1990 water use. With O&M and raw water, the total 1990 cost per thousand gallons would be $3.62 per thousand gallons. In 2010, or at 0.533 mgd water use, the total cost would be $3.41 per 1,000 gallons.

5.2.2.2 O.H. Ivie Regional System II

Ivie Regional System II would serve Millersview-Doole WSC, and the cities of Melvin and Eden. The projected annual water requirement of this potential regional water supply system is 0.892 mgd (see Table 5.2.2-2).

The system (shown in Figure 5.2.2-2) would be comprised of an intake and a 1.8 mgd water treatment plant located at Ivie Reservoir and a transmission system to transport treated water to the participating entities. The transmission system would include a 10-inch transmission to the Salt Gap standpipe to serve Millerview-Doole WSC with an



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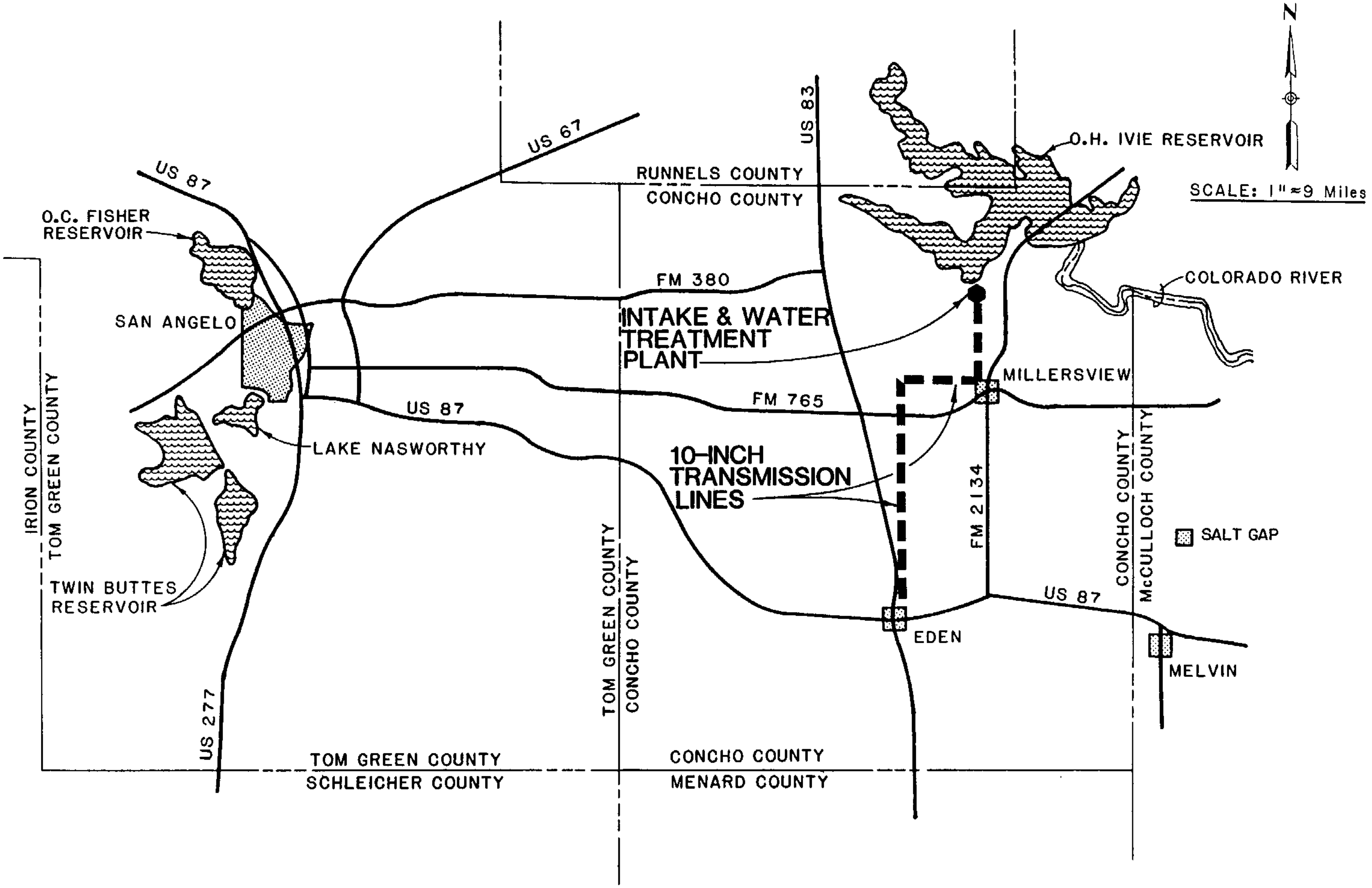
REGIONAL WATER SUPPLY PLAN

O.H. IVIE REGIONAL SYSTEM I

FIGURE 5.2.2-1



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REGIONAL WATER SUPPLY PLAN

O.H. IVIE REGIONAL SYSTEM II

FIGURE 5.2.2-2

Table 5.2.2-2
Current and Projected Water Use for O.H. Ivie Regional System II

Participant	Water Use[1] in 1990 (mgd)	Projected Water Requirements in 2010[1] (mgd)
Eden	0.247	0.328
Melvin	0.028	0.031
Millersview-Doole WSC	0.472	0.533
TOTAL	0.747	0.892

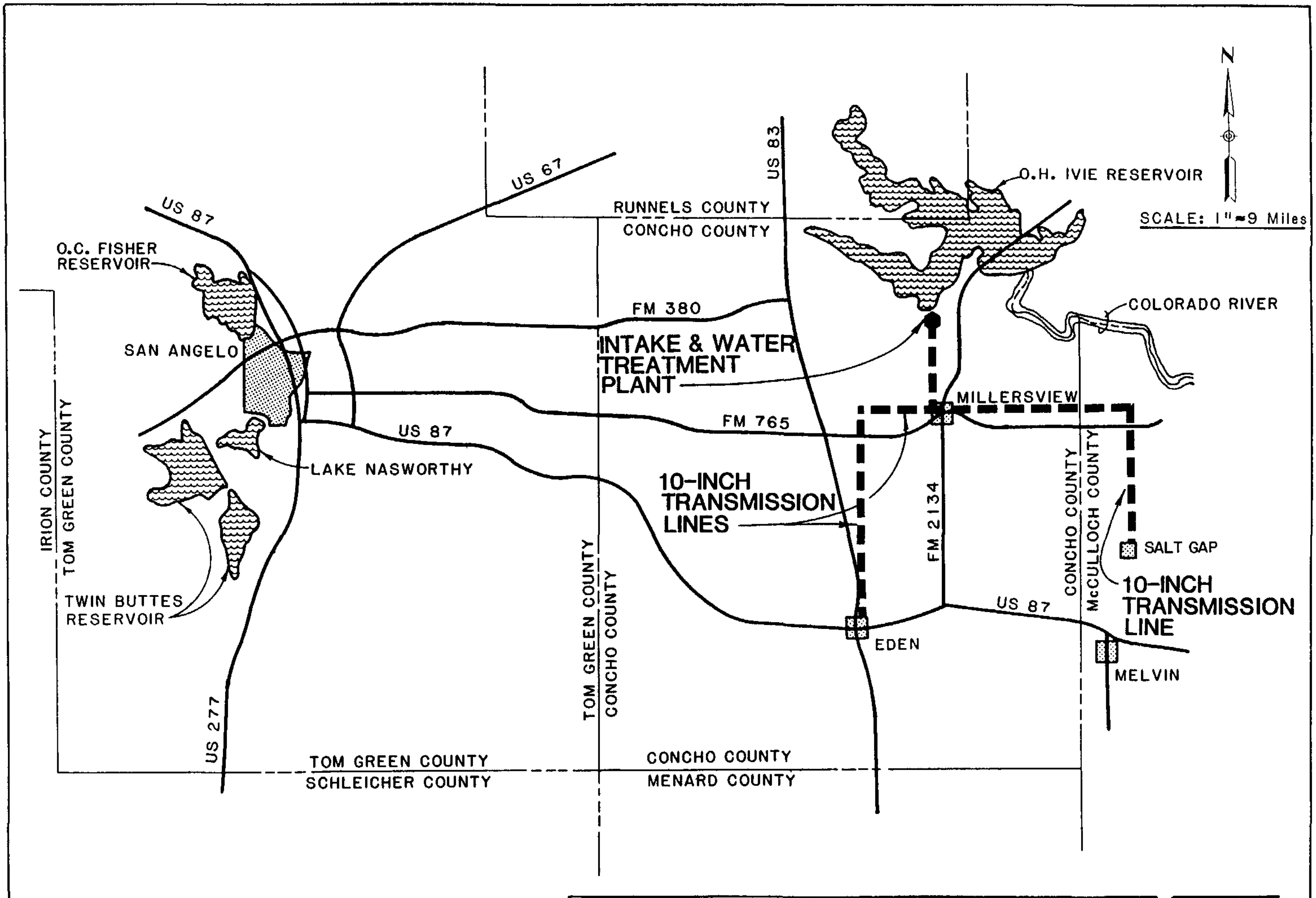
[1] From Table 2.4-1

interconnection to serve Melvin and a 10-inch transmission line to serve the City of Eden.

The estimated total project cost for the Ivie Regional System II is $8,000,000, with an estimated operation and maintenance cost of $1.34 per 1,000 gallons of treated water, including the cost of raw water. If the project cost is financed at eight percent interest for 25 years, without capitalized interest, the annual debt service would be $750,000, which equates to $2.75 per 1,000 gallons of 1990 water use. With O&M and raw water costs, the total 1990 cost would be $4.09 per thousand. By 2010, or when water requirements reach 0.824 mgd, the total cost per thousand would reduce to $3.70.

5.2.2.3 O.H. Ivie System III

Ivie System III would serve only the City of Eden, which, as Table 5.2.2-2 shows, used 0.247 mgd in 1990 and has a 2010 projected water need of 0.328 mgd. The system to serve the City of Eden (Figure 5.2.2-3) would be comprised of an intake and a 0.66 mgd water



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REGIONAL WATER SUPPLY PLAN

O.H. IVIE REGIONAL SYSTEM III

FIGURE 5.2.2-3

treatment plant at Ivie Reservoir and a 10-inch transmission line from the plant to the City of Eden.

The estimated total project cost for Ivie System III is $4,600,000, with an estimated operation and maintenance cost of $2.34 per one thousand gallons of treated water, including the cost of raw water. If the project cost is financed at eight percent for 25 years, without capitalized interest, the annual debt service would be $431,000, which equates to $4.78 per 1,000 gallons of 1990 water use. With O&M and raw water cost, the total 1990 cost would be $7.12 per thousand. By 2010, or when the requirements reach 0.260 mgd, the total cost per thousand gallons would reduce to $5.94.

5.2.3 Lake Brownwood

Lake Brownwood, completed in 1933, is owned by the Brown County Water Control and Improvement District Number 1. Their water rights permit allows 16,800 acre-feet to be used for municipal and industrial purposes and 50,590 acre-feet for irrigation. As is the case with O.H. Ivie Reservoir, there have not been any discussions with Brown County WCID #1, but Lake Brownwood does appear to be a potential water supply source for the North San Saba Water Supply Corporation. Based on discussions with Brown County WCID #1, the cost of raw water from Lake Brownwood would be $0.57/1,000 gallons. In order to serve North San Saba WSC, raw water would have to be released from Lake Brownwood into Pecan Bayou, and the water would subsequently be diverted for treatment at or near the confluence of Pecan Bayou and the Colorado River.

North San Saba Water Supply Corporation is projected to need 0.116 mgd of water in 2010 and its 1990 use was estimated to be 0.081 mgd.

The estimated total project cost for this system would be $3,000,000 with a projected operation and maintenance cost of $1.37 per 1,000 gallons, including raw water costs. If the project cost is financed at eight percent interest for 25 years, without capitalized interest, the annual debt service would be $281,000, which equates to $9.50 per 1,000 gallons of 1990 water use. With O&M and raw water costs, the total 1990 cost would be $10.87 per thousand. By 2010, or when the water requirement reaches 0.116 mgd, the total cost per thousand gallons would reduce to $7.59.

5.2.4 City of San Angelo

For the past 10 years, the City of San Angelo has been supplying Millersview-Doole WSC an average of 27.4 acre-feet of water per year (0.025 mgd). This water is used primarily in the western parts of the Corporation's service area. San Angelo's water supply sources include Lake O.C. Fischer, Lake Twin Buttes, Lake Nasworthy, a well field in Concho County, and it will soon be supplied from O. H. Ivie Reservoir. Based on a preliminary analysis of their projected annual requirements and the capacities of their water supply sources, it appears the City of San Angelo may have water available to sell to the participants of this study, but there have not been any discussions with San Angelo about their interest in selling additional water to any of the participants.

Millersview-Doole WSC, which presently obtains water from San Angelo, has a projected annual water requirement in 2010 of 0.533 mgd. Others that could possibly receive service from San Angelo include Eden and Melvin, which have projected 2010 requirements of 0.328 mgd and 0.031 mgd, respectively. To determine how much it would cost for San Angelo to serve these three participants with the 0.892 mgd they will need in

2010, a detailed evaluation of existing pipelines, pump stations, storage tanks, and connecting links would have to be made in order to determine the necessary modifications. The evaluation of the systems and the necessary modifications is beyond the scope of this study. However, it appears service from San Angelo may be a viable alternative, and Millersview-Doole WSC, Eden, and Melvin should consider this as a future option.

5.2.5 San Saba Reservoir

The proposed San Saba Reservoir Project, located 14 miles west of San Saba on the San Saba River, is a potential water supply source for the study area as well as the surrounding areas. If it is built to its maximum capacity, the project would have a surface area at the top of the flood control pool of 18,700 acres, would hold 901,000 acre-feet of water, and would have a sediment pool of 26,000 acre-feet. The water supply pool would have a surface area of 7,200 acres and a capacity of 280,000 acre-feet. The estimated firm yield of the reservoir is 27,400 acre-feet of water per year (24.4 mgd). Projected 2010 annual water requirements for all the participants of this study are 3.0 mgd. Thus, the San Saba Reservoir could serve as a water supply source for all of the entities of this study as well as some neighboring entities.

Construction cost of the reservoir, including flood control storage, is estimated at $144 million (1962 prices adjusted for inflation).[2] Cost estimates for a water supply project, without flood control capacity, are not available, but based on splitting cost of the reservoir pro-rata between flood control and water supply (based on water in each pool), the cost of

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[2] "Texas Basins Project," Volume I, page 87, Bureau of Reclamation, U.S. Department of the Interior, Amarillo, Texas, February, 1965.

a water supply project could be expected to be about $45 million. This would equate to $0.47 per 1,000 gallons. In addition to the cost of the reservoir, a regional water system, including water treatment, pumping, transmission, and storage facilities, would have to be planned, constructed, and operated in order to supply the participants of this study with water from this source. It is estimated that such a regional water supply would cost $28,000,000 to construct and $0.92/1,000 gallons to operate and maintain. If the regional water supply system project cost is financed at eight percent interest for 25 years, without capitalized interest, the annual debt service would be $2,623,040, which equates to $2.79 per 1000 gallons of 1990 water use. With O&M, the total 1990 cost including raw water, would be $4.18 per thousand. By 2010, or when the water requirement reaches 3.0 MGD, the total cost per thousand gallons would reduce to $3.58.

As of the date of this study, the San Saba Reservoir project has received only preliminary planning attention. Therefore, it can only be considered a potential water supply for the distant future, i.e., after the year 2010, since it usually takes more than 20 years to plan and develop such a project if there are no legal and environmental problems. To begin the process, a local sponsor would need to obtain the necessary permits, arrange financing, and build the project. Since planning of the San Saba Reservoir has not advanced beyond the preliminary stages, it is not possible to provide additional information in this study about the potential of this long range project as a future water supply for the participants of this study.

5.2.6 Summary

For the participants of this study, five potential sources of surface water have been identified which could either replace or supplement their existing sources. For most potential sources, a number of possible regional systems have been identified. Table 5.2.6-1 lists the potential sources and regional systems identified in this chapter.

5.3 Ground Water Sources

At the present time, the City of San Saba has the only potential source of ground water that might be used to either replace or supplement the existing sources for the participants of this study.

The City of San Saba has four wells that serve as its primary source of drinking water. Based on the capacities of these wells and the projected annual requirement of the City, excess capacity is available. However, the gross alpha and combined radium concentrations exceed the present MCL of the TDH Drinking Water Standards, but not the proposed MCL of the Radionuclide Rule. It should be pointed out that no information is available on the concentration of radon for the City of San Saba's well supply. Based on the analysis from the participants of this study, the concentration of radon probably exceeds the MCL for radon in the Radionuclide Rule.

North San Saba WSC has arranged with the City of San Saba to obtain an emergency water supply from the City and is presently constructing a pipeline to connect to San Saba's water distribution system. North San Saba WSC should consider obtaining enough water from the City to meet their projected annual requirement, which is 0.116 mgd annually in 2010. However, in this study, there have not been any contacts with San Saba regarding additional water supplies for North San Saba WSC, nor have the conveyance and storage facilities been evaluated to determine if existing facilities and pumping capacities would be adequate to implement such an option.

Table 5.2.6-1
Projected Cost of Potential Surface Water Sources and Regional Water Systems to Serve Study Participants
(cost per 1,000 gallons based on 2010 water use)

Participants	Lake Brady		O.H. Ivie Reservoir			Lake Brownwood	City of San Angelo	San Saba Reservoir
	I	II	I	II	III			
Brady	1.65	1.89						3.58
Live Oak Hills	1.65	1.89						3.58
Richland WSC	1.65	1.89						3.58
Rochelle WSC	1.65	1.89						3.58
Millersview-Doole WSC		1.89	3.41	3.70			NA	3.58
Melvin		1.89		3.70			NA	3.58
Eden				3.70	5.94		NA	3.58
North San Saba WSC						7.59		3.58

SECTION 6

6.0 SAFE DRINKING WATER ACT REQUIREMENTS

6.1 Introduction

The Safe Drinking Water Act (SDWA), originally signed into law on December 16, 1974, mandated the establishment of the first national regulations for drinking water that were to apply to all public water systems in the United States. The U.S. Environmental Protection Agency (EPA) was authorized to set regulations for drinking water, conduct research and special studies as required, and administer the implementation of the act. State governments, via their health departments or environmental agencies, were to assume the major responsibility for implementation and enforcement of the requirements of the SDWA in their individual states. The local public water systems were obligated by law to assume the actual day-to-day responsibility of meeting the regulations.

To improve the SDWA, in particular the process for setting regulations, and to initiate the protection of groundwater, Congress amended most of the 1974 act and added six new sections in 1986. The SDWA Amendments of 1986, signed into law on June 19, 1986, mandated the establishment of a variety of new drinking water regulations with a very specific timetable. The 1986 Amendments included several water quality related regulations, as follows:

* Maximum contaminant level goals (MCLGs) and maximum contaminant levels (MCLs) must be established for 83 contaminants listed in the Advanced Notice of Proposed Rulemaking published March 4, 1982 and October 5, 1983. EPA was required to set MCLGs and MCLs for these 83 contaminants, although up to seven substitutes were allowed.

* MCLGs and MCLs must be established for 25 contaminants selected from a priority list to be prepared by EPA and updated every three years.

- For all public water systems utilizing surface water sources, EPA must establish criteria requiring filtration of the water as a minimum treatment technique.
- Disinfection of drinking water is required for all public water systems.

Specific time lines were specified in the Act for development of regulations to implement the various requirements of the 1986 Amendments. In addition, EPA was to establish regulations covering monitoring of additional unregulated contaminants, so data could be developed on their occurrence and the health risks they present could be evaluated.

The process for developing regulations was modified as follows:

1. The previously established recommended maximum contaminant levels (RMCLs) were redefined as MCLGs. MCLGs are non-enforceable health-based goals which must be set at a level so no known or anticipated adverse effect on human health occurs. An adequate margin of safety must be allowed without regard to cost.
2. EPA is to publish MCLGs and promulgate MCLs for each contaminant that, in the judgement of EPA, may have an adverse health effect if it occurs or is anticipated to occur in any public water system. The MCLG should be proposed at the same time that the MCL for a contaminant is promulgated.
3. MCLs are enforceable standards. EPA is required to set the MCL for a contaminant as close to the MCLG as feasible, using the best technology, treatment techniques, and other available means. Cost is taken into consideration.
4. For each contaminant, EPA is to establish the technology, treatment technique, and other means feasible for metering the MCL, which is to be referred to as the best available technology (BAT). Public water systems may use any "appropriate technology," including BAT, that is acceptable to the State and results in compliance with an MCL.
5. EPA can establish the use of a treatment technique in lieu of an MCL, if it is determined that monitoring for the contaminant is not economically or

technologically feasible.

The 1974 SDWA required water systems to notify their customers when drinking water standards were violated. The 1986 Amendments directed EPA to revise the original public notification requirements to provide for different types and frequencies of notices based on the type of violation. In addition, the new public notification requirements were to take into account the seriousness of any potential adverse health effects. EPA is to determine the mandatory language that must be used in each public notice for each violation of each new regulatory phase.

With the 1986 Amendments, Congress also established a specific schedule for EPA to follow in implementing the new regulations. However, the established development and review process for proposed and final regulations to be promulgated by EPA is extensive and involves public participation at several levels. Congress underestimated the time needed to develop sound regulations, so most actions have exceeded the originally established time schedule.

Congress established the general guidelines of the 1986 Amendments, and EPA, utilizing these guidelines, has developed specific rules for the implementation of the requirements. In this chapter, the specific regulations impacting the participants of this study will be identified, as well as the anticipated schedule for implementation of the regulations.

6.2 Volatile Organic Chemicals (VOCs) Rule

The final Volatile Organic Chemicals (VOC) Rule was originally published on July

8, 1987, and on July 1, 1988, corrections were published to the rule. The VOC regulation establishes MCLGs and MCLs for the eight VOCs listed in Table 6.2-1.

Table 6.2-1
MCLGs and MCLs for Volatile Organic Chemicals

Contaminant	MCLG (mg/l)	MCL (mg/l)
Benzene	0.000	0.005
Carbon Tetrachloride	0.000	0.005
para-Dichlorobenzene	0.075	0.075
1,2-Dichloroethane	0.000	0.005
1,1-Dichloroethylene	0.007	0.007
1,1,1-Trichloroethane	0.200	0.20
Trichloroethylene	0.000	0.005
Vinyl Chloride	0.000	0.002

The major impact of this particular rule on the participants of this study will be the monitoring requirements, which include the eight VOCs listed above as well as 51 additional contaminants. The initial monitoring requirement is one sample per quarter for one year at each entry point to the distribution system, regardless of whether any VOCs are initially detected. Composite samples of up to five sampling points are allowed. Compliance with the MCLs is determined by a running annual average of quarterly samples for each sample location. If the average is greater than the MCL, then the system is out of compliance.

The initial monitoring requirements were phased in based on the size of the population served by the public water system. For systems with a service population greater than 3,300, the initial sampling and analysis should have been completed by December, 1989; and for systems with a service population of less than 3,300, the initial sampling and analysis is to be completed by December, 1991. In Texas, the Texas Department of Health (TDH) typically performs this analysis for public water systems as part of their routine sampling program.

Repeat monitoring and sampling for all sources must take into account the previous monitoring results and the vulnerability of the source. The vulnerability of each water system is determined by the state, which bases its conclusion on several factors, including: previous monitoring results; number of people served by the system; proximity of the system to commercial or industrial use, disposal, or storage of VOCs; and level of protection given to the water source, such as a watershed management or wellhead protection program. A system is considered vulnerable for a period of three years after any positive sample of one or more VOCs. If no VOCs are detected and the source is determined to be nonvulnerable to possible contamination, then the repeat monitoring frequency can be reduced to every three years.

In addition to the monitoring requirements for the eight regulated VOCs, monitoring requirements for 51 additional contaminants were also included. The contaminants to be monitored can be placed into three general groups:

- Thirty-four compounds that must be monitored for all systems;
- Two compounds that must be monitored by systems whose water supply is determined to be vulnerable to possible contamination; and
- Fifteen compounds that must be monitored by systems at the discretion of the state.

The monitoring and sampling schedule for these 51 additional contaminants is the same as the schedule for VOCs. The initial monitoring requirements were phased in based on the size of the population served by the public water system. For systems with a service population greater than 3,300, the initial sampling and analysis should have been completed by December, 1989; and for the systems with a service population of less than 3,300, the initial sampling and analysis is to be completed by December, 1991. In Texas, TDH typically performs this analysis for public water systems as part of their routine sampling program. Repeat sampling and monitoring must be performed every five years.

Based on the analyses completed to date by TDH, no measurable concentration of any of the eight regulated VOCs and recommended 51 additional contaminants has been detected in the areas covered by this study. Therefore, the only impact of this Rule on the participants of this study will be the increased cost of monitoring and analysis performed by TDH.

6.3 Fluoride Rule

The Fluoride Rule was originally published on November 14, 1985, and finalized on April 2, 1986. It established both the MCLG and MCL at 4.0 mg/l to protect against crippling skeletal fluorosis. The secondary maximum contaminant level (SMCL) was set at 2.0 mg/l to protect against objectionable dental fluorosis, which is not considered an adverse health effect by EPA. System compliance with the fluoride MCL is based on each sampling point; in other words, if any sampling point is out of compliance, the entire system is deemed to be out of compliance.

Public water systems utilizing groundwater must monitor for fluoride once every three years. TDH may reduce the monitoring requirements to once every 10 years if it determines that the system is not likely to exceed the MCL. However, it is recommended that systems practicing fluoridation monitor the fluoride level daily.

The general public notification requirements of non-compliance apply to this rule. In addition, community water systems that exceed the SMCL, but not the MCL, are required to give special notice annually using the mandatory language that informs the customer of the significance of exceeding the SMCL.

Based on the analyses completed to date by TDH, the fluoride concentrations in all of the water systems participating in this study are below the MCL with only Millersview-Doole WSC exceeding the SMCL. Therefore, the only impact of this rule on the participants of this study is the increased cost of monitoring and analysis performed by TDH.

6.4 Surface Water Treatment Rule

Known as the Surface Water Treatment Rule (SWTR), the regulations for filtration, disinfection, turbidity, *Giardia lamblia*, viruses, *Legionella*, and heterotrophic bacteria were proposed on November 27, 1987, and promulgated on June 29, 1989. The rule applies to all public, community and noncommunity, water systems that use surface water sources or ground water sources under the direct influence of surface water. This rule assumes that all surface water sources and groundwater sources under the direct influence of surface water can be potentially contaminated by *Giardia lamblia* and other protozoa, viruses, and pathogenic bacteria.

This rule would only impact the participants of this study if TDH determines that the groundwater supply sources are under the direct influence of surface water. If such a determination is made, then the affected entity would have to provide treatment of the water source in accordance with this rule.

6.5 Total Coliform Rule

Total coliforms were included in the list of 83 contaminants that were to be regulated in the SDWA 1986 Amendments. EPA developed a proposed rule for total coliform, which was published on November 3, 1987. Additional regulatory options were published on May 6, 1988, and the final rule was promulgated on June 29, 1989. Beginning January 1, 1991, all public water systems in the nation must comply with this rule.

In the final rule for total coliforms, the MCL for water systems analyzing at least 40 samples per month is that no more than 5.0 percent of the monthly samples may be positive for total coliform. The MCL for water systems analyzing less than 40 samples per month requires that no more than one sample per month may be positive for total coliform. The frequency of monthly routine sampling is based on the population served by the water system.

All public water systems must sample according to a written plan which includes a map of the water distribution system showing the location of the sample sites. The plan is subject to review and revision by TDH, which determines if the plan is adequate.

The biggest impact of the Total Coliform Rule on the participants of this study is the repeat sample monitoring requirements. For each routine sample that tests positive for total coliform, three or four repeat samples, depending on the number of monthly routine samples, must be collected and analyzed for total coliforms. At least one repeat sample must be taken from the same location as the original positive sample. Two or three repeat samples must be taken within five service connections of the original sample location, one upstream, one downstream, and, if required, the other somewhere in between. The repeat samples must be collected within 24 hours of the notification of the original results. The repeat samples are required to determine whether the original positive sample is indicative of contamination of the system, or of a nondistribution system problem. If there is a nondistribution problem, then TDH can invalidate the sample for compliance calculation purposes.

If total coliforms are detected in any repeat sample, the utility must collect another set of repeat samples, using the same procedure as before, unless the MCL has been violated and the utility has notified TDH. If there is a violation, TDH may reduce or eliminate the repeat sampling requirement for the remainder of the month.

For compliance calculations, all positive total coliform samples count except for samples that are invalidated by TDH. TDH can invalidate a sample, if:

- The laboratory performing the analysis acknowledges that improper sample analysis caused the positive results;
- The contamination is a domestic or nondistribution system plumbing problem if the repeat sample taken at the same tap was positive while the other repeat samples taken nearby are negative; or
- TDH has substantial grounds to indicate that the positive result is because of some circumstance or condition not related to the quality of drinking water in the distribution system.

Whenever a total coliform positive sample is invalidated, TDH must provide written documentation that states the specific cause of the positive sample as well as the action that has been or will be taken by the utility to correct the problem.

If a total coliform sample tests positive, whether routine or repeat, then a fecal coliform or *E. coli* analysis must be performed on the positive sample to determine the extent of the problem. If fecal coliform or *E. coli* is detected, then the utility must notify TDH immediately.

Based on the population served by each participant's water distribution system, if more than one sample, whether routine or repeat, per month tests positive for total coliform, then the utility is out of compliance with this rule. If a routine total coliform positive sample tests positive for fecal coliform or *E. coli*, the repeat total coliform sample tests positive, and the original sample is not invalidated, the utility is in violation of the MCL also. Due to its severity, a violation of this rule is an acute violation, requiring public notification with the appropriate mandated language via the TV and radio within 72 hours and the newspaper within 14 days of the violation.

This rule could have a significant impact on the participants of this study. Since the number of routine monthly bacteriological samples to be taken for the Total Coliform Rule is the same as that required in the present Drinking Water Standards, there will no immediate impact to the participants. However, if one sample is tested positive, the utility must take the required repeat samples for testing. If a repeat sample tests positive and the original sample is not invalidated, then public notification is required. Public notification with the mandatory language will clearly have a negative impact on the utility.

6.6 Lead and Copper Rule

On August 18, 1988, the original requirements to minimize lead and copper in drinking water were proposed. EPA received many comments from the public, as well as numerous congressional inquiries, on the proposed rule. The final rule was scheduled to be proposed in December, 1990, but this deadline slipped due to comments on the proposed rule. Finally, on May 6, 1991, the Lead and Copper Rule was proposed.

For water sampled at the customer's tap, the proposed MCLG is 0 and 1.3 mg/l for lead and copper, respectively. In lieu of an MCL for these constituents, a treatment technique has been specified. The action level triggers for initiating the treatment technique requirements, will be a maximum concentration of 0.015 mg/l for lead and 1.3 mg/l for copper in 90% of the samples taken at the customer's tap. The treatment technique for lead includes: (1) optimal corrosion control treatment; (2) source water treatment; (3) public education; and (4) lead service line replacement. The treatment technique for copper includes: (1) optimal corrosion control treatment; and (2) source water treatment.

For medium-sized systems (3,300 to $\leq$ 50,000 population), the schedule for determining compliance with the Lead and Copper Rule begins with an initial sampling period starting July, 1992. For the small-sized systems (< 3,300 population), the initial monitoring period begins in July, 1993. Table 6.6-1 provides the general monitoring requirements for the various stages of the Lead and Copper Rule.

Table 6.6-1
Lead and Copper Monitoring

Monitoring Period	Parameters	Location	Frequency
Initial Monitoring	Lead and Copper	Targeted high risk interior taps	Every six months
After Installation of Corrosion Control	Lead and Copper	Targeted high risk interior taps	Two consecutive six month monitoring periods
After State Specifies Parameter Values for Optimal Corrosion Control	Lead and Copper	Targeted high risk interior taps	Two consecutive six month monitoring periods
Reduced Monitoring After 1 Year of Compliance	Lead and Copper	Targeted high risk interior taps	Once every year
After 3 Years of Compliance	Lead and Copper	Targeted high risk interior taps	Once every three years

If a small or medium-sized system exceeds the action levels as proposed, the system must submit its recommendation for optimal corrosion control treatment to the Texas Department of Health. TDH may approve the system's recommendation, require the installation of an alternative corrosion control treatment, or require the system to conduct a corrosion control treatment study. The corrosion control study will determine the effectiveness of adjusting the pH and alkalinity as well as calcium concentration of the treated water to reduce the acidity or increase the scale forming potential of the treated water. In addition, the effectiveness of the addition of a corrosion inhibitor, such as sodium hexametaphosphate, to minimize the concentration of lead and copper at the consumer's tap will be determined. Table 6.6-2 provides the regulatory schedule for the Lead and Copper Rule for medium-sized systems. The regulatory schedule for small-sized systems in general follows the steps of the medium-sized systems, but it starts with initial monitoring beginning July, 1993.

Based on the source water and at-the-tap monitoring, TDH may determine that source water treatment is necessary to *minimize* the levels of lead and copper at the tap. For any size system, the system would have 36 months to install the necessary treatment at the water source and conduct follow-up monitoring. This time period may be extended for medium and small systems because: (1) these systems comprise the vast majority of public water systems and the large number of evaluations that must be completed by TDH will place a strain on their resources; and (2) greater TDH involvement and technical assistance will be required to assist medium and small systems in determining optimum corrosion control treatment.

In addition to lead and copper, systems that exceed the action levels will be required to monitor the general quality of the treated water as it enters the representative taps throughout the distribution system. Table 6.6-3 provides the general monitoring requirements for determining compliance with the Lead and Copper Rule based on the water quality parameters specified.

In Appendix C, a copy of the EPA Fact Sheet on the Lead and Copper Rule is included to provide more detailed information.

Table 6.6-2
Lead and Copper Rule
Regulatory Schedule for Medium-Size Systems
(3,300 to 50,000 people)

With Corrosion Study	Date	No Corrosion Control Study
Lead & Copper Rule Promulgated	May 6, 1991	Lead & Copper Rule Promulgated
Begin monitoring	July, 1992	Begin monitoring
Treatment Technique requirements take effect	November, 1992	Treatment Technique requirements take effect
State requires system to conduct study	January, 1994	
	July, 1994	State designates optimal corrosion control treatment
Complete study & submit to State	July, 1995	
State designates optimal corrosion control treatment	January, 1996	
	July, 1996	Complete installation of corrosion control treatment
	July, 1997	Complete follow-up monitoring
Complete installation of corrosion control treatment	January, 1998	State designates water quality parameters
Complete follow-up monitoring	January, 1999	
State designates water quality parameters	July, 1999	

6.7 Synthetic Organic and Inorganic Chemical Rule (Phase II)

EPA proposed regulations for 30 synthetic organic chemicals (SOCs) and eight inorganic chemicals (IOCs) on May 22, 1989. EPA issued the final rule on January 30,

Table 6.6-3
Lead and Copper Rule
Water Quality Parameter Monitoring

Monitoring Period	Parameters	Location	Frequency
Initial Monitoring	pH, alkalinity, orthophosphate or silica, calcium, conductivity, and temperature	Taps and entry point(s) to distribution system	Every six months
After Installation of Corrosion Control	pH, alkalinity, orthophosphate or silica, and calcium	Taps	Every six months
	pH, alkalinity dosage rate and concentration, and inhibitor dosage rate and residual	Entry point(s) to distribution system	Biweekly
After State Specifies Parameter Values for Optimal Corrosion Control	pH, alkalinity, orthophosphate or silica, and calcium	Taps	Every six months
	pH, alkalinity dosage rate and concentration, and inhibitor dosage rate and residual	Entry point(s) to distribution system	Biweekly
Reduced Monitoring	pH, alkalinity, orthophosphate or silica, and calcium	Taps	Every six months
	pH, alkalinity dosage rate and concentration, and inhibitor dosage rate and residual	Entry point(s) to distribution system	Biweekly

1991; it will become effective 18 months thereafter. The proposed MCLGs and MCLs for the 30 SOCs and eight IOCs are given in Table 6.7-1.

The proposed monitoring requirements for groundwater sources are as follows:

1. For the IOCs:
   a. The monitoring schedule for barium, cadmium, chromium, mercury, and selenium will be every three years.
   b. The monitoring schedule for asbestos is once every nine years.
   c. The initial monitoring for nitrate-nitrite will be annually, unless the concentrations measured are greater than 50 percent of the MCL, for which quarterly sampling is required.

2. For the SOCs:
   a. The initial monitoring for all SOCs is quarterly for one year.
   b. Repeat monitoring frequency is dependent on the occurrence of one of the listed contaminants. If after one round of sampling, no contaminants are detected, then the monitoring frequency will be reduced to two samples for medium-sized systems and one sample for small-sized systems every three years.

3. For the pesticides, herbicides, and PCBs:
   a. The initial monitoring for all SOCs is quarterly for one year.
   b. Repeat monitoring frequency is dependent on the occurrence of one of the listed contaminants. If after one round of sampling, no contaminants are detected, then the monitoring frequency will be reduced to two samples for medium-sized systems and one sample for small-sized systems every three years.

4. For the Volatile Organic Chemicals (VOCs):
   a. For the initial monitoring period, quarterly samples are required for one year.
   b. A vulnerability assessment is required to determine the repeat sampling frequency. If the supply source is determined to be nonvulnerable, then monitoring is not required.
   c. If the supply source is determined to be vulnerable, then the initial monitoring is quarterly for the first year. If any of the VOCs are detected, then the repeat monitoring frequency continues on a quarterly basis. If none are detected, then four quarterly samples are required to be taken every three years for systems with more than 500 connections and every five years for systems with less than 500 connections.

Table 6.7-1
Contaminants, MCLGs, and MCLs Proposed in the SOC and IOC Rule (Phase II)

Contaminant	MCLG (mg/l)	MCL (mg/l)
Inorganics		
Asbestos	7 million fibers/l	7 million fibers/l
Barium	2	25
Cadmium	0.005	0.005
Chromium	0.1	0.1
Mercury	0.002	0.002
Nitrate[1] (as Nitrogen)	10	10
Nitrite[1] (as Nitrogen)	1	1
Selenium	0.05	0.05
Volatile organics (solvents)		
cis-1,2-Dichloroethylene	0.07	0.07
1,2-Dichloropropane	0	0.005
Ethylbenzene	0.7	0.7
Monochlorobenzene	0.1	0.1
o-Dichlorobenzene	0.6	0.6
Styrene	0.1	0.005
Tetrachoroethylene	0	0.005
Toluene	1	1
trans-1,2-Dichloroethylene	0.1	0.1
Xylene	10	10
Pesticides, herbicides, PCBs		
Alachlor	0	0.002
Aldicarb	0.003	0.003
Aldicarb sulfone	0.003	0.003
Aldicarb sulfoxide	0.003	0.003
Atrazine	0.003	0.003
Carbofuran	0.04	0.04
Chloradane	0	0.002
Dibromochloropropane (DBCP)	0	0.0002
2,4-D	0.07	0.07
Ethylene dibromide (EDB)	0	0.0005
Heptachlor	0	0.0004
Heptachlor epoxide	0	0.0002
Lindane	0.0002	0.0002
Methoxychlor	0.04	0.04
PCBs	0	0.0005
Pentachlorophenol	0.001	0.001
Toxaphene	0	0.003
2,4,5-TP (Silvex)	0.05	0.05
Water Treatment Chemicals		
Acrylamide	0	Treatment Technique[2]
Epichlorohydrin	0	Treatment Technique[2]

[1] Total nitrate plus nitrate MCLG and MCL = 10 mg/l (as Nitrogen).
[2] Treatment technique requirement limits the amount of the chemical used to treat drinking water.

5. For the water treatment chemicals, monitoring is not required since the MCL is a treatment technique which limits the use of the chemicals that could possibly contain these contaminants.

Compliance with the MCL for each of the contaminants is based on the annual running average for each sample point for systems monitoring on a quarterly (or more frequently) basis. For systems monitoring annually or less frequently, compliance is based on a single sample. For nitrate and nitrite, if the first sample exceeds the MCL, another sample must be taken within 24 hours. Compliance for nitrate and nitrite is based on the average of these two samples.

Based on the analyses completed to date by TDH, the concentrations of these contaminants in all of the water systems participating in this study are below the MCL. Therefore, the only impact of this rule on the study participants will be the increased cost of monitoring and analysis performed by TDH.

6.8 SOCs AND IOCs RULE (Phase V)

The Phase V rule will set regulations for 24 of the last 25 contaminants of the 83 mandated for regulation by the SDWA 1986 Amendments. The original final statutory deadline was June 19, 1989, but the rule was not published until July 25, 1990, and the final rule is not expected to be promulgated until March, 1992. The tentative MCLGs and MCLs for the 18 SOCs and six IOCs are shown in Table 6.8-1.

Compliance requirements will parallel the Phase II SOC-IOC Rule. The unique aspect of the Phase V rule is the lack of data on the occurrence of most of the contaminants of this rule. However, EPA is required by the SDWA 1986 Amendments to regulate these

Table 6.8-1
Tentative Contaminants, MCLGs, and MCLs to be Proposed in the SOCs and IOCs Rule (Phase V)

Contaminant	MCLG (mg/l)	MCL (mg/l)
Synthetic Organic Chemicals		
Dalapon	0.2	0.2
Di(2-ethyl hexyl)adipate	0.5	0.5
Di(2-ethyl hexyl)phthalate	0.00	0.004
Dichloromethane	0.00	0.005
Dinoseb	0.007	0.007
Diquat	0.02	0.02
Endothall	0.1	0.1
Endrin	0.002	0.002
Glyphosate	0.7	0.7
Hexachlorobenzene	0.00	0.001
Hexachlorocyclopentadiene	0.05	0.05
Oxamyl (vydate)	0.2	0.2
PAHs (Benzo(a)pyrene)*	0.00	0.0002
Picloram	0.5	0.5
Simazine	0.001	0.001
1,2,4-Trichlorobenzene	0.009	0.009
1,1,2-Trichloroethane	0.003	0.005
2,3,7,8-TCDD (Dioxin)	0.00	5×10^{-8}
Inorganic Chemicals		
Antimony	0.03	0.005
Beryllium	0.00	0.001
Cyanide	0.2	0.2
Nickel	0.1	0.1
Sulfate	400	400
Thallium	0.0005	0.001

*EPA is considering establishing MCLGs and MCLs for six additional PAHs classified as human carcinogens: benz(a)anthracene, benzo(b)fluoranthene, benzo(k-fluoranthene, chrysene, dibenz(a,h)anthracene, and indenopyrene.

contaminants, regardless of their occurrence, so monitoring requirements will most likely be set at some minimum frequency.

It should be noted that many of the organic contaminants listed in the Phase V Rule are included as additional contaminants for monitoring under the VOC Rule (Phase I) and the SOC-IOC Rule (Phase II).

Based on the analyses completed to date by TDH, the concentrations of these contaminants in all of the water systems participating in this study are below the proposed MCL. Therefore, the only impact of this rule on the participants of this study will be the increased cost of monitoring and analysis performed by TDH.

6.9 Radionuclide Rule (Phase III)

The proposed Phase III regulation will develop MCLGs and MCLs for the radionuclides found in drinking water. A proposed rule was scheduled to be published in September, 1990, but due to delays, it was not proposed until June 17, 1991.

The rule sets the following MCLGs and MCLs for radon-222, radium-226, radium-228, uranium, and gross alpha (adjusted) and beta particle emitters.

	MCLG	MCL
Radon	Zero	300 pCi/l
Radium-226	Zero	20 pCi/l
Radium-228	Zero	20 pCi/l
Uranium	Zero	30 pCi/l
Gross Alpha (adjusted)	Zero	15 pCi/l
Gross Beta	Zero	4 mrem ede/yr

Based on the analyses completed to date by TDH, the concentration of radon in all of the water systems participating in this study is above the proposed MCL, while the concentration of radium-266 is below the proposed MCL. The only study participants who have drinking water with a concentration of radium-228 that exceeds the proposed MCL are Millersview-Doole Water Supply Corporation, North San Saba Water Supply Corporation, and Richland Water Supply Corporation - Brady System.

This rule will have a major impact on the participants of this study. Depending on what the final MCL is established for the radionuclides, every system could possibly be in violation of at least one MCL. Therefore, each system will probably have to provide some form of treatment to reduce the concentration levels of the contaminants to meet the SDWA 1986 Amendments. The treatment could be as simple as providing aeration to remove radon or as complex as installing reverse osmosis or ion exchange to remove radium, or obtaining water from a new water supply source.

In Appendix D, a copy of the EPA Fact Sheet on the Radionuclide Rule is included to provide more detailed information.

6.10 Disinfection/Disinfection By-Products Rule (Phase VIa)

The SDWA 1986 Amendments require EPA to regulate 25 additional contaminants beyond the original 83, every three years beginning in 1991. Contaminants to be regulated under the Disinfection/Disinfection By-Products (D/DBP) Rule will satisfy a major portion of this regulatory requirement. The remainder of the required 25 contaminants will be regulated in a separate rule to be known as Phase VIb. EPA activity began on this rule in the spring of 1989, when EPA developed a strawman proposal outlining its initial reasoning on the rule. The D/DBP Rule is expected to be proposed in June, 1993 and the final rulemaking completed in June, 1995.

The major thrust of the D/DBP Rule will be monitoring requirements and MCLs designed to reduce human exposure to disinfectants and their by-products. This will be done by changing water treatment practices.

The D/DBP Rule will probably be the most complex rule developed by EPA because of its interface with other rules and requirements of the amendments. For instance, the SWTR will require that a water system meet certain minimum CT values (disinfection), turbidity requirements, and residual disinfectants in the distribution system. The proposed Lead and Copper Rule may require some communities to increase the pH of the treated water to protect the piping and household plumbing, but this higher pH promotes trihalomethane (THM) formation, which is a disinfectant by-product. In order to meet the minimum disinfection requirements of SWTR for free chlorine, higher CT values are required at higher pH values. Therefore, water utilities will be required to meet the mandatory disinfection requirements and the Lead and Copper Rule and, at the same time, meet the restrictions placed on disinfectant application by the D/DBP Rule.

The impact of this rule on the study participants cannot yet be determined, but it will probably be minimal, as long as the primary water source is groundwater. The greatest impact will be the increased costs for the additional sampling and monitoring, which will probably be done by TDH.

6.11 Groundwater Disinfection

The SDWA 1986 Amendments require EPA to set mandatory disinfection requirements for all public water systems, while disinfection of surface water sources is regulated by the SWTR. It was originally anticipated that groundwater disinfection would be covered with the D/DBP Rule. The schedule for the promulgation of this rule has yet to be determined.

Although the Groundwater Disinfection Rule has not yet been published, numerous publications have indicated what the rule might contain. The rule is expected to set a level of disinfection, which will probably be similar to the SWTR, with a minimum CT value to be maintained in order to provide the necessary inactivation of viruses. Continuous monitoring of the disinfectant residual entering the distribution system will probably be required for systems serving more than 3,300 people, and a single grab sample will be required for systems with less than 3,300 people. A minimum residual disinfectant level in the distribution system must be found in 95 percent of the distribution system samples.

Disinfection of all public water systems has been required in Texas since the development of the Rules and Regulations for Public Water Systems. Therefore, the impact of this rule on the participants of this study will probably be minimal. Its greatest impact will be the continuous monitoring of the disinfectant residual entering the distribution system.

SECTION 7

7.0 CONCLUSIONS AND RECOMMENDATIONS

7.1 Conclusions

1. The population for each of the study participants is projected to increase slightly by the year 2010. The projected annual growth rates range from 0.3% to 1.6%. In addition, the annual water requirements, taking conservation into consideration, will also increase slightly by the year 2010.

2. The existing water systems for all of the participants meet or exceed the minimum requirements of the Texas Department of Health, except for the high concentrations of Combined Radium-226 and Radium-228 and Gross Alpha in the supply sources. In addition, the iron and manganese concentrations in the water for a number of the participants exceed the recommended MCL. To alleviate any problems with iron and manganese, a sequestering agent is injected into the water.

3. Various methods to reduce the concentrations of radium and radon were identified and evaluated. Based on this evaluation, the sodium cation exchange treatment process appears to be the most practical method to reduce the concentration of radium. A number of other treatment processes, although they are either untested and new or very expensive, appear to be potential methods that should be considered when the participants decide to reduce the high levels of radium.

4. The potential alternate water supply sources and associated regional water supply systems were identified and evaluated. Based on this evaluation, the cost of

constructing any of the regional water supply systems to make use of alternate water sources was much higher than the cost of constructing the most practical treatment method to reduce the concentration of radium in existing water supply sources.

5. The existing and future requirements of the 1986 Amendments to the Safe Drinking Water Act were reviewed. Based on this review, the greatest primary impact of the new rules and regulations will be the cost to meet the Radionuclide Rule. A secondary impact will be the increase in annual assessment charged by the Texas Department of Health to cover the increased testing requirements. The Radionuclide Rule will establish the MCL for radium-226, radium-228, radon, uranium, gross alpha (adjusted), and gross beta. The proposed MCL for radium-226 and radium-228 will be 20 pCi/l each, and for radon, it will be 300 pCi/l. If the MCLs are established at these levels in the final Rule, then only Millersview-Doole Water Supply Corporation, Richland Water Supply Corporation, and North San Saba Water Supply Corporation will need to provide some form of treatment to reduce the concentration of radium below the new regulatory level. If the MCL for radon is established at 300 pCi/l, all of the participants will be required to treat their water supplies to reduce the concentration of radon below the new regulatory level.

7.2 Recommendations

1. It is recommended that the participants of this study request that the bilateral agreement between the Texas Department of Health (TDH) and the participants be stayed until the MCLs for Radium-226, Radium-228, Radon, Gross Alpha (adjusted), and Gross Beta are promulgated in the Radionuclide Rule, and until rules and procedures are established by TDH for disposal of naturally occurring radioactive wastes (NORM) that have been removed from water supplies.

2. After promulgation of the Radionuclide Rule and the regulations for disposal of NORM, the participants should verify the least expensive method for meeting the MCLs. It appears the participants should plan to have the treatment systems in place within five to six years.

3. The costs of the options to remove the radium by treatment and of alternate water supply sources are shown in Table 7.2-1. Based on the costs shown, all of the study participants should construct the recommended treatment option, which is the sodium cation exchange process, to remove the radium from the water.

Table 7.2-1
Total Project Cost Summary

Treatment Options	**Cost ($/1,000 gallons)**
Sodium Cation Exchange	$0.55
Lime-Soda Ash Softening	2.10
Reverse Osmosis	2.36
Point of Use	2.61
Alternate Supply Options	
Lake Brady Regional System I	1.65
Lake Brady Regional System II	1.89
O.H. Ivie Regional System I	3.41
O.H. Ivie Regional System II	3.70
O.H. Ivie Regional System III	5.94
Lake Brownwood	7.59
San Saba Reservoir	3.58

APPENDIX A

The Heart of Texas Water Conservation and Drought Contingency Plan

THE HEART OF TEXAS
WATER CONSERVATION
AND
DROUGHT CONTINGENCY PLAN

INTRODUCTION

The Hickory Underground Water Conservation District No. 1 (District) is the contractor on a municipal regional water supply study to investigate means to reduce radiation levels in area municipal water supplies. The study was funded jointly by the Texas Water Development Board and the eight (8) participating water suppliers: the City of Brady, the City of Eden, Rochelle community, Melvin community, Millersview-Doole Water Supply Corporation, Richland Water Supply Corporation, North San Saba Water Supply Corporation and Live Oak Hills - a subdivision. While the District is the principal administrator of the grant, it has no direct regulatory authority over the participants other than rules and regulations promulgated by the District concerning the use of water from the Hickory aquifer as put forth in Chapter 52 of the Texas Water Code.

The area served by the participating water suppliers is in eastern Tom Green, Concho, McCulloch and San Saba Counties. Service is provided in a 100 mile line east to west from San Saba to just east of San Angelo and in a 25 mile line north and south from the northern edge of McCulloch county to just south of Brady.

Between 1950 and 1980 the area experienced a population loss of nearly 30%. Projections by the Texas Water Development Board to the year 2010 would indicate a stabilization and even a slight growth (.5% - 3%) for the area. Population coupled with projected water use does not show significant increase. Since all entities are dependent on groundwater, it would appear the main concern for all these entities is the continued availability of a dependable groundwater source rather that acquisition of new supplies/sources. Consequently, any Water Conservation and Drought Contingency Plan should reflect these circumstances.

It is the goal of this document to set forth a plan that will include reasonable guidelines that will provide for long-term efficient water use. Even though the size of the participants vary by a factor of 100, there are essentially two (2) types of systems. One is a "loop" system in which water is pumped from wells and distributed to customers and then the waste water is returned to a central waste water treatment plant for final treatment. Brady and Eden are the only entities that have this characteristic. A Conservation and Drought Contingency Plan has been approved for Brady and will be used as a model. The other type of system is a "one way" in which the water is pumped from a well(s) to customers and then disposed at the point of use in a septic system.

SYSTEM ANALYSIS, POPULATION AND WATER REQUIREMENTS

Table 1 provides a summary of each systems wells, capacity and well depth. The City of Brady and Brady Lake are shown separately because they are totally independent. Appendix A1-9 shows the service area for each of the entities.

TABLE 1

System Summary by Entity

ENTITY	WELL #	CAPACITY (GPM)	DEPTH (FEET
City of Brady	1	*	2114
City of Brady	2	*	2100
City of Brady	3	950	2082
City of Brady	4	550	2127
City of Brady	5	1,000	2092
City of Brady	7	1,000	2250
Brady Lake	6	70	2225
Brady Lake	8	390	2460
Eden	1	200	42
Eden	2	400	4160
Eden	3	150	35
Eden	4	150	35
Eden	5	75	35
Eden	6	350	4200
Live Oak Hills	1	45	1230
Melvin	1	195	2500
Melvin	2	170	2480
Millersview-Doole	1	*	2700
Millersview-Doole	2	420	3207
Millersview-Doole	3	410	3305
North San Saba	1	237	3480
Richland	2	310	2640
Rochelle	1	45	2350
Rochelle	2	20*	300
Rochelle	3	20*	300

*Not in normal use - Standby only

As previously mentioned, the area has experienced an out migration in population since 1950. Table 2 shows historic population data for 1950 and 1980 for Concho, McCulloch and San Saba Counties as well as Texas Water Development Board projections for the Area for 1990, 2000, 2010. While there is a modest growth projected for the area by 2010, most of the growth should occur in McCulloch county. It is of interest to note the preliminary 1990 census data indicated exactly the same as 1980 (9735). This should easily lead to the conclusion that the population projections are optimistic.

TABLE 2
Historic and Projected County Populations

County	1950	1980	1990		2000		2010	
			low	high	low	high	low	high
CONCHO	5078	2915	2586	2587	2487	2612	2343	2487
MCCULLOCH	11701	8735	9055	2084	9299	9334	9555	9617
SAN SABA	8666	6204	5510	5605	5733	5979	6047	6363
TOTAL	25,445	17,854	17,161	17,276	17,519	17,925	17,945	18,467

Source: Texas Water Development Board

This limited growth is also reflected in projected requirements for each entity. The largest projected water increase is that of Brady (11.8%) to the year 2010. However, that figure is in jeopardy with the loss of G. Rollie White Downs horse racing facility and Loadcraft Trailer Manufacturing (the largest employer is the City). Table 3 shows projected water requirements incorporating potential water conservation measures:

TABLE 3
Projected Annual Water Requirements

Participant	1990	2000		2010	
		Low	High	Low	High
	------million gallons per day---------				
Brady	1.710	1.682	1.968	1.656	1.934
Eden	0.200	0.210	0.250	0.220	0.260
Live Oak Hills Sub.	0.008	0.008	0.009	0.008	0.011
Melvin	0.028	0.026	0.032	0.025	0.031
Millersview-Doole WSC	0.472	0.462	0.541	0.450	0.533
North San Saba WSC	0.081	0.078	0.093	0.077	0.091
Richland WSC (Brady)	0.064	0.062	0.072	0.060	0.071
Rochelle WSC	0.032	0.031	0.035	0.030	0.035

The nature of use among the six (6) one way systems: Live Oak Hills, Melvin, Millersview-Doole, North San Saba, Richland and Rochelle is primarily domestic. Several customers additionally water stock with the water supplied by the entity. All of the customers are metered. In fact, some customers are metered and paying a monthly minimum to be hooked up to the system even though they are currently, not receiving water. This type of customer is either relying on a previously dug well of questionable longevity or reserving a place for a future home site. Any substantial rate change would cause these individuals to discontinue service which could have two (2) negative results. First, it would create additional financial hardships on the remaining customers. Secondly, they (the individuals leaving the system) would resort to drinking water that is not periodically checked for quality.

Brady and Eden are the only loop systems. Again the majority of water use is household use. However, both entities have industrial/commercial customers that represent a larger-than-average use. Examples for each are: Brady - Roddie Wool Scouring, Brady Independent School District, nursing homes and motels. For Eden large users are: The Eden Detention Center, Eden Independent School District and a motel. All Customers are metered.

PUBLIC INVOLVEMENT

Public access to involvement varies substantially among the entities. On one end Live Oak Hills is a private owned subdivision and does not have regular public meeting. However, due to the small size of the development; it is more likely that operators and customers can have a personal dialogue regarding needs and policies of the system. At the other end is the City of Brady. Brady has regularly scheduled council meetings on the first and third Tuesday of the month. These meetings are subject to open meetings laws; therefore, interested parties have the opportunity to address the council and make their views known. In all cases the circumstances are such that there is a two way communication between the supplier and the customer.

WATER CONSERVATION PLAN

I. EDUCATION AND INFORMATION

Due to the difference in size, projected growth and nature of customer different techniques should be implemented to provide users with information to improve water use efficiency.

New Customer will receive a Texas Water Development Board publication entitled "A Homeowner's Guide To Water Use And Water Conservation". (Appendix B) Brady and Eden may additionally choose to provide a water conservation kit to new customers that includes flow restrictors for shower heads, displacement bags for the water closet and dye tablets for leak detection. If possible low-flow shower heads should be furnished.

Regular Activities should be published in the local paper periodically by Brady and Eden.

Additionally Information will be available to interested parties from the Texas Water Development Board as evidenced in a brochure entitled "Water Conservation - Literature and Material" (Appendix C). This brochure will be on display at the business office for each entity.

Water Conservation literature can be obtained from:
Texas Water Development Board
P.O. box 13231, Capitol Station
Austin, Texas 78711-3231

II. PLUMBING CODES

Only Brady and Eden have the authority to implement water codes. A copy of the plumbing code adopted as part of Brady's Conservation and Drought Contingency Plan is included as a reference (Appendix D). As little population increase is projected for this area, this activity would yield little return for the remaining entities.

III. RETROFIT PROGRAMS

Access to water conservation literature and water conservation kits should help customers understand the benefits of retrofit with water conservation devices. Recently, Brady implemented a sewer charge based on use which should additionally encourage conservation. However, with little increase in population and water use projected, marginal return would likely be realized from this activity for most entities.

IV. WATER RATE STRUCTURES

There is quite a variation in water rates for the entities. Table 4 show the water rates for the eight (8) participants.

TABLE 4
WATER RATES

ENTITY	RATE
BRADY	1st 1000 gallons $ 4.09 over 1000 gallons $ .80/1000
EDEN	1st 2000 gallons $14.00 over 2000 gallons $ .80/1000
LIVE OAK HILLS	1st 3000 gallons $16.00 over 3000 gallons $1.50/1000
MELVIN	1st 2000 gallons $ 2.54 over 2000 gallons $1.27/1000
MILLERSVIEW-DOOLE	1st 2000 gallons $22.50 2000 - 7000 gallons $2.50/1000 over 7000 gallons $3.00/1000
NORTH SAN SABA	1st 4000 gallons $20.00 4000-15000 gallons $1.00/1000 15000-30000 gallons $1.50/1000 30000-45000 gallons $2.00/1000 45000-60000 gallons $3.00/1000 over 60000 gallons $5.00/1000
RICHLAND	1st 1000 gallons $25.00 1000-4000 gallons $3.00/1000 over 4000 gallons $1.75/1000
ROCHELLE	1st 1000 gallons $10.00 over 1000 gallons #2.00/1000

Considering current and projected population, economic conditions and water need; the water rate is probably the most effective tool for encouraging water conservation. The best way to promote conservation is a step rate to increase with water usage such as Millersview-Doole or North San Saba. Another acceptable rate structure is the flat rate in which there is a minimum charge for a certain amount of water. Any additional water has a flat rate. Examples of this type are Brady, Eden, Live Oak Hills, Melvin and Rochelle.

V. METERING

All of the customers receiving water from the entities are metered. Periodic meter testing should be completed according to the following schedule:

Production Meters - Once a Year
Meters > 1" - Once a Year
Meters < 1" - Once Every 10 Years

VI. WATER CONSERVATION LANDSCAPING

Only the Cities of Brady and Eden have the authority to pass regulations regarding efficient use of water through landscaping. The City of Brady indicated in its Water Conservation Plan it intends to sponsor a xeriscape demonstration project. For the remaining entities, water conservation landscaping can be promoted by sharing literature available at their business office.

VII. LEAK DETECTION AND REPAIR

Finding and repairing water leaks is a high priority for all entities involved. Leak detection programs include the following components:

1. Audits during billing for those accounts that have had previous meter service.
2. Constant monitoring of the system for leaks.
3. Visual inspection by employees for situations suggesting a leak.
4. Trained personnel to repair leaks.
5. Information or assistance from the Texas Water Development Board on leak detection.

VIII. RECYCLING AND REUSE

Only Brady and Eden have opportunities for recycling; the other systems do not receive any return flow. Brady has indicated in its Conservation Plan it will visit with certain large users about conversion to recycled water.

IX. IMPLEMENTATION AND ENFORCEMENT

The utilization of an effective water rate structure is the best means of implementing water conservation by the entities involved. Brady and Eden have the authority to pass plumbing codes. Brady has already adopted a code requiring installation of toilets that use no more than 3.5 gallons per flush.

X. CONSERVATION PLAN ANNUAL REPORT

The District has no legal authority over the entities involved likewise, when the Regional study is concluded in April 1991, the group will be dissolved from any formal association. Therefore, it will be difficult to file an Annual Report to the Executive Administrator. However, a copy of the Heart of Texas Water Conservation and Drought Contingency Plan will be retained by the District for future reference.

XI. WHOLESALE CUSTOMERS

None of the entities have wholesale customers.

DROUGHT CONTINGENCY PLAN

I. TRIGGER CONDITIONS

A. LEVEL I - Public water suppliers may declare LEVEL I drought conditions and initiate appropriate contingency plans for existing conditions that include the following:

1. Average daily water consumption reads 90% of system design capacity; and/or
2. Peak period water demand reaches 90% of system design capacity; and
3. Recent and projected climatic conditions are thought to be conducive to excessive eater demands.

B. LEVEL II - Public water suppliers may declare and initiate LEVEL II drought contingency plans for existing conditions that include the following:

1. Water system components including production, storage, pumping and distribution are operating outside of normal design limits due to component failure or diminished capacity; and/or
2. Average daily water consumption approaches 100% of design capacity; and/or
3. Peak period water demand approaches 100% of design capacity; and/or
4. Diminished service to any consumer results in service levels outside of regulatory limits; and/or
5. An imminent threat to water systems components exists to due potential failure.

II. DROUGHT CONTINGENCY MEASURES

Because of the different nature of the two types of systems involves, a variety of strategies will be necessitated.

A. STEP I - all managers should restrict the use of water for outdoor sprinkling, watering of lawns, shrubs, driveways and automobiles. Brady and Eden can additionally restrict water to certain areas by days and to certain houses. This situation will be in effect until the manager lifts restrictions.
All managers should:

1. Inform the public through the media that a trigger condition LEVEL I has been reached and voluntary water reduction should be started.
2. During winter months, request water users to insulate pipes rather than running water to prevent freezing.
3. Monitor water pressure in the distribution system and water levels n the storage tanks.

Brady and Eden should additionally:

4. Notify major commercial water users of the situation and request voluntary water use restrictions.
5. Implement the following lawn watering schedule: Customers with even numbered addresses water on even days. And customers with odd numbered addresses water on odd days. Watering shall occur only between the hours of 9:00 p.m. to 9:00 a.m.

A. STEP II - all managers should totally ban outdoor use for sprinkling, watering lawns, shrubs, driveways and automobiles. Brady and Eden can totally restrict certain commercial and civic activities related to water use. Restrictions will remain in effect until formal notice from the managers.

All managers should:
1. Continue implementation of all relevant actions in preceding phase.
2. Prohibit pavement washing and only allow car and window washing with a bucket.

Brady and Eden should:
3. Prohibit the following water uses that are not essential for public health or safety:
   a. street washing
   b. water hydrant flushing
   c. filling swimming pools
   d. athletic field watering
   e. golf course watering
   f. commercial car/truck washing
4. Adopt a fining mechanism with each day of non-compliance considered a separate offense.

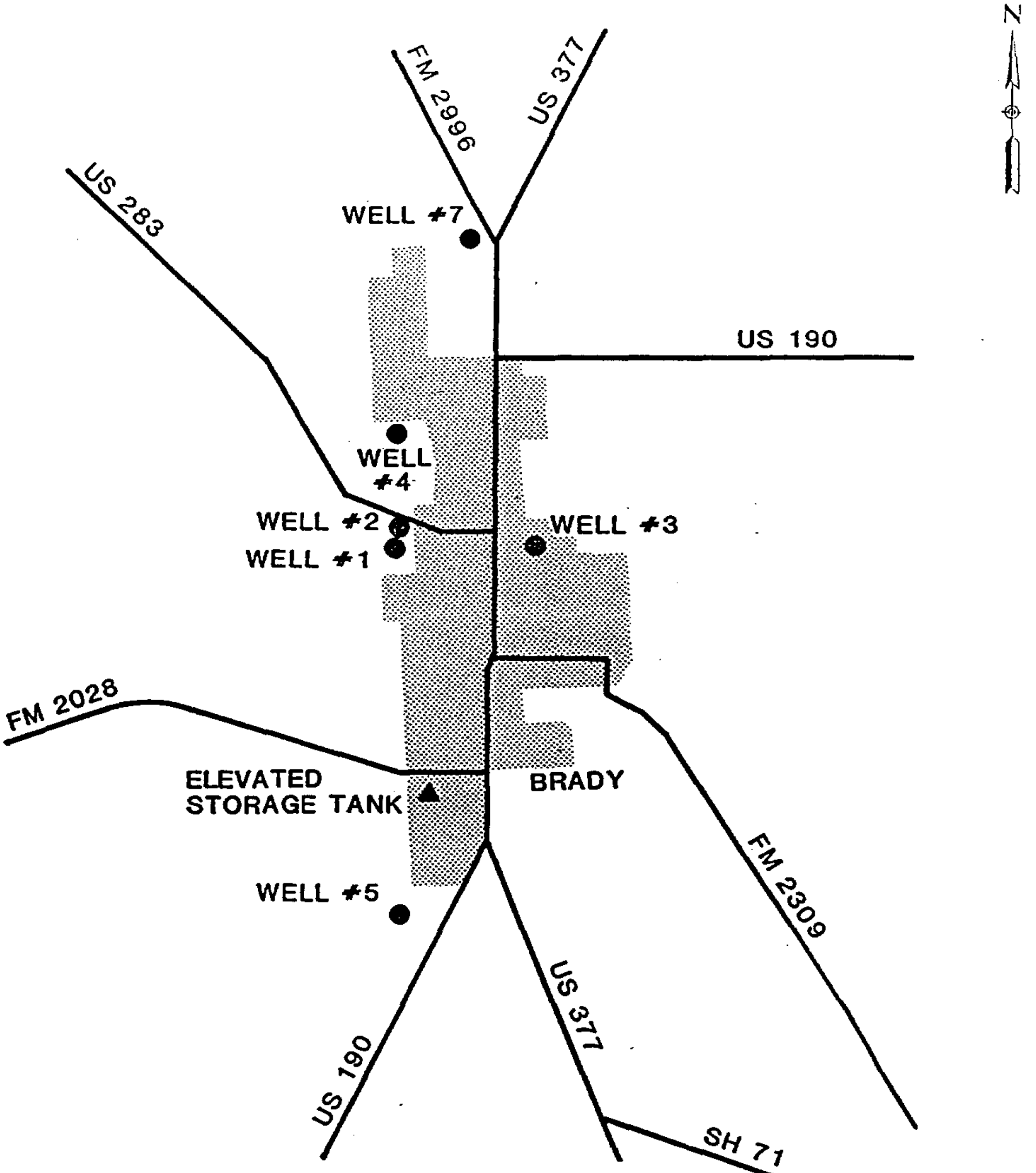
I. INFORMATION AND EDUCATION

Once trigger condition have been reached and emergency measures taken, a variety of information avenues will be taken, including:

A. Posting a notice of drought conditions at key locations such as: business offices, post office, banks and court houses
B. Articles in newspapers of general circulation
C. Public notices over local radio stations

II. TERMINATION NOTIFICATION

Termination of the drought measures will occur when trigger conditions that initiated the drought measures have subsided and an emergency situation no longer exists. Consumers will be informed of the termination of the drought measures in the same manner they were informed of the initiation of drought measures.



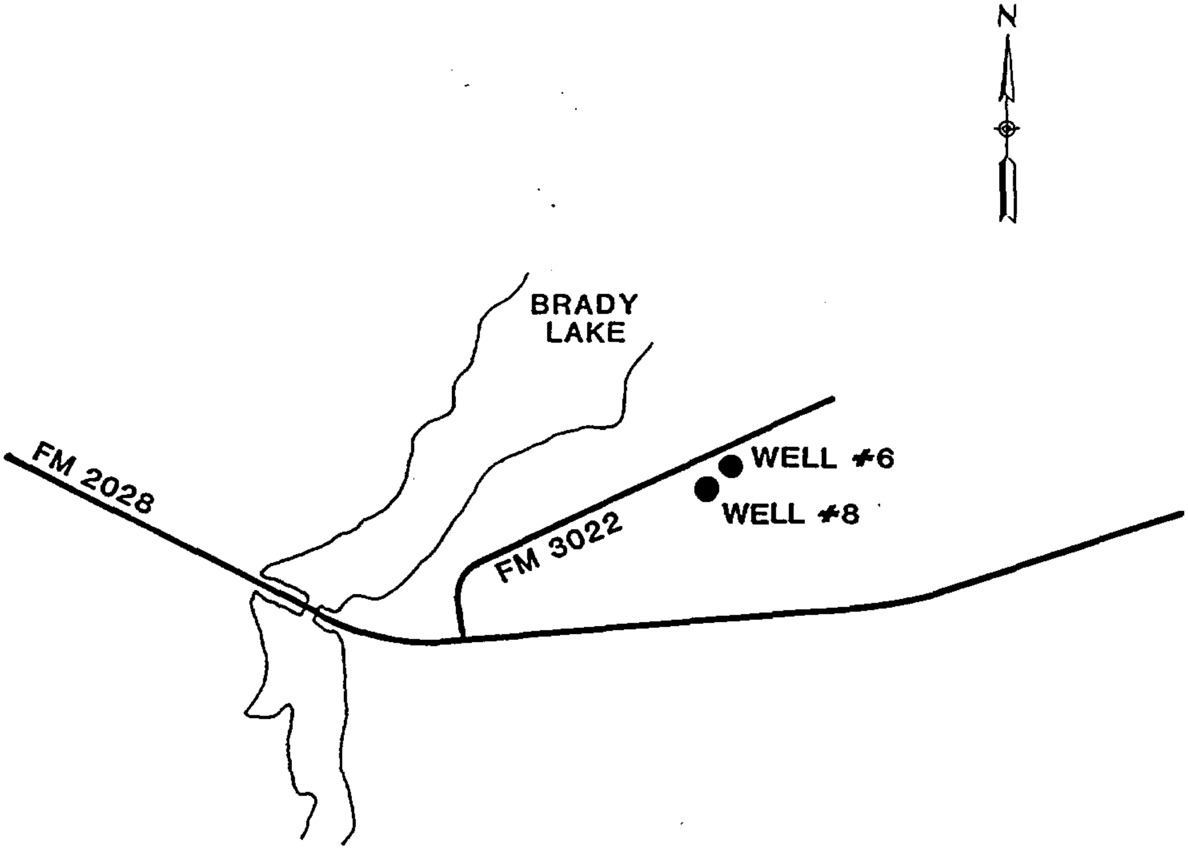
Pump Station No.	Well. No.	High Service	Ground Storage
1	1 & 2	*	*
2	3	1-800 gpm	30,000
3	4	1-1,000	25,000
4	5	1-1,250 1-1,000 1-750	1,000,000
5	7	2-500 gpm	

NOTE:
DISTRIBUTION LINES ARE NOT SHOWN.

HDR Engineering, Inc.

MAJOR DISTRIBUTION SYSTEM COMPONENTS
CITY OF BRADY

Figure A-1



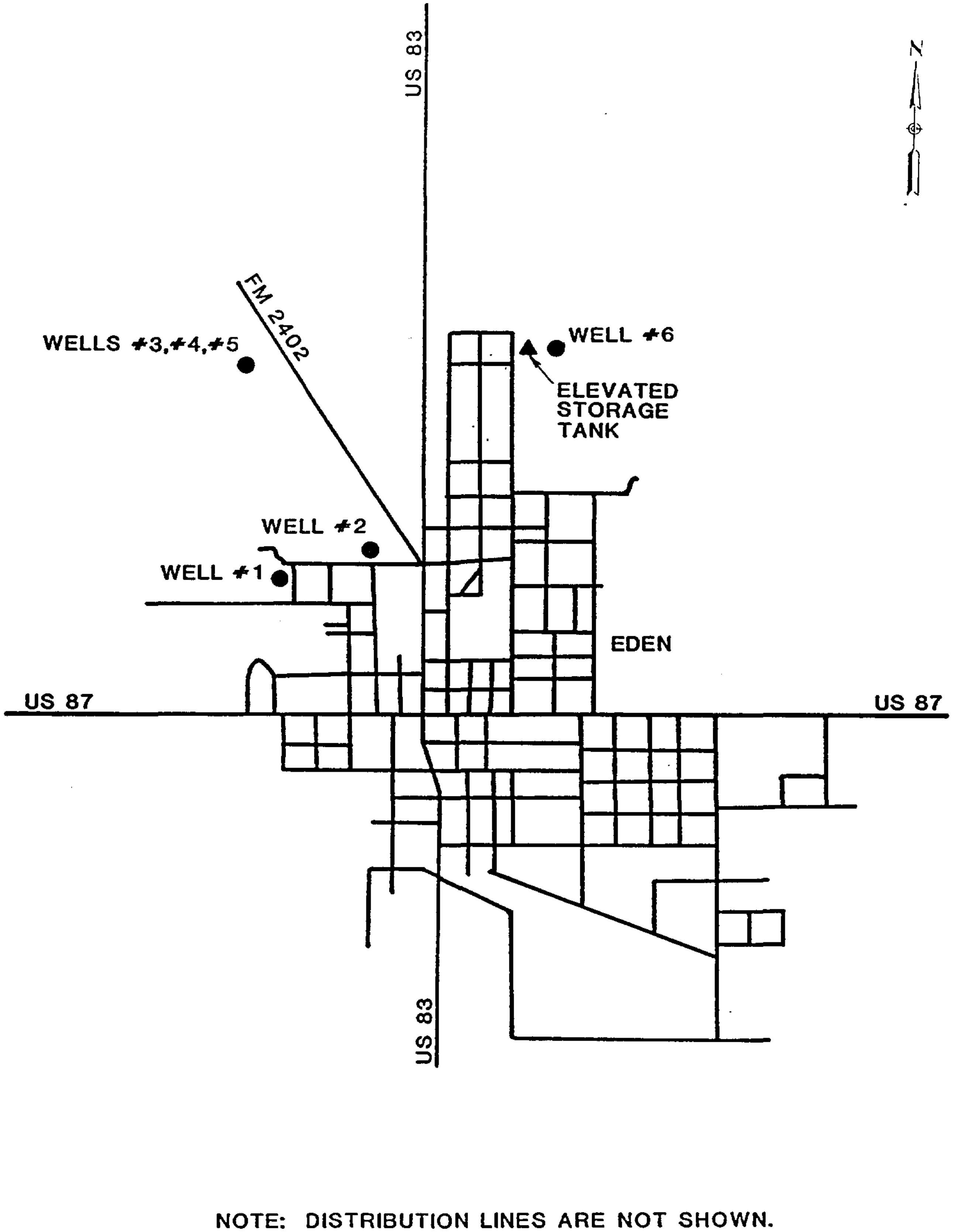
Well No.	High Service	Ground Storage	Pressure Tank
6 & 8	2-172 gpm	50,000	2,500

NOTE:
DISTRIBUTION LINES ARE NOT SHOWN.

HDR Engineering, Inc.

MAJOR DISTRIBUTION SYSTEM COMPONENTS
CITY OF BRADY - BRADY LAKE

Figure A-2

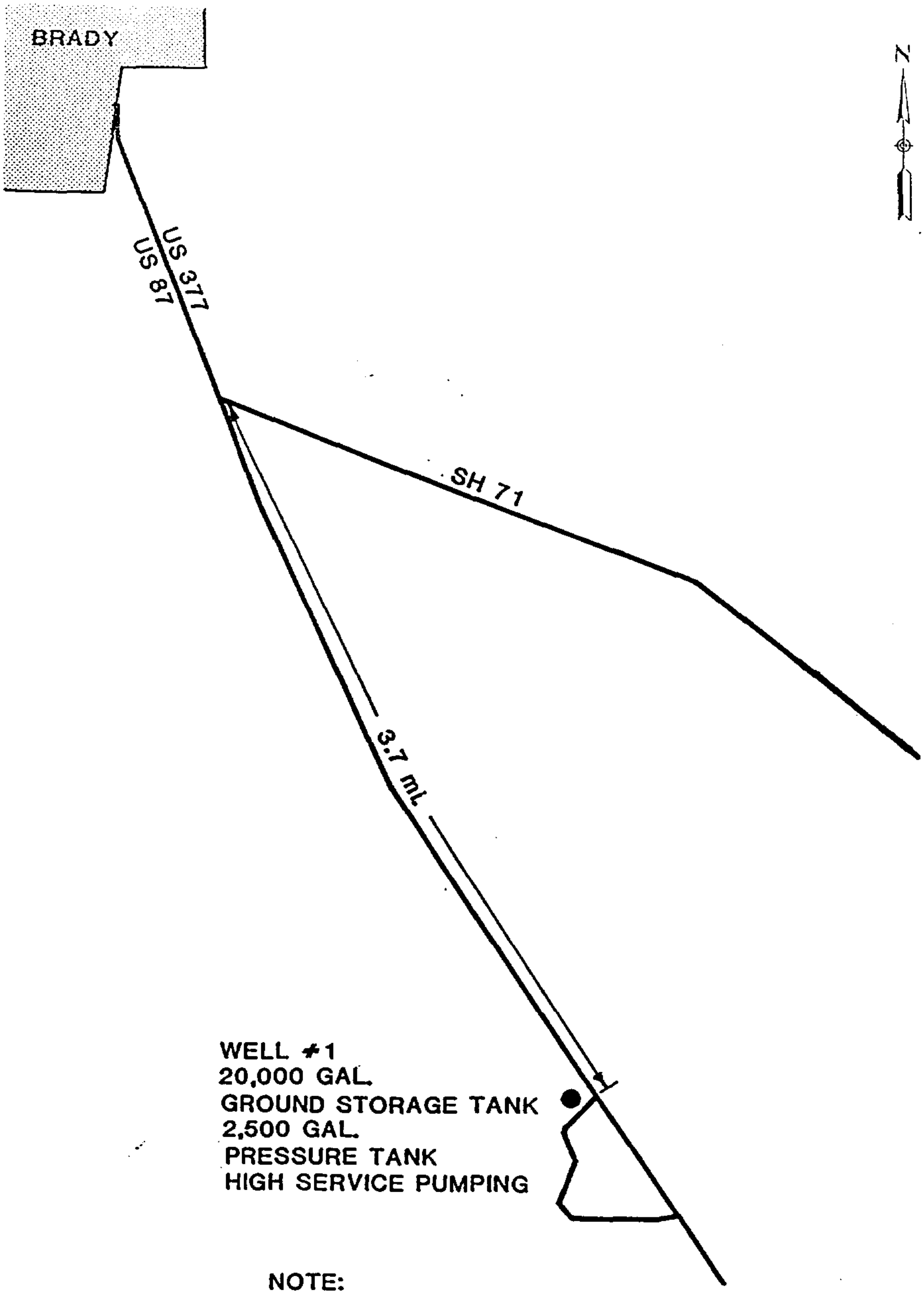


NOTE: DISTRIBUTION LINES ARE NOT SHOWN.

HDR Engineering, Inc.

MAJOR DISTRIBUTION SYSTEM COMPONENTS
CITY OF EDEN

Figure A-3

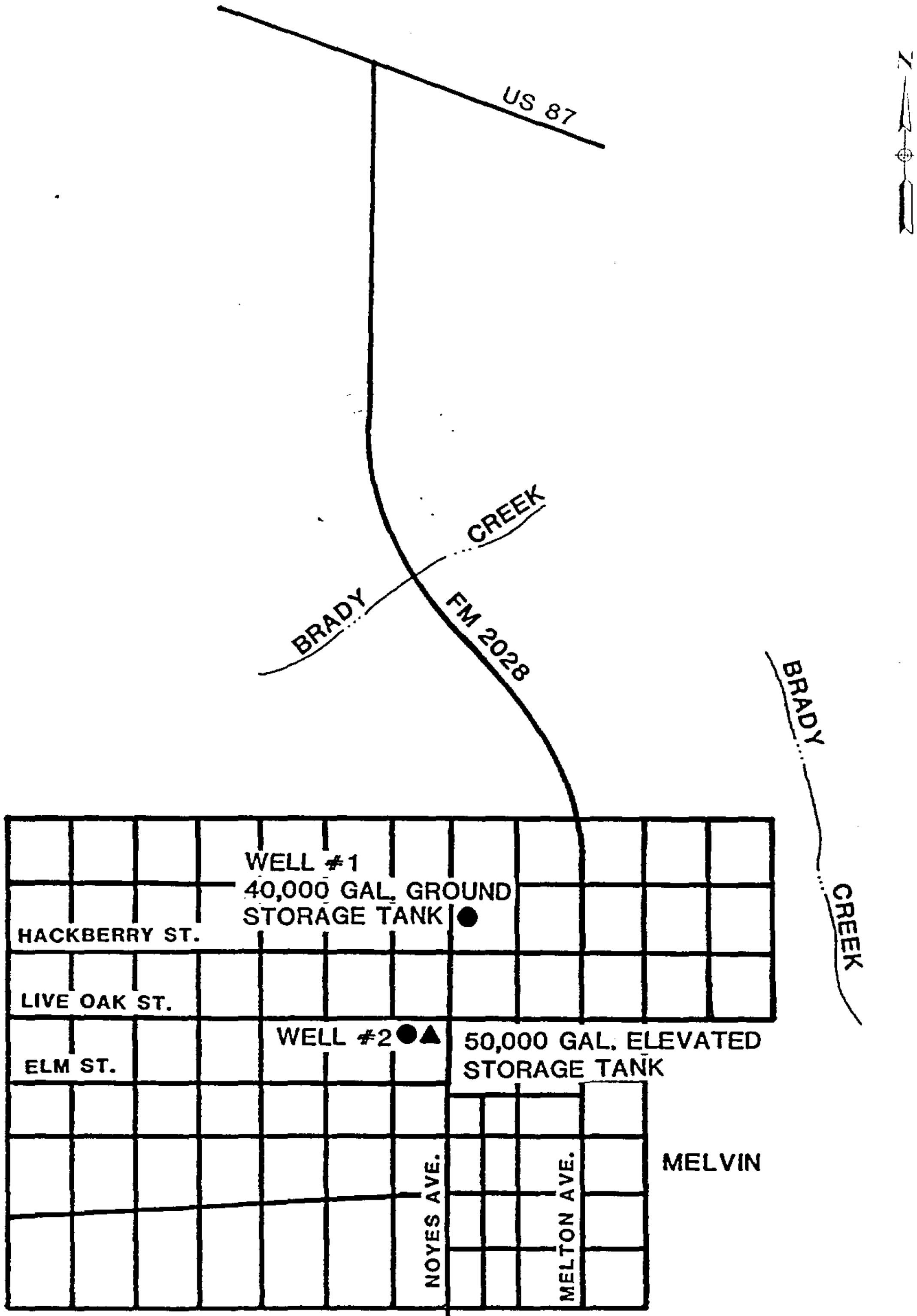


NOTE:
DISTRIBUTION LINES ARE NOT SHOWN.

HDR Engineering, Inc.

MAJOR DISTRIBUTION SYSTEM COMPONENTS
LIVE OAK HILLS SUBDIVISION

Figure
A-4



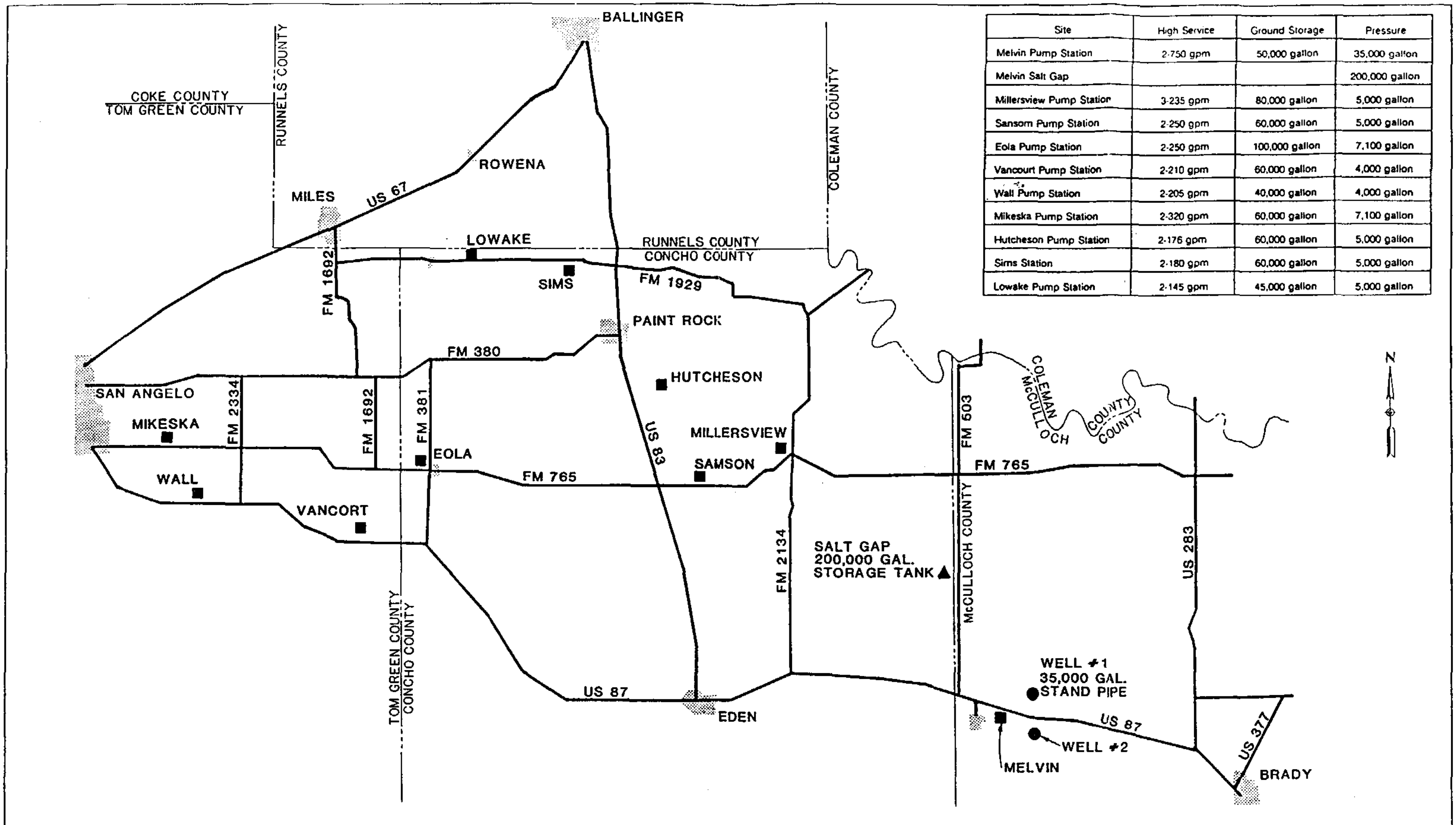
NOTE:
DISTRIBUTION LINES ARE NOT SHOWN.

HDR

HDR Engineering, Inc.

MAJOR DISTRIBUTION SYSTEM COMPONENTS
CITY OF MELVIN

Figure
A-5



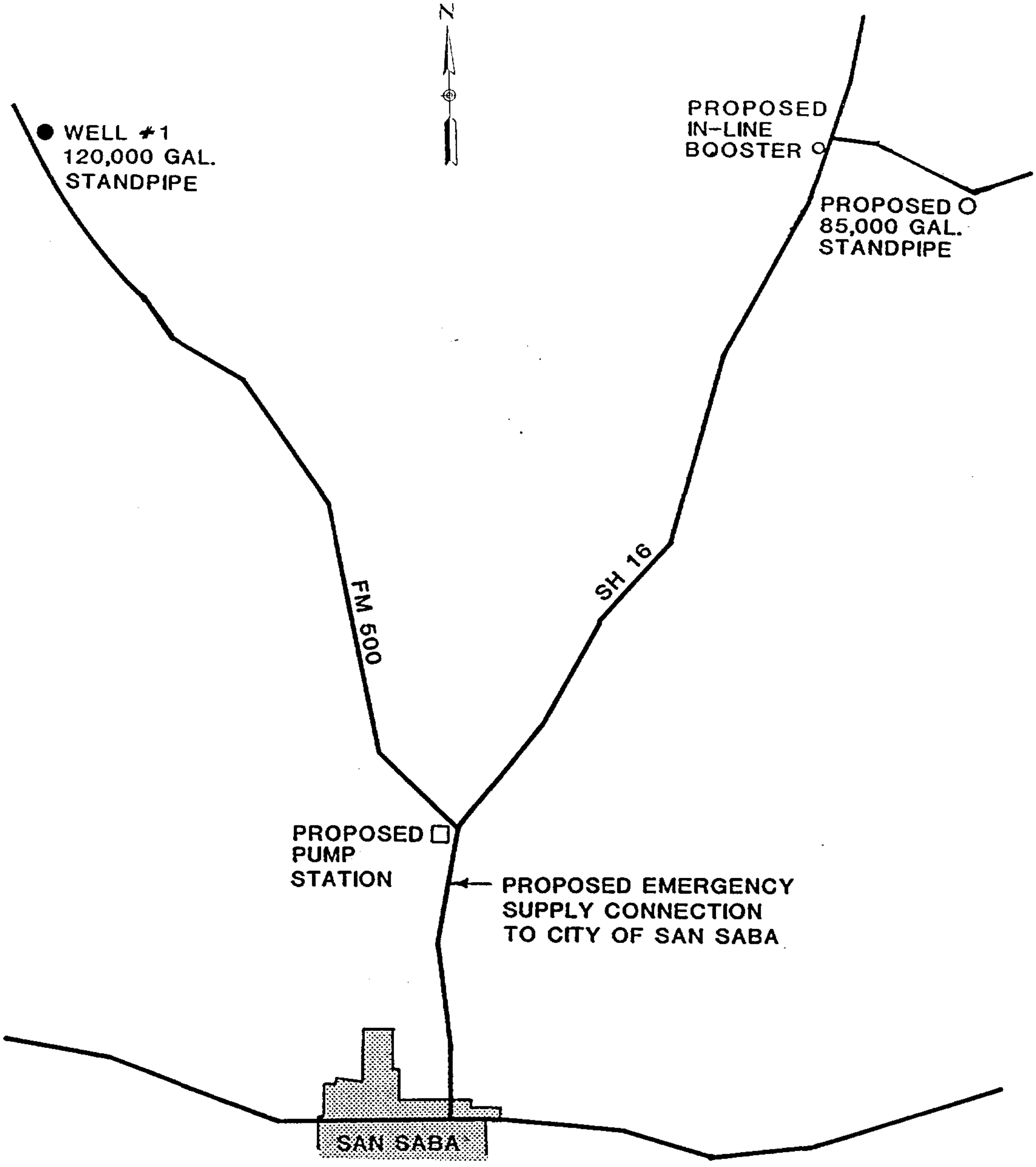
Site	High Service	Ground Storage	Pressure
Melvin Pump Station	2-750 gpm	50,000 gallon	35,000 gallon
Melvin Salt Gap			200,000 gallon
Millersview Pump Station	3-235 gpm	80,000 gallon	5,000 gallon
Sansom Pump Station	2-250 gpm	60,000 gallon	5,000 gallon
Eola Pump Station	2-250 gpm	100,000 gallon	7,100 gallon
Vancourt Pump Station	2-210 gpm	60,000 gallon	4,000 gallon
Wall Pump Station	2-205 gpm	40,000 gallon	4,000 gallon
Mikeska Pump Station	2-320 gpm	60,000 gallon	7,100 gallon
Hutcheson Pump Station	2-176 gpm	60,000 gallon	5,000 gallon
Sims Station	2-180 gpm	60,000 gallon	5,000 gallon
Lowake Pump Station	2-145 gpm	45,000 gallon	5,000 gallon

NOTE:
DISTRIBUTION LINES ARE NOT SHOWN.

HDR Engineering, Inc.

MAJOR DISTRIBUTION SYSTEM COMPONENTS
MILLERS VIEW - DOOLE WATER
SERVICE CORP.

Figure
A-6

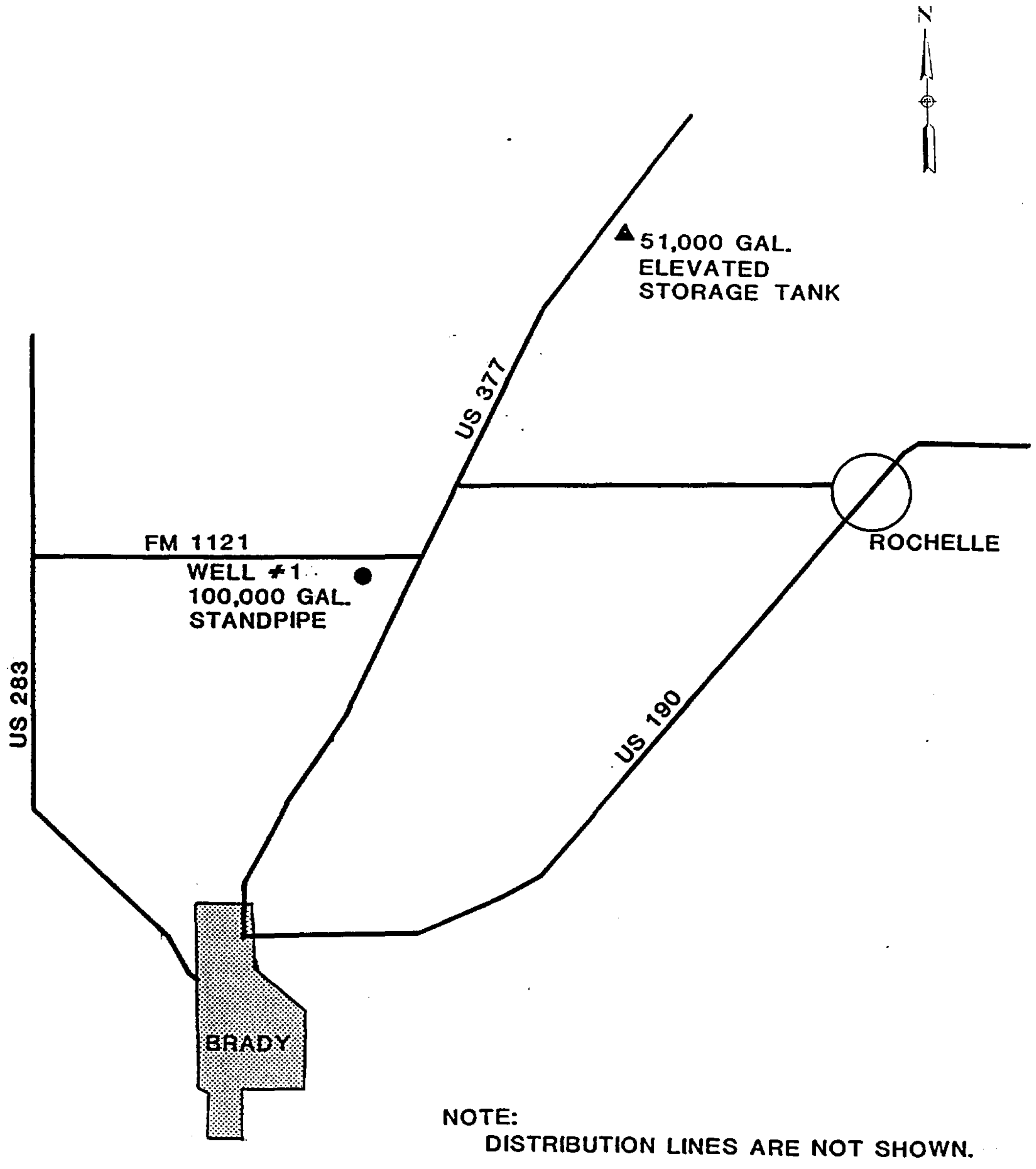


NOTE:
DISTRIBUTION LINES ARE NOT SHOWN.

HDR Engineering, Inc.

MAJOR DISTRIBUTION SYSTEM COMPONENTS
NORTH SAN SABA WATER SERVICE CORP.

Figure A-7

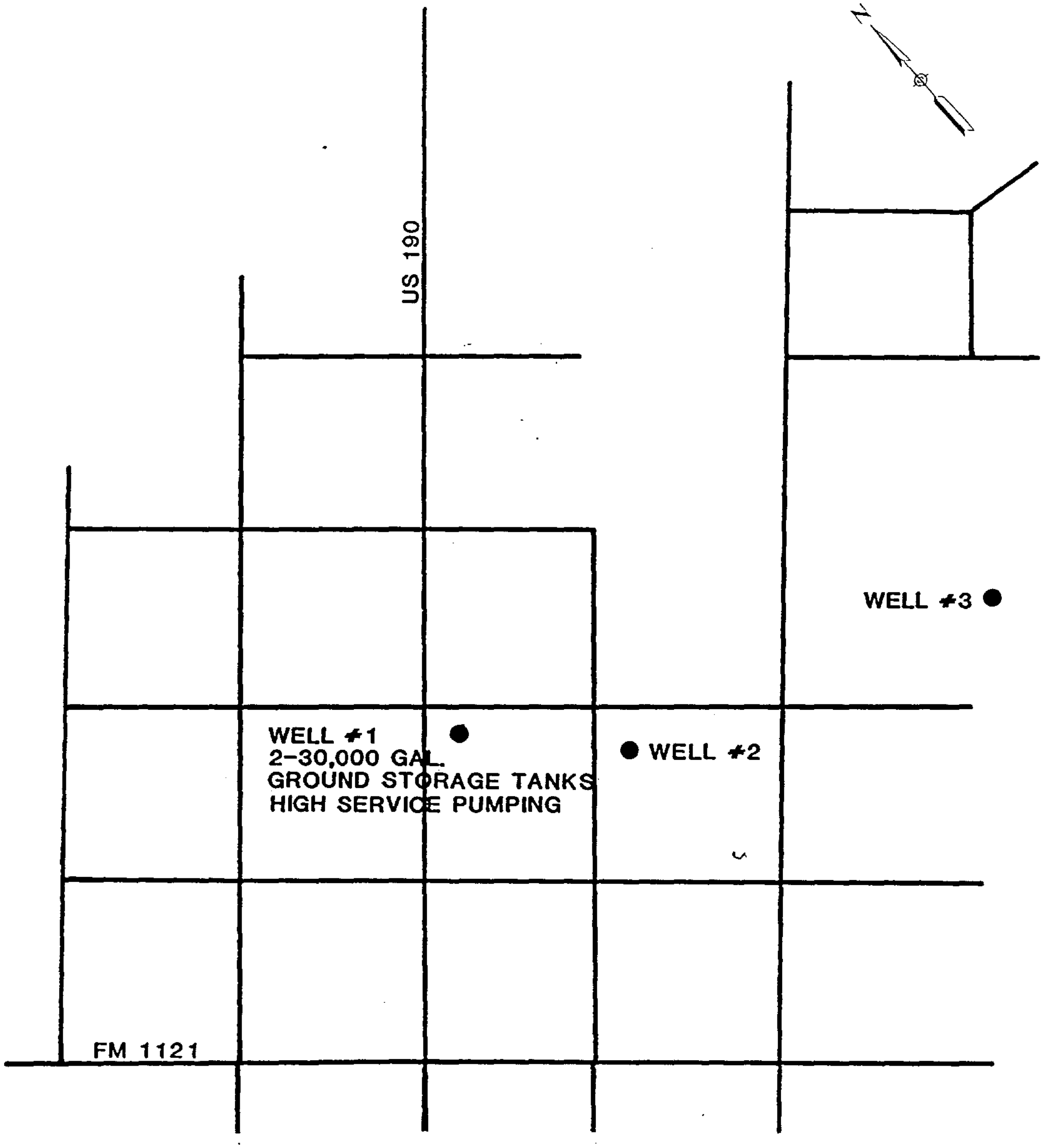


NOTE:
DISTRIBUTION LINES ARE NOT SHOWN.

HDR
HDR Engineering, Inc.

MAJOR DISTRIBUTION SYSTEM COMPONENTS
RICHLAND WATER SERVICE CORP. -
BRADY SYSTEM

Figure
A-8



NOTE:
DISTRIBUTION LINES ARE NOT SHOWN.

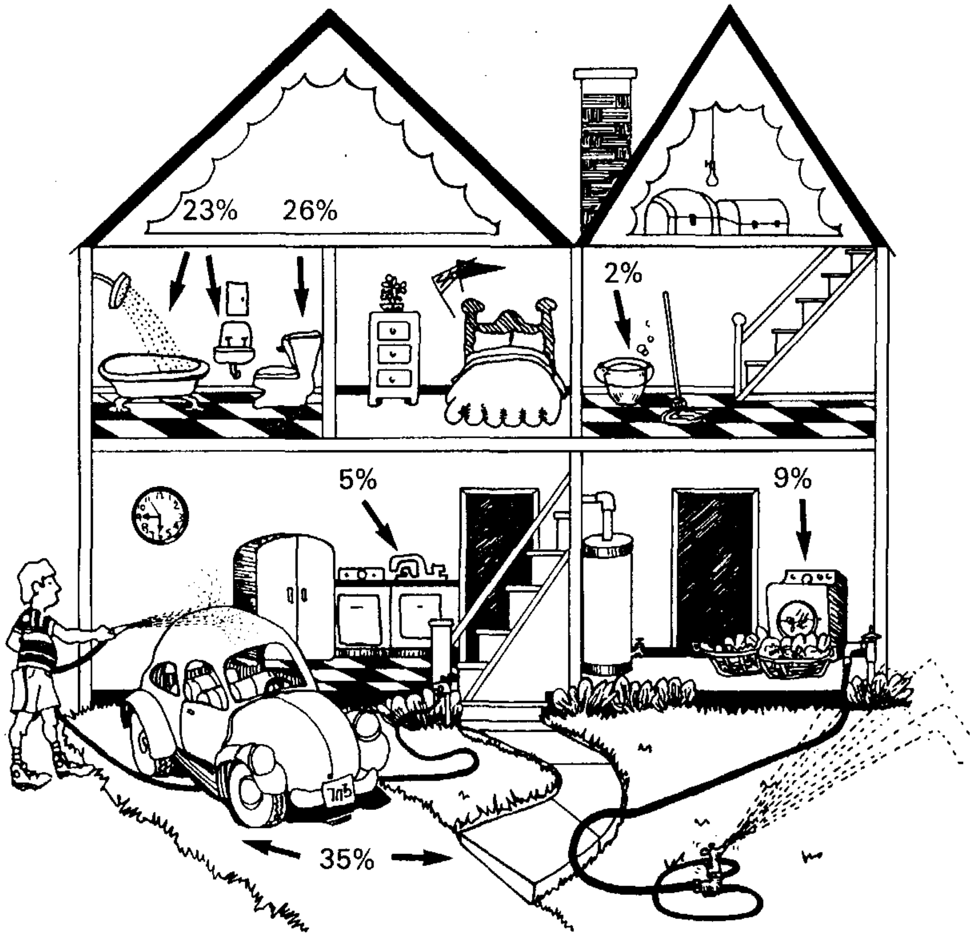
HDR
HDR Engineering, Inc.

MAJOR DISTRIBUTION SYSTEM COMPONENTS
ROCHELLE WATER SERVICE CORP.

Figure A-9

A HOMEOWNER'S GUIDE TO WATER USE AND WATER CONSERVATION

TEXAS WATER DEVELOPMENT BOARD



TYPICAL HOME WATER USE

Appendix
B

Please keep this booklet as a year-round reference for conserving water and saving money.

A copy of this publication may be obtained by writing to:

CONSERVATION

Texas Water Development Board
P.O. Box 13231
Austin, Texas 78711-3231
Published in 1986

A HOMEOWNER'S GUIDE TO WATER USE AND WATER CONSERVATION

TEXAS WATER DEVELOPMENT BOARD

SEPTEMBER 1986

CONTENTS

Page

"Hunger is bitter, but the worst
Of human pangs, the most accursed
. . . is Thirst."

Eliza Cook (1818-1889)
Melaia

". . . by means of water we give
life to everything."

Boethius (c. 470-525)
The Koran

"Water!" . . . "Everywhere you turn here,
you're involved with the lack of water."

Lady Jessica
Dune

INTRODUCTION

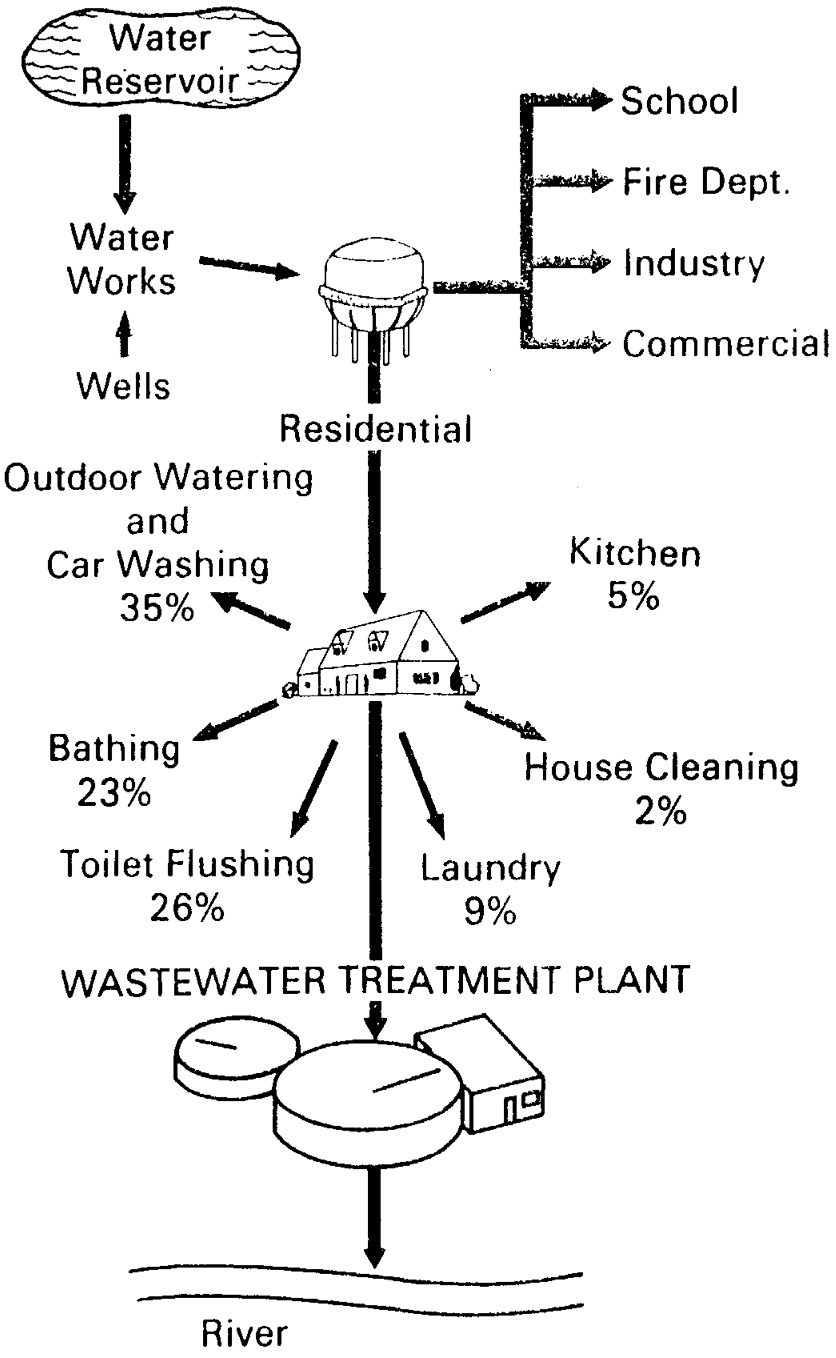
Whether you are a new homeowner or you have lived in your home for many years, the information contained in this booklet can help you better understand your home's water supply and drainage systems. The booklet is designed to provide suggestions on ways to conserve water, thereby saving money, and ways to maintain or even enhance property values.

Water conservation is defined as the efficient use of water so that unnecessary or wasteful uses are eliminated. In Texas, more ground water is drawn out of the aquifers each year than is recharged through rainfall and streamflow. As a result, groundwater levels have dropped dramatically. Demand for water from the state's streams and reservoirs is rapidly approaching the available long-term supply. Across the state, many communities are regularly required to limit water use at some time during the year.

While some additional water supplies can be developed by constructing new reservoirs, in many locations the demand for water will still equal or exceed the available long-term supply. For these towns and cities, water conservation can make the difference between adequate supplies and shortages. Importantly, the way water is used and, in some instances, wasted must be rethought.

This booklet explains seven easy ways that water can be conserved and money can be saved at the same time. Useful information is also provided on how home plumbing systems work, how to measure home water use, how simple repairs can be made, and how water-saving devices are installed.

Water Use in a Typical Community



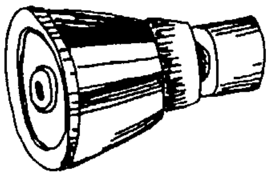
GETTING STARTED— SEVEN EASY WAYS TO CONSERVE WATER AND SAVE MONEY AT HOME

There are dozens of ways to conserve water and save money around the home. As a starting point, seven of the easiest ways to reduce water use at home are listed below:

1. **Install A Water-Saving Showerhead.**

An effective water-saving showerhead should have a flow rate of 3 gallons or less a minute. The standard showerhead uses 5-10 gallons a minute. Therefore, a showerhead using only 2.5 gallons a minute can save as much as 75 gallons of water during a normal 10-minute shower.

To determine whether a low-flow showerhead needs to be installed, the flow rate of the current showerhead should be checked by using the water meter (see page 8) or by putting a gallon container under the showerhead while the water is on and measuring the time it takes to fill the container at the usual shower setting. If it fills in less than 20 seconds, the flow is greater than 3 gallons per minute. Instructions for installing a new water-conserving showerhead are on page 13.

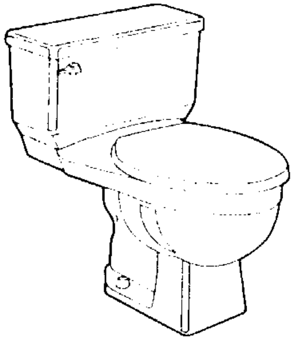


Low-flow showerheads can save money while providing a satisfying shower

Low-flow showerheads can be purchased at most department, hardware, and plumbing stores. Models costing from under $3 are available. A showerhead can usually be installed in ten minutes.

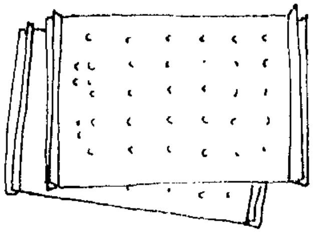
2. **Place Displacement Devices In The Toilet.**

There are three types of displacement devices that can be used in toilets, but they should not be used in newer, low-water use toilets which use less than 3.5 gallons per flush. To determine the capacity of the toilet tank, either turn off the water supply valve to the toilet (usually located on the pipe behind the toilet) *or* hold the float ball up so that the tank does not refill, and flush the toilet. Measure the capacity of the tank by filling it to the normal level with a gallon milk container or other measuring device.

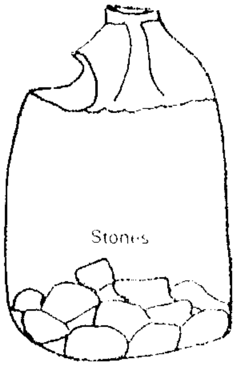


Displacement devices placed in the toilet can save both water and money

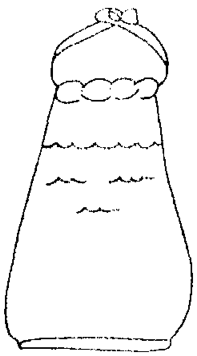
Toilet Dams can be installed in toilet tanks to reduce the amount of water used, typically saving 1/2 to 2 gallons per flush (see page 14 for installation instructions). Toilet dams are available from many utilities or from most plumbing supply stores from under $5 per pair.



Toilet Dams



Plastic Bottle



Toilet Bag

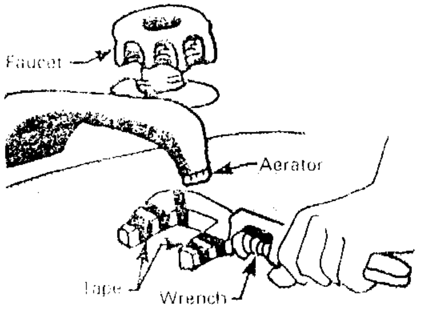
A Plastic Bottle filled with water and weighted down with a few stones can accomplish the same purpose as dams. It is important to place the bottle in the toilet tank where it will not interfere with the flushing mechanism. A plastic bottle in the tank will save 1/2 to 1 gallon per flush.

A Toilet Bag, which is available free of charge from many utilities, can also be used in place of dams. A displacement bag in the tank will save 1/2 to 1 gallon per flush.

Note: Bricks *should not* be used in the tank because small pieces can break off and permanently damage the plumbing system.

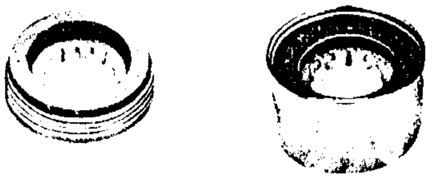
3. Install Faucet Aerators

The standard faucet flow rate is 5 gallons a minute. A low-flow aerator can reduce this flow to approximately 2.5 gallons a minute while still providing adequate water for washing and rinsing. Installing aerators on the kitchen sink and lavatory faucets will save hot water and cut water use by as much as 60 gallons a month for a typical family of four.



To remove an aerator, place adhesive or duct tape on the wrench or plier teeth and turn clockwise

Most aerators have either internal or external threads. Before purchasing aerators, the location of the threads and the diameter of each spout should be determined. If the faucet already has a standard aerator (not a low-flow type), it can be removed and taken to the store to insure that the correct low-flow size is purchased. Aerators are available for less than a dollar from most stores that sell plumbing fixtures.



External Threads

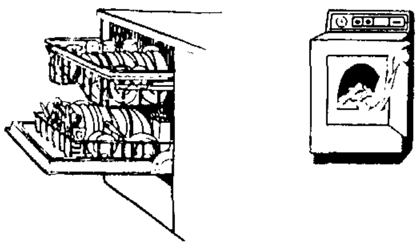
Internal Threads

Faucet aerators reduce the flow by 50%

Note: If the kitchen has a portable dishwasher that must be connected to the faucet, do not install a low-flow aerator since the reduced flow may affect the performance of the dishwasher and result in dishes that are not properly cleaned.

4. Change Use Patterns

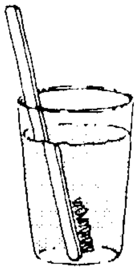
The washing machine and dishwasher should only be used when fully loaded. This practice can save at least two loads or approximately 60 gallons each week.



Run the washing machine and dishwasher only when they have a full load

5. Examine Personal Water Use Habits

For example, by simply changing toothbrushing habits, as much as 14 gallons of water a day can be saved. Instead of allowing the tapwater to run while brushing, use a cup, and run the tap just to rinse the toothbrush. The same method can be used to conserve additional water when shaving, washing hands, or taking a shower. Likewise, when washing vegetables or rinsing dishes in the kitchen, a pan should be used instead of running water constantly.



Use a cup for mouth rinsing when brushing teeth

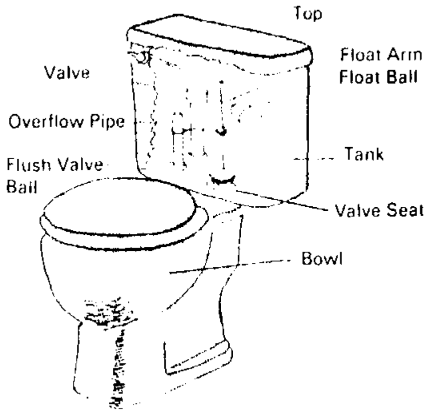
6. Find Water Leaks

According to water utilities, leaks can easily account for 10% of a water bill and waste both water and energy if the source is a hot water tap.

Toilet Leaks: When a toilet leaks, water escapes from the tank into the bowl. Toilets are notorious for hidden or silent leaks, because leaks are seldom noticed unless the toilet "runs" after each flush (which can waste 4-5 gallons a minute).

At home, a silent leak caused by a corroded flush valve can waste as much as 10 gallons of water an hour or over 7,000 gallons a month from a plumbing failure that cannot be seen or heard. For water costing $1.20 per 1,000 gallons, over $100 per year becomes "money down the drain."

To determine if the toilet is leaking, look first at the toilet bowl after the tank has stopped filling. If water is still running into the bowl or if water can be heard running, the toilet is leaking. Oftentimes, however, the toilet may have a "silent leak." To test for a silent leak, mix a few drops of food coloring or place a dye capsule or tablet (available from many utilities and hardware stores) into the water in the toilet tank after the water has stopped filling and the tank is full. DO NOT flush the toilet. Wait for about 10 minutes, and if the dye or food coloring appears in the toilet bowl, the toilet has a silent leak. Leaks of this type are usually caused by a defective flush valve (flapper) ball or a corroded or scaled valve seat. Replacement balls and valves, which can be installed in less than 30 minutes, are available from most hardware and plumbing stores for under $3.

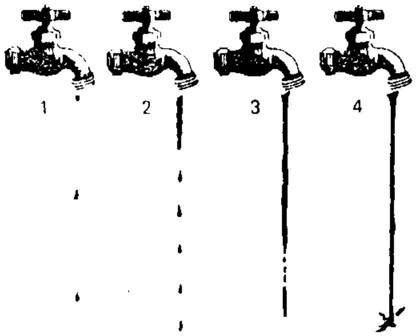


Faucet Leaks: Faucet leaks are obvious. However, seldom used taps in the basement or outside the house should be periodically checked. The cause of faucet leaks is frequently a worn washer that can be replaced with two or three hand tools. Replacement washers can be purchased from most hardware and variety stores and cost only a few cents.

— To determine if leaks that can not be seen exist in the home, refer to page 8 on water meters.

— For instructions on fixing leaks, turn to pages 11-13.

Water Wasted in One Month From Leaks



	gallons/month
1. a slow steady drip (100 drops a minute)	350
2. a fast drip	600
3. a small stream	2,000-2,700
4. a large stream	4,600

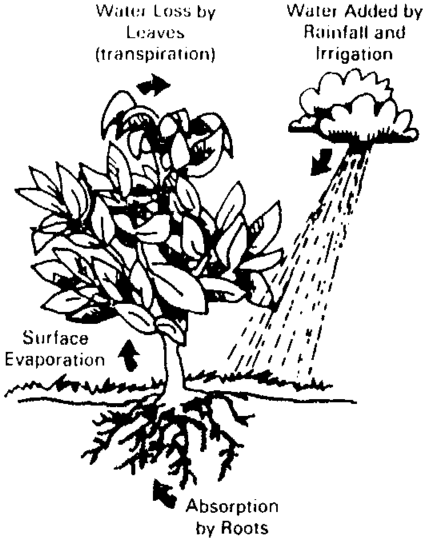
7. Use Efficient Outside Watering Practices

a. Plant native vegetation. Once established, which usually takes two years, these plants require less frequent watering.

b. Water grass only when needed. If grass springs up after being walked on, it does not need watering.

c. Soak grass long enough for water to reach the roots. WATER SLOWLY AND DEEPLY.

d. Water during the cool, early morning hours to minimize water loss by evaporation and discourage disease. Do not water on windy days.



The Plant/Soil/Water Cycle.

e. Use sprinklers that produce droplets instead of mist and that have a low trajectory. This type of sprinkler will lose less water by evaporation and be less affected by the wind.

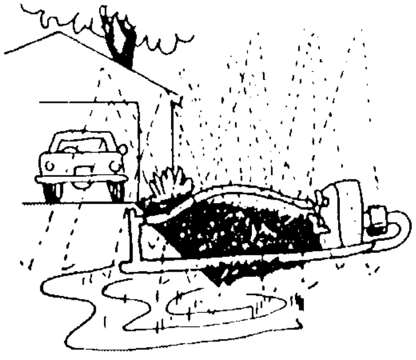
f. Use drip irrigation systems for shrubs, flowerbeds, and other frequently watered areas.

Drip irrigation soaks the root system without any runoff

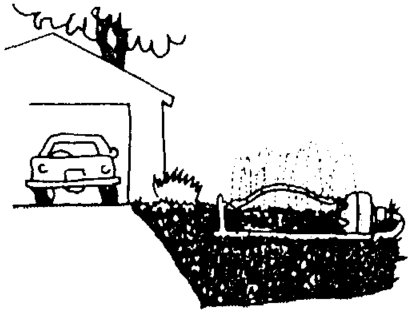
g. During the summer, keep the grass about 2"-3" high. This height shades the root system and holds soil moisture better than does a closely clipped lawn.

h. Do not water streets and sidewalks. Adjust the hose or sprinkler until it waters just the grass or shrubs. For flowerbeds, shrubs, and trees, use drip or soaker systems.

i. If water is rationed during a drought, give priority to trees and shrubs which are more expensive and harder to replace than grass and annual plants.



Incorrect location for the sprinkler = wasted water



Correct location for the sprinkler = no runoff

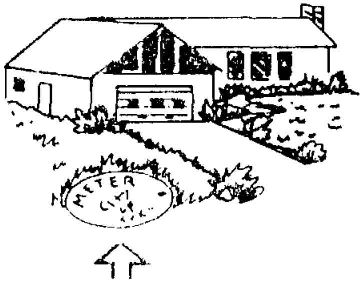
j. During a severe drought when outside watering is prohibited, water plants with "gray water" saved from bathing, dishwashing, and clothes washing, if this is permitted by the city or local health department.

MEASURING WATER USE

How To Read A Water Meter And Use It To Measure Leaks

If the house has a water meter, the entire plumbing system can be checked for an undetected leak in five easy steps:

1. Find the water meter. (It may be outdoors or hidden in a dark corner of the basement).



The water meter is often located in the front yard near the street

2. Turn off all running water and water-using appliances, and do not flush the toilet.

3. Read the dial (or dials) and record the reading.

4. After 15 to 20 minutes, re-check the meter.

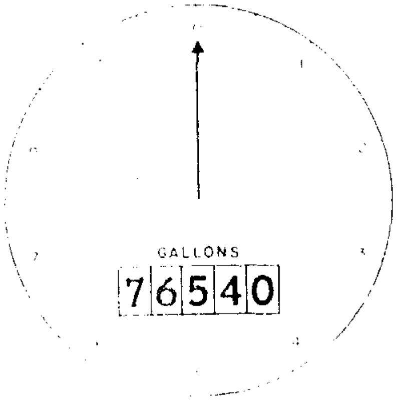
5. If no water has been turned on or used and the reading has changed, a leak is occurring. The rate (gallons per minute) of the leak can be determined by dividing the number of gallons by the elapsed time.

If the leak cannot be found and fixed, a plumber should be called. Before calling, check all toilets for silent leaks by following the instructions on page 6.

Water meters are not all alike. The three shown on these pages are the types most commonly used in Texas. The water bill shows the amount of water, usually in thousands of gallons, used every month. By following the directions next to the meter that matches the one in the home, weekly or even daily usage can be monitored and recorded.

Reading The Type A Meter

A Type A meter should be read like the odometer (mileage counter) on a car. The example meter reads 765400.



TYPE A

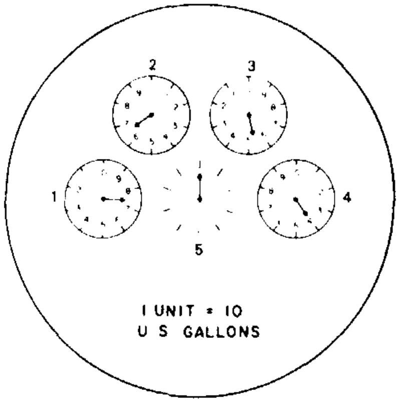
The actual total amount of water used since the example meter started at zero is 765,400 gallons.

If the reading were recorded and a week later the meter read 766500, then 1,100 gallons of water would have been used during the week.

Based on the example meter, the reading on the monthly bill should show 765 if the utility company charges to the nearest thousand gallons used.

Reading The Type B Meter

A Type B meter has five clock-like circles. The circles *are not* numbered 1-2-3-4-5 as in this illustration. Rather, the circles on the example meter have been numbered to show where to begin reading.



TYPE B

Start at Circle 1 and read each circle in clockwise order, finishing with Circle 5. If the arrow points between numbers, read the smaller number.

In this example: Circle 1 = 7; Circle 2 = 6; Circle 3 = 5; Circle 4 = 4; Circle 5 = 0. Therefore, the meter reads 76540. The actual total amount of water used since meter B started with each arrow pointing at zero is 765,400 gallons.

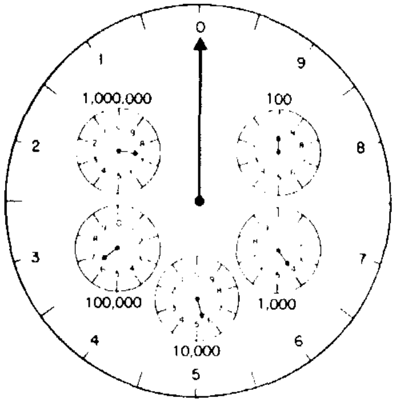
If this reading was recorded and in a week the meter read 76650, then 1,100 gallons of water would have been used that week.

The reading on the monthly bill would be 765 if the utility charges to the nearest thousand gallons used.

Reading The Type C Meter

A Type C meter is read by starting at the circle marked 1,000,000 and finishing with the outside circle. If the arrow points between numbers, read the smaller number.

In this example: Circle 1,000,000 = 7; Circle 100,000 = 6; Circle 10,000 = 5; Circle 1,000 = 4; Circle 100 = 0; Outside Circle = 0. The meter reads 765400. The actual total amount of water used since meter C started with all arrows pointing at zero is 765,400 gallons.



TYPE C

If this reading was recorded and in a week the meter read 766500, then 1,100 gallons of water would have been used that week.

The reading on the monthly bill would show 765 if the utility charges to the nearest thousand gallons used.

How To Read A Water And Sewer Bill

A typical water bill should contain the following information:

1. The amount of water used.
2. The beginning and ending dates of the billing period.
3. The beginning and ending meter readings.
4. The dollar amount being billed.

If the home has central sewer service, the monthly sewer and water charges will almost always be on the same bill.

Water and sewer service are generally billed at a fixed rate for the first 1,000 to 3,000 gallons used and a separate rate per 1,000 gallons after the minimum is used. For example, the water rate for a residential customer could be $8.00 for the first 2,000 gallons used and $1.50 per 1,000 gallons above the 2,000 gallon base amount. The corresponding sewer rate could be $10.00 for 2,000 gallons and $1.80 per 1,000 gallons above the 2,000 gallon base amount.

More than likely, sewer lines serving the home are unmetered. In Texas, many utilities average individual home water consumption over December, January, and February and use the average for these three months, or the actual monthly water use if that is lower, to determine the monthly sewer gallonage for billing purposes. The sewer rates are then multiplied by this gallonage to determine the sewer bill. The reason that December, January, and February are chosen is that all water usage will generally be inside the house during the winter months and, therefore, will flow into the sewer system. Other utilities simply charge a fixed monthly amount for sewer service.

The water and sewer consumption volume will usually be shown on the monthly bill as two or three numbers. When examining household use from a utility bill, make sure that the bill is checked carefully to determine the units (i.e., 1000 gallons/month, 100 gallons/month, or 100 cubic feet/month) that the utility uses as a basis for bill charges. Billing questions can be answered by calling the local utility.

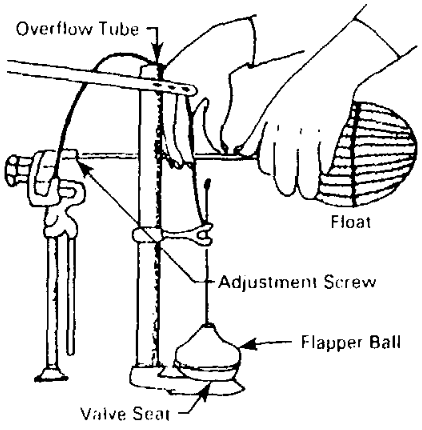
Some utilities have adopted rate structures specifically designed to promote water conservation. Such rate structures usually increase the charge for each 1,000 gallons that are used above the overall average amount of water used each month. For example, the minimum monthly bill could be $8.00 for the first 2,000 gallons, the next 6,000 gallons could be billed at $1.50 per 1,000 gallons, and use above 8,000 gallons could be billed at $2.00 per 1,000 gallons. This structure is called an increasing block rate. Some utilities also use seasonal rates that charge higher rates during the summer out-of-doors watering period. Questions about the rate structure can be answered by calling the local utility.

EASY REPAIRS THE HOMEOWNER CAN MAKE

Repairing Toilet Leaks

Some types of toilet leaks are relatively easy to fix. Other leaks may require the services of a plumber. Several simple repair steps that can be done without contacting a plumber are listed below.

1. If the water is too high in the toilet tank and is spilling into the overflow tube, the float can be adjusted by turning the adjustment screw or by very gently bending the float arm down so that the water shuts off at a level slightly below the top of the overflow tube. Ideally, the water level should be set so that it is about even with the fill line on the back of the toilet tank.



2. A frequent problem that causes a toilet to leak is a worn flapper ball or a flapper ball that does not seat properly into the valve seat. If the flapper ball is worn, it can be removed and replaced with a new flapper ball. If a flapper ball is being replaced, care should be taken to note how the chain is adjusted before the old ball is removed. Also, the valve seat should be checked for scale or corrosion and cleaned if necessary. If cleaning does not work, a retrofit valve seat, which is now available from most plumbing or hardware stores, can be obtained and installed.

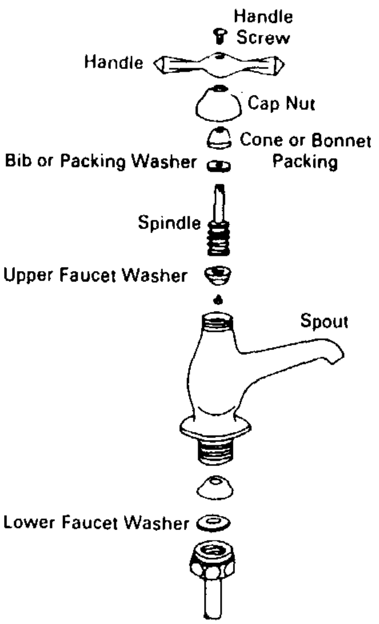
3. If the handle needs to be jiggled to keep the toilet from "running," the guide-wire or the handle itself may be sticking. If the handle is sticking, adjust the nut that secures it in the toilet tank. If that does not work, the handle may have to be replaced.

4. If none of the preceding steps solve the problem, a plumber should be contacted to repair the toilet.

Repairing Faucet Leaks

Leaky faucets, which can develop even in new houses, are wasteful and are a nuisance. With a few simple tools, a leaky faucet can be repaired in less than an hour.

Most water faucets in houses today are compression-type units in which a washer is compressed over a pipe opening when the faucet is closed, thus closing off the water. All compression-type faucets may not look alike, but all are similar in their operation and repair.



The exact point where a leak appears on a faucet is a good clue to finding its cause:

A spout drip is usually caused by a worn upper faucet washer or a corroded seat.

Leaks at the stem result from a loose cap nut or worn cone or bonnet packing.

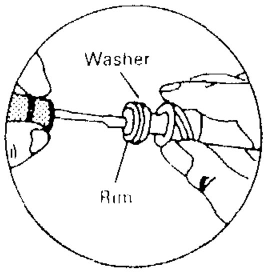
A cap leak, or water oozing below the cap nut, indicates a worn bib or packing washer.

A leak at the base of the faucet results from water seeping past a worn lower faucet washer.

Faucets With Washers:

To repair a leaky compression type faucet, use the following steps:

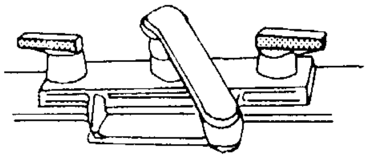
- Turn off the water supply at the valve nearest the faucet. Next, open the tap to drain the faucet.
- Remove the handle screw and lift the handle off the spindle.
- Unscrew the cap nut. It is probably best to use a protective cover of adhesive tape or a rag to avoid marring the finish.
- Unscrew the stem with finger pressure and lift it out.
- Remove the screw from the bottom end of the spindle. Scrape away all worn washer parts. Be careful not to damage the rim.



- Install a new washer. Take either the old washer or the complete spindle unit with you to purchase the correct size and shape (flat or conical) replacement washer. Double-check to make sure the replacement is like the worn washer.
- Check the seat (which is located down inside the faucet) to make sure it is not pitted or rough. If the seat is scarred or rough, a seat-dressing kit to grind the seat to a smooth finish should be used.
- A leak at the stem usually means that the packing inside the cap nut needs replacing. To replace the packing, pry out the old packing washer with a screwdriver. If a washer is used, replace it with a new one. If there is no washer, wrap the spindle tightly with "packing wicking." String can be used if commercial wicking is not available.
- Reassemble the faucet. Tighten the cap nut just enough to prevent leaking. Screwing the nut down too tightly causes rapid wear on the stem.
- Turn the shutoff valves to the on position and check the faucet for leaks.

Mixing Faucets:

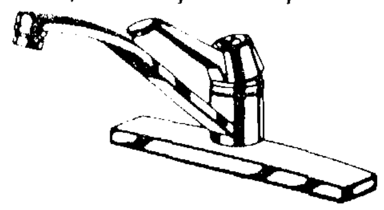
A mixing faucet may look more complicated than a single faucet, but repairs are made in much the same way. Actually, a mixing faucet is two separate units with a single spout. Mixing faucets are used on sinks, bathtubs, and laundry tubs. Repairs must be made separately on each faucet unit. Follow the same steps listed above, but *remember* to turn off both faucet valves before beginning work.



Mixing Faucet

Washerless And Single-Lever Faucets:

Every washerless and single-lever faucet model is a little different. When repairs are required, homeowners can purchase a repair kit for their model which includes instructions and the parts that generally will be worn. By replacing all the parts at once, the faucet should function for several years without needing further repair.



Single-Lever Faucet

WATER-SAVING DEVICES THAT THE HOMEOWNER CAN INSTALL

Installing a Low-Flow Showerhead

In some cases, a newly purchased low-flow showerhead may not fit on the existing pipe. This is often the case if the neck ends in a ball joint. Most necks can be unscrewed and replaced, or an adaptor can be used. Adaptors or replacement necks are readily available in plumbing and most hardware stores.

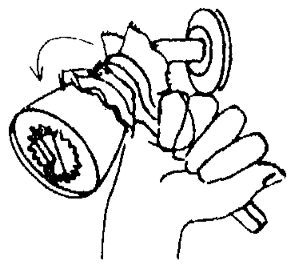
When installing the new showerhead, teflon tape or pipe joint compound should be applied to the exposed threads of the new neck so that the joint will be sealed and provide a leak-proof connection.

If the shower neck that has been on for a few years and the neck joint is too stiff to unscrew with moderate pressure, consider having a plumber replace it.

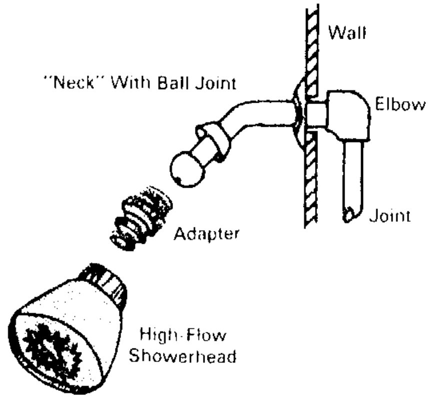
Installing Shower Flow Restrictors

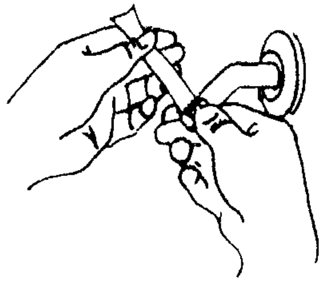
Restricting devices fit into the space between the showerhead and the shower neck. These devices range from a simple washer with a small hole in it to small chrome-plated pressure compensating fittings. While restrictors reduce water flow, low-flow showerheads produce a flow that is more satisfying to most individuals.

To Change Showerheads:

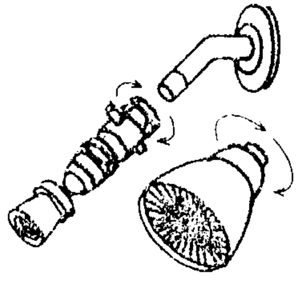


1. Remove old "high-flow" showerhead . . . turn counter-clockwise





2. Cover exposed threads of "neck" with pipe joint compound or teflon tape to form a tight seal



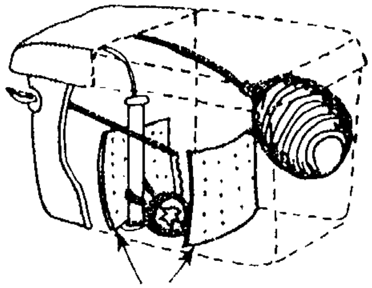
3. To install the "low-flow" showerheads screw onto the "neck" clockwise

Installing Toilet Dams, Toilet Bags, and Plastic Bottles

Many of the toilets in homes today use from 5 to 7 gallons per flush. Several effective retrofit devices are available that can reduce the volume of water used with each flush by 2 to 3 gallons. While these devices can be used in some of the low flush toilets on the market today that use from 3 to 4 gallons per flush, they generally perform better when used in the older 5 to 7 gallon per flush models. Since the height of water in the toilet tank (and not just the volume) causes the bowl to flush, the purpose of displacement devices is to reduce the volume of water used in each flush without affecting water height in the tank. The following three types of devices have proven to be effective:

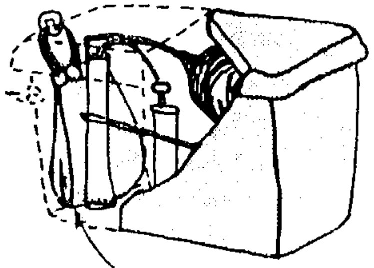
Toilet Dams:

To install a dam, flex or bend the dam and insert it into the tank as illustrated. The dam should fit tightly against the tank sides and bottom and should curve outwards away from the plumbing fixtures in the tank. Most tanks work best when a single dam is used.


Toilet Dam

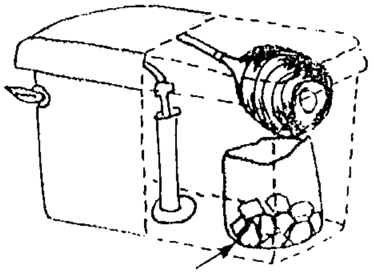
Toilet Bags:

The bag should be filled with water, the top of the bag should be securely sealed with the bag clamp, and the bag should be hung in the tank by using the bag clip and hanger that are provided. Make sure the bag is located in the tank so it does not interfere with the operation of the toilet tank.


Toilet Bag

Plastic Bottles:

Plastic bottles can be filled with water and weighted down with a few stones and placed in the most open portion of the tank. This will save as much water as is displaced by the bottle. Make sure the bottle does not interfere with the operation of the moving parts of the tank.


Plastic Bottle

ALL SUCH DEVICES SHOULD BE CHECKED PERIODICALLY TO INSURE THAT THEY REMAIN IN PLACE.

GETTING TO KNOW THE PLUMBING SYSTEM

Draining The Water Supply System

There are two types of plumbing systems in every house. There is a water supply system and a drainage system.

Occasionally, the water system may need to be drained to perform major plumbing repairs or to prevent the pipes from freezing if the house is to be left unattended for long periods in cold weather.

To drain the water from the pipes, follow these steps:

1. Turn off the water supply at the main shut-off valve to the house. The shut-off valve is usually located in a valve box which is buried in the ground between the water meter and the house. The plate covering the valve box can often be removed by hand or with a screwdriver.

2. Turn off the gas or the electric current to the hot water heater. This is ***extremely important*** because the heater will be seriously damaged if the water is drained while the heater is operating.

3. Next, flush all toilets and open every faucet in the house.

4. If your house is heated by a hot water or steam system, turn off the furnace and open all radiator valves (if the radiators have individual valves). Remove the air-escape valve from one or more radiators on the highest floor of the house and drain the furnace and water heater.

5. Open the drain valve on the main supply line. If no valve exists, determine where the lowest point in your system is and disconnect a fitting there to allow any remaining water to run out of the pipes. A plumber may be needed to install a suitable drain valve in plastic or copper piping.

6. If the house is supplied from a well, drain both the above-ground pump lines and the tank. Turn off the pump and drain it completely.

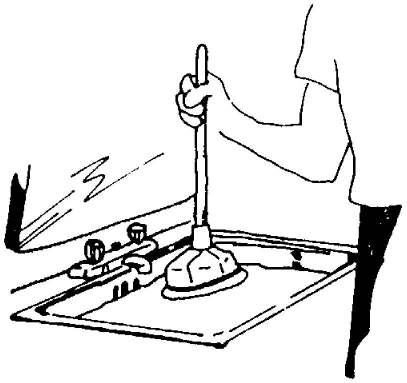
7. Pour a mixture of anti-freeze and water into the trap of every sink, wash basin, bath tub, and shower stall and into the toilet bowls. Prepare this mixture, according to the directions on the can, for the lowest possible temperature expected in the local area.

Unclogging The Drains

If water backs up into any of the household fixtures, the drain is probably clogged at some point. To unclog the drain, the following steps should be followed.

1. Bail or siphon as much water as possible out of the fixture.

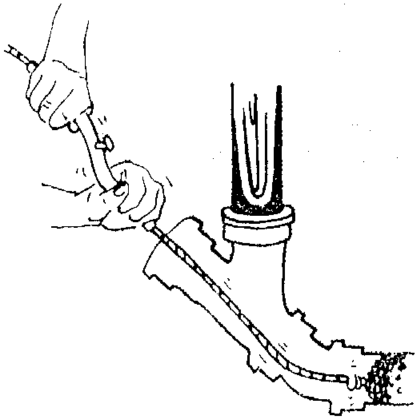
2. Use a plunger (plumbers helper) to try to unclog the drain by placing the plunger over the clogged drain and working it up and down vigorously. If that does not work and the fixture is a


Use a plunger to break up line blockages

bathroom sink or bathtub, a hand, a wet sponge, or duct tape can be placed over the overflow vent to seal it off, and the plunger action can be repeated.

3. If Step 2 does not work, pour a caustic drain cleaner into the drain. The manufacturer's instructions should be followed completely. Allow the product to set for the recommended time and then run cold water down the drain.

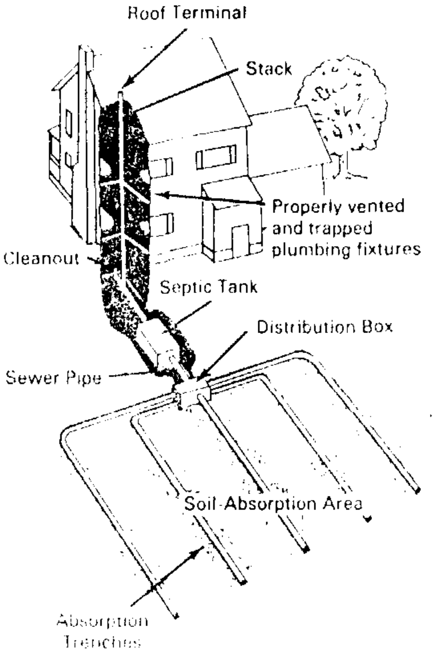
4. If Step 3 does not work, it is time to either call a plumbing service or try to clean out the line without help. To clean a line, the point of line blockage will need to be determined, the cleanout will need to be located, and a drain auger will need to be used to clear the blockage.



Use a drain auger to clear a line blockage

Septic Tanks And Drainfields

Houses in rural areas or on the fringe of an urban area without centralized sewage system use a septic system to dispose of sewage. A properly designed septic system usually consists of a septic tank and a drainfield. By reducing water use, the life of the drainfield can be prolonged.



A typical septic system (with trenches)

Maintenance

A few basic steps can prevent expensive repairs and lengthen the life of the system.

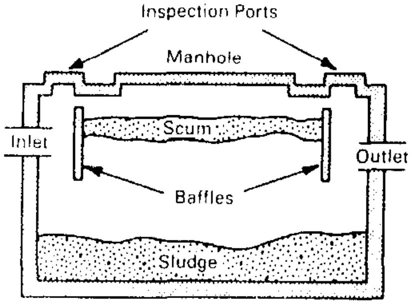
— Follow the water conservation methods discussed in this pamphlet. This will help prevent saturation of the drainfield.

— Do not use a garbage disposal system in the sink and do not use the toilet as a trash can. Large amounts of organic matter can cause a rapid buildup of sludge or scum which will require more frequent cleaning and could cause the septic system to fail.

— The use of bacterial additives, yeast, and other "activators" can also help septic system performance.

Cleaning

The frequency of cleaning depends on the size of the septic tank and the number of people it serves. With ordinary use and care, a septic tank may require cleaning every two or three years. In many cases, septic tanks can be satisfactorily operated for even longer without cleaning.



Cross Section of a Septic Tank

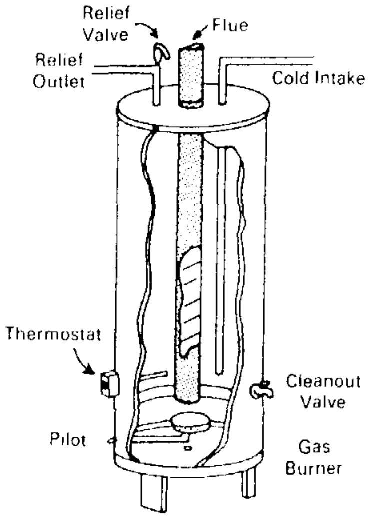
To determine when to clean the tank, an annual check should be performed. When the bottom of the scum is within 3 inches of the bottom of the outlet baffels or the top of the sludge is within 3 inches of the bottom of the outlet, the tank should be cleaned. The accumulated solids should be pumped out by a licensed, commercial septic tank cleaning company.

The Hot Water Heater

Hot water is a simple luxury that is often taken for granted. Knowing about a water heater can help trouble shooting when problems occur and could prevent more serious problems should the water heater fail.

Conserving Energy

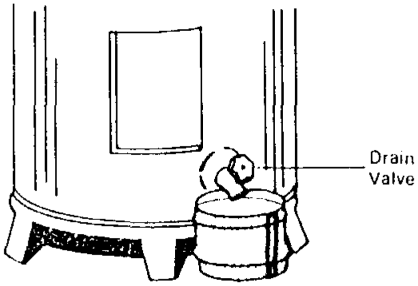
The thermostat controls the temperature of the hot water. Try setting the thermostat at around 125° Fahrenheit unless the kitchen has an automatic dishwasher. The lower temperature will save energy and wear and tear on the equipment, since hot water is more corrosive to pipes. The lower temperature will also protect small children from accidental scalding. If there is an automatic dishwasher which does not have a water heater option, the thermostat should be set at a minimum of 140° F so that the dishwasher will clean effectively.



The hot water heater will use less energy if it is wrapped in an insulating blanket. The hot water pipe on top of the tank should also be wrapped in pipe insulation. The cold water pipe should be insulated up to the shut-off valve.

Maintenance

Approximately every six months, about two quarts of water should be drained from the faucet-like drain valve located at the bottom of the tank. To drain the tank, either attach a garden hose and flush water outside or down a nearby drain or catch the water in a bucket. Either way, water should be drained until it runs clear. Periodic draining will prevent an accumulation of sediment and improve the efficiency and life of the heater. Also, the heater may make less noise as a result.



Every water heater should have a pressure relief valve to prevent a buildup of pressure from occurring because of a malfunctioning thermostat. Test the valve by lifting the test lever on the valve. If water runs out, the valve is functioning.

Check the turn-off valve located on the cold water intake line to the heater to be sure it closes smoothly. This valve is used to shut the cold water off to the heater and hot water off to the rest of the house for repairs or in an emergency situation.

WATER SAVING IDEAS

When Remodeling or Replacing Fixtures and Appliances

1. Hot Water Pipes:

Where possible, hot water pipes from the hot water heater to fixtures and appliances should be insulated during remodeling. This will reduce the time between turning the water on and the time hot water comes out of the faucet and reaches a constant temperature.

2. New or Replacement Fixtures and Appliances:

Low-water using fixtures should be installed when remodeling or replacing fixtures.

a. Toilets: Install toilets that use 3.5 gallons or less per flush.

b. Showerheads: Low-flow showerheads should flow at 3 gallons or less per minute.

c. Appliances: Water-saving dishwashing and clothes-washing machines should be installed. Be sure to ask the salesperson about the water efficiency of appliances when shopping for replacement appliances.

3. Pools and Hot Tubs:

a. Run the filter backwash onto the lawn rather than down the sewer.

b. Purchase covers for hot tubs and pools to reduce evaporation losses.

When Adding New Landscape Or Redesigning The Yard

When planning to add new landscaping or redesign existing landscaping, the following suggestions should be considered because it may be possible to save 50% or more of the water needed to maintain a traditional lawn.

1. If a landscape architect or gardener is to be hired, select one who is experienced in Xeriscape.

WHAT IS XERISCAPE?

XERISCAPE (ZIR-I-SKĀP) N. The conservation of water and energy through creative landscaping. (From the Greek word Xeros, meaning "dry".) Xeriscaping, Xeriscaped, Xeriscaper.

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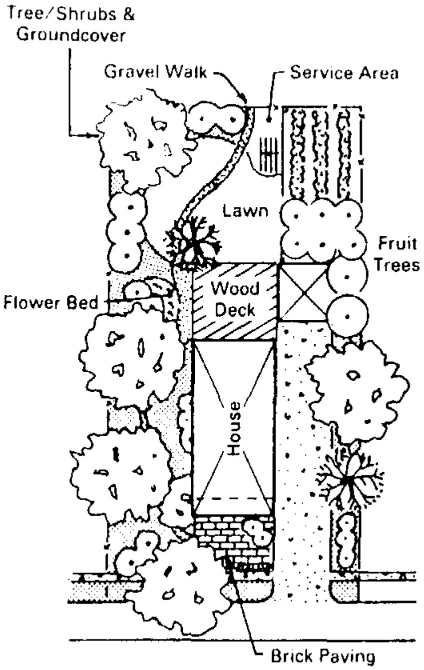
2. Design the yard to reduce the grassed areas to only that amount which will actually be used for recreation and entertainment. Front and side yards are most frequently just for show and are logical areas that can be completely or partially converted from lawns to native grasses, groundcovers, and shrubs.



A Xeriscape design for the yard

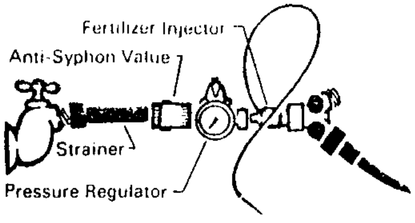
3. Use native grasses, groundcovers, shrubs, and trees. There are many beautiful varieties of native species that can be used in landscaping that are preferable to imported species. The advantage of native species is that, once they are established (usually about two years), they do not need to be watered as frequently (about 1/3 to 1/2 as often), and they usually will survive a dry period without any watering.

4. Irrigation for Lawn, Shrubs, and Trees:

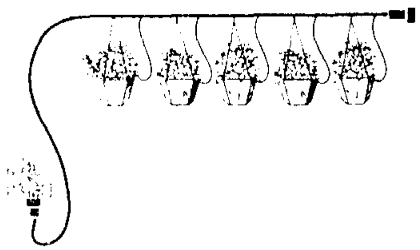
a. When installing an irrigation system for the lawn, the sprinkler heads should be low-angle spray heads that sprinkle the grass without spraying the water high in to the air or allowing the water to drift onto the sidewalks and streets. The heads should produce droplets of water instead of a mist.

Drip Irrigation Hose
and Emitters

b. The preferable irrigation system for shrubs beds and trees is a drip-type system. There are



Typical drip irrigation arrangement. The fertilizer injector and pressure regulator are optional.



Drip irrigation systems can be used for hanging baskets

several varieties, including soaker hoses, bubblers, and "leaky pipe." If a sprinkler system is installed for shrubs, an upright pipe extension may be needed if low-angle spray heads are to be used. This is done to spray evenly without obstructions. For more information, contact a licensed landscape irrigator or a reputable dealer.

c. Automatic controls will allow the proper watering time and will minimize waste.

d. Regular spacing between spray heads will provide uniform coverage.

5. Soil: Shape the soil to protect against erosion and use conditioners to promote water penetration and retention.

a. Shape the soil into earthen basins around all shrubs.

b. If the original soil is rocky, shallow, or a heavy clay, the soil can be improved by adding 2 to 4 inches of organic material or topsoil which is compatible with the soil type.

c. If your original soil is a heavy clay or fine sand, the soil can be improved by tilling organic material such as peat, compost, decomposed rice hulls, and rotted manure into the soil.

6. Watering:

a. Plants: During summer months, most plants will need about 1 inch of water every 5 to 7 days.

b. Lawns: The frequency of watering depends on the type of grass. Since evaporation and transpiration rates vary across Texas, the frequency of watering also depends on the location.

APPENDIX:

CALCULATING THE TOTAL SAVINGS POSSIBLE THROUGH CONSERVATION

The average homeowner can save both water and money by practicing conservation. The three following tables illustrate some of the savings, both in water use and energy use, that are possible. Taking a few minutes to compare the possible savings with past utility bills will show how much money can be saved in the home.

Estimated Energy and Monetary Savings Associated with Residential Water Conservation

		Amount of energy saved		Value of energy saved	
Device	Hot water saved[1]	Gas water heaters[3]	Electric water[5]	Gas[6]	Electric[7]
	(Gal/day/D.U.)[2]	(Therms/year/D.U.)[4]	(Kw-hr/year/D.U.)	(Dollars/year/D.U.)	
Showerhead, 3.0 gpm	8.0	22.9	541	12.6	32.4
Water saving dishwashers	4.7	13.6	320	7.5	19.2
Water saving clothes-washing machines	2.4	6.8	160	3.7	9.6
Subtotal	15.1	43.3	1,021	23.8	61.2
Insulation of hot water pipes	4.7	13.6	320	7.5	19.2
Total	19.8	56.9	1,341	31.3	80.4

[1] 140°F water saved as follows: shower 3.4 gallons per capita per day (gpcd); dishwasher 2.0 gpcd; washing machines 1.0 gpcd; thermal pipe insulation 2.0 gpcd.

[2] D.U. = dwelling units; 2.37 persons per dwelling unit.

[3] 79 percent efficiency. Source: The California Appliance Efficiency Program—Revised Staff Rept. California Energy Resources Conservation & Devel. Comm. Conservation Div. (Nov. 1977).

[4] One Therm 100,000 BTU.

[5] 98 percent efficiency. Source: ibid.

[6] $0.55/therm. [7] $0.06/kw-hr.

Water Conserving Retrofit Devices

Application	Device	Function	Water savings	Unit Water savings gpcd[1]	Estimated Estimated cost $	Service life years
Toilet	Two displacement bottles	Reduces flush volume	0.5 gal/flush	2.3	0-0.20	5
Toilet	Water closet dam	Reduces flush volume	1.0 gal/flush	4.5	1.50-3.00	5
Toilet	Dual-flush	Variable-flush volume	3.5 gal/flush	15.7	15.00	15
Shower	Flow restrictor	Limits flow to 3 gpm[2]	1.5 gpm	6.7	0.50	5
Shower	Reduce-flow shower head	Limits flow to 3 gpm	1.5 gpm	6.7	3.00-20.00	15
Shower	Reduce-flow shower head with cutoff valve	Limits flow to 2.5 gpm	2 gpm	8.0	5.00-20.00	15
Shower	Cutoff valve	Facilitates "navy" shower"	—	—	2.50-5.00	15
Faucets	Aerator	Reduces splashing, enhances flow aesthetics creates appearance of greater flow	—	0.5	0.50-2.00	15
Hot water pipes	Insulation	Reduces warm-up time	—	0.5	0.50/ft	25
Water hook-up	Pressure-reducing valve	Reduces available water pressure at fixtures and, hence, flow rate	—	3.0	85.00	25

[1] gpcd—gallons per capita per day

[2] gpm—gallons per minute

Water Conserving Devices for New Construction

Application	Device	Function	Water savings	Estimated unit water savings gpcd[1]	Estimated additional cost $	Service life years
Toilet	Low-flush, 3.5 gal/flush	Reduced flush volume	1.5 gal/flush	7.5	0	25
Toilet	Low-flush, 2.5 gal/flush	Reduced flush volume	2.5 gal/flush	12.5	0	25
Toilet	Low-flush, 1.0 gal/flush	Reduced flush volume	4.0 gal/flush	20.0	*	25
Shower	Reduced-flow shower head	Reduces shower flow rate to 3.0 gpm	1.5 gpm[2]	6.7	0	15
Shower	Reduced-flow shower head with cutoff valve	Reduces shower flow rate to 2.5 gpm	2.0 gpm	8.0	0	15
Shower	Cutoff valve	Facilitates "navy shower"	—	—	2.50-5.00	15
Faucet	Aerator	Reduces splashing, enhances flow aesthetics, creates appearance of greater flow	—	0.5	0.50-2.00	15
Water hook-up	Pressure-reducing valve	Reduces available water pressure at fixtures and, hence, flow rate	—	3.0	45.00	25
Appliances	Water-efficient dish-washing appliances	Reduced water requirement	6-gal/cycle	2.0	0	15
Appliances	Water-efficient clothes-washing machine	Reduced water requirement	14-gal/cycle	3.5-7.0	70.00	15

*Some are expensive, but others are available at costs comparable to 3.5 gallon per flush models.

[1] gpcd—gallons per capita per day

[2] gpm—gallons per minute

MAINTENANCE/REPLACEMENT RECORD

DEVICES AND REPLACEMENTS INSTALLED

	Date	Cost
Low-Flow Showerheads Installed		
Low-Flow Faucet Aerators Installed		
Toilet Tank Displacement Devices Installed		
Fixture/Appliance Replacements		

ANNUAL OR SEMI-ANNUAL LEAK CHECKS AND REPAIRS

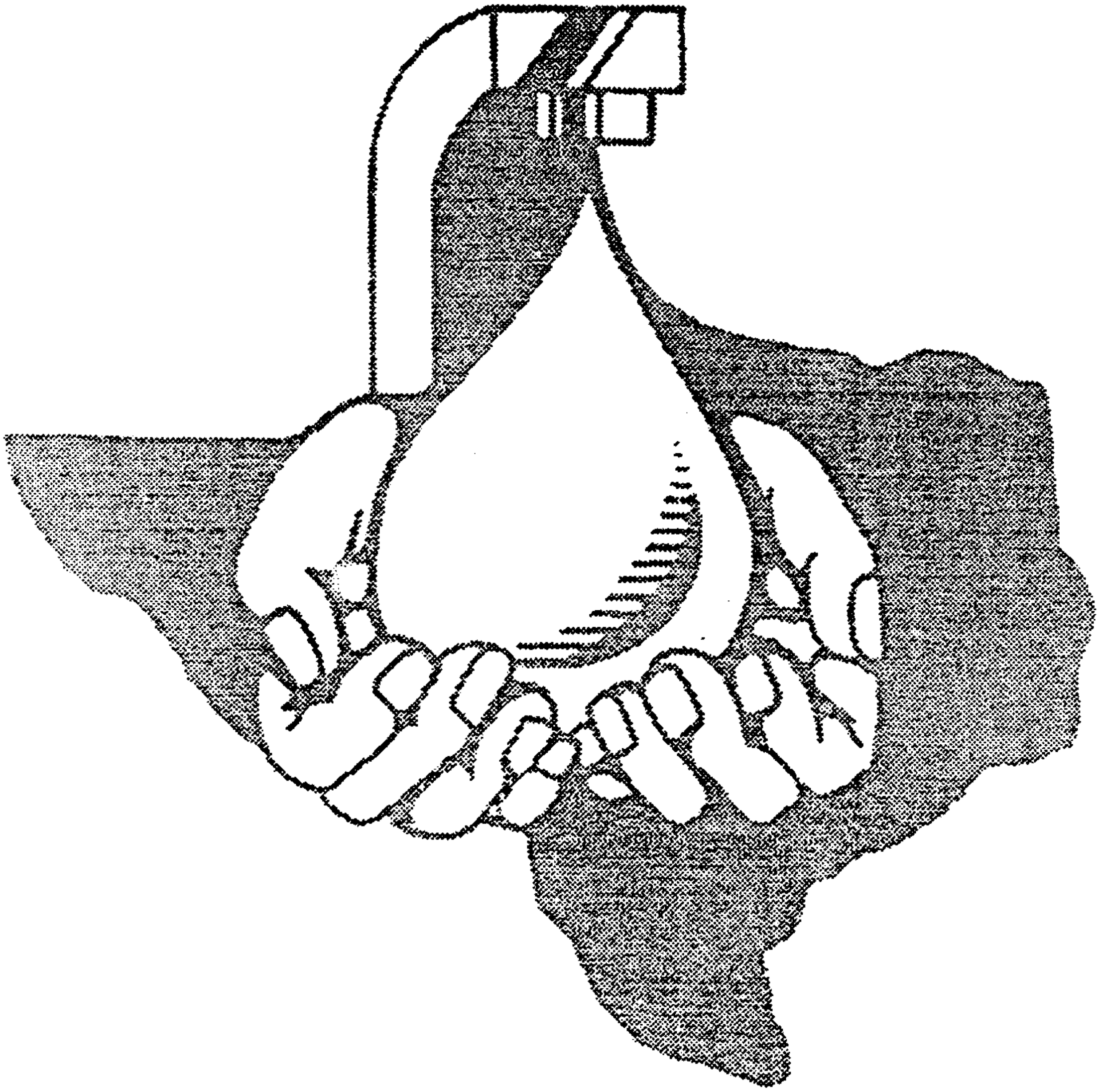
Date	Number of Faucet Leaks	Number of Toilet Leaks	Cost of Repairs

OTHER ANNUAL MAINTENANCE

	Date	Date	Date	Comments
Drain Hot Water Heater				
Check Septic Tank Scum and Sludge Levels				

Texas Water Development Board

WATER CONSERVATION
LITERATURE AND MATERIAL



Appendix
C

Publications, Prices, Order Forms

TEXAS WATER DEVELOPMENT BOARD
Conservation Literature

Copies of the following publications and materials can be obtained at no charge up to the quantity indicated depending on the category of the requestor (see Categories A,B,C, and D listed below). Most publications are available for sale when larger quantities than the number of free copies are desired.

CATEGORIES

Category A includes:

- Texas cities and towns;
- Texas Water and/or Wastewater Utilities and Districts;
- Other Texas water-related districts;
- Texas Water Supply Corporations;
- Texas River Authorities; and
- Texas State Agencies;

These may obtain free copies up to the maximum listed for Category A, not to exceed a total of 1,500 pieces, and may purchase additional copies at the listed price. They may also have their name and address printed on purchased publications and may purchase print negatives for printing locally. Use FORM I to order.

Category B includes:

- Texas schools and universities;
- Texas non-water related political subdivisions; and
- Texas non-profit clubs and organizations interested in water conservation.

These may obtain free copies up to the maximum quantity listed for Category B, not to exceed a total of 300 pieces, and may purchase additional copies at the listed price. Use FORM II to order.

Category C includes:

- Individual Texas residents;
- Private Texas businesses; and
- Out-of-state cities and state governments.

These may obtain free copies up to the maximum quantity listed for Category C, not to exceed a total of 30 pieces, and may purchase additional copies at the listed price. Use FORM II to order.

Category D includes:

- Individual out-of-state residents; and
- Private out-of-state businesses.

These may obtain single free copies of each item listed and may purchase up to 100 additional copies at the listed price. Use FORM II to order.

To make a request, write:

CONSERVATION
Texas Water Development Board
P.O. Box 13231, Capitol Station
Austin, Texas 78711-3231

Please be sure to use the proper order form. For more information, please call (512) 445-1467.

Agricultural Conservation Literature

Title	Number of Free Copies Per Category A / B / C	Unit Price After Free Copies
Agricultural Water Conservation in Texas: TWDB, Pamphlet, 8 pp.	50 / 50 / 10	NA
Have Your Irrigation System Evaluated Free: TWDB, Pamphlet, 4 pp.	350 / 100 / 10	$0.05
LEPA Irrigation: TWDB, Pamphlet, 6 pp.	350 / 100 / 10	0.05
Drip Irrigation: TWDB WC-8, Pamphlet, 6 pp.	250 / 100 / 10	0.10
Conserving Water in Irrigated Agriculture: TWDB, Booklet, 12 pp.	150 / 100 / 10	0.16
Furrow Dikes: HPUWCD #1, Pamphlet, 4 pp.	100 / 100 / 10	NA
Soil Moisture Monitoring: HPUWCD #1, Pamphlet, 4 pp.	100 / 100 / 10	NA
Center Pivot Irrigation: TAEX, Pamphlet, 4 pp.	350 / 100 / 10	NA
Surge Flow Irrigation: TAEX L-2220, Pamphlet, 4 pp.	350 / 100 / 10	NA
Surge Irrigation: SCS, Pamphlet, 6 pp.	350 / 100 / 10	NA

Abbreviations:

AWWA	American Water Works Association
EPA	U.S. Environmental Protection Agency
HPUWCD #1	High Plains Underground Water Conservation District No. 1
NA	Not available for sale from TWDB. Address of publishing agency provided on request.
OP	Out of Print - Call for information
SCS	USDA - Soil Conservation Service
TAEX	Texas Agricultural Extension Service
TDA	Texas Department of Agriculture
TWDB	Texas Water Development Board

Municipal Conservation Materials Available for Use in Education and Information Programs

Title	Number of Free Copies Per Category A / B / C	Unit Price After Free Copies	
		9,999 or Less	10,000 or More
Water...Half-A-Hundred Ways To Save it: TWDB WC-1, Pamphlet 8 pp.	500 / 100 / 10	$0.04	$0.03
Municipal and Commercial Water Conservation Services: TWDB WC-2, Pamphlet, 8 pp.	OP / OP / OP	NA	NA
A Homeowner's Guide to Water Use and Water Conservation: TWDB WC-3, Booklet, 22 pp.	250 / 100 / 10	0.25	0.23
How to Save Water Inside the Home: TWDB WC-4, Pamphlet, 8 pp.	500 / 100 / 10	0.07	0.06
Water Saving Ideas for Business and Industry: TWDB WC-5, Pamphlet, 8 pp.	OP / OP / OP	0.04	0.03
How to Save Water Outside the the Home: TWDB WC-6, Pamphlet, 8 pp.	500 / 100 / 10	0.07	0.06
Texas Sesquicentennial Native Plant Landscape (Located in Austin): TDA/TWDB WC-7, Pamphlet, 8 pp.	5 / 5 / 5	NA	NA
Drip Irrigation: TWDB WC-8, Pamphlet, 6 pp.	250 / 100 / 10	0.10	0.09
Coloring Poster for Children: TWDB WC-9, Poster, 1 pp.	OP / OP / OP	0.04	0.04
Water Conservation Coloring Book No. 1: TWDB WC-10, Booklet, 4 pp.	500 / 100 / 10	0.05	0.04
Water Conservation Coloring Book No. 2: TWDB WC-11, Booklet, 4 pp.	500 / 100 / 10	0.05	0.04
Lawn Watering Guide: TWDB WC-12 Card, 3.5" X 5"	500 / 100 / 10	0.04	0.03

Municipal Conservation Materials Available for Use in Education and Information Programs - Continued

Title	Number of Free Copies Per Category A / B / C	Unit Price After Free Copies	
		9,999 or Less	10,000 or More
A Directory of Water Saving Plants and Trees for Texas: TWDB WC-13, Booklet, 42 pp.	50 / 25 / 2	0.55	0.52
Xeriscape-Principles, Benefits: TWDB WC-14, Pamphlet, 4 pp. Standard Size 6" X 11".	250 / 100 / 10	0.19	0.16
Xeriscape-Principles, Benefits: TWDB WC-14A, Pamphlet, 4 pp. Size 3.5" X 7.5".	500 / 100 / 10	0.07	0.06
Toilet Tank Leak Detector Tablets: 2 Tablets per packet	100 / 25 / 1	NA	NA
Plastic Ruler: TWDB, 6" X 1 1/4"	250 / 100 / 10	NA	NA
Landscape Water Conservation... Xeriscape: TAEX B-1584, Booklet, 12 pp.	50 / 25 / 10	NA	NA

Guidebooks and Conservation Planning Literature

Title	Number of Free Copies Per Categories A / B / C	Unit Price After Free Copies
The Authority of Cities, Water Utilities, and Water Districts to Regulate and Enforce Water Conservation Measures: TWDB GB-1, Guidebook, 5 pp.	10 / 1 / 1	—
A Guidebook for Reducing Unaccounted for Water: TWDB GB-2, Guidebook, 33 pp.	10 / 1 / 1	—
Guidelines for Municipal Water Conservation and Drought Contingency Planning and Program Development: TWDB GB-3, Guidebook, 36 pp.	10 / 1 / 1	—
Guidelines for Water Reuse EPA600/8-80-036: EPA GB-4, Guidebook, 105 pp.	10 / 1 / 1	—
Sources of Leak Detection Equipment and Services: TWDB GB-5, Guidebook, 2 pp.	10 / 1 / 1	—
Sources of Water Saving Devices: TWDB GB-6, Guidebook, 20 pp.	10 / 1 / 1	—
Texas Water Resources and Conservation: TWDB GB-7, Guidebook, 38 pp.	10 / 1 / 1	—
Water Conservation And Drought Contingency Plan Development Procedures: TWDB GB-8, Guidebook, 58 pp.	10 / 1 / 1	—
Example Xeriscape Incentives and Ordinances: TWDB GB-9, Guidebook, 58 pp.	10 / 1 / 1	—
The Texas Water Education Network Directory: TWDB GB-10, Guidebook, 23 pp.	1 / 1 / 1	—
Summary of Water for Texas: TWDB C-20, Pamphlet, 8 pp.	10 / 1 / 1	—
Water Planning in Texas: TWDB C-18, Booklet, 27 pp.	10 / 1 / 1	—
Texas Water Development Board (Funding Programs): TWDB, Pamhplet, 4 pp.	10 / 1 / 1	—

Publications and Audiovisual Materials Available for Loan From Texas Water Development Board (TWDB)*

The following water conservation publications and audiovisual materials are available for loan of up to two weeks from TWDB. To borrow any of these write to: CONSERVATION, Texas Water Development Board, Capitol Station, Austin, Texas 78711-3231.

Publications

Title	Originally Published By	Description	Length
Water Audit and Leak Detection Guidebook	California Dept. of Water Resources	Book	142 pages
Example Brochures and Promotional Material	Compiled by TWDB	Ringbinder	32 pages
Municipal Water Conservation Workshop Notebook	TWDB	Ringbinder	6 sections

Audiovisual Materials

Title	Originally Published By	Description	Length
The Alternative is Conservation	Water Films	16mm Film or VCR/VHS Format	28 minutes
Water Follies	AWWA	16mm Film or VCR/VHS Format	7.5 minutes
Orangutans (Public Service Annoucement)	AWWA	16mm Film or VCR/VHS Format	30 seconds
Gooney Birds (Public Service Annoucement)	AWWA	16mm Film or VCR/VHS Format	30 seconds
Tanks (Public Service Announcement)	AWWA	16mm Film or VCR/VHS Format	30 seconds
Spot Announcements	Lower Colorado River Authority	Audio Cassette	30 seconds

*The films, video cassettes, and publications are provided for review purposes only. Permission to use any of this material for print or broadcast must be obtained from the producer or publisher of the material.

FOR CATEGORY A ONLY

FORM I

TWDB WATER CONSERVATION LITERATURE
Order Form

Please print or type your order. The name and address of your organization can be printed on paid literature orders of 1,000 or more copies of a brochure. A charge of $25.00 must be paid for each title ordered with name and address. Names and addresses cannot be printed on free copies. Checks or money orders made payable to the Texas Water Development Board must be received with your order.

	Title	Quantity Free	Quantity Purchased	Unit Price	Subtotal
1.					
2.					
3.					
4.					
5.					
6.					
7.					
8.					
	Total	*			

Charge for Name and Address on Literature ________ @$25.00 ________

Postage** ________

Total ________

Organization Name: ________________

____

Address: ________________

City, State, Zip: ________________

Contact Person: ________________

Telephone: ________________

Name and Address as it is to be printed on literature:

____

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*Not to exceed 1,500 pieces.

**The TWDB will ship orders up to 400 pounds free at book rate or cheapeast rate available. The Board requests that the utility arrange for shipping or pay the incremental difference on orders exceeding 400 pounds. Call (512) 445-1467

Send Order to:
CONSERVATION
Texas Water Development Board
P.O. Box 13231, Capitol Station
Austin, Texas 78711-3231

FOR CATEGORIES B, C, AND D FORM II

TWDB WATER CONSERVATION LITERATURE
Order Form

Please print or type your order. Checks or money orders made payable to the Texas Water Development Board must be received with your order.

	Title	Quantity Free	Quantity Purchased	Unit Price	Subtotal
1.					
2.					
3.					
4.					
5.					
6.					
7.					
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	Total	*			
		Postage**			
				Total	

Organization Name: ____________________

____

Address: ____________________

City, State, Zip: ____________________

Contact Person: ____________________

Telephone: ____________________

*Not to exceed 300 pieces for Category B or 30 pieces for Category C.

**The TWDB will ship orders up to 50 pounds free at the book rate through the U.S. Postal Service. The Board asks that the requestors arrange for shipping or pay the incremental difference on orders exceeding 50 pounds. Call (512) 445-1467 for information.

Send Order to:
CONSERVATION
Texas Water Development Board
P.O. Box 13231, Capitol Station
Austin, Texas 78711-3231

BRADY, TEXAS

MCCULLOCH COUNTY
WATER CONSERVATION
AND
DROUGHT CONTINGENCY PLAN

Appendix
D

INTRODUCTION

The City of Brady is a Home Rule City that underwent a complete Charter review and modification in 1989. As provided by the Charter, the City is operated with a Mayor/Council form of Government in which a Mayor and five (5) Council are elected form the general public. The Mayor runs the Council meetings; however, does not vote on matters. All matters relating to City activities from the paving of streets to wastewater treatment plant renovations must be approved by Council. All administrative functions and work approved by Council is carried out by the City Manager.

The City of Brady provides water service to an area approximately 40 square miles. It is the area granted to the City of Brady under Texas Water Commission certificate of convenience and necessity No. 11121. The area covered by this certificate is the City of Brady, McCulloch County, Texas and areas immediately adjacent to the City. It is more specifically identified on the Commission's official service area map, WRS-160 (Appendix A).

While population in Brady has had modest growth, the extension of water service to new areas has put an extra demand on the delivery system to the point that production is approaching capacity. The goals of the following Water Conservation and Drought Contingency Plan are two fold:

1) Increased awareness of the value of the limited resource of water in a west Texas climate subject to periodic dry cycles.

2) With the limited financial resources of the community, utilize conservation and drought contingency planning to delay capital expenditures for replacement or additions to water service facilities.

UTILITY EVALUATION

A. Population of service area - 7,800

B. Size of service area (estimated) - 40 square miles

C. Breakdown of connections in service area:

2450 - residential 429 - commercial 13 - industrial

D. Rate of new connections per year - 20 residential

E. Water use information:

1. Average annual production - 787,263,000 gal/yr.
2. Average monthly production - 65,605,000 gal/mo.
3. Monthly sales: January 1988 - December 1988

	Residential	Commercial-Institutional	Industrial	Total
January	17,051,000	10,002,000	4,868,000	31,921,000
February	19,621,000	9,745,000	4,578,000	33,944,000
March	21,123,000	9,616,000	4,008,000	34,747,000
April	30,020,000	12,338,000	4,219,000	46,577,000
May	41,746,000	14,693,000	4,491,000	60,930,000
June	29,631,000	13,553,000	3,364,000	46,548,000
July	46,418,000	15,082,000	3,979,000	65,479,000
August	40,975,000	14,983,000	4,851,000	60,809,000
September	48,946,000	15,522,000	4,390,000	68,858,000
October	31,845,000	13,723,000	4,754,000	50,322,000
November	25,885,000	11,124,000	3,536,000	40,545,000
December	19,735,000	10,620,000	5,005,000	35,360,000
Total	372,996,000	151,001,000	52,043,000	576,040,000

4. Peak daily use - 4,100,000 gpd
5. Peak to average use ratio - 2.19

G. Peak daily capacity - 5,529,600 gpd

H. High Volume Customers:

1. Roddie Wool Scouring - 36,315,000 gal/yr.
2. Loadcraft Inc. - 8,912,000 gal/yr.
3. Brady Independent School District - 7,570,000 gal/yr.
4. Housing Authority (Site E) - 4,922,000 gal/yr.
5. Leisure Lodge Inc. - 4,464,000 gal/yr.
6. Housing Authority (Site F) - 4,401,000 gal/yr.
7. Sunset Inn - 2,533,000 gal/yr.
8. C H W Corporation - 2,155,000 gal/yr.
9. Heart of Texas Racing, Inc. - 1,655,000 gal/yr.

I. Population and water use projections - See Appendix B

J. Percent of metered connections

Residential - 100% Commercial - 100% Industrial - 100%

K. Existing water rate structure

Gallons	In Town	Out of Town
1st 1000	$ 4.09	$ 6.14
2nd 1000	1.82	2.73
3rd 1000	1.37	2.05
4th 1000	1.37	2.05
> 4000	.80	1.20

L. Average annual revenue from water sales - $570,000

M. Average annual revenue from wastewater rates - $210,000

N. Average annual cost of operation - $100,000

O. Applicable local regulations

The only applicable regulations are those adopted by the Council.

P. Applicable State or Federal Regulations

The City of Brady is a Municipal Corporation of the State of Texas and, therefore, must comply with rules of the following agencies:

TEXAS WATER COMMISSION
TEXAS WATER DEVELOPMENT BOARD
TEXAS DEPARTMENT OF HEALTH
ENVIRONMENTAL PROTECTION AGENCY

PUBLIC INVOLVEMENT

The City of Brady conducts council meetings on the first and third Tuesday of the month. These meetings are subject to the open meetings laws; therefore, interested parties are notified of the contents of the meeting and have the opportunity to address the Council to make their views known.

WATER CONSERVATION PLAN

I. EDUCATION AND INFORMATION

The City of Brady will provide water conservation through a variety of informational and educational means. The following are examples of programs that will be utilized, subject to availability of material and success of the program:

New Customers will receive a water conservation kit that includes flow restrictors for shower heads, displacement bags for the water closet and dye tablets for leak detection plus a one page flyer on water conservation explaining the general provisions of the water conservation and drought contingency plan and the benefit to Brady plus conservation tips around the home.

Water Conservation Week will be declared by the Mayor or City Council the first week of November, which is just prior to the three (3) month period used to establish sewer rates. During that period:

The one page flyers will be mailed to all customers.
Several public service announcements will be aired on the radio.
Water conservation tips will be given to the newspaper.

Regular Information will be given to the radio and newspapers regarding some aspect of water conservation or explaining how to utilize material provided in the water conservation kit.

Youth Water Education and Conservation will be promoted by providing films like "Water Follies" and "The Alternative Is Conservation" and other audio/visual materials that may be appropriate to grades K-6 regarding water conservation. It is intended this information will be provided to the schools so that the teachers may utilize the material in their Earth Science or water units.

Water Evaluation Test - A simple test will be developed and distributed to customers twice a year the first year (1990) and once a year there after. The test will be designed to cover various aspects about the Brady water system and general water use/conservation characteristics of West Texas. The purpose of the test is threefold: 1) to determine how much the public knows about Brady's water system, use and conservation 2) areas where information is lacking and more education should be targeted and 3) help promote water conservation. A sample questionnaire is included in Appendix C.

Additional Information will be available to interested parties through a host of literature provided at City Hall, an extensive bibliography, or in pre-printed brochures available from:

Texas Water Development Board
P.O. Box 13231, Capital Station
Austin, Texas 78711-3231
(Appendix D)

II. PLUMBING CODES

The City of Brady has adopted the Texas State Board of Plumbing Examiners plumbing code; therefore, there is no specific set of regulations for Brady. A water conservation amendment is included (Appendix F). New construction and bathroom remodeling should be required to use toilets that use no more than 3.5 gallons per flush. By 1995, the plumbing codes need to be reviewed and amendments made as necessary. This work should be done under the direction of the water/utility directors.

III. RETROFIT PROGRAM

Educational and informational programs will help notify local citizens of the advantage of installing water saving devices. A proposed rate structure cited below should help promote efficient water use.

Water conservation kits will be available at City Hall, Chamber of Commerce and other specified locations for interested parties.

IV. WATER RATE STRUCTURE

At present, the City of Brady has a decreasing water rate structure. A flat rate structure that has a minimum monthly service charge plus a uniform rate for all water used will be adopted. This structure is consistent with current utility regulatory practice. However, the final figures have not been determined as the City is having to consider a major capitol improvement program for the entire water distribution system. Consequently, the City wishes to evaluate all of the data and make the necessary changes once.

V. METERING

The City currently meters 100% of the customers. Currently the City billing department notes abnormal usage patterns and reports them to the Water Department. They are routinely replaced. A program is in place to replace all master meters and eventually have chart recorders at each well to monitor pumpage characteristics and demand current. The City will establish the following meter testing schedule:

1) Production Meters - test once a year
2) Meters larger than 1" - test once a year
3) Meters 1" and smaller - test every ten years

VI. WATER CONSERVATION LANDSCAPING

To promote water saving landscaping literature will be available describing the benefits and process of xeriscaping. Also local nurseries will be contacted and asked to support this program by making available low water demand vegetation suitable for the climate in this area.

The Water Conservation Plan Committee has been allocated a parcel of City owned property to set up a xeriscape demonstration project.

VII. LEAK DETECTION AND REPAIR

The City is continually monitoring and repairing leaks. Examples of ongoing and future efforts are:

1) Audits by the computer billing program that identifies those accounts that show abnormally high usage after a meter has been repaired or a service call for a water leak has been indicated for that account.

2) Constant monitoring of facilities to look for leaks.

3) Visual inspection by meter readers looking for conditions that would suggest a leak.

4) Develop and implement a 10-Year program to examine entire distribution system of City for leaks and replace as necessary.

5) Have Municipal Team from Texas Water Development Board visit City to demonstrate technique for identifying leaks.

6) Consider purchase of "sniffer" equipment that will aide in leak detection.

VIII. RECYCLING AND REUSE

The City will enter into discussions with Roddie Wool Scouring Plant regarding the possible use of treated effluent for the Plant. Currently the Scouring Plant is the largest user of water from the Brady system. Other potential customers for treated effluent are G. Rollie White Downs and road construction crews for dust containment.

IX. IMPLEMENTATION AND ENFORCEMENT

The Water Conservation Plan will be enforced by the following methods:

1) The plan will be adopted by City Council, then non-compliance will be handled in the same manner as other administrative functions.

2) City water hook ups will not be given to customers who do not meet the requirements of the water conservation plumbing fixtures.

X. CONSERVATION PLAN ANNUAL REPORT

A committee should be appointed by the City Council to file an Annual Report with the Executive Administrator of the Texas Water Development Board addressing the progress and effectiveness of the Water Conservation Plan. The report will include:

A. Public information which has been issued
B. Public response
C. An analysis of water consumption data and comments on the apparent effectiveness of the Water Conservation Plan.
D. Implementation progress and status of Water Conservation Plan

XI. CONTRACTS WITH OTHER POLITICAL SUBDIVISIONS

Any contract with another political subdivision of the State of Texas will be approved only if that entity adopts the City of Brady's Water Conservation and Drought Contingency Plan.

DROUGHT CONTINGENCY PLANS .

I. TRIGGER CONDITIONS

A. Level I - The Brady City Council may declare and direct the City Manager to initiate Level I drought contingency plans under appropriate existing conditions including the following:

1). Recent and projected climatic conditions are in the opinion of the council to be conducive to excessive water system demands; and/or

2) Average daily water consumption reaches 90% of system design capacity; and/or

3) Peak period water demand reaches 90% of system design capacity; and

4) Water system components including production, storage, pumping and distribution are operating within normal design limits.

B. Level II - The Brady City Council may declare an emergency and direct the City Manager to initiate Level II drought contingency plans under appropriate existing conditions including the following:

1) Water system components including production, storage, pumping an distribution are operating outside of normal design limits due to component failure or diminished capacity; and/or

2) Average daily water consumption approaches 100% of design capacity; and/or

3) Peak period water demand approaches 100% of design capacity; and/or

4) Diminished service to any consumers results in service levels outside of regulatory limits; and/or

5) An imminent threat to water systems components exists due to natural or manmade disaster or imminent threat of component failure due to service levels or physical condition.

II. DROUGHT CONTINGENCY MEASURES:

In order to try and flatten the peak demand curve, a two-tiered scheme of water restrictions will be implemented by the City Manager as necessitated by the conditions.

1. Step I is where the City Manager can restrict the use of water for outdoor sprinkling, watering of lawns, shrubs, driveways and automobiles to certain areas of the City by days and to certain hours. Said restrictions will remain in effect until the City Manager lifts the restrictions. More specifically stated shall be:

A. Inform the public through the news media that a trigger condition Level I has been reached, and that they should look for ways to voluntarily reduce water use. Specific steps which can be taken will be provided thru the news media.

B. Notify major commercial water users of the situation and request voluntary water use reductions:

C. The following mandatory lawn watering schedule shall be implemented:

> Customers with even numbered addresses may water on even days numbers of a month. Customers with odd numbered addresses may water on odd days of the month. Watering shall occur only between the hours of 9:00 p.m. to 9:00 a.m..

D. During winter months, request water users to insulate pipes rather than running water to prevent freezing; and

E. The City will begin monitoring water pressure in the distribution system and water levels in the storage tanks.

2. Step II is where the City Manager will ban the use of water totally for outdoor sprinkling, watering lawns, shrubs, driveways and automobiles. Said restriction will remain in effect until the City Manager lifts the ban. More specifically stated as follows:

A. Continue implementation of all relevant actions in preceding phase;

B. Car washing, window washing, pavement washing prohibited except when a bucket is used:

C. The following public water uses, not essential for public health or safety, are prohibited.

1) Street washing
2) Water hydrant flushing
3) Filling pools
4) Golf course watering
5) Athletic field watering
6) Commercial car/truck washes

3. The curtailment will be effective upon the City Manager's giving notice of curtailment to the customers within the City; the posting of a notice of curtailment and notifying the news media of curtailment and as stated in Section III.

4. The curtailments will be terminated upon the City Managers giving notice of termination.

5. The City Council can amend, add or delete any of these Rules and Regulations in the same manner they would handle an ordinance as provided in the City Charter.

6. Any violation of the Rules and Regulations adopted by the City shall carry a maximum penalty of $200 per violation.

In addition to the existing Drought Contingency Plan, the City will enact the following Drought Contingency measures.

A. LEVEL ONE DROUGHT CONTINGENCY MEASURES

1. Inform public by giving notice of a Level I drought to the customers within the City; the posting of the notice, and notifying news media of the drought.

2. Included in the information to the public will be the recommendation that water users look for ways to conserve water.

3. Public will be advised of the trigger condition situation daily.

4. The Step I curtailment will be enacted.

B. LEVEL TWO DROUGHT CONTINGENCY MEASURES

1. Public will be informed as mentioned above.
2. The Step II curtailment will be enacted.
3. Certain industrial and commercial water users which are not essential to the health and safety of community will be prohibited from water use.
4. Public will be advised of the trigger conditions daily.

III. <u>INFORMATION AND EDUCATION</u>

Once trigger conditions and emergency measures have been reached, the public will be informed of the conditions, and measures to be taken. The process for notifying the public includes:

A. Posting the Notice of Drought conditions at City Hall, Post Office, Banks, Court House and major supermarkets.

B. General circulation of Newspapers.

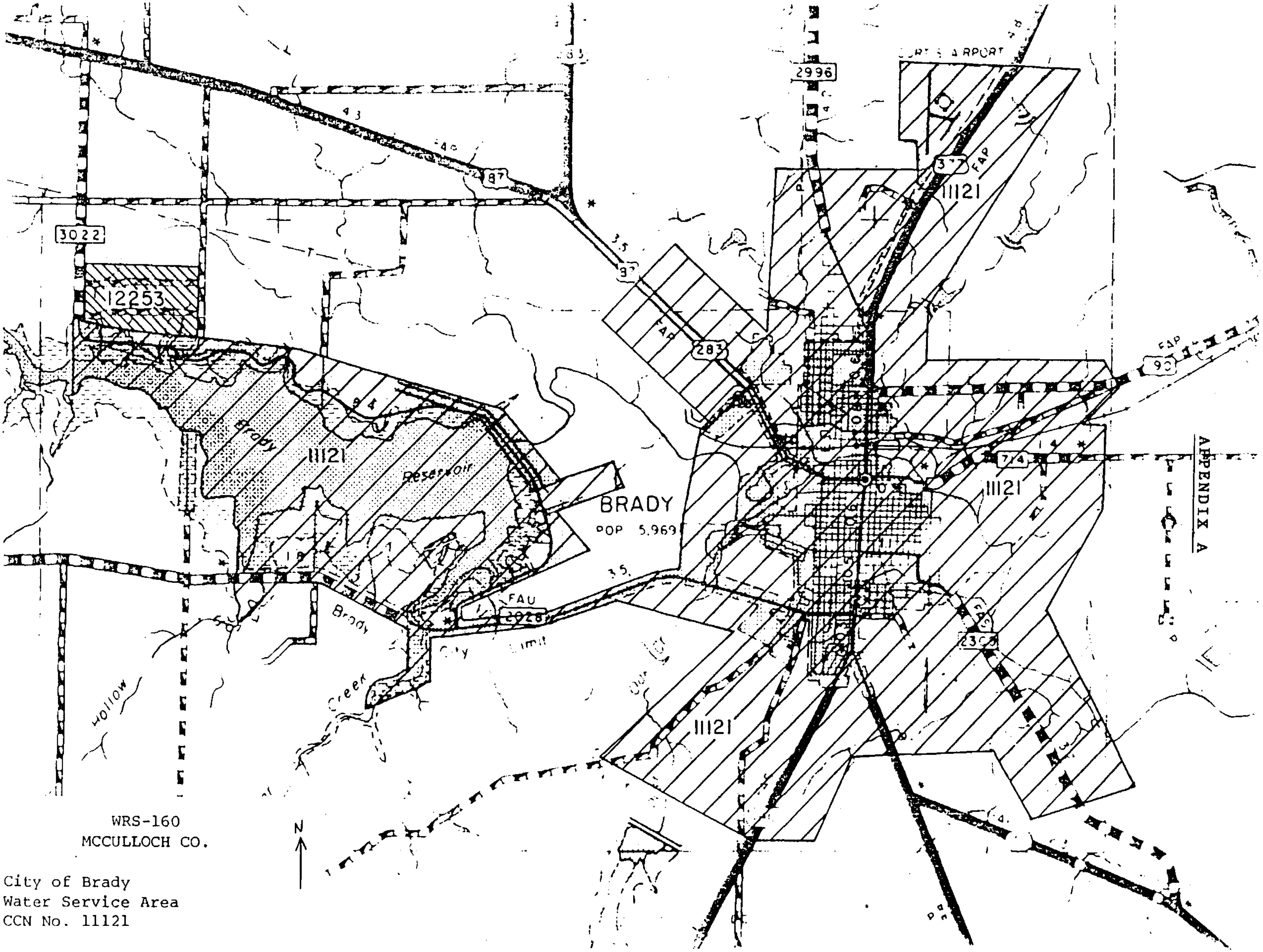
C. Notifying local radio stations.

D. Direct mailing to customers explaining need for provisions of Drought Contingency Plan which will include steps to be taken to insure compliance with the plan.

IV. TERMINATION NOTIFICATION

Termination of the Drought measures will take place when the trigger conditions which initiated the drought measures have subsided, and an emergency situation no longer exists. The public will be informed of the termination of the drought measures in the same manner that they were informed of the initiation of the drought measures through the City Manager.

APPENDIX A



WRS-160
MCCULLOCH CO.

City of Brady
Water Service Area
CCN No. 11121

Scale: 1" = 1 mile 3/10/89

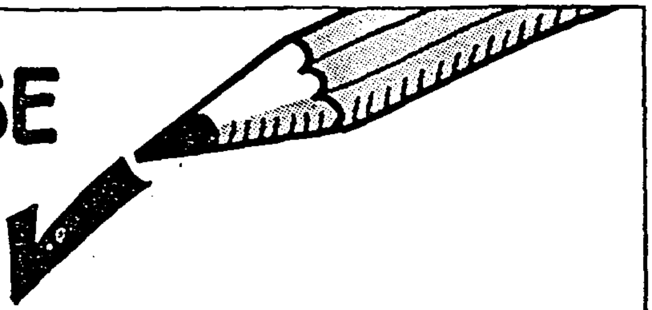
APPENDIX B

PROJECTED WATER REQUIREMENTS

Year	Estimated Average Population (Planning Area)	Projected Average Daily Usage (MGD)	Projected Peak-Day Usage (MGD)	Projected Maximum-Hour Usage (MGD)
XXX	Pop	Pop X .0003	Pop X .0006	Pop X .0018
1989	7847	2.35	4.71	14.12
1990	7931	2.38	4.76	14.28
1991	7961	2.39	4.78	14.33
1992	7992	2.40	4.80	14.39
1993	8022	2.41	4.81	14.44
1994	8053	2.42	4.83	14.50
1995	8084	2.43	4.85	14.55
1996	8115	2.43	4.87	14.61
1997	8146	2.44	4.89	14.66
1998	8177	2.45	4.91	14.72
1999	8209	2.46	4.93	14.78
2000	8240	2.47	4.94	14.83
2001	8274	2.48	4.96	14.89
2002	8308	2.49	4.98	14.95
2003	8343	2.50	5.01	15.02
2004	8377	2.51	5.03	15.08
2005	8412	2.52	5.05	15.14
2006	8447	2.53	5.07	15.20
2007	8482	2.54	5.09	15.27
2008	8517	2.56	5.11	15.33
2009	8553	2.57	5.13	15.40
2010	8588	2.58	5.15	15.46

APPENDIX C

HOME WATER USE SURVEY



Here is a chance for you to find out some information about your family's water use. You may be able to answer some questions yourself, but you will probably need to talk to the other members of your family in order to complete all items.

GENERAL INFORMATION

1. What type of home do you live in? (check one)
   ______ house ______ apartment/condominium ______ mobile home
2. How many people are living at home? ______
3. What are the ages of the children living at home?
   ______ ______ ______ ______ ______ ______
4. How many gallons of water were used in your home during the last billing period? (Ask your parent(s) to show you the latest water bill).
   ______
5. How many days were there in that billing period? ______

SPECIFIC WATER USE - Outside the House

1. What size is your lot compared with other lots on your street? (check one)
   ______ larger ______ the same ______ smaller
2. What type of plants do you have? (check all that apply)
   ______ lawn or ground cover ______ flowers and/or shrubbery
   ______ vegetable garden and/or fruit trees ______ none
3. Outside watering for the months April to September (estimate).
   ______ number of watering minutes per day (total number of minutes each hose is run every watering day).
   ______ number of watering days per week

SPECIFIC WATER USE - Inside the House

1. Dishwasher (answer only if you have one)
   a. How many times per week is the dishwasher run? ______
   b. How full is the dishwasher usually loaded?
   ______ full ______ ½ full ______ less than ½ full
2. Washing Machine (answer only if you have one)
   a. How many loads per week are usually washed? ______
   b, How full is the washer usually loaded?
   ______ full ______ ½ full ______ less than ½ full
3. How many of each of the following do you have in your home?
   ______ sinks ______ showers ______ bathtubs ______ toilets
4. How many showers per week are taken in your house? ______
5. How many tub baths per week are taken in your house? ______
6. How many minutes is your family's average shower? ______
7. How many times each day is a toilet flushed in your home? ______
8. Is there any other place where a significant amount of water is used in and/or around your home? (Examples: automatic sprinklers, hot tub, pool, etc.)

APPENDIX D

TEXAS WATER DEVELOPMENT BOARD
WATER CONSERVATION LITERATURE

Single copies of all of the following publications and materials can be obtained at no charge. The * indicates those publications that are available free to political subdivisions in small quantities. Larger quantities can be obtained through special arrangement or at the cost of printing. To make a request, write: CONSERVATION, Texas Water Development Board, Capitol Station, Austin, Texas 78711-3231

Agricultural Conservation Literature

Title	Published By	Description	Length
Agricultural Water Conservation in Texas*	TWDB	Pamphlet with Tear-out	8 pages
Have Your Irrigation System Evaluated Free*	TWDB	Pamphlet	4 pages
LEPA Irrigation*	TWDB	Pamphlet	6 pages
Drip Irrigation*	TWDB	Pamphlet	6 pages
Plastic Ruler*	TWDB	6" x 1 1/4"	-
Furrow Dikes*	HPUWCD #1	Pamphlet	4 pages
Soil Moisture Monitoring*	HPUWCD #1	Pamphlet	4 pages
Center Pivot Irrigation Systems L-2219*	TAEX	Pamphlet	4 pages
Surge Flow Irrigation L-2220*	TAEX	Pamphlet	4 pages
Surge Irrigation*	SCS	Pamphlet	6 pages
Coloring Poster for Children*	TWDB	Coloring Poster	1 page
Water Conservation Coloring Book* (No. 1)	TWDB	Booklet	4 pages

Attachment A

Contents of the Municipal Water Conservation Workshop Notebook

The notebook is distributed to participants at Board-sponsored Municipal Water Conservation Workshops. In addition, single copies of the notebook can be provided to cities and utilities. Single copies of selected materials from the notebook can also be provided.

Title	Published By	Description	Length
Section 1: The Need for Conservation			
Texas Water Resources and Conservation	TWDB	Paper	38 pages
Section 2: Water Conservation Techniques			
Efficient Use of Water in the Garden and Landscape (B-1496)	TAEX	Booklet	20 pages
Xeriscape	City of Austin	Booklet	20 pages
Water Pressure Reducing Valves	Watts Regulator	Booklet	21 pages
Texas Native Tree and Plant Directory, 1986	TDA	Book	162 pages
Sources of Leak Detection Equipment and Services	TWDB	List	2 pages
Sources of Water Saving Devices	TWDB	List	21 pages
Locating and Reducing Unaccounted For Water Through The Use Of The Water Audit And Leak Detection	TWDB	Guidebook	30 pages
Water Rate Design Emphasizing Conservation Rate Structures	TWDB	Guidebook	30 pages
Model Water Ordinances	TWDB	Guidebook	25 pages
The Authority of Cities, Water Utilities, and Water Districts to Regulate and Enforce Water Conservation Measures	TWDB	Guidebook	25 pages

Section 3. Alternate Sources

The Cost of Conventional Water Supply Development and Treatment	TWDB	Paper	9 pages
Potential for Utilization of Brackish Groundwater	TWDB	Paper	21 pages
Guidelines for Water Reuse EPA-600/8-80-036	EPA	Book	105 pages

Section 4: Workshop Exercise

Example Problem	TWDB	Loose-leaf	15 pages

Section 5: Plan Elements

Guidelines for Municipal Water Conservation and Drought Contingency Planning and Program Development	TWDB	Loose-leaf	36 pages

Section 6: Plan Development

Water Conservation and Drought Contingency Plan Development Procedures	TWDB	Loose-leaf	58 pages

PUBLICATIONS AND AUDIOVISUAL MATERIALS AVAILABLE FOR LOAN FROM TEXAS WATER DEVELOPMENT BOARD (TWDB) (a)

The following water conservation publications and audiovisual materials are available for a loan of up to two weeks from TWDB. To borrow any of these write to: CONSERVATION, Texas Water Development Board, Capitol Station, Austin, Texas 78711-3231.

Publications

Title	Published By	Description	Length
Water Audit and Leak Detection Guidebook	California Dept. of Water Res.	Book	142 pages
Example Brochures and Promotional Material	Compiled by TWDB	Ringbinder	32 pages
Regional Teachers Guide Supplements	California Dept. of Water Res.	Books	Nos. 1-7

Audiovisual Materials

Title	Published By	Description	Length
The Alternative is Conservation	Water Films	16mm Film VCR/VHS Format	28 minutes
Water Follies	American Water Works Assoc. (AWWA)	16mm Film VCR/VHS Format	7.5 minutes
Orangutans (Public Service Announcement)	AWWA VCR/VHS Format	16mm Film VCR/VHS Format	30 seconds
Gooney Birds (Public Service Announcement)	AWWA VCR/VHS Format	16mm Film VCR/VHS Format	30 seconds
Tanks (Public Service Announcement)	AWWA VCR/VHS Format	16mm Film VCR/VHS Format	30 seconds
Spot Announcements	Lower Colorado River Authority	Audio Cassette	30 seconds

(a) The films, video cassettes, and publications are provided for review purposes only. Permission to use any of this material for print or broadcast must be obtained from the producer or publisher of the material.

Municipal Conservation Literature

Title	Published By	Description	Length
Water Half-A-Hundred Ways To Save It*	TWDB	Pamphlet	8 pages
Water Saving Ideas For Business and Industry*	TWDB	Pamphlet	8 pages
How To Save Water Outside The Home*	TWDB	Pamphlet	8 pages
How To Save Water Inside The Home*	TWDB	Pamphlet	8 pages
Toilet Tank Leak Detector Tablets*	TWDB	2 Tablets	-
Municipal and Commercial Water Conservation Services	TWDB	Pamphlet with Tear-out	8 pages
A Homeowner's Guide To Water Use and Water Conservation	TWDB	Booklet	22 pages
Guidelines for Municipal Water Conservation and Drought Contingency Planning and Program Development	TWDB	Loose-leaf	36 pages
How to Xeriscape	NXC	Pamphlet	10 pages
Texas Sesquicentennial Native Plant Landscape	TDA/TWDB	Pamphlet	8 pages
Municipal Water Conservation Workshop Notebook (See Attachment "A" for a Description of Contents)	TWDB	Notebook	6 sections
Water Conservation Coloring Book* (No. 2)	TWDB	Booklet	4 pages

Texas Water Resources and Planning Literature

Title	Published By	Description	Length
TWDB Report 294 - Surveys of Irrigation in Texas	TWDB	Book	243 pages
Summary of Water for Texas (C-20)	TDWR	Pamphlet	8 pages
Water Planning in Texas	TDWR	Booklet	27 pages
Texas Water Development Board (Funding Programs)	TWDB	Pamphlet	4 pages
Water For Texas (GP-4-1)	TDWR	Books	
Volume 1 (Comprehensive Plan)	[Available for purchase only from the Texas Water Commission, P.O. Box 13087 Austin, Texas 78711]		72 pages
Volume 2 (Technical Appendix)			530 pages
Texas Water Facts	TDWR	Booklet	12 pages

Abbreviations:

HPUWCD #1	High Plains Underground Water Conservation District No. 1
NXC	National Xeriscape Council, Inc.
SCS	USDA - Soil Conservation Service
TAEX	Texas Agricultural Extension Service
TDA	Texas Department of Agriculture
TDWR	Texas Department of Water Resources
TWDB	Texas Water Development Board

APPENDIX E

A RESOLUTION ADOPTING THE CITY OF BRADY WATER CONSERVATION AND DROUGHT CONTINGENCY PLAN

Whereas, To assure that there will be water for essential uses and to avoid waste, either of this resource or of public funds used in construction and operation of water facilities, a Water Conservation and Drought Contingency Plan is necessary; and

Whereas, At the request of the City Council, the Citizen's water committee has drafted such a plan; and

Whereas, The Council has reviewed said plan, found it to be consistent with the goals and objectives of the City, and considers its adoption to be in the best interest of the citizens of Brady.

BE IT THEREFORE RESOLVED BY THE CITY COUNCIL OF THE CITY OF BRADY, TEXAS, that the "Water Conservation and Drought Contingency Plan" attached hereto be and is hereby adopted.

PASSED AND APPROVED this the ______ day of ______________, 1990.

H.L. Gober, Jr.
Mayor

ATTEST:

Dorene Patterson, City Secretary

APPENDIX F

A RESOLUTION ADOPTING WATER CONSERVATION PLUMBING CODES FOR THE CITY OF BRADY

WHEREAS, The City of Brady has adopted general guidelines as put forth in the Texas State Board of Plumbing Examiners Plumbing Code; and

WHEREAS, The City of Brady desires to adopt certain standards that are complimentary to the "Water Conservation And Drought Contingency Plan" adopted previously by the City Council; and

WHEREAS, At the request of City Council, The Citizen's Water Committee has prepared such amendments; and

WHEREAS, The Council has reviewed said amendments, found them to be consistent with the goals and objectives of the City, and considers adoption to be in the best interest of the citizens of Brady.

BE IT THEREFORE RESOLVED BY THE CITY COUNCIL OF THE CITY OF BRADY, TEXAS, that the "Water Conservation Plumbing Amendment" attached hereto be and is hereby adopted.

PASSED AND APPROVED this the _______________ day of__________, 1990.

____
H. L. GOBER, JR.
MAYOR

ATTEST:

____
DORENE PATTERSON, CITY SECRETARY

WATER CONSERVATION PLUMBING AMENDMENTS

The following standards are to be used for new construction and remodeling of bathroom areas:

101 - General

Automatic flushing devices of the siphonic design shall not be used to operate urinals

102 - Water Closets

Water closets, either flush tank or flushometer operated, shall be designed, manufactured and installed to be operable and adequately flushed with no more than 3.5 gallons per flushing cycle.

103 - Urinals

Urinals shall be designed, manufactured and installed to be operable and adequately flushed with no more than 1.5 gallons per flush.

104 - Lavatory Facilities

104.1 - Public Facilities

Faucets for public lavatories shall be equipped with outlet devices that have self-closing valves to limit the flow of water.

104.2 - Private Facilities

Faucets for private lavatories shall have aerators attached that reduce the flow of water delivered.

105 - Shower Heads

Shower heads shall be installed with flow restriction devices to limit the flow of water delivered.

APPENDIX B

Radon Analysis Results

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PEG-03 MCCULLOCH

WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

BRADY CITY OF

P O BOX 351
BRADY TX 76825

WATER SUPPLY #: 1540001
LABORATORY NO: EP100230
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 3A 371 HICKORY 1:15 PM
SOURCE: WELL #3
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON NOT TESTED - LAB ACC		PCI/L	

**COPY - CENTRAL OFFICE*
REG-03 MCCULLOCH

WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

BRADY CITY OF

P O BOX 351
BRADY TX 76825

WATER SUPPLY #: 1540001
LABORATORY NO: EP100231
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 3B 371 HICKORY 1:15 PM
SOURCE: WELL #3
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	668.00	PCI/L	44.0

**COPY - CENTRAL OFFICE*
REG-03 MCCULLOCH

WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

BRADY CITY OF

P O BOX 351
BRADY TX 76825

WATER SUPPLY #: 1540001
LABORATORY NO: EP100232
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 4A 371 HICKORY 1:27PM
SOURCE: WELL #4
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	535.00	PCI/L	38.0

COPY - CENTRAL OFFICE
REG-03 MCCULLOCH

WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

BRADY CITY OF

P O BOX 351
BRADY TX 76825

WATER SUPPLY #: 1540001
LABORATORY NO: EP100233
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 4B 371 HICKORY 1:28PM
SOURCE: WELL #4
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	576.00	PCI/L	39.0

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WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

BRADY CITY OF

P O BOX 351
BRADY TX 76825

WATER SUPPLY #: 1540001
LABORATORY NO: EP100234
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 5A 371 HICKORY 2:41PM
SOURCE: WELL #5
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	587.00	PCI/L	39.0

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WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

BRADY CITY OF

P O BOX 351
BRADY TX 76825

WATER SUPPLY #: 1540001
LABORATORY NO: EP100235
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 5B 371 HICKORY 2:41PM
SOURCE: WELL #5
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	641.00	PCI/L	44.0

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WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

BRADY CITY OF

P O BOX 351
BRADY TX 76825

WATER SUPPLY #: 1540001
LABORATORY NO: EP100236
SAMPLE TYPE: PLANT

COLLECTOR REMARKS: 5A-WD 371 HICKORY 2:22PM
SOURCE: WELL #5
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON NOT TESTED - LAB ACC		PCI/L	

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TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
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AUSTIN, TEXAS 78756

BRADY CITY OF

P O BOX 351
BRADY TX 76825

WATER SUPPLY #: 1540001
LABORATORY NO: EP100237
SAMPLE TYPE: PLANT

COLLECTOR REMARKS: 5B-WD 371 HICKORY 2:22PM
SOURCE: WELL #5
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON NOT TESTED - LAB ACC		PCI/L	

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DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

BRADY CITY OF

P O BOX 351
BRADY TX 76825

WATER SUPPLY #: 1540001
LABORATORY NO: EP100236
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 7A 371 HICKORY 1:42PM
SOURCE: WELL #7
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON NOT TESTED - LAB ACC		PCI/L	

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WATER ANALYSIS REPORT
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AUSTIN, TEXAS 78756

BRADY CITY OF

P O BOX 351
BRADY TX 76825

WATER SUPPLY #: 1540001
LABORATORY NO: EP100239
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 7B 371 HICKORY 1:43PM
SOURCE: WELL #7
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON NOT TESTED - LAB ACC		PCI/L	

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WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

BRADY LAKE WATER SYSTEM
C/O CITY OF BRADY
PO BOX 351
BRADY TX 76825

WATER SUPPLY #: 1540005
LABORATORY NO: EP100242
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 6A 371 HICKORY 3:15PM
SOURCE: WELL #6
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	456.00	PCI/L	39.0

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WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

BRADY LAKE WATER SYSTEM
C/O CITY OF BRADY
PO BOX 351
BRADY TX 76825

WATER SUPPLY #: 1540005
LABORATORY NO: EP100243
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 6B 371 HICKORY 3:15PM
SOURCE: WELL #6
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	506.00	PCI/L	40.0

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WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

BRADY LAKE WATER SYSTEM
C/O CITY OF BRADY
PO BOX 351
BRADY TX 76825

WATER SUPPLY #: 1540005
LABORATORY NO: EP100244
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 8A 371 HICKORY 3:07PM
SOURCE: WELL #8
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	558.00	PCI/L	44.0

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WATER ANALYSIS REPORT
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DIVISION OF WATER HYGIENE
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BRADY LAKE WATER SYSTEM
C/O CITY OF BRADY
PO BOX 351
BRADY TX 76825

WATER SUPPLY #: 1540005
LABORATORY NO: EP100245
SAMPLE TYPE: DISTRIBUTION

COLLECTOR REMARKS: 8B 371 HICKORY 3:07PM
SOURCE: WELL #8
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	663.00	PCI/L	51.0

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REG-03 CONCHO

WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

EDEN CITY OF

P O BOX 915
EDEN TX 76837

WATER SUPPLY #: 0480001
LABORATORY NO: EP100316
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 1A EDEN 12:05PM
SOURCE: WELL #1
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	696.00	PCI/L	46.0

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WATER ANALYSIS REPORT

TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

EDEN CITY OF

P O BOX 915
EDEN TX 76837

WATER SUPPLY #: 0480001
LABORATORY NO: EP100317
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 1B EDEN 12:05PM
SOURCE: WELL #1
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	647.00	PCI/L	46.0

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EDEN CITY OF

P O BOX 915
EDEN TX 76837

WATER SUPPLY #: 0480001
LABORATORY NO: EP100318
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 2A EDEN 11:50AM
SOURCE: WELL #2
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	422.00	PCI/L	37.0

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WATER ANALYSIS REPORT
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EDEN CITY OF

P O BOX 915
EDEN TX 76837

WATER SUPPLY #: 0480001
LABORATORY NO: EP100319
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 2B EDEN 11:50AM
SOURCE: WELL #2
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	378.00	PCI/L	38.0

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WATER ANALYSIS REPORT
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EDEN CITY OF

P O BOX 915
EDEN TX 76837

WATER SUPPLY #: 0480001
LABORATORY NO: EP100320
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 3A EDEN 12:18PM
SOURCE: WELL #3
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	811.00	PCI/L	56.0

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WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

EDEN CITY OF

P O BOX 915
EDEN TX 76837

WATER SUPPLY #: 0480001
LABORATORY NO: EP100321
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 3B EDEN 12:18PM
SOURCE: WELL #3
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	736.00	PCI/L	49.0

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WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

EDEN CITY OF

P O BOX 915
EDEN TX 76837

WATER SUPPLY #: 0480001
LABORATORY NO: EP100322
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 4A EDEN 12:24PM
SOURCE: WELL #4
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	720.00	PCI/L	51.0

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WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

EDEN CITY OF

P O BOX 915
EDEN TX 76837

WATER SUPPLY #: 0480001
LABORATORY NO: EP100323
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: RADON)48 EDEN 12:24PM
SOURCE: WELL #4
DATE COLLECTED / / DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	611.00	PCI/L	45.0

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REG-03 CONCHO

WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

EDEN CITY OF

P O BOX 915
EDEN TX 76837

WATER SUPPLY #: 0480001
LABORATORY NO: EP100324
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 5A EDEN 12:29PM
SOURCE: WELL #5
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	672.00	PCI/L	48.0

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REG-03 CONCHO

WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

EDEN CITY OF

P O BOX 915
EDEN TX 76837

WATER SUPPLY #: 0480001
LABORATORY NO: EP100325
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 5B EDEN 12:29PM
SOURCE: WELL #5
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	634.00	PCI/L	44.0

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WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

EDEN CITY OF

P O BOX 915
EDEN TX 76837

WATER SUPPLY #: 0480001
LABORATORY NO: EP100326
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 6A EDEN 12:50PM
SOURCE: WELL #6
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	176.00	PCI/L	26.0

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AUSTIN, TEXAS 78756

EDEN CITY OF

P O BOX 915
EDEN TX 76837

WATER SUPPLY #: 0480001
LABORATORY NO: EP100327
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 6B EDEN 12:50PM
SOURCE: WELL #6
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	166.00	PCI/L	25.0

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EDEN CITY OF

P O BOX 915
EDEN TX 76837

WATER SUPPLY #: 0480001
LABORATORY NO: EP100328
SAMPLE TYPE: PLANT

COLLECTOR REMARKS: 10A EDEN COOLING TOWER EF
SOURCE: WELL #2&6
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	< 100.00	PCI/L	

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EDEN CITY OF

P O BOX 915
EDEN TX 76837

WATER SUPPLY #: 0480001
LABORATORY NO: EP100329
SAMPLE TYPE: PLANT

COLLECTOR REMARKS: 108 EDEN COOLING TOWER EF
SOURCE: WELL #2&6
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME		RESULT	UNITS	+/-
RADON	<	100.00	PCI/L	

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WATER ANALYSIS REPORT
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LIVE OAK HILLS SUBDIVISION
C/O M J WOOTEN
108 WEST GRAYSON
LLANO TX 78643

WATER SUPPLY #: 1540012
LABORATORY NO: EP100240
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 1A LIVE OAK 4:25PM
SOURCE: WELL #1
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	547.00	PCI/L	43.0

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WATER ANALYSIS REPORT
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LIVE OAK HILLS SUBDIVISION
C/O M J WOOTEN
108 WEST GRAYSON
LLANO TX 78643

WATER SUPPLY #: 1540012
LABORATORY NO: EP100241
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 13 LIVEOAK 4:25PM
SOURCE: WELL #1
DATE COLLECTED 9/ 1/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	522.00	PCI/L	43.0

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WATER ANALYSIS REPORT
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AUSTIN, TEXAS 78756

NORTH SAN SABA WSC
P.O.BOX 568

SAN SABA TX 76877

WATER SUPPLY #: 2060003
LABORATORY NO: EP100228
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 1A 371 HICKORY 10:22 AM
SOURCE: WELL #1
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	770.00	PCI/L	51.0

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REG-01 SAN SABA

WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
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AUSTIN, TEXAS 78756

NORTH SAN SABA WSC
P.O.BOX 568

SAN SABA TX 76877

WATER SUPPLY #: 2060003
LABORATORY NO: EP100229
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 1B 371 HICKORY 10:23 AM
SOURCE: WELL #1
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	726.00	PCI/L	47.0

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WATER ANALYSIS REPORT
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DIVISION OF WATER HYGIENE
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AUSTIN, TEXAS 78756

RICHLAND WSC - BRADY SYSTEM
C/O LARRY COTTRELL - PRESIDENT
P O BOX 217
RICHLAND SPRING TX 76877

WATER SUPPLY #: 1540008
LABORATORY NO: EP100224
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: #1A 371 HICKORY 8:47 AM
SOURCE: WELL #1
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	610.00	PCI/L	40.0

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WATER ANALYSIS REPORT
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DIVISION OF WATER HYGIENE
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RICHLAND WSC - BRADY SYSTEM
C/O LARRY COTTRELL - PRESIDENT
P O BOX 217
RICHLAND SPRING TX 76877

WATER SUPPLY #: 1540008
LABORATORY NO: EP100225
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 1B 371 HICKORY 8:48AM
SOURCE: WELL #1
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	700.00	PCI/L	45.0

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WATER ANALYSIS REPORT
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DIVISION OF WATER HYGIENE
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AUSTIN, TEXAS 78756

MELVIN CITY OF
PO BOX 777

MELVIN TX 76858

WATER SUPPLY #: 1540003
LABORATORY NO: EP100312
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 1A MELVIN 10:29AM
SOURCE: WELL #1
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	343.00	PCI/L	32.0

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DIVISION OF WATER HYGIENE
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AUSTIN, TEXAS 78756

MELVIN CITY OF
PO BOX 777

MELVIN TX 76858

WATER SUPPLY #: 1540003
LABORATORY NO: EP100313
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 1B MELVIN 10:29AM
SOURCE: WELL #1
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	293.00	PCI/L	30.0

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WATER ANALYSIS REPORT
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MELVIN CITY OF
PO BOX 777

MELVIN TX 76858

WATER SUPPLY #: 1540003
LABORATORY NO: EP100314
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 2A MELVIN 10:48AM
SOURCE: WELL #2
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	< 100.00	PCI/L	

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AUSTIN, TEXAS 78756

MELVIN CITY OF
PO BOX 777

MELVIN TX 76858

WATER SUPPLY #: 1540003
LABORATORY NO: EP100315
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 28 MELVIN 10:48AM
SOURCE: WELL #2
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	< 100.00	PCI/L	

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WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
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MILLERSVIEW-DOOLE WSC
C/O V C WHITWORTH - PRESIDENT
P O BOX 121
MILLERSVIEW TX 76862

WATER SUPPLY #: 0480015
LABORATORY NO: EP100306
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 2A MILLERSVILLE DOOLE
SOURCE: WELL #2
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	547.00	PCI/L	39.0

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WATER ANALYSIS REPORT
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MILLERSVIEW-DOOLE WSC
C/O V C WHITWORTH - PRESIDENT
P O BOX 121
MILLERSVIEW TX 76862

WATER SUPPLY #: 0480015
LABORATORY NO: EP100307
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 2B MILLERSVILLE DOOLE
SOURCE: WELL #2
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	598.00	PCI/L	41.0

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WATER ANALYSIS REPORT
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MILLERSVIEW-DOOLE WSC
C/O V C WHITWORTH - PRESIDENT
P O BOX 121
MILLERSVIEW TX 76862

WATER SUPPLY #: 0480015
LABORATORY NO: EP100308
SAMPLE TYPE: PLANT

COLLECTOR REMARKS: 3A-WD MILLERSVILLE DOOLE
SOURCE: WELL #3
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	311.00	PCI/L	34.0

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MILLERSVIEW-DOOLE WSC
C/O V C WHITWORTH - PRESIDENT
P O BOX 121
MILLERSVIEW TX 76862

WATER SUPPLY #: 0480015
LABORATORY NO: EP100309
SAMPLE TYPE: PLANT

COLLECTOR REMARKS: 38-WD MILLERSVILLE DOOLE
SOURCE: WELL #3
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	287.00	PCI/L	31.0

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MILLERSVIEW-DOOLE WSC
C/O V C WHITWORTH - PRESIDENT
P O BOX 121
MILLERSVIEW , TX 76862

WATER SUPPLY #: 0480015
LABORATORY NO: EP100310
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 3A MILLERSVILLE DOOLE
SOURCE: WELL #3
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	634.00	PCI/L	45.0

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WATER ANALYSIS REPORT
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DIVISION OF WATER HYGIENE
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MILLERSVIEW-DOOLE WSC
C/O V C WHITWORTH - PRESIDENT
P O BOX 121
MILLERSVIEW TX 76862

WATER SUPPLY #: 0480015
LABORATORY NO: EP100311
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 3B MILLERSVILLE DOOLE
SOURCE: WELL #3
DATE COLLECTED 9/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	576.00	PCI/L	41.0

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WATER ANALYSIS REPORT
TEXAS DEPARTMENT OF HEALTH
DIVISION OF WATER HYGIENE
1100 WEST 49 TH STREET
AUSTIN, TEXAS 78756

ROCHELLE WATER SUPPLY CORPORATION
C/O RONNIE NOWLIN - PRESIDENT
P O BOX 191
ROCHELLE TX 76872

WATER SUPPLY #: 1540004
LABORATORY NO: EP100226
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 1A 371 HICKORY 9:14AM
SOURCE: WELL #1
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	422.00	PCI/L	34.0

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AUSTIN, TEXAS 78756

ROCHELLE WSC
P.O.BOX 191

ROCHELLE TX 76872

WATER SUPPLY #: 1540005
LABORATORY NO: EP100305
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: #2B ROCHELLE 8:27AM
SOURCE: WELL #2
DATE COLLECTED 2/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	207.00	PCI/L	25.0

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ROCHELLE WATER SUPPLY CORPORATION
C/O RONNIE NOWLIN - PRESIDENT
P O BOX 191
ROCHELLE TX 76872

WATER SUPPLY #: 1540004
LABORATORY NO: EP100227
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 1B 371 HICKORY 9:15 AM
SOURCE: WELL
DATE COLLECTED 9/10/90 DATE RECEIVED 9/11/90 DATE REPORTED 9/19/90

CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	639.00	PCI/L	46.0

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ROCHELLE WATER SUPPLY CORPORATION
C/O RONNIE NOWLIN - PRESIDENT
P O BOX 191
ROCHELLE TX 76872

WATER SUPPLY #: 1540004
LABORATORY NO: EP100304
SAMPLE TYPE: RAW SAMPLE

COLLECTOR REMARKS: 2A ROCHELLE 8:27AM
SOURCE: WELL #2
DATE COLLECTED 2/12/90 DATE RECEIVED 9/13/90 DATE REPORTED 9/19/90

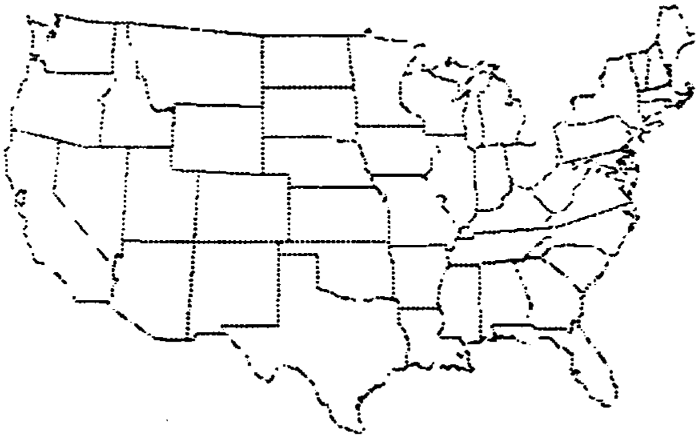
CONSTITUENT NAME	RESULT	UNITS	+/-
RADON	168.00	PCI/L	23.0

APPENDIX C

EPA Fact Sheet: National Primary Drinking Water Regulations for Lead and Copper, May, 1991

Fact Sheet:

National Primary Drinking Water Regulations for Lead and Copper



May 1991

Office of Ground Water and Drinking Water
U.S. Environmental Protection Agency
Washington, DC

Summary

	MCLG (mg/L)	Action Level (mg/L)*
Lead	0	0.015
Copper	1.3	1.3

*Measured in 90th percentile at consumers' taps

These regulations will:

- Establish Maximum Contaminant Level Goals (MCLGs) of zero for lead and 1.3 milligrams per liter (mg/L) for copper.
- Establish a treatment technique requirement for lead that includes:
  1) optimal corrosion control treatment,
  2) source water treatment,
  3) public education, and
  4) lead service line replacement.
- Establish a treatment technique requirement for copper that includes:
  1) optimal corrosion control treatment, and
  2) source water treatment.
- The treatment technique requirements are triggered by exceedances of the lead action level of 0.015 mg/L or the copper action level of 1.3 mg/L measured in the 90th percentile.

These final rulemakings establish:

- Two additional National Primary Drinking Water Regulations (NPDWRs), bringing the total to 58.
- Two additional inorganic standards, bringing the total to 13.

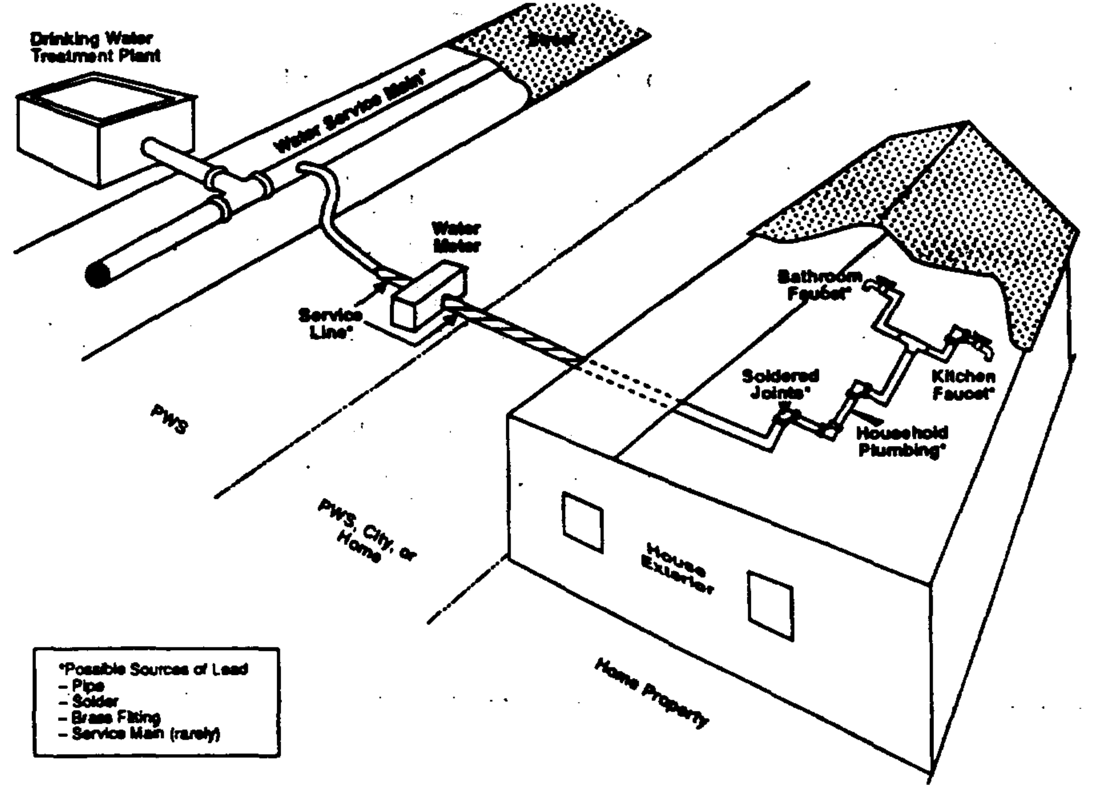
These rules also include provisions for:

- Analytical methods and laboratory performance requirements;
- Best Available Technologies (BAT) for complying with the treatment technique requirements;
- Mandatory health effects language to be used by systems when notifying the public of violations;
- System recordkeeping and reporting requirements; and
- State recordkeeping, reporting and primacy requirements.

Health Effects and Sources of Lead and Copper

Contaminant	Low Level Health Effects	Sources in Drinking Water
Lead	**Children:** Altered physical and mental development; interference with growth; deficits in IQ, attention span, and hearing; interference with heme synthesis **Women:** Increased blood pressure; shorter gestational period **Men:** Increased blood pressure	**Corrosion of:** Lead solder and brass faucets and fixtures Lead service lines (20% of public water sytems) **Source water** (1% of systems)
Copper	Stomach and intestinal distress; Wilson's Disease	**Corrosion of:** Interior household and building pipes **Source water** (1% of systems)

Public Water System (PWS) and Homeowner Plumbing



Regulatory Impact

Benefits

- These regulations will:
  - reduce the exposure of approximately 130 million people to lead in drinking water.
  - result in an additional 570,000 children having their blood lead level reduced to below 10 micrograms per deciliter (µg/dl).
- Nationwide health benefits when translated into avoided medical costs are estimated to be:
  - between $2.8 and $4.3 billion per year for corrosion control and source water treatment.
  - between $70 and $240 million per year for lead service line replacement.
- Nationwide material benefits attributable to reduced corrosion of water distribution systems and household plumbing systems are estimated to be $500 million per year.

Costs

- Capital costs are estimated to be between $2.9 and $7.6 billion.
- Operation and maintenance costs are estimated to be $240 million per year.
- Total annualized costs are estimated to be between $500 and $790 million per year.
- For large systems (serving >50,000 persons), corrosion control treatment required by this rule is estimated to cost $1 per household per year.
- For smaller systems (serving ≤ 50,000 persons), corrosion control treatment is estimated to cost $2 to $20 per household per year.
- Total annualized costs for lead service line replacement are estimated to be $80 to $370 million per year.
- Tap water monitoring will be required for 79,000 community and nontransient non-community water systems.
- Monitoring costs are estimated to be:
  - $40 million per year nationwide.
  - $0.10 per household per year for large systems and less than $3 per household per year for smaller systems.
- State implementation costs are estimated to be $40 million per year.

Treatment Technique Requirements

Corrosion Control Treatment

- Systems must collect tap water samples for lead and copper from high risk homes.

Corrosion Control Studies

- Systems triggered into the corrosion control treatment requirements may first have to conduct studies to compare the effectiveness of:
  - pH and alkalinity adjustment (reduces the acidity of the water);
  - calcium adjustment (promotes the formation of protective coatings inside pipes and plumbing); and
  - addition of phosphate or silica-based corrosion inhibitor (forms protective coating inside pipes and plumbing).
- All large water systems (serving >50,000 people) are required to conduct such studies.
- Small and medium-size water systems (serving ≤50,000 people) that exceed the lead or copper action level are required to first submit a recommendation for optimal corrosion control treatment to the State.
- The State will either approve the recommended treatment or require the installation of an alternative treatment. The State may, as an alternative, require small and medium-size water systems to conduct the corrosion control treatment studies described above.
- Any system that conducts corrosion control studies must recommend an optimal corrosion control treatment to the State on the basis of study results and monitoring data.
- States will either approve a system's recommendation or designate an alternative treatment as optimal.

Corrosion Control Treatment

- Once treatment is specified by the State, systems will have 24 months to install optimal corrosion control treatment and 12 months to collect follow-up samples.
- States will assign values for a set of water quality parameters that constitute optimal corrosion control treatment:
  - pH;
  - alkalinity;
  - calcium, when carbonate stabilization is used;
  - orthophoshate, when an inhibitor with a phosphate compound is used; and
  - silica, when an inhibitor with a silicate compound is used.
- A system must continue to operate within the water quality parameters established by the State.

Source Water Treatment

- All public water systems that exceed the tap water lead or copper action level must collect source water samples and submit the data with a treatment recommendation to the State.
- States may specify one of the following treatments, or an alternative treatment at least as effective, for the system to install: 1) ion exchange, 2) reverse osmosis, 3) lime softening or 4) coagulation/filtration.
- Once treatment is specified by the State, systems will have 24 months to install source water treatment and 12 months to collect follow-up source water samples.
- States will review follow-up source water monitoring results and assign maximum permissible lead and copper concentrations in source water entering the distribution system.
- Systems must continue to deliver water to all entry points in the distribution system that does not exceed the maximum permissible lead and copper concentrations established by the State.
- Source water monitoring will be standardized to 3/6/9 year cycles after treatment is installed or the State determines no treatment is necessary.

Treatment Technique Requirements

Public Education

- Informs the public about the adverse health effects of lead and explains the steps people can take in their homes to reduce their exposure to lead in drinking water (i.e., flushing the tap; cooking with cold water rather than hot; checking for lead solder in new plumbing; and testing their water for lead).
- All public water systems exceeding the lead action level must deliver the EPA-developed public education program to their customers within 60 days.
- Every 12 months, systems must deliver:
    - bill stuffers to their customers and brochures to all institutions in the community frequented by women and children (i.e., health departments, hospitals, clinics, etc.), and
    - the public education material to the editorial departments of major newspapers serving the community.
- Every 6 months, systems must submit a public service announcement on lead in drinking water to major television and radio stations serving the community.
- Every 12 months, non-transient non-community water systems must post information notices in each building served by the system and deliver brochures to all of the system's customers.
- The public education program must be delivered by a water system for as long as the system exceeds the lead action level.

Lead Service Line Replacement

- All public water systems that continue to exceed the lead action level after installing optimal corrosion control treatment and source water treatment must replace lead service lines that contribute in excess of 15 parts per billion (ppb) to total tap water lead levels.
- A system must replace seven percent of its lead lines each year or demonstrate that the lines not replaced contribute less than 15 ppb of lead to drinking water at the tap.
- A system must replace the entire lead service line unless it can demonstrate that it does not control the entire line. Water systems must offer to replace the owner's portion of the service line.
- A system that exceeds the lead action level after installing optimal corrosion control treatment and source water treatment has 15 years to replace all lead service lines.

Tap Water Monitoring

Lead and Copper

Start Dates for Monitoring

January 1992	Large Systems (>50,000)
July 1992	Medium-size Systems (3,300 to ≤ 50,000)
July 1993	Small Systems (≤ 3,300)

Monitoring Period

1 monitoring period is equivalent to 6 months

(2 per calendar year, i.e., January to June and July to December)

Sample Site Location

- Tap water samples must be collected at high risk locations:
  - homes with lead solder installed after 1982,
  - homes with lead pipes,
  - homes with lead service lines.

Sample Collection Methods

- First flush tap water samples must stand motionless for at least six hours before the samples are collected.
- One liter of water must be drawn from the cold water kitchen or bathroom tap.
- Systems may collect samples or enlist residents to collect samples. Residents fill the container supplied by the water system according to directions and leave the container for the system to pick up.

Tap Water Monitoring

Lead and Copper

Number and Frequency

Base Monitoring:

- All public water systems are required to collect one sample for lead and copper analysis from the following number of sites during each six month monitoring period.

Table 1. Tap Sampling for Lead and Copper

System Size (Population)	No. of Sampling Sites (Initial Base Monitoring)	No. of Sampling Sites (Reduced Monitoring)
>100,000	100	50
10,001 to 100,000	60	30
3,301 to 10,000	40	20
501 to 3,300	20	10
101 to 500	10	5
≤100	5	5

Reduced Monitoring:

- All public water systems that meet the lead and copper action levels or maintain optimal corrosion control treatment for two consecutive six month monitoring periods may reduce the number of tap water sampling sites (see Table 1) and their collection frequency to once per year.
- All public water systems that meet the lead and copper action level, or maintain optimal corrosion control treatment, for three consecutive years may reduce the tap water sampling sites (see Table 1) and their collection frequency to once every three years.

Monitoring

Water Quality Parameters (other than lead and copper)

- In addition to lead and copper, all large water systems, and those small and medium-size systems that exceed the lead or copper action level, will be required to monitor for the following water quality parameters:
  - pH,
  - alkalinity,
  - calcium,
  - conductivity,
  - orthophosphate,
  - silica, and
  - water temperature.
- These parameters are used to identify optimal treatment, and once treatment is installed, to determine whether a system remains in compliance with the rule.
- Systems are required to maintain water quality parameters within State specifed ranges.
- Systems will have to monitor water quality parameters at two separate locations:
  - representative taps throughout the distribution system (systems can use total coliform sampling sites), and
  - entry points to the distribution system.

Base Monitoring:

- All large water systems (>50,000) and those small and medium-size water systems (≤50,000) that exceed the lead or copper action level must collect two tap samples for each applicable water quality parameter from the following number of sites during each six month monitoring period.

Table 2. Tap Sampling for Water Quality Parameters

System Size (Population)	No. of Tap Sampling Sites (Initial Base Monitoring)	No. of Tap Sampling Sites (Reduced Monitoring)
>100,000	25	10
10,001 to 100,000	10	7
3,301 to 10,000	3	3
501 to 3,300	2	2
101 to 500	1	1
≤100	1	1

Monitoring

Water Quality Parameters (other than lead and copper)

- All large water systems, and those small and medium-size water systems that exceed the lead or copper action level, must collect one sample for each applicable water quality parameter at each entry point to the distribution system every two weeks.

- All large water systems, and those small and medium-size systems that exceed the lead or copper action level, after installing optimal corrosion control treatment must continue to collect:

    - two samples for each applicable water quality parameter at each of the sampling sites specified above every six months, and

    - one sample for each applicable water quality parameter at each entry point to the distribution system every two weeks.

Reduced Monitoring:

- All water systems that maintain State-specified water quality parameters reflecting optimal corrosion control for two consecutive six month monitoring periods may reduce the number of tap samples collected (see Table 2) during each six month monitoring period.

- All water systems that maintain State-specified water quality parameters reflecting optimal corrosion control for three consecutive years may reduce the number of tap samples collected (see Table 2) and the frequency with which they are collected to once per year.

Monitoring Schedules

Lead and Copper Monitoring

Monitoring Period	Parameters [1]	Location	Frequency
Initial Monitoring	Lead and Copper	Targeted high risk interior taps	Every six months[1,2]
After Installation of Corrosion Control	Lead and Copper	Targeted high risk interior taps	Two consecutive six month monitoring periods
After State Specifies Parameter Values for Optimal Corrosion Control	Lead and Copper	Targeted high risk interior taps	Two consecutive six month monitoring periods[3]
Reduced Monitoring	Lead and Copper	Targeted high risk interior taps	Once every year[4]
	Lead and Copper	Targeted high risk interior taps	Once every three years

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[1] Large systems collect tap water samples for two-six month monitoring periods before conducting comparative corrosion control treatment studies.

[2] Small and medium-size systems collect tap water samples for lead and copper until they exceed the lead or copper action level and are triggered into the treatment technique requirement.

[3] Systems meeting the lead and copper action level, or maintaining optimal corrosion control treatment specified by the State, for two consecutive six month monitoring periods may reduce tap water sampling to once per year and collect the reduced number of samples indicated in Table 1 on page 7.

[4] Systems meeting the lead and copper action level, or maintaining optimal corrosion control treatment specified by the State, for three consecutive years may reduce tap water sampling to once every three years and collect the reduced number of samples indicated in Table 1 on page 7.

Monitoring Schedules

Water Quality Parameter Monitoring

Monitoring Period	Parameters [1]	Location	Frequency
Initial Monitoring	pH, alkalinity, orthophosphate or silica [2], calcium, conductivity, temperature	Taps and at entry point(s) to distribution system	Every six months
After Installation of Corrosion Control	pH, alkalinity, orthophosphate or silica [2], calcium [3]	Taps	Every six months
	pH, alkalinity dosage rate and concentration (if alkalinity adjusted as part of corrosion control), inhibitor dosage rate and inhibitor residual [4]	Entry point(s) to distribution system	Biweekly
After State Specifies Parameter Values for Optimal Corrosion Control	pH, alkalinity, orthophosphate or silica [2], calcium [3]	Taps	Every six months
	pH, alkalinity dosage rate and concentration (if alkalinity adjusted as part of corrosion control), inhibitor dosage rate and inhibitor residual [4]	Entry point(s) to distribution system	Biweekly
Reduced Monitoring	pH, alkalinity, orthophosphate or silica [2], calcium [3]	Taps[5,6]	Every six months
	pH, alkalinity dosage rate and concentration (if alkalinity adjusted as part of corrosion control), inhibitor dosage rate and inhibitor residual [4]	Entry point(s) to distribution system	Biweekly

---

[1] Small and medium-size systems have to monitor for water quality parameters only during monitoring periods in which the system exceeds the lead or copper action level. Large systems must monitor water quality parameters during each monitoring period.

[2] Orthophosphate must be measured only when an inhibitor containing a phosphate compound is used. Silica must be measured only when an inhibitor containing silicate compound is used.

[3] Calcium must be measured only when calcium carbonate stabilization is used as part of corrosion control.

[4] Inhibitor dosage rates and inhibitor residual concentrations (orthophosphate or silica) must be measured only when an inhibitor is used.

[5] Systems maintaining optimal corrosion control treatment specified by the State for two consecutive six-month monitoring periods may reduce tap water sampling of water quality parameters to once per year and collect the reduced number of samples indicated in Table 2 on page 8.

[6] Systems maintaining optimal corrosion control treatment specified by the State for three consecutive years may reduce tap water sampling of water parameters to once every three years and collect the reduced number of samples indicated in Table 2 on page 8.

Regulatory Schedule for Large Systems (>50,000 people)

Date	Activities (system activities denoted by ▶)
May 1991	National Primary Drinking Water Regulations (NPDWRs) for lead and copper promulgated
January 1992	▶ Begin tap water and distribution system monitoring
November 1992	▶ Treatment technique requirements take effect
January 1993	▶ Complete tap water and distribution system monitoring
July 1994	▶ Complete corrosion control studies and recommend treatment to State
January 1995	State designates optimal corrosion control treatment
January 1997	▶ Complete installation of corrosion control treatment
January 1998	▶ Complete follow-up monitoring and submit results to State[1]
July 1998	State designates water quality parameters

---

[1] Systems that continue to exceed action level begin 15-year lead service line replacement program.

Regulatory Schedule for Medium-size Systems (3,300 to ≤50,000 people)

Date	Activities (system activities denoted by ▶)
May 1991	National Primary Drinking Water Regulations (NPDWRs) for lead and copper promulgated
July 1992	▶ Begin tap water monitoring
November 1992	Treatment technique requirements take effect
January 1993	▶ Recommend optimal corrosion control treatment[1]
January 1994	State requires system to conduct corrosion control studies[2]
July 1994	State designates optimal corrosion control treatment[3]
July 1995	▶ Complete corrosion control studies and recommend treatment to State[2]
January 1996	State designates optimal corrosion control treatment for system conducting treatment studies[2]
July 1996	▶ Complete installation of corrosion control treatment[3]
July 1997	▶ Complete follow-up monitoring and submit results to State[3,4]
January 1998	▶ Complete installation of corrosion control treatment[2]
	State designates water quality parameters[3]
January 1999	▶ Complete follow-up monitoring and submit results to State[2,4]
July 1999	State designates water quality parameters[2]

[1] Assumes system exceeds lead or copper action level during first monitoring period.

[2] Medium-size systems required to conduct comparative treatment studies.

[3] Medium-size systems which State specifies optimal treatment without studies.

[4] Systems that continue to exceed action level begin 15-year lead service line replacement program.

Regulatory Schedule for Small Systems (≤3,300 people)

Date	Activities (system activities denoted by ▶)
May 1991	National Primary Drinking Water Regulations (NPDWRs) for lead and copper promulgated
November 1992	Treatment technique requirements take effect
July 1993	▶ Begin tap water monitoring
January 1994	▶ Recommend optimal corrosion control treatment to State[1]
January 1995	State requires system to conduct corrosion control studies[2]
January 1996	State designates optimal corrosion control treatment[3]
July 1996	▶ Complete corrosion control studies and recommend treatment to State[2]
January 1997	State designates optimal corrosion control treatment [2]
January 1998	▶ Complete installation of corrosion control treatment[3]
January 1999	▶ Complete installation of corrosion control treatment[2]
	▶ Complete follow-up monitoring and submit results to State[3,4]
July 1999	State designates water quality parameters[3]
January 2000	▶ Complete follow-up monitoring and submits results to State[2,4]
July 2000	State designates water quality parameters[2]

[1] Assumes system exceeds lead or copper action level during first monitoring period.

[2] Small systems required to conduct comparative treatment studies.

[3] Small systems which State specifies optimal treatment without studies.

[4] Systems that continue to exceed action level begin 15-year lead service line replacement program.

For More Information

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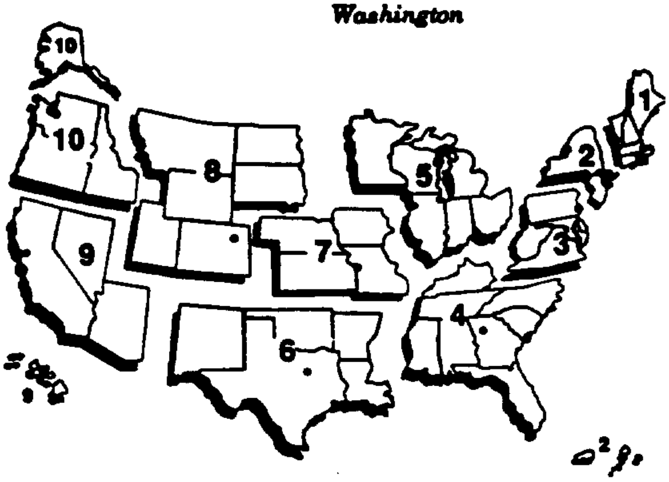


EPA Safe Drinking Water Hotline
☎ 1-800-426-4791

APPENDIX D

EPA Fact Sheet: National Primary Drinking Water Regulations for Radionuclides, June, 1991

United States
Environmental Protection
Agency

Office of Water
WH-550

570 9-91-700
June 1991

EPA

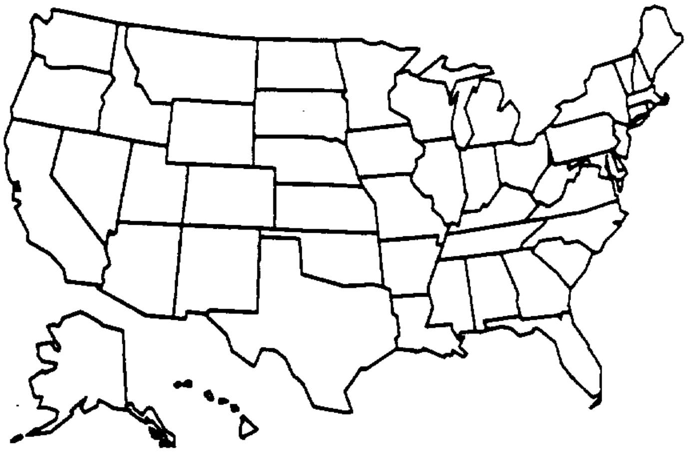
Radionuclides in Drinking Water

F•A•C•T•S•H•E•E•T

National Primary Drinking Water Regulations for Radionuclides

Proposed Rule

June 1991



**Office of Ground Water and Drinking Water
U.S. Environmental Protection Agency
Washington, DC**

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SUMMARY

These regulations will:

Propose Maximum Contaminant Level Goals (MCLGs) and Maximum Contaminant Levels (MCLs) for four radionuclide contaminants and two categories of radionuclides.

- The four radionuclide contaminants are: radium-226, radium-228, radon-222, and uranium.
- The two categories of radionuclides are adjusted gross alpha emitters, and beta and photon emitters.
  - The category of adjusted gross alpha emitters regulates alpha emitters and is gross alpha measurement less uranium and less radium-226.
  - The category of beta and photon emitters regulates total beta and photon emitters (mostly man-made).

When this proposed rulemaking is final:

- These rules will establish
  - Four revised radionuclide standards; and
  - Two new radionuclide standards (radon and uranium) for a total of six.

These proposed rules also include additional provisions for:

- Monitoring, analytic methods and laboratory requirements;
- Best Available Technologies (BATs) for compliance with the MCLs and for the purpose of issuing variances:
  - aeration
  - reverse osmosis
  - anion exchange
  - ion exchange
  - coagulation/filtration
- Mandatory health effects language to be used by systems when notifying the public of violations;
- State reporting, recordkeeping and primacy requirements; and
- Unregulated contaminant monitoring for lead-210, the first long-lived progeny of radon-222.

EPA is seeking comment from the public in this proposed rulemaking on:

- A number of alternative MCLs for radon, uranium, and radium; and
- A variety of technical and policy issues.

Key Dates	
June 1991	Standards for 4 radionuclide contaminants and 2 categories of radionuclides proposed
September 1991	**Public hearings in Washington, D.C. and Chicago, IL**
September 1991 (approximate)	Close of public comment period (90 days after Federal Register publication)
April 1993 (22 months after proposal)	**Standards for 4 radionuclide contaminants and 2 categories of radionuclides promulgated**
October 1994 (18 months after promulgation)	• Standards for 4 radionuclide contaminants and 2 categories of radionuclides effective • State adoption complete
January 1996	**First monitoring compliance period begins**

Regulatory Impact

As proposed, these regulations, when implemented, will reduce the exposure of 20 million consumers to the regulated contaminants and result in avoidance of an estimated 83 cancer cases per year.

- This includes:
  - Avoidance of an estimated 80 cancer cases per year due to reduced radon exposure of an estimated 17 million customers of public water systems served by ground-water sources;
  - Avoidance of an estimated 3 cancer cases per year due to radium; and
  - Reduced exposure of approximately 875,000 people to kidney toxicity risk.
- Regulation of radon is expected to result in the most violations, increased cost to public water systems, and provide the greatest health benefits.

National Costs of Proposed Radionuclide MCLs

	Rn-222	Ra-226	Ra-228	Uranium	AGA*	Beta Emitters	TOTAL
Proposed MCL**	300	20	20	20***	15	4****	
Systems Affected	26,000	70	40	1,500	130	0	28,000
Treatment Cost							
Total Capital ($M)	1,600	190	40	350	230	0	2,400
Annual O&M ($M)	70	20	3	30	20	0	140
Total Annual Cost ($M)	180	30	6	60	40	0	310
Monitoring ($K/Yr)*****	5,000	3	860	3	640	250	6,800
State Implementation							
Initial ($M)	NA						15-28
Annual ($M)	NA						10-19
Annual Household Cost by System Size							
Very Small (25-500)	120	630	650	580	770	0	
Small (501-3,300)	30	150	150	180	340	0	
Medium (3,301-10,000)	6	90	90	80	200	0	
Large (over 10,000)	5	60	60	40	140	0	

** Adjusted Gross Alpha.*

*** MCLs are expressed in pCi/L unless otherwise noted.*

**** MCL for Uranium is expressed in µg/L.*

***** MCL for Beta Emitters is expressed in millirems effective dose equivalent per year (mrem ede/yr).*

****** Gross Alpha is used as a screen for Radium-226 and uranium.*

Total Costs

- Total costs to all public water systems will be approximately $317 million per year. Approximately 75% of these costs will be borne by systems serving fewer than 10,000 people.

- State implementation costs will be $15-28 million initially and $10-19 million in future years.

Monitoring and Treatment

- Monitoring requirements will be standardized, with monitoring required every three, six, or nine years depending on the system's vulnerability to the particular contaminant.

- Nationally, it will cost approximately $7 million per year for systems to monitor.

- Nationally, monitoring for lead-210 will cost systems an addition $8 million one-time cost.

- Monitoring for radionuclides will be required for approximately 80,000 systems.

  ### Ground Water

  - Approximately 68,000 community and non-transient non-community public water systems with ground-water or mixed surface and ground-water sources must monitor for radon, radium-226, radium-228, uranium, and adjusted gross alpha emitters. Vulnerable systems also must monitor for beta and photon emitters.

  ### Surface Water

  - Approximately 12,000 community systems and non-transient non-community public water systems with surface water sources must monitor for radium-226, radium-228, uranium, and adjusted gross alpha emitters. Vulnerable systems also must monitor for beta and photon emitters.

- Nationally, approximately 28,000 or 35% of affected public water systems will be required to provide treatment or find an alternative source of water.

  - Treatment will cost approximately $3 to $800 per household annually depending upon system size, degree of contamination, and other factors.

  - It will cost systems $310 million per year to provide treatment.

  - At State's option, extendable exemptions based on costs may be allowed for systems with less than 500 service connections, as long as the level does not exceed unreasonable risk to health and alternative sources are not feasible.

Radionuclide National Primary Drinking Water Regulations

		EPA Standards					
Contaminants	**Drinking Water Health Effects**	**Proposed MCLG**	**Proposed MCL[1]**	**Current MCL**	**Sources**	**Analytic Method**	**BAT[2]**
Radium-226	cancer	zero	20 pCi/l	5 pCi/l combined with radium-228	naturally occurring	RE; RC	IE; LS; RO
Radium-228	cancer	zero	20 pCi/l	5 pCi/l combined with radium-226	naturally occurring	RC; LS	IE; LS; RO
Radon-222	cancer	zero	300 pCi/l	-	naturally occurring	LS; LC	AER
Uranium	kidney, cancer	zero	20 µg/l[3]	-	naturally occurring	RC; FL; AS	C/F; AE; LS; RO[4]
Adjusted gross alpha emitters	cancer	zero	15 pCi/l	15 pCi/l	naturally occurring and man-made	GA/B; GA	RO
Gross beta and photon emitters	cancer	zero	4 mrem ede/yr	4 mrem/yr any organ or whole body	man-made and naturally occurring		IE; RO
- radioactive cesium						PREC	
- radioactive iodine						PREC	
- radioactive strontium 89.90						PREC; RC	
- titium						LS	
- gamma and photon emitters						GRS	

Analytic Methods Key:

RE = Radon Emanation
RC = Radiochemical
LS = Liquid Scintillation
LC = Lucas Cell
FL = Fluorometric
AS = Alpha Spectrometry
GA/B = Gross alpha and/or beta activity
GA = Gross alpha activity
PREC = Precipitation
GRA = Gamma Ray Spectrometry

Best Available Technology Key:

IE = Ion Exchange
LS = Lime Softening
RO = Reverse Osmosis
AER = Aeration
C/F = Coagulation/Filtration
AE = Anion Exchange

[1] *pCi/l is an activity measurement of radioactive decay (1 pCi = 2.2 disintegrations per minute); µg/l is a mass measurement; mrem is measurement of effective radiation dose to organs).*

[2] *Except as noted, BAT for the purpose of issuing variances is the same as BAT for compliance.*

[3] *20 µg/l is based on kidney toxicity. 20 µg/l is the equivalent of 30 pCi/l.*

[4] *Coagulation/Filtration and Lime Softening are not BAT for small systems (those with less than 500 connections) for the purpose of granting variances.*

Summary of Proposed Monitoring Requirements for Radionuclides

Contaminant	Base Requirements		System is out of compliance IF:	Increased Frequency Triggered by Non-compliance	Waiver Conditions	Reduced Frequency (with waiver)
	Initial	Repeat				
Gross Beta* (Vulnerable systems)	Quarterly	Quarterly	Average of 1 sample + 1 mandatory confirmation sample > MCL.	Monthly if out of compliance. Return to base when 3 mo. < MCL.	None	None
Tritium, Strontium-90	Annually	Annually				
Radon - GW	YR. 1 - INITIAL PERIOD	Annually	YR. 1 - INITIAL PERIOD	YR. 1 - INITIAL PERIOD	State Discretion; Consistently Meeting MCL	Waiver reduces to 1 per 3-year period
	Quarterly		Annual average of quarterly samples > MCL	Continue quarterly if out of compliance. Annually when average of 4 quarters < MCL		
	YRS 2 & 3 - INIT. PER.		ALL OTHER YEARS	ALL OTHER YEARS		
	Annually		1 sample > MCL	Quarterly if out of compliance. Return to annually when average of 4 quarters < MCL.		
Gross Alpha** **Ra-226, Ra-228, Uranium** (Compositing is permitted)	GW/SW- Annually	GW/SW - 1 per 3-year period	Average of 1 sample + 1 mandatory confirmation sample > MCL	GW/SW - Annually if out of compliance. Return to 1 per 3-year period when 3 years < MCL	State Discretion; Consistently Meeting MCL	Waiver reduces to 1 per 9-year period
Unregulated Contaminant Pb-210	One Sample	N/A	N/A	N/A	N/A	N/A

* Gross beta is a screen for Ra-228 and an MCL standard.

** Gross alpha is a screen for Ra-226 and uranium, and an MCL standard.

Radon in Water

The proposed radon regulation will provide a substantial public health benefit compared with other drinking water regulations and other environmental regulatory programs administered by EPA.

- The regulation will reduce the exposure to radon of 17 million consumers whose household water comes from a public water system served by ground-water sources.

- The regulation, when implemented, may result in an estimated avoidance of about 80 cancer cases per year.

Radon comes from the natural break-down of uranium in soil, rock and water.

- Radon is a volatile gas and is not a problem in drinking water from surface water sources.

- Radon may be present in drinking water from ground-water sources.

  - Public water supplies with ground-water or mixed ground and surface water sources would be required to monitor for radon and to provide treatment or find an alternative source of water if the radon MCL is exceeded.

  - Radon also may be present in drinking water from private wells not regulated by EPA.

The primary health hazard comes from breathing air containing radon.

- Breathing air containing radon can damage lung tissue and increase the risk of lung cancer.

  - Radon in water generally accounts for about 5% of the total indoor air concentration in homes with ground-water sources of drinking water.

  - EPA estimates that indoor radon may result in 8,000-40,000 lung cancer deaths annually.

  - The higher the level of radon, the greater the risk of developing lung cancer.

  - Smokers exposed to radon may have up to ten times the risk of contracting lung cancer as never-smokers exposed to the same radon levels.

  - Radon present in drinking water is released into indoor air during household water use such as showering and washing clothes.

  - As an average, every 10,000 pCi/l radon in water contributes about 1 pCi/l radon to indoor air for an average house.

  - On average, outdoor levels of radon range from 0.2 pCi/l to 0.7 pCi/l.

- Drinking water contaminated with radon may add some minor risk of developing stomach or other internal organ cancer. This risk is small compared to the risks from breathing indoor air containing radon.

EPA is taking action to protect public health from radon in air and radon in water to the highest degree possible, given existing mitigation technologies.

- Central water treatment for radon by public water systems is affordable and technologically feasible. Under the Safe Drinking Water Act, EPA is proposing to regulate radon at a level public water systems can achieve—300 pCi/l. Radon at this level contributes about 0.03 pCi/l of radon in air. This corresponds to an estimated lifetime cancer risk of approximately 2×10^{-4} (2 in 10,000) and is consistent with EPA's risk guidelines for regulating drinking water contaminants. This risk may vary for individuals.

- Concerning radon in air, EPA has proposed to keep its current action level of 4 pCi/l, corresponding to an estimated lifetime risk of $1\text{-}5 \times 10^{-2}$ (1-5 in 100) because the Agency believes that level can be achieved consistently and economically in homes, given current technology. The new *Citizen's Guide*, scheduled for publication in Fall 1991, also will provide information on testing methods to determine radon levels in homes.

All homes should be tested for radon in air.

- Most radon in household air comes from soil gas which seeps into the home through the foundation.

- Information about radon testing and mitigation of household air can be found in the following EPA publications:

  - *A Citizen's Guide to Radon: What It Is and What To Do About It*; and
  - *Radon Reduction Methods: A Homeowner's Guide.*

- Information about radon in air can be requested by calling 1-800-SOS-RADON or by contacting the State Radon office.

Some homes with private wells also should be tested for radon in water.

- After testing the air for radon and finding levels above EPA's current action guideline of 4 pCi/l, homeowners with private wells should consult with their State drinking water office to obtain information on laboratories to test for radon in water, and test the water. After testing, homeowners should select the mitigation strategy that is most cost-effective for reducing radon exposure for the individual home.
  - In the majority of cases, the appropriate mitigation strategy will be controlling the soil gas contribution to indoor air before treating the water to remove radon.
- If treatment of radon in private wells is appropriate, the most effective treatment is to remove the radon from the water before it enters the home. This is called point-of-entry (POE) treatment.
- Two basic types of POE water treatment are available.
  - **Aeration Systems**

    Forced air bubblers remove radon gas from the water.
  - **Granular Activated Carbon (GAC) Systems**

    Filters remove radon from the water. Homes with high levels of radon in water should not use GAC to remove radon.
- Treatment at the tap is called point-of-use (POU) treatment. POU treatment will fail to reduce the most important risk from radon in water, breathing radon in indoor air transferred from water.
- Information about radon in drinking water can be requested by calling the Safe Drinking Water Hotline (1-800-426-4791) or contacting the State Drinking Water Office.

FOR MORE INFORMATION ABOUT RADON IN INDOOR AIR

EPA Radon Information Hotline
☎ 1-800-SOS-RADON

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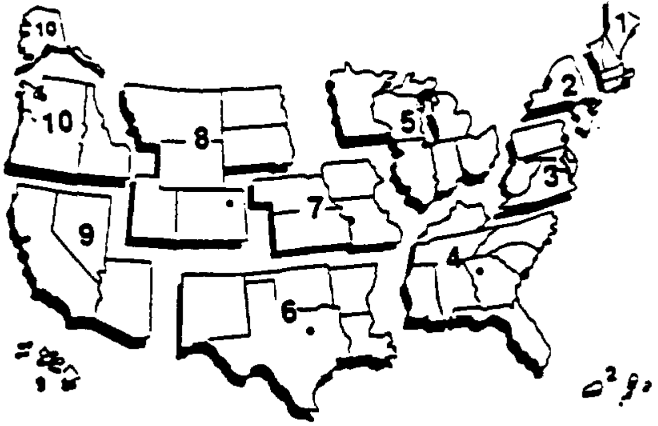

FOR MORE INFORMATION ABOUT RADIONUCLIDES AND RADON IN DRINKING WATER

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Protection Branch
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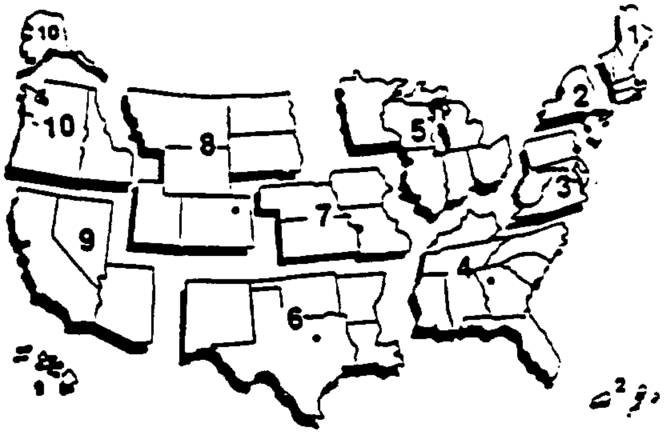

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APPENDIX E

Selected Articles on Radium and Radon Treatment

Removing Barium and Radium Through Calcium Cation Exchange

Anthony G. Myers, Vernon L. Snoeyink, and David W. Snyder

The removal of barium (Ba) and radium (Ra), which are found in many groundwater sources, was achieved in laboratory studies with an ion exchange process. In the studies, a strong acid resin in the calcium form effectively removed Ba^{2+} and Ra^{2+} to meet standards. The resin was regenerated with reclaimed brine, which would reduce disposal problems in actual practice. Total hardness, alkalinity, pH, and other water quality parameters remained unchanged. Additional research is needed, however, to refine the process and establish the costs.

This article describes an ion exchange process that can be used to remove barium (Ba) and radium (Ra) from water. When all of the water must be treated by ion exchange to achieve the maximum contaminant level (MCL) of 1 mg Ba/L or 5 picoCuries (pCi)/L of Ra^{226} plus Ra^{228}, this process can be used to produce a water that is lower in sodium and less corrosive. Additionally, a smaller quantity of spent brine is produced that must be disposed of because a portion of the brine can be treated and reused. The process, shown in Figure 1, uses a calcium (Ca)-form strong acid ion exchange resin column in parallel with a conventional sodium (Na)-form strong acid resin column. The divalent Ra^{2+}, Ba^{2+}, Ca^{2+}, and magnesium (Mg^{2+}) ions are exchanged for Na^+ in the portion of the flow that passes through the Na-form column. If present, Ba^{2+} and Ra^{2+} are exchanged for Ca^{2+} in the fraction of water passing through the Ca-form column, and the total hardness in this fraction is essentially unchanged. Calcium chloride brine ($CaCl_2$) is used to regenerate the Ca-form column, and the spent brine can be reused after precipitating and removing the Ba and Ra. Barium and Ra in the spent sodium chloride (NaCl) brine can also be precipitated and removed before disposal, but this brine cannot be recycled in its entirety because of the depletion of Na^+ ions when it is used. In addition to describing the process, the data presented in this article show how the Ca-form column should be operated and what procedure should be followed to reclaim the $CaCl_2$ brine.

Barium in excess of the MCL is found in some waters in northern Illinois and Iowa.[1] Concentrations of 2–7 mg Ba/L are typical; a high of 18–20 mg Ba/L has been reported. Radium, another groundwater contaminant, occurs as Ra^{226} and Ra^{228} and is found more frequently in groundwater than Ba. Groundwater sources in Illinois, Iowa, Florida, Texas, Wisconsin, and some Rocky Mountain states contain Ra in excess of the MCL. Typical concentrations are 5–50 pCi/L.[2-5]

Lime softening, reverse osmosis, and ion exchange are effective treatment processes for Ba and Ra removal.[1,6] However, because of the costs involved and ease of operation, many small communities choose ion exchange or no treatment at all.

Although strong acid Na cation exchange resins remove Ra and Ba along with hardness ions, the effluent is corrosive because its Ca concentration is very low.[7,8] To avoid corrosion problems, many communities blend raw water with Na cation exchange column effluent to stabilize the finished water.[6] For waters containing high concentrations of Ba or Ra, this blending procedure may cause drinking water to exceed the MCL; in such situations the proposed process may be of use.

The research reported in this article was conducted to develop operating parameters for the Ca cation exchange resin column and the procedures for reclaiming spent Ca brine. The performance of the process was then determined through several cycles of exhaustion, regeneration, and brine reclamation.

Materials and methods

Column studies. The strong acid cation exchange resin used in this study was a polystyrene-based resin,* cross-linked with divinylbenzene with sulfonate functional groups. This resin was placed in 2.45-cm ID glass columns with PTFE stopcocks. Heat-resistant glass wool was used to support the resin. Water and brine solutions were pumped through the column with a variable-speed pump.† Influent water was prepared by mixing calcium carbonate ($CaCO_3$), magnesium carbonate ($MgCO_3$), sodium carbonate (Na_2CO_3), and hydrated barium chloride ($BaCl_2 \cdot 2H_2O$) with deionized water in a 230-gal (870-L) stainless steel tank. Carbon dioxide and nitrogen gas were used to adjust the pH; nitrogen was used to strip carbon dioxide when it was necessary to raise the pH. When Ra was used instead of Ba, it was added by disposable pipette from a 1-μCi/L Ra stock solution, preserved in 4 *M* hydrochloric acid (HCl). Effluent water was collected in a 53-gal (200-L) polystyrene tank. Regeneration brine was prepared by mixing hydrated

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*Duolite C-20, Diamond Shamrock, Redwood City, Calif.
†Fluid Metering Inc., Oyster Bay, N.Y.

calcium chloride ($CaCl_2 \cdot 2H_2O$) with deionized water.

Jar tests. A synthetic spent brine was prepared from $CaCl_2 \cdot 2H_2O$, $BaCl_2 \cdot 2H_2O$, and deionized water. Powdered hydrated calcium sulfate ($CaSO_4 \cdot 2H_2O$) was added to this brine and mixed in 1-L glass beakers with a six-paddle stirrer. The $CaSO_4$ dissolved to provide sulfate (SO_4^{2-}) for barium sulfate ($BaSO_4$) precipitation. The mixture was vacuum filtered through 0.45-μm membrane filters.

Chemical analyses. Barium was analyzed by atomic absorption spectrophotometry. Sodium chloride was added to each sample in concentrations of 1000–2000 mg/L to suppress ionization. Samples containing high concentrations of Ca were diluted to less than 0.017 *M* Ca to avoid interferences. All samples were analyzed at concentrations less than or equal to 10 mg Ba/L. Calcium and hardness were analyzed by the ethylenediaminetetraacetate (EDTA) titrimetric method and alkalinity by HCl titration.[9] Magnesium was analyzed by atomic absorption spectrophotometry. In some instances, the concentration of Mg was determined by subtracting Ca and Ba concentrations from total hardness. The pH readings were made with a pH meter. Sulfate was measured with an ion chromatograph.‡ Conductivity was measured with a portable conductivity meter. Readings were taken in micromhos per centimetre and multiplied by 0.64 to convert to milligrams per litre of total dissolved solids (TDS). Radium analyses were made at the University of Iowa Hygienic Laboratory. The procedure involved determination of Ra by alpha counting after precipitation with $BaSO_4$. Lead (Pb) and Ba carriers were added to the sample containing alkaline citrate, then sulfuric acid was added to precipitate Ra, Ba, and Pb as sulfates. The precipitate was purified by washing with nitric acid, dissolving in alkaline EDTA, and reprecipitating as Ra–Ba sulfate after adjusting the pH to 4.5.[10]

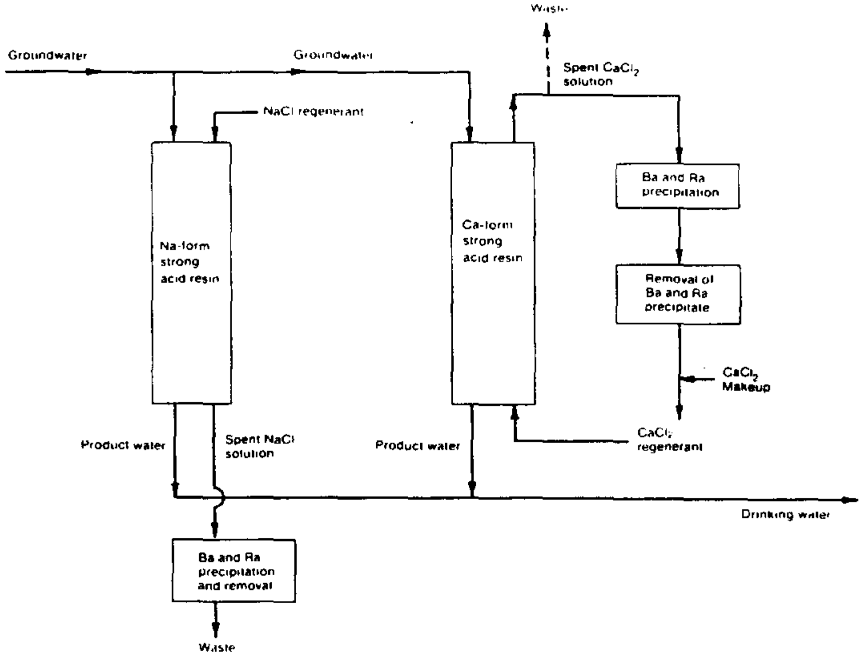

Figure 1. Process schematic

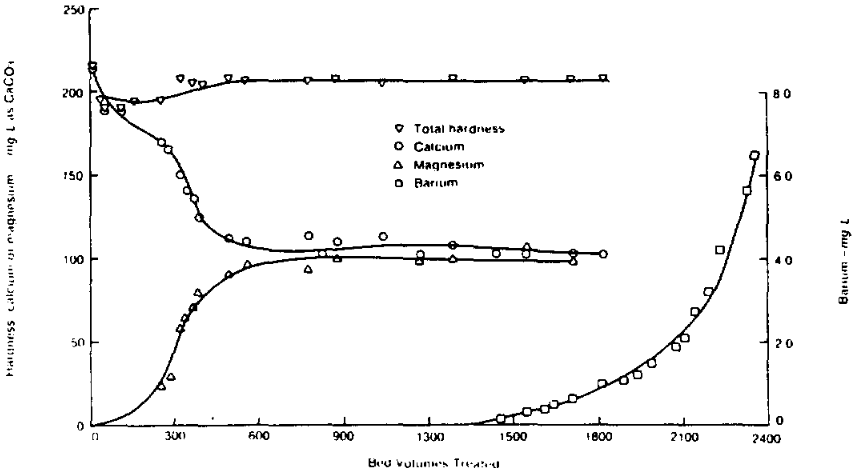

Figure 2. Divalent cations in the effluent from a virgin strong acid calcium-form resin column

Results and discussion

Calcium cation exchange column effluent. The influent water described in Table 1 was pumped downflow through 22 in. (56 cm) of resin in the Ca form at a loading rate of 2.5 gpm/sq ft (6.2 m/h). This equates to a loading of 1.35 gpm/cu ft (2.97 L/s/m³). The effluent composition for this run is shown in Figure 2. Barium was removed well in exchange for Ca, which is consistent with the high selectivity of the strong acid resin for Ba over Ca;[12] Mg was also removed. The total quantity of Mg removed was about the same as the quantity of Ba removed, even though the concentration of Mg was about seven times that of Ba. This behavior of the resin is consistent with its preference for Ba over Mg. The removal of Mg can be expected only in the first few cycles of use if the $CaCl_2$ brine is to be reused, because Mg will build up in the brine. Total hardness, pH, and alkalinity were nearly unchanged throughout the run.

The effect of empty bed contact time (EBCT) on the virgin resin breakthrough curve for Ba^{2+} is shown in Figure 3. Column lengths were 4.3 in. (11 cm), 21.8 in. (55.5 cm), and 22 in. (56 cm); loading rates were 2.1 gpm/sq ft (5.1 m/h), 5.4 gpm/sq ft (13.2 m/h), and 2.5 gpm/sq ft (6.1 m/h), respectively. Empty bed contact times were 1.3 min, 2.5 min, and 5.5 min, respectively.

In order to plot the results on the same graph, the number of bed volumes (BV) of water processed through the 21.8-in. (55.5-cm) column was multiplied by the concentration ratio 20:15. The results show that the column utilization, i.e., the capacity of the column used by Ba compared with the total capacity of the column at an EBCT of 1.3 min, was about half that at 2.5 min and 5.5 min. Thus, the zone of active mass transport in the column at 1.3 min EBCT occupies most of the column, whereas at the longer times, it is a small portion of the total length of the column. The column capacities for Ba at a breakthrough concentration of 1 mg/L were 32, 67, and 64 mg Ba/g dry resin (H^+ form) for the 4.3-in. (11-cm), 21.8-in. (55.5-cm), and 22-in. (56-cm) columns, respectively.

Calcium regenerant concentration. A 4.3-in. (11-cm) resin column was set up to determine the optimum concentration of

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[‡] Dionex Autolon System 12 Analyzer, Santa Clara, Calif.

$CaCl_2$ for regeneration. Influent water with a composition similar to that shown in Table 1 was applied downflow at a rate of 2.1 gpm/sq ft (5.1 m/h), thus giving an EBCT of 1.3 min. Brine was pumped countercurrent (upflow) at 0.26 gpm/sq ft (0.65 m/h). The EBCT of the brine was 10 min. Rinse water (19.5 BV) was applied upflow. Approximately 37 mg Ba/g dry resin (H^+ form) and 14.6 g Ba/L resin (Ca^{2+} form) were on the resin prior to regeneration. The resin column was regenerated with various concentrations and dosages of $CaCl_2$. Figure 4 shows how the $CaCl_2$ dosage and concentration affect the amount of Ba in the effluent after regeneration. Five BV were processed after regeneration before the effluent samples were taken. A $CaCl_2$ concentration of 0.85 *M* produced the lowest effluent concentration of Ba over the range of regenerant dosages. Both higher and lower regenerant concentrations were less effective.

The existence of an optimum $CaCl_2$ concentration is not surprising because such an optimum exists for the NaCl brine that is used to regenerate strong acid resin.[13] There may be several reasons for the existence of an optimum. At a relatively low concentration of brine, the concentration of Ca^{2+} may not be sufficiently high to drive the regeneration reaction

$$R\text{-}Ba + Ca^{2+} \rightleftharpoons R\text{-}Ca + Ba^{2+}$$

far to the right, thus reducing regeneration efficiency. At a high concentration of brine, the resin will shrink. This shrinking takes place from the outside of the resin bead and moves toward the center of the bead as the brine migrates into the resin.[13] The Ba^{2+} from the interior of the resin bead must then migrate through narrower pores after exchange has taken place. The rate of diffusion through narrow pores is less than the rate through wide pores,[11] and thus the removal of Ba^{2+} is impeded. Other factors that may contribute to reduced efficiency at a high concentration of $CaCl_2$ are contact time and viscosity. The contact time for a given dosage decreases as the concentration of brine increases, so less time is allowed for removal of Ba^{2+}, and the opportunity for short-circuiting increases. Short-circuiting is also likely to increase as the viscosity of the brine increases with higher concentrations.

Calcium regenerant dosages. Experiments were then conducted to determine the capacity of the Ca-form resin for Ba after different dosages of 0.85 *M* $CaCl_2$ regenerant. A 21.6-in. (55-cm) resin column was used, which is approximately the same as resin depths used in practice. The influent water was pumped downflow at 5.7 gpm/sq ft (14.2 m/h), thus giving an EBCT of 2.3 min. The water composition was similar to that given in Table 1, except that the Ba^{2+} concentration was 23 mg/L. Countercurrent regeneration was carried out at 0.26 gpm/sq ft (0.65 m/h), a contact of 0.14 gpm/cu ft (0.3 L/s/m^3), and 5.7 BV of rinse water were applied at a rate of 0.54 to 1.6 gpm/sq ft (1.3 to 3.9 m/h). The EBCT for the brine was 51 min. After each regeneration, the resin was reloaded to a total of approximately 70 mg Ba/g dry resin (H^+ form), or 30.6 g Ba/L resin (Ca^{2+} form).

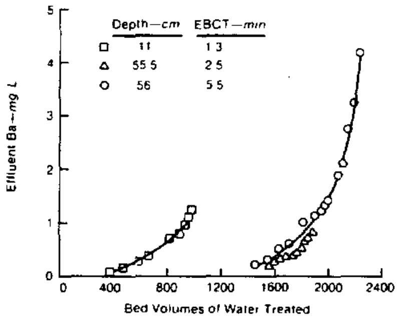

Figure 3. Effect of contact time and column length on barium breakthrough curves

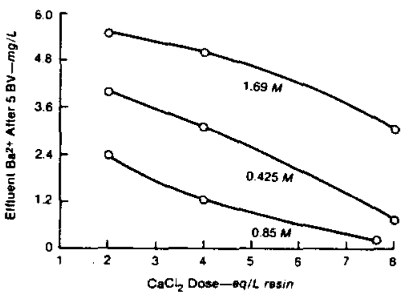

Figure 4. Barium leakage versus regenerant dosage at various $CaCl_2$ concentrations

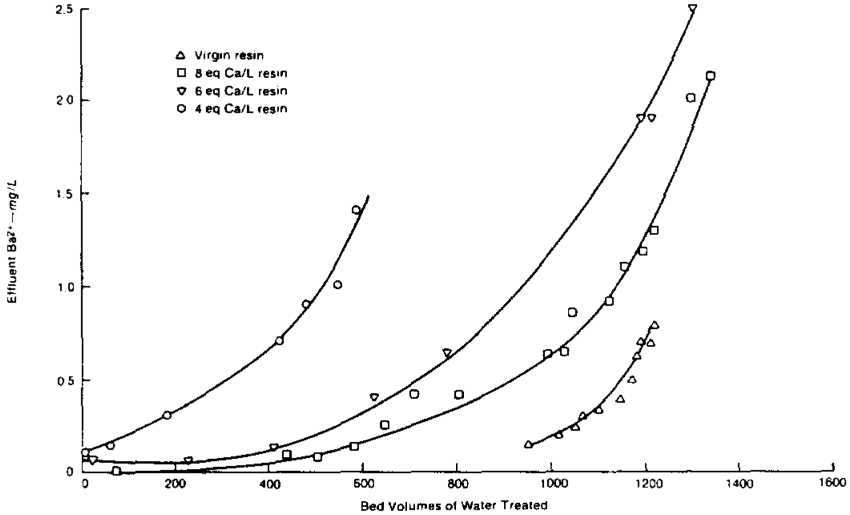

Figure 5. Barium breakthrough curves after different regenerant dosages

Barium breakthrough curves after various regenerant dosages are shown in Figure 5. The influent Ba concentration to the virgin resin column was 20 mg/L and to all other columns it was 23 mg/L. As shown in the figure, the capacity of the column for Ba increases as the regenerant dosage increases. It is also of interest to note the effect of contact time on column performance. The data in Figure 4 shows that the effluent concentration for the 4.3-in. (11-cm) column was about 1.1 mg/L after a regenerant dose of 4 eq/L of resin, whereas the data in Figure 5 show an effluent concentration immediately after regeneration of 0.1 mg/L for the 21.6-in. (55-cm) column regenerated with the same dose. The EBCTs for the process cycle were 1.3 and 2.3 min, respectively. The longer contact time for the process cycle undoubtedly contributed to the lower leakage from the longer column.

Another difference between the results for the 4.3-in. (11-cm) column shown in Figure 3 and the results for the 21.6-in. (55-cm) column shown in Figure 4 is brine contact time. At a regenerant dose of 4 eq Ca^{2+}/L resin, the brine EBCT for

TABLE 1

Composition of influent water

Parameter	Concentration
Total hardness	215 mg as $CaCO_3$/L
Calcium	105 mg as $CaCO_3$/L
Magnesium	95 mg as $CaCO_3$/L
Barium	15 mg/L
Sodium	~23 mg/L
Chloride	~10 mg/L
Total alkalinity	250 mg as $CaCO_3$/L
pH	7

TABLE 2

Barium removal from calcium brine by addition of $CaSO_4$

$CaSO_4$:Ba *mol ratio*	Ba^{2+} Removal *percent*		$CaSO_4$ Used *percent*		SO_4^{2-} Remaining After 60 min *mg/L*
	30 min	60 min	30 min	60 min	
1.00	95.12	95.93	95.12	95.93	2.5
1.02	97.90	98.93	95.98	96.99	8.0
1.04	97.08	99.72	93.35	95.88	83
1.06	98.05	99.63	92.50	93.99	24
1.08	98.22	99.70	90.94	92.31	28

TABLE 3

Barium removal from brines with different calcium concentrations

Calcium—*M*	$CaSO_4$:Ba *mol ratio*	Ba^{2+} Removal *percent*
0.4	1.02	87.23
0.6	1.02	90.55
0.8	1.02	89.92

TABLE 4

Barium removal from brines with different barium concentrations

Barium—*g/L*	$CaSO_4$:Ba *mol ratio*	Ba^{2+} Removal *percent*
4	1.02	98.83
6	1.02	95.17
8	1.02	93.45
10	1.02	78.39

TABLE 5

Wastewater composition in cyclic column runs

Parameter	Rinse Water Only		Brine Plus Rinse Water	
	Average	Range	Average	Range
TDS—*mg/L*	4476	4571-4931	44 516	42 085-52 013
Barium—*mg/L*	179	112-250	2 818	2 168-3 591
Wastewater produced—*gal/100 gal (L/100 L) treated*	0.49	0.37-0.67	0.74	0.58-0.98

the 4.3-in. (11-cm) column was 10 min and for the 21.6-in. (55-cm) column was 51 min. Regeneration efficiency, i.e., the equivalents of Ba^{2+} removed per equivalents of Ca^{2+} applied, increased from 3.7 percent to 8.6 percent with the increase in EBCT. However, the 21.6-in. (55-cm) column contained 73 mg Ba^{2+}/g dry resin before regeneration, whereas the 4.3-in. (11-cm) column contained only 32 mg Ba^{2+}/g dry resin. Therefore, more Ba^{2+} was left on the 21.6-in. (55-cm) column after regeneration (19 mg Ba/g dry resin) than on the 4.3-in. (11-cm) column (9.6 mg Ba/g dry resin). In spite of this, more leakage occurred in the 4.3-in. (11-cm) column because of the shorter EBCT of the process cycle and the lower utilization of the column.

Analysis of the data in Figure 5 shows that as the regenerant dosage increases, the regeneration efficiency decreases and resin capacity increases. This relationship is shown in Figure 6. Dosages of Ca regenerant ranged from 18.3-36.7 lb $CaCl_2$/cu ft resin (293-588 kg $CaCl_2/m^3$ resin), which can be compared with typical Na-form column regenerant dosages of 5-20 lb NaCl/cu ft (80-320 kg $NaCl/m^3$).[13] Lower $CaCl_2$ dosages would increase efficiency, but excessive leakage of Ba would likely occur.

The importance of brine contact time is illustrated by the curves in Figure 7. The 21.6-in. (55-cm) columns were regenerated with loading rates of 0.27 gpm/sq ft (0.65 m/h), a contact of 0.14 gpm/cu ft (0.3 $L/s/m^3$) with an EBCT of 51 min; and with a loading rate of 1.4 gpm/sq ft (3.5 m/h), a contact of 0.75 gpm/cu ft (1.65 $L/s/m^3$) with an EBCT of 10 min.

Barium removal from calcium brines. The high concentrations of Ba^{2+} in the $CaCl_2$ brine prevent brine reuse, and treatment may be necessary before disposal. Removal of the Ba^{2+} as $BaSO_4$ precipitate is possible, however, and if this can be done, reuse may be possible. If SO_4^{2-} is added to the brine, $BaSO_4$, which is very insoluble ($K_{SO} = 10^{-10}$), can be formed. However, addition of SO_4^{2-} in the form of a salt, such as Na_2SO_4, will cause the solubility product of both $BaSO_4$ and $CaSO_4$ ($K_{SO} = 10^{-4.59}$) to be exceeded initially, thus increasing the Na_2SO_4 required and the quantity of sludge produced. Because the spent brine is slightly undersaturated with $CaSO_4$, solid $CaSO_4$ can be added; as it dissolves, the SO_4^{2-} required to precipitate the Ba^{2+} becomes available, and some Ca^{2+} enters solution. A supersaturated solution of $CaSO_4$ cannot be produced by this means, however. After separation of solids, the brine will be suitable for reuse if SO_4^{2-} and Ba^{2+} concentrations are low. The proper $CaSO_4$ dosage will remove most of the Ba^{2+}, but the sulfate concentration must be low so that the resin will not be fouled during regeneration.

Solid $CaSO_4$ was added to spent brine in 1-L jar tests in order to determine the optimum dosage and reaction conditions. Table 2 shows Ba^{2+} removal at various $CaSO_4$ dosages at 30- and 60-min and mixing conditions at 100 rpm. The brine originally contained 0.6 *M* Ca^{2+} and 6000 mg Ba^{2+}/L. Each mixing jar contained 300 mL brine. The Ba^{2+} and SO_4^{2-} concentrations remaining after precipitation are shown in Figure 8 as a function of the $CaSO_4$:Ba^{2+} mol ratio. Separate jar tests at 40 rpm showed much less Ba removal. For example, a $CaSO_4$ to Ba mol ratio of 1.1:1 removed only 62.5 percent of the Ba after 60 min of mixing.

Thus, mixing is important for good Ba removal. Poor removal at the slow mixing rate was likely to have been caused by a slower rate of $CaSO_4$ dissolution when there was little turbulence. It is also expected that the smaller the particle size of the $CaSO_4$, the more rapidly $BaSO_4$ will be precipitated.

The concentration of SO_4^{2-} rises sharply as the dosage of $CaSO_4$ exceeds that required to precipitate Ba^{2+} as shown in Figure 8. The SO_4^{2-} concentration is controlled by SO_4^{2-} solubility—a maximum of about 680 mg/L. The desired molar dosage of $CaSO_4$ is about 10 percent greater than the initial concentration of Ba^{2+}. At this point the SO_4^{2-} concentration is about 100 mg/L.

Table 3 shows that the concentration of Ca in the brine does not have a major effect on Ba removal. The original Ba concentration was the same as in Table 2, but Ba removal is slightly lower because of less vigorous mixing. These data indicate that concentration of the brine Ca will not be a major factor in

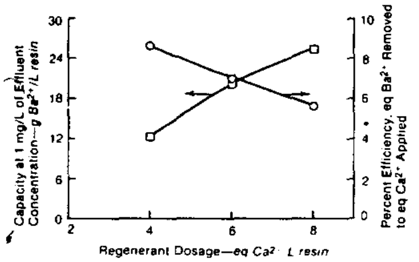

Figure 6. Regeneration efficiency and column capacity at various regenerant dosages

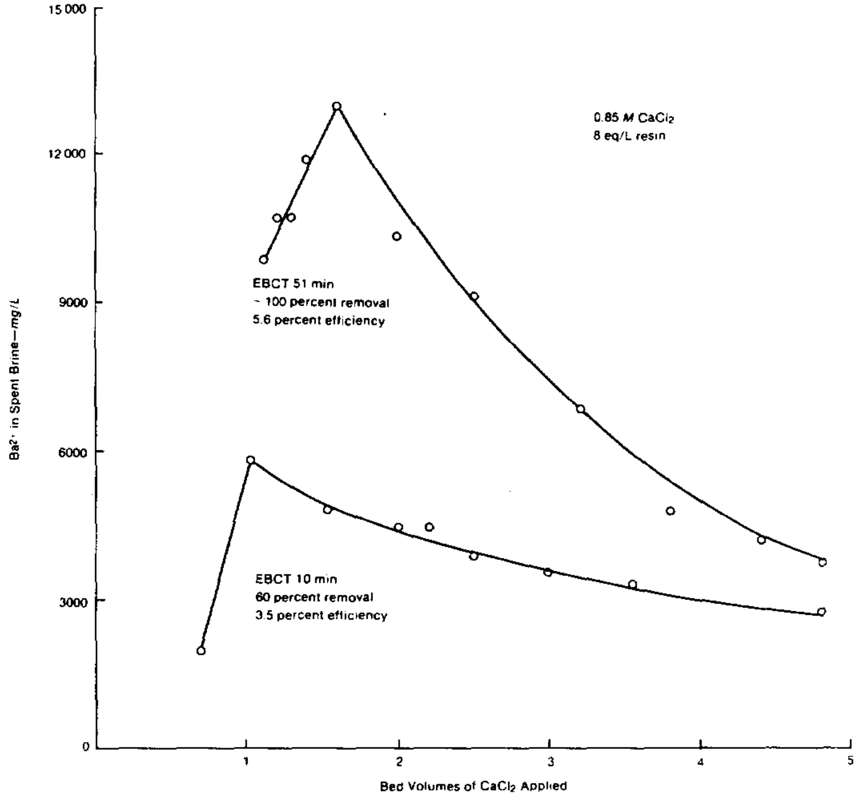

Figure 7. Barium concentration in spent regenerant at two contact times

brine reclamation. However, higher concentrations of Ba in a 0.6 *M* Ca brine result in lower removal, as shown in Table 4. All four samples were mixed at 100 rpm for 30 min. The reason for the lower removal has not been established but may have been caused by the precipitation of $BaSO_4$ onto the $CaSO_4$ before it dissolved. Thus, the concentration of Ba in spent brine must be considered when setting the $CaSO_4$ dosage and mixing conditions.

Barium removal from brine by using $CaSO_4$ in a column was ineffective when treating Ca brine for reuse. At the front of the column, $BaSO_4$ was precipitated, and the brine dissolved $CaSO_4$ as it continued to pass through the column. The effluent SO_4^{2-} concentration was near the limit imposed by $CaSO_4$ solubility. Also when Ba^{2+} broke through the column, $BaSO_4$ precipitated in the effluent collection container, thus requiring a solids separation step.

Barium removal in cyclic column runs. Five exhaustion-regeneration-brine reclamation cycles were performed to determine whether reclaimed brine would effectively regenerate the resin. The sixth cycle consisted of exhaustion only. Influent water, with a composition similar to that in Table 1 and a Ba^{2+} concentration of 20 mg/L,* was pumped at 5.7 gpm/sq ft (14.2 m/h) through a 22.8-in. (58-cm) resin column. The EBCT was 2.5 min. Regenerant (975 mL, or 3.6 BV) was pumped countercurrent (upflow) at a rate of 0.29 gpm/sq ft (0.7 m/h). The last regeneration was cocurrent (downflow). A regenerant dose of 6 eq Ca/L resin (27.5 lb $CaCl_2 \cdot 2H_2O$/cu ft resin [440.5 kg $CaCl_2 \cdot 2H_2O/m^3$ resin]) was applied at a concentration of 0.85 *M*. Dilution of the regenerant brine occurred by mixing with rinse water during regeneration, so the concentration and dosage were lower for some cycles. Additional $CaCl_2$ was added in later cycles to restore the 0.85 *M* Ca concentration. The brine was reclaimed by precipitation with $CaSO_4$ after each regeneration and reused throughout the five cyclic column runs. A $CaSO_4$ to Ba mol ratio of 1.1:1 was usually required to remove most of the Ba^{2+}.

Barium breakthrough curves for the six exhaustion runs are shown in Figures 9 and 10. Capacities at a breakthrough of 2 mg Ba^{2+}/L ranged from 50 to 75 mg/g dry resin in the H^+ form (21.6 to 32.2 g/L resin in the Ca^{2+} form), depending on regeneration conditions. Capacities at a breakthrough concentration of 1 mg/L were slightly less for all runs except for run 4, for which an effluent concentration of 1 mg/L was not achieved. Curve 4 in Figure 9 shows high Ba^{2+} leakage because of the high (705 mg/L) Ba concentration in the reclaimed regenerant. This high Ba^{2+} concentration was attributable to the use of insufficient $CaSO_4$ during brine reclamation. Regenerant Ba concentrations of ~100 mg/L do not produce this leakage, as shown by curves 3 and 5. Since the same volume of regenerant (975 mL) was applied in all regenerations, lower regenerant Ca^{2+} concentrations resulted in lower dosages, and less Ba^{2+} was removed from the resin. The effect of removing successively less Ba^{2+} from the resin was a successively lower column capacity for runs 2, 3, and 4. The addition of supplemental $CaCl_2$ to increase the regenerant concentration to 0.85 *M* and the regenerant dose to 6 eq/L for regeneration after run 4 restored the capacity of the column to near that for virgin resin.

The resin was regenerated with the same strength $CaCl_2$ brine after runs 1 and 4, but the capacity of the resin was about 30 percent higher after the latter (curves 2 and 5 in Figure 9). A possible reason for this behavior is the Mg^{2+} ion that accumulates on the resin and in the brine through successive cycles. Figure 11 shows that the Mg^{2+} concentration in the reclaimed brine has increased to a plateau of nearly 4000 mg/L. The concentration in the brine is proportional to the amount on the resin after regeneration, so the proportion of the column in the Mg^{2+} form increased significantly between runs 1 and 5. Because of the relative selectivity of the resin, Mg^{2+} can be displaced more easily than Ca^{2+}, so

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*The influent concentration for the fourth and fifth runs was 29 mg/L instead of 20 mg/L. The bed volumes processed to a particular effluent concentration were thus multiplied by the ratio 29:20 before plotting the data for these runs in Figure 7.

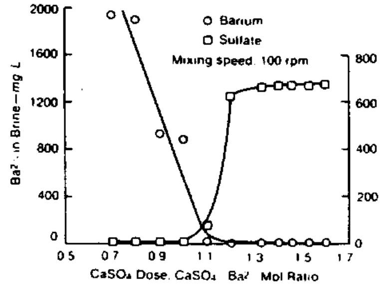

Figure 8. Barium precipitation from brine with $CaSO_4$

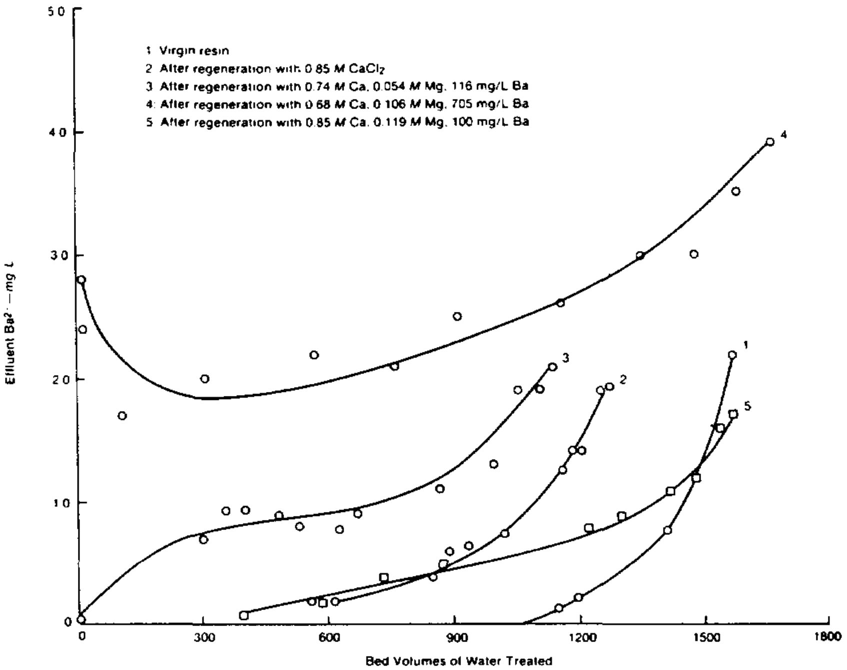

Figure 9. Barium breakthrough curves for cyclic column runs

the higher capacity is expected. Additional experiments, however, are necessary to confirm this reason.

Cocurrent regeneration is less effective than countercurrent regeneration because Ba remains on the resin at the effluent end of the column after the former. Figure 10 shows the high Ba^{2+} leakage that results after cocurrent regeneration and the low leakage that results after countercurrent regeneration. The same strength and dosage of regenerant were used in both cases. Cocurrent regeneration of Na-form resins is common, and, generally, leakage is not a problem, but this may be caused by the greater difference in selectivity between Na^+ and Ba^{2+} compared with the difference between Ca^{2+} and Ba^{2+}. If cocurrent regeneration of the Ca-form column is to be practiced, a larger regenerant dosage may be necessary to remove enough Ba^{2+} to stop excessive leakage. Brine reclamation and reuse could make this alternative economically feasible.

Characteristics of the wastewater during the cyclic column runs are shown in Table 5. Rinse water consisted of the portion of regeneration wastewater that was not saved for reclamation and reuse. If spent brine is not reclaimed but is disposed of with the rinse water, both the Ba^{2+} and the TDS concentrations of the combined wastewater will increase by factors of 16 and 10, respectively. In addition, the volume of wastewater will increase 50 percent. Reusing spent brine can significantly reduce wastewater disposal problems. However, the $BaSO_4$ sludge produced during reclamation of spent brine must also be disposed of.

Radium removal in cyclic column runs. The ability of a Ca-form resin column to remove Ra from water was evaluated by performing four exhaustion-regeneration cycles. The fifth cycle consisted of exhaustion only. Influent water was similar to that shown in Table 1, except that 43 pCi Ra^{226}/L replaced the Ba. Exhaustion in the 4.7-in. (12-cm) column was carried out at 1.35 gpm/sq ft (3.4 m/h) for 500 BV. Termination of the run at 500 BV was arbitrary; additional runs are needed to establish the BV to breakthrough. Regenerant brine consisted of 0.85 *M* Ca^{2+} and 0.2 *M* Mg^{2+}. The brine loading rate was 0.29 gpm/sq ft (0.7 m/h), and the dose was 6 eq Ca/L resin (27.5 lb $CaCl_2 \cdot 2H_2O$/cu ft resin [440.5 kg $CaCl_2 \cdot 2H_2O/m^3$ resin]). The spent brine was not reclaimed for reuse. Rinse water volume was 8.5 BV.

The average effluent in each of the five exhaustion runs was ≤0.5 pCi Ra^{226}/L (98.8 percent Ra removal). The amount of Ra placed on the column during exhaustion was 45 pCi Ra/g dry resin (0.02 μCi Ra/L resin). Radium capacity at breakthrough was not determined. However, the high Ra removal indicates that the Ca cation exchange process can also be used to treat water with a high Ra content. The brine reclamation process would have to be modified by adding $BaCl_2$ along with $CaSO_4$ to coprecipitate Ra with $BaSO_4$.[14] The effectiveness of this method for removal of Ra^{2+} from brine was not tested. Water supplies containing both Ba^{2+} and Ra^{2+} could also be treated by Ca cation exchange. If sufficient Ba^{2+} is present in the spent brine, the addition of solid $CaSO_4$ will coprecipitate Ra^{2+} and Ba^{2+} as Ra-$BaSO_4$.

Summary and conclusions

A strong acid resin in the Ca form achieved excellent removal of Ba^{2+} and Ra^{2+} from water containing 15-30 mg Ba^{2+}/L and 43 pCi Ra^{226}/L. The resin was regenerated easily with $CaCl_2$ brine, and after repeated exhaustion-regeneration cycles using reclaimed brine, the only water quality change observed during the production cycle was the reduction of the Ba^{2+} and Ra^{226} concentrations. Total hardness, alkalinity, pH, and other parameters were essentially unchanged. Operation of this column in parallel with a strong acid, Na-form column will enable the production of water that meets the standards for Ba and Ra and has the hardness desired for distribution. The capacity of the virgin Ca-form column for Ba^{2+} was approximately 75 mg Ba^{2+}/g dry resin (H^+ form), or 32 g Ba^{2+}/L resin (Ca form). The capacity of the column after several exhaustion-regeneration cycles depends on the dosage of regenerant.

The $CaCl_2$ regenerant concentration

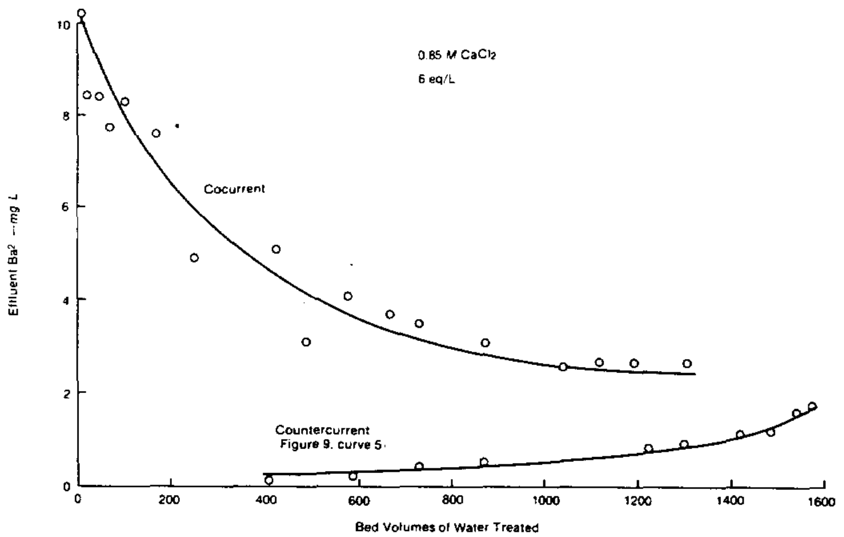

Figure 10. Barium breakthrough curves after cocurrent and countercurrent regeneration

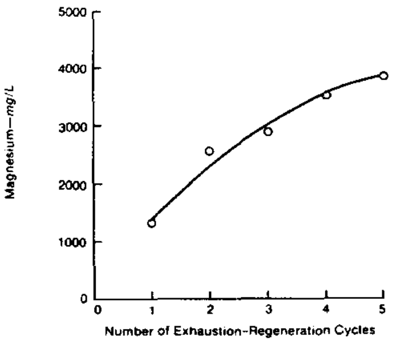

Figure 11. Magnesium buildup in reclaimed brine

of 0.85*M* was found to be significantly more effective at a given dosage than higher (1.69*M*) or lower (0.42*M*) concentrations. The amount of Ba^{2+} removed from the resin increased and Ba^{2+} leakage from the column decreased as the regenerant dose increased. However, regeneration efficiency decreased as the regenerant dose increased. A dose of 6 eq 0.85*M* $CaCl_2$/L of resin removed 92-100 percent of the Ba^{2+} on the column at an EBCT of 51 min. Cocurrent regeneration resulted in much more leakage than countercurrent regeneration, but further studies with higher regenerant dosages, possibly followed by mixing of the bed after regeneration, may result in good operation with cocurrent regeneration.

The spent $CaCl_2$ brine can be reclaimed for reuse. Addition of 10 percent molar excess solid $CaSO_4$ resulted in reduction of the Ba^{2+} concentration to about 100 mg/L. The $CaSO_4$ dissolved, and $BaSO_4$ precipitated. Reuse of brine was possible after removal of the precipitate by filtration. The concentration of Mg^{2+} in the brine increased through successive cycles until a plateau was reached. This resulted in a column that was partially in the Mg^{2+} form at the beginning of an exhaustion run, but this does not pose a problem because Ba^{2+} and Ra^{2+} can replace Mg^{2+} more easily than Ca^{2+}. If the brine contains Ra^{2+} as well as Ba^{2+}, the Ra^{2+} will coprecipitate on the $BaSO_4$ and thus also be removed. However, a Ba salt, such as $BaCl_2$, will have to be added along with the $CaSO_4$ to spent $CaCl_2$ brine containing only Ra^{2+} in order to achieve Ra^{2+} removal.

Additional research is needed to refine the process. In particular, the best procedure to precipitate and separate Ba and Ra needs to be established. A procedure is also needed to control the precipitation process to ensure that the Ba^{2+} and Ra^{2+} have been removed and that too much SO_4^{2-} has not been added. The brine reclamation process should significantly reduce the brine disposal problem, but ways of disposing of the precipitate must be found, and the cost involved in using the process must be established.

Acknowledgment

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#3

Conclusions and Recommendations

The particle-counting technique has been found to be useful in pilot-plant tests aimed at selecting water treatment processes and optimizing operating parameters for unit processes. Particle counts are useful in conjunction with other water quality and operating parameters.

One area for future research is the development of a convenient method for distinguishing inorganic from organic particulates in water supplies. Further work on disinfection is also needed to define those circumstances in which particulates in the water supply may shield pathogens from a disinfectant.

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The Efficiency of Water Treatment Processes in Radium Removal

Dorothy L. Bennett

In Illinois, one of the few states in which naturally occurring radioactive materials affect public water supplies, a study was undertaken to evaluate the effectiveness of conventional water softening processes in removing radium.

The Interim Primary Standards of the Safe Drinking Water Act have focused attention on water contaminants that are potentially hazardous to the health of the consumer. Illinois is one of the few states which will be affected by the regulations for naturally occurring radioactive materials, specifically radium. A study was conducted to investigate the effectiveness of conventional water softening processes in lowering radium levels to minimize the exposure of those people living in areas of naturally occurring radium.

The Problem in Illinois

Gross alpha activity has been monitored in Illinois by the Environmental Protection Agency for a number of years. Results indicate that 72 public water supplies have gross alpha activity in excess of 15 pCi/l; 147 range from 5.0 to 14.9 pCi/l; and 115 range from 3.1 to 5 pCi/l. In general, these supplies lie across the northern part of the state in a band that is relatively narrow at the eastern edge and broadens as it goes west (Fig. 1). There are a few isolated spots in southern Illinois, but they are very small compared to the northern area.

Past studies[1] have indicated that at least part of the alpha activity in these areas is due to naturally occurring deposits of radium found primarily in wells 1000 or more feet deep.[2]

Since these deep wells are the major source of water in this area of the state, it will be necessary to reduce the levels of radium which reach the consumer. Unfortunately, little information is available on satisfactory methods for doing this.

Because of the large number of water supplies in Illinois that will be affected by the regulations in the Interim Primary Standards of the Safe Drinking Water Act, the Illinois Environmental Protection Agency (IEPA) was asked to participate in a USEPA-sponsored study to determine the effect of conventional water softening processes on radium.

Selection of Water Supplies

Criteria used to select the water supplies to be included in this study were (1) that the raw water contain sufficient radium-226 so that changes in concentration could be measured accurately, and (2) that the water plant have some type of softening equipment. It was planned to have as many types of softening methods as possible represented, and to have water treatment plants of various sizes. With the help of Argonne Nation Laboratories, water supplies were ident fied which have radium-bearing well Those selected were Dwight Correctio al Center, Herscher, Lynwood, Peru, an Elgin. The correctional center, whic serves a population of approximate 250, has one softener unit that uses nat ral green sand. Herscher and Lynwoo which serve populations of approx mately 1000 and 4000, respectively, bo utilize styrene-based zeolite for softe ing. In addition, Herscher has an ir removal filter ahead of the softener. Bo Peru and Elgin use lime softening a serve populations greater than 10 000.

Sampling Procedures

Schedule. Since the main purpose the study was to determine the effect conventional water treatment on radiu content, sampling was done after ea step of the treatment process. Samples raw, aerated, and softened water we collected in each of the ion exchan softening plants. At Herscher samp were also collected after the iron filt Because the ion exchange softeners op ate on a definite cycle, samples w collected shortly after the softeners w regenerated, at the approximate m point of the cycle, and at a time n breakthrough. Samples were collect three times at each location at appro mately one-week intervals.

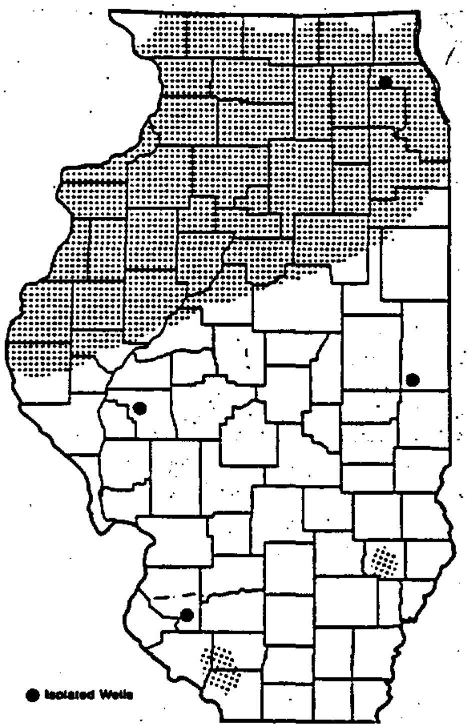

Fig. 1. Areas in Illinois Where Natural Radioactivity May Be Expected

TABLE 1

Radium-226 Removal Efficiency With Ion Exchange Softening

Sample Source	Percentage Removal			
	First Sampling	Second Sampling	Third Sampling	Average
Plant 1 (natural green sand)				
regenerated	88.9	88.4	91.6	89.6
midpoint	96.0	92.7	90.6	93.1
near breakthrough	98.2	70.2	83.7	84.0
Plant 2 (styrene-based zeolite)				
regenerated	92.0	95.3	86.0	91.1
midpoint	97.2	97.8	95.7	96.9
near breakthrough	85.4	84.5	83.8	84.6
Plant 3 (styrene-based zeolite)				
regenerated	97.3	97.0	94.7	96.3
midpoint	98.3	97.6	98.2	98.0
near breakthrough	97.3	96.5	98.2	97.3

TABLE 2

Radium-226 Removal Efficiency With Lime Softening

Sample Source	pH	Percentage Removal			
		First Sampling	Second Sampling	Third Sampling	Average
Plant 4	8.4	92.4	70.6	77.9	80.3
Plant 5	10.2	89.9	86.0	87.0	87.6

At the lime softening plants, samples were limited to raw and filtered water. Again, samples were collected three times at one-week intervals.

Preservation and quality control. To ensure the integrity of the samples during transport, the sampling containers were prepared with the proper preservatives by the laboratories that would be performing the various portions of the analyses. Argonne National Laboratories, which performed the radium-226 analysis, prepared the bottles by using disodium EDTA as a preservative to prevent the plating of the radium on the container walls. Every tenth sample was collected and analyzed in duplicate.

Samples were submitted to the IEPA laboratory in Chicago for mineral, trace metal, and gross alpha and beta analyses. The containers for this portion of the analyses were provided by the agency laboratory. Redistilled nitric acid was added to the bottles used for trace metal analyses to prevent plating. Since the bottles were disposable, representative samples from each batch of bottles were acidified and filled with distilled water. After a few days' storage, all of the appropriate analyses were performed on the acidified distilled water blank to ensure that containers were uncontaminated. The portion of the sample used for the remainder of the mineral analyses was collected in an untreated polyethylene bottle.

All samples were refrigerated during transport to minimize changes in nitrate and alkalinity. Every tenth sample was collected and analyzed in duplicate.

Radium Removal Efficiency

Radium removal, in general, was good with both ion exchange and lime softening. The removal efficiency for ion exchange ranged from 70.2 to 98.2 per cent. Removal at the lime softening plants did not appear to be quite as effective, ranging from 70.6 to 92.4 per cent (Tables 1, 2).

Not evident from these tables is that the water from the ion exchange softeners is reduced to near-zero hardness. In order to stabilize this water before putting it into the distribution system, unsoftened water is blended with the softened water. This, of course, raises the radium content of the water reaching the consumer. However, in each of the cases tested, even with the blending the radium-226 content of the finished water stayed within the range of 0.36 to 2.37 pCi/L.

At most of the lime softening plants, the water is softened only to the point desired and blending is unnecessary. Also, in the lime softening plants the treatment is constant, while in the ion exchange plant breakthrough was observed at about the same time as the calcium-magnesium breakthrough. The amount of lime fed was based on the hardness to be removed to bring the finished water to the desired quality. No laboratory analyses of the lime were made to determine its radium content. However, readings of the dry lime taken with a field instrument indicated no significant radioactivity.

The plant at Herscher had an anthrafilt iron filter ahead of the softener. This filter removed from 47.5 to 54.5 per cent of the radium. Since this treatment was used at only one plant, it is impossible to conclude that iron removal is an effective means of removing radium; more study will be required to see whether this is generally true or is a phenomenon peculiar to this particular plant.

Field Study

In addition to the laboratory testing, a field survey was conducted to observe the plants in operation and to determine the levels of radiation in each plant. Although radiation levels in the plants were elevated above normal background, it was not felt that the levels present a hazard to the operators. Levels within the plant should not produce an occupational exposure of more than 25 to 100 mrem/year, whereas radiation workers are allowed exposure of 5000 mrem/year.

During the field study a survey of the iron filters and softeners was made to determine whether or not a buildup was occurring within the tank. Readings were taken against the side walls of each tank and a rate profile developed. The rate profile of the softeners at Lynwood (Fig. 2) does indicate some buildup in each of the plant's three softeners. Figure 3 shows the condition of the iron filters and softener at Herscher. Significant quantities of radioactive materials are building up at approximately 0.6 metre from the bottom of the filters, with much lower levels evident in the softener.

Waste Problem

Another question was what happened to the radium that was removed. Presumably, it was in the waste or was being deposited on the zeolite or anthrafilt. To find out, samples of backwash and brine rinse water, resin, anthrafilt, and lime sludge were analyzed.

It was found that much of the radium removed from the water was released from the ion exchange softeners. Samples of backwash, brine, and rinse water were composited. Radium content of the wastewater varied according to the radium content of the raw water. Dwight Correction Center, which had a low radium content in the raw water (average 3.25 pCi/l), had an average of 26 pCi/l of radium-226 in the wastewater. Lynwood and Herscher, each with approximately 15 pCi/l of radium-226 in the raw water, had averages of 72 and 106 pCi/l of radium-226 in the wastewater respectively. The precision of these values, however, is questionable because of sampling technique: Samples were collected with a small sump pump whose lines clogged frequently, making it impossible to composite samples evenly over the regeneration cycle.

A much better evaluation of the wastewater was made during one regeneration cycle at the Herscher plant. Samples were collected every six minutes, starting with the backwash and ending with the fast rinse. Figure 4 shows clearly that very little radium was released during the backwash. As the brine came through, the radium was released rapidly, and the quantity rose very sharply to a peak at 315 pCi/l. The drop-off was also very sharp, finally leveling off near zero. When calculations were made taking into consideration the rate of flow, radium-226 in the wastewater was found to average 54 pCi/l over the regeneration cycle. During this cycle 16 000 litres of water were used.

Samples of anthrafilt and ion exchange resin were collected for evidence of radium buildup in the softeners and filters. The rate profiles developed during the field visit were used to determine the depth of highest activity so that samples could be collected at that level. In some cases this was impossible because the entry port was on the side of the unit, and the sampling devices could not be inserted to the proper depth.

Analysis of these solid materials presented additional problems. Argonne Laboratories attempted to extract the radium by various methods. A sample of the natural green sand from Dwight Correctional Center was divided into separate portions, one extracted with hydrochloric acid and one with EDTA. The USEPA examined a similar sample by gamma spectrometry using a germanium-lithium detector. Results showed considerable variations, with the acid-extracted sample showing a radium-226 content of 34.22 ± 0.55 pCi/g and the EDTA extract showing 28.66 ± 0.46 pCi/g. The gamma scan showed 46.4 ± 5.5 pCi/g. The USEPA examination also included the radium-228 content, which indicated the presence of 59 ± 2.2 pCi/g.

The highest levels of activity were found in the materials from Herscher. The anthrafilt samples showed 111.6 pCi/g of radium-226 and 38.9 pCi/g of radium-228. The zeolite resin contained 43 pCi/g of radium-226 and 15 pCi/g of radium-228. In contrast, the plant at Lynwood, which had approximately the same radium content in the raw water and did not have the iron filters ahead of the softeners, had a radium-226 content of 9.6 pCi/g and a radium-228 content of 6.6 pCi/g. This difference reflects the buildup that occurs over time. The plant at Herscher has been in operation for about ten years, and the Lynwood plant for only two.

The information gathered is incomplete because the number of samples collected and analyzed had to be limited. However, the results clearly indicate that there is a radium buildup in both the anthrafilt and the zeolite. More study will be needed to determine how large a reservoir of radioactive material is being built up within the filters and softeners, and the rate at which the buildup occurs.

Similar evaluations of the lime sludge from Elgin and Peru were made. As expected, the radium was precipitated along with the calcium and magnesium. The results of these tests were not very consistent, ranging from 1.26 to 30 pCi/g. At first glance this may not seem to be very much radium but, considering the amount of sludge produced, it is a significant amount. The plant at Elgin, for example, uses 1900 to 3800 kg of lime each day. It is estimated that 2 kg of sludge are produced for each kilogram of lime used. At this particular lagoon the radium-226 content in the sludge ranged from 6 to 30 pCi/g. At the lowest and highest values, between 2.28×10^7 and 2.28×10^8 picocuries of radium are deposited in this lagoon each day.

The supernatant liquid from the lagoon was not analyzed for radium. However, the gross alpha count was low, ranging from 2.1 to 4.1 pCi/l.

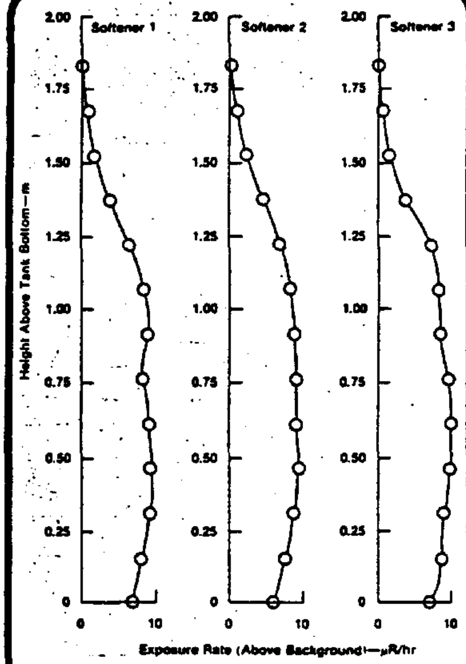

Fig. 2. Exposure Rate Profiles: Lynwood, Ill.

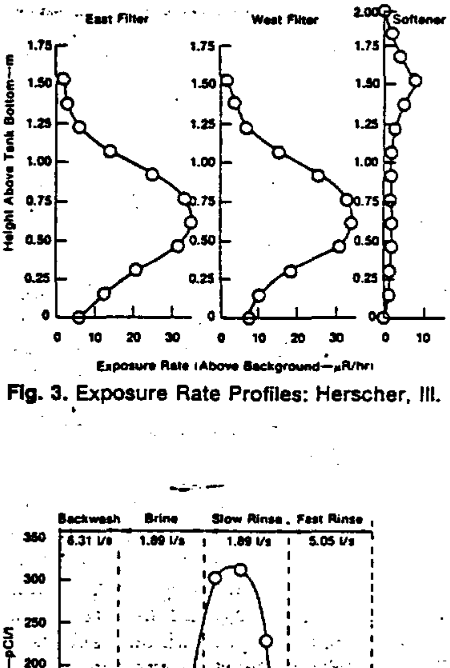

Fig. 3. Exposure Rate Profiles: Herscher, Ill.

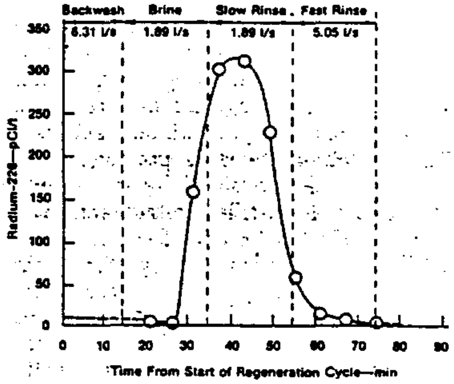

Fig. 4. Concentrations of Radium-226 in Softener Wastewater During Regeneration Cycle: Herscher, Ill.

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Evaluating Various Adsorbents and Membranes for Removing Radium From Groundwater

Dennis Clifford, Winston Vijjeswarapu, and Suresh Subramonian

Field studies were conducted in Lemont, Ill., to evaluate specific adsorbents and reverse osmosis (RO) membranes for removing radium from groundwater. A radium-selective complexer and barium-sulfate-loaded alumina appeared to have the best potential for low-cost adsorption of radium from raw water or ion exchange brines, provided that the problem of the ultimate disposal of spent media can be solved. A new, low-pressure (70-psig) RO module achieved radium and total dissolved solids (TDS) rejections of 91 and 87 percent, respectively. Central treatment with standard or low-pressure RO modules would be effective for radium removal—but expensive and unnecessary if TDS reduction is not also required. Point-of-use treatment using RO is effective for removing radium and may be cost effective for very small communities.

According to a recent survey,[1] an estimated 500 community water supplies exceed the US Environmental Protection Agency (USEPA) allowable maximum contaminant level (MCL) for radium in drinking water: 5 pCi/L for the sum of Ra-226 and Ra-228.[2] These are groundwater supplies, and in many cases alternative sources are not available at a reasonable cost. Thus, as enforcement orders are implemented, many communities will have to install treatment to reduce radium levels.

This is a partial report of the results of a USEPA-funded 15-month bench- and pilot-scale field study in Lemont, Ill., of the possible techniques for radium removal. The University of Houston-USEPA mobile drinking water treatment research facility, built in 1980 for studying the removal of inorganic contaminants from small community water supplies,[3] was used for the study. It was modified in 1986 to provide for safe operation with radionuclides and was moved to Lemont in December 1986, where it remained until the completion of this study in April 1988.

Two proven full-scale methods for radium removal are sodium ion exchange softening and lime–soda softening.[4-6] With these traditional processes, however, radium removal is an adjunct to softening, and the processes have not been optimized for radium removal. Recent modifications to the traditional softening methods have been reported by Snoeyink and co-workers[7-9] but have not yet been tested in field situations. Accordingly, one of the main objectives of the present study was to optimize the ion exchange softening process for radium removal using a typical radium-contaminated groundwater. This involved extensive experimentation with various resins, regenerants, and operating techniques, and the results of the ion exchange portion of the study will be reported in a future article.

Reverse osmosis (RO) hyperfiltration[4,10] has also been shown to be an excellent method for radium removal, but, again, radium removal has been studied only as an adjunct to the primary purpose of the process, which is reduction of total dissolved solids (TDS). Conventional processes are generally considered to be too expensive to be used solely for the purpose of removing a trace contaminant like radium whose MCL translates to an allowable radium concentration of 0.000005 mg/L based on Ra-226. Membrane technology has been advancing at a rapid pace, however, and one of the objectives of the present study was to evaluate the new lower pressure membranes for radium removal, softening, and TDS reduction. Also, point-of-use (POU) treatment systems utilizing RO modules as the main treatment device were evaluated because such devices are being considered as possible means of compliance with the provisions of the Safe Drinking Water Act.[11]

The focus of this article is the use of packed beds containing granular- or bead-form radium-specific adsorbents. One such adsorbent, the radium-selective complexer (RSC),* has been studied for the decontamination of uranium-tailing pond effluents[12] and radium-contaminated drinking water.[13,14] In spite of the successful use of RSC in these studies, several important questions remained unanswered. These were addressed in the present study and include:

- What does a typical radium breakthrough curve look like for low leve radium-contaminated feedwater?
- What is the ultimate radium capacity of RSC in drinking water applications?
- Is prefiltration necessary for low iron waters like the Lemont ground water?

The major unanswered question, however, was not addressed here and deals with the ultimate disposal of the exhausted adsorbent. (It is considered nonregenerable although this has no yet been proved.) Using RSC or any effective radium adsorbent presents a dilemma—the more effective the adsorbent, the more highly loaded it become and, consequently, the more difficult is its disposal. In fact the disposal problem with this very effective radium adsorbent was so significant that an announcement was made in December 1987 tha the product would no longer be commercially available through the Dow Chemical Company.[15] The fact that it commercial future is uncertain shoul not detract from its proved effectivenes for radium removal, which was furthe verified and quantified in this study.

In addition to RSC, other radium selective adsorbents were studied in Lemont. These were generally two types manganese-impregnated beads and barium-sulfate-impregnated ($BaSO_4$-loaded activated alumina. The former produc was offered for testing by its manufacturers, whereas the $BaSO_4$-loaded alumina (patent applied for) was develope specifically for radium removal fro drinking water in the University o

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*XSF 43230.00, Dow Chemical Co., Midland, Mich.

TABLE 1
Analyses for Ra-226, Ra-228, and Ra-224

ID	Date Sampled	Date Analyzed	Sample Age *days*	Method, Isotope, and Laboratory	Activity* *pCi/L*
a+	1/16/87	2/26/87	45	Ra-226, 903.1, USEPA	11.3
b+	2/03/87	3/09/87	37	Ra-226, 903.1, Univ. of Houston	12.0
c+	3/20/87	3/20/87	0	Ra-226, gamma spectroscopy, Argonne	11.8
d+	3/31/87	4/29/87	28	Gross radium alpha, 900.1, Univ. of Houston (Lemont)	11.8
d	1/16/87	2/26/87	45	Ra-228, 904.0, USEPA	6.1
e	3/20/87	3/20/87	0	Ra-228, gamma spectroscopy, Argonne	6.8
f‡	3/20/87	3/20/87	0	Ra-224, gamma spectroscopy, Argonne	6.4

*Previously reported values for samples from this well were Ra-226 = 12.8 pCi/L and Ra-228 = 5.6 pCi/L.

+Ra-226 analyses (a, b, and c) by the standard radon emanation technique are in agreement when performed by the University of Houston, USEPA, and Argonne. The gross radium alpha result (d) agrees with the Ra-226 analyses (a, b, and c) after a long holding time.

‡Analyses were performed by using adsorption from acidified 20-L samples onto RSC followed by gamma counting.

TABLE 2
*Chemical analysis of Lemont, Ill., groundwater supply**

Parameter	Concentration *mg/L*	Concentration *meq/L*
Alkalinity as $CaCO_3$	227 (276)†	5.54
Hardness as $CaCO_3$	245 (237)	4.90
pH	7.2 (7.4)	
Total dissolved solids	439	
Silica	7.5	
Calcium	70 (67)	3.49
Magnesium	17 (22)	1.4
Strontium	6.1	0.07
Barium	<0.2 (0.015)	
Iron	0.22	0.008
Manganese	0.009	0.0003
Sodium	63	2.74
Potassium	12.5	0.32
Copper	0.018	0.001
Zinc	0.018	0.001
Aluminum	0.067	
Lead	0.034	0.0003
Total cations		8.06
Bicarbonate	338.0	5.54
Chloride	31.0	0.87
Sulfate	80.0	1.67
Fluoride	1.41	0.074
Nitrate + nitrite as N	0.06	0.001
Total anions		8.15

*All analyses were done by Enviro-test/Perry Laboratories Inc., Downers Grove, Ill.; sample from Houston Street well on 1/21/87.

†Values in parentheses were obtained in UH/USEPA trailer in Lemont.

TABLE 3
Analysis of backwash water

Element	Raw Water	Softener Backwash	Filter Backwash
Ca—*mg/L*	70.0	82.3	75.4
Mg—*mg/L*	17.0	18.1	18.3
Sr—*mg/L*	6.1	5.5	4.3
Ba—*mg/L*	<0.2	0.3	0.2
Mn—*mg/L*	0.009	4.9	2.4
Fe—*mg/L*	0.22	480.0	276.0

TABLE 4
*Ra-226 to Ra-228 ratio in water and adsorbent samples**

Sample	Ra-226 *pCi/L*	Ra-228 *pCi/L*	Ratio
Lemont water	11.8	6.45	1.82
Ion exchange effluent	5.26	2.97	1.77
Filter backwash	184.0	102.0	1.80
Exhausted RSC	751,000	429,000	1.75
Exhausted $BaSO_4$ alumina	419,000	250,000	1.68
Exhausted cation resin	30,340	16,215	1.87

*All determinations were made by the gamma spectroscopy method.[24]

Houston Environmental Engineering Laboratories. It is not commercially available. Plain activated alumina was also experimented with as a radium adsorbent, primarily as a control for comparison with the $BaSO_4$-loaded alumina that proved to be so effective.

Materials and methods

Both the research trailer and the field researchers' living trailer were located immediately adjacent to the Houston Street well pump house located under the 300,000-gal water tower. The research trailer's water supply was the Lemont distribution system, which is supplied by two wells: the Houston Street well and the State Street well. Source water for the experiments was predominantly chlorinated groundwater from the Houston Street well, which was always in use. The State Street well, although not a significant contributor to the trailer feedwater, was also contaminated with radium and was of approximately the same composition as the Houston Street well. Wastewater from the research trailer was discharged into the sanitary sewer.

Adsorption columns. The pilot-scale adsorption experiments were performed using 8-in.-diameter clear acrylic, 2-in.-diameter glass, or 1-in.-diameter glass columns. Media depth was generally 30–40 in. (76–102 cm), and the empty bed contact time (EBCT) was in the range of 0.5 to 10 min. (Additional column operating details are given with the respective breakthrough curves.) Exhaustion was usually downflow, although upflow was experimented with in some cases in which the specific adsorbents collapsed during downflow operation. (This collapse occurred only with the cellulose-based adsorbents.) Feedwater was pumped through the main flow system using a plastic (PVC) impeller centrifugal pump. In addition to PVC, the materials of construction of the adsorption flow system were glass, PTFE, polyethylene, fluoroelastomer, and stainless steel. Complete details regarding the flow systems in the research trailer can be found in the report of the design, construction, and operation of the facility.[3]

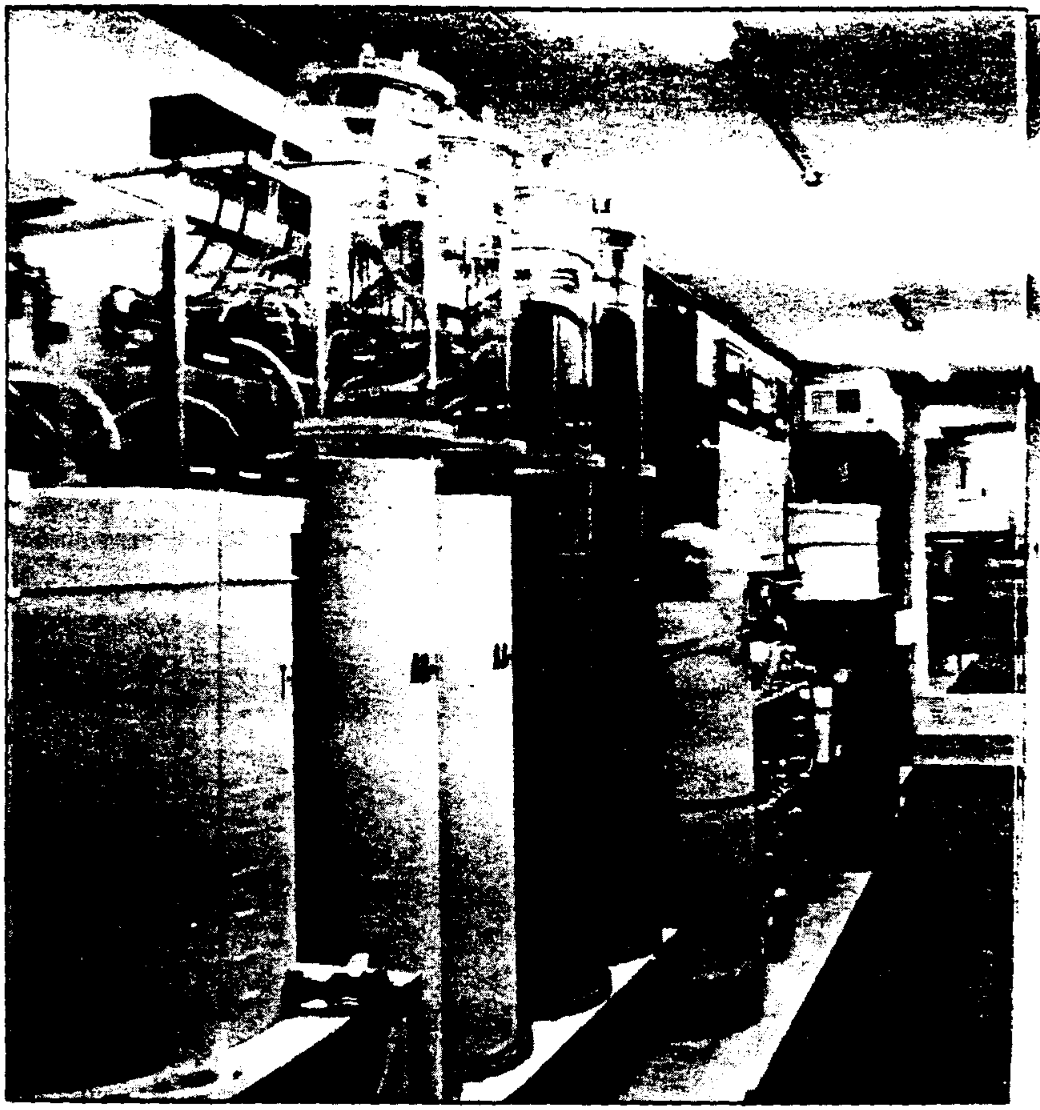
Prefiltration. A 12-in.-diameter deep-bed filter containing 30 in. (76 cm) of 12 × 28-mesh aluminosilicate AG medium was used for all but a few of the early experiments that demonstrated suspended solids were clogging and fouling the resins and adsorbents. Further, these solids tended to adsorb a significant portion (approximately 30 percent) of the gross alpha activity and radium. The filter velocity varied, depending on the number and size of columns in use, but in no case did it exceed 2 gpm/sq ft (5 m/h). Prefiltration was not used ahead of the POU system, which contained its own prefilter.

RO equipment. Three types of RO modules were tested: (1) RO-1—a hollow-fiber (HF) polyamide membrane type with a rated flow of 2,100 gpd at 50 percent recovery and 400 psig feed pressure.[*] (2) RO-2—a spiral-wound (SW) thin-film-composite (TFC) type rated at 1,800 gpd of product water at 10 percent recovery and 225 psig feed pressure,[†] and (3) RO-3—an experimental low-pressure SW TFC type rated at 1,700 gpd at 15 percent recovery and 70 psig feed pressure.[‡] Module RO-1 had been used successfully in five previous studies over a period of six years, and module RO-2 had been used in one previous study. Module RO-3 was new, and, to determine reproducibility, two different modules of the same construction were tested.

The RO flow system consisted of dechlorination with sodium sulfite (Na_2SO_3), pH adjustment with hydrochloric acid, deep-bed filtration with the AG-medium filter, 5-μm cartridge filtration, pressurization with a nine-stage pump, and, finally, RO hyperfiltration. The feed pH was adjusted to the value recommended by the manufacturer after calculation by a computer program using the chemical composition of the feedwater as the primary input. Only single modules were used in each test; thus 50 percent recovery was required for the HF module whereas only 12–15 percent recovery was possible with the SW modules. In general, five-day continuous tests were performed with sampling twice daily.

Radium-selective complexer. The RSC is a standard polystyrene divinylbenzene strong acid cation resin in the sodium form containing a barium sulfate precipitate.[16] Thus, it is not really a complexer but rather a carrier of $BaSO_4$ microcrystals that scavenge radium from solution. The preparation of a class of $BaSO_4$-loaded resins is described in a Canadian patent granted to the Dow Chemical Company in 1984.[17]

For the experiments described here, 2,000 cm^3 of the 20 × 50-mesh beads were loaded into a 2-in.-diameter glass column to a depth of 40 in. (102 cm). The column was operated at an exhaustion flow rate of 37.5 gpm/sq ft for an EBCT of 0.67 min. (The EBCT is based on the initial volume of the RSC. During exhaustion, this strong acid cation resin changed from the sodium to the calcium form, with a resulting 25 percent loss in volume.) This exhaustion rate was greater than the recommended rate[18] but was deemed necessary to shorten the run time to a reasonable length. A previous study with the complexer[12] showed no significant change in radium removal efficiency in the range 2–20 gpm/sq ft during upflow operation. Furthermore, the RSC run lasted in excess of 40 days—a more than adequate time for the complexer near the top of the bed to come to near equilibrium with the feedwater.



Possible radium removal techniques were assessed in a research trailer supplied by water from the Lemont, Ill., system.

Manganese-containing adsorbents. Several samples of experimental adsorbent beads[§] were tested in Lemont. One such adsorbent, CYC, is described as "a nontoxic, noncorrosive, and insoluble cellulose derivative interstitially containing an insoluble metal oxide." RadiSorb, a polymeric adsorbent containing an internal oxidizing catalyst and MnO_2,[**] known to be an excellent adsorber of radium and barium,[19] was evaluated for radium removal. Two different lots of the same material were tested because the first sample disintegrated during testing. These MnO_2-containing adsorbents were tested in 1-in.-diameter glass columns loaded to a depth of 30 in. (76 cm) and backwashed extensively before use to wash out fines.

Activated alumina adsorbents. The plain activated alumina used in the Lemont radium adsorption experiments was untreated 28 × 48-mesh F-1 alumina.[*] This low temperature activated alumina[20] is a mixture of gamma and amorphous alumina. The $BaSO_4$-loaded alumina was prepared in a 1-in.-diameter glass column by slowly rinsing a 30-in.-deep bed of plain alumina with 10 bed volumes (BV) of 0.25 *N* sulfuric acid (H_2SO_4) followed by 10 BV of 0.25 *N* barium chloride ($BaCl_2$). Exposing the alumina to H_2SO_4 converted its ligand exchange sites to the relatively highly preferred sulfate form.[21] Adding the $BaCl_2$ resulted in the precipitation of $BaSO_4$ in the micropores of the alumina. The proposed reaction is

$$\begin{aligned} 2 \equiv \mathrm{Al \cdot HSO_4} + \mathrm{BaCl_2} \rightarrow 2 \\ \equiv \mathrm{Al \cdot Cl} + \mathrm{BaSO_4} + \mathrm{H_2SO_4} \end{aligned} \qquad (1)$$

Excess $BaSO_4$ was washed off the alumina by extensive rinsing with radium-free RO product water. The $BaSO_4$ loading procedure was not optimized in Lemont, however. An extensive study of the preparation of $BaSO_4$-loaded

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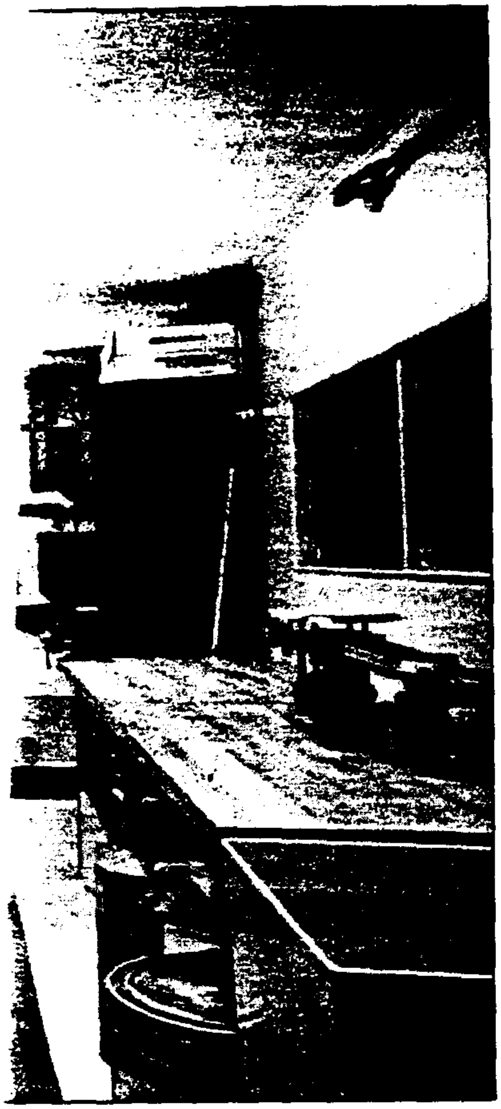
[*] Model B9 0420-021, DuPont, Wilmington, Del.
[†] Model BW-30-4030, Film-Tec/Dow, Midland, Mich.
[‡] Model NF 70-4040, Film-Tec/Dow, Midland, Mich.
[§] Isoclear Systems Corp., Yorkville, Ill.
[**] Grindl and Associates, Dallas, Texas



alumina and its radium removal performance is under way at the University of Houston.

POU system. Spiral-wound cellulose-acetate (CA) and TFC membranes were tested in the POU treatment system.[†] The system was installed under the laboratory sink in the research trailer and was operated in a normal fashion; i.e., water was drawn intermittently as needed. This system provided water for the laboratory demineralizer cartridges and for general lab use. In addition, it provided water for the two field researchers for drinking and cooking.

Water treated in the POU system passed through a fiber-wound cartridge filter, a granular activated carbon (GAC) filter, the RO module, and a final GAC filter before being stored in a pressurized storage tank. The system operated at line pressure (50–60 psig) and had a nominal capacity of 8 gpd of product water. Samples of the feedwater, cartridge filter effluent, GAC effluent, and product water were taken weekly and were analyzed for radium, hardness, and conductivity.

Analysis for radium isotopes. The reported levels of radium isotopes in the Lemont water were 12.8 pCi/L of Ra-226 and 5.6 pCi/L of Ra-228.[22] In fact, one reason for choosing Lemont for this field study was the presence of a significant level of Ra-226 activity, which is easier to determine analytically than is Ra-228. These isotopes were assumed to be chemically and physicochemically indistinguishable. Therefore, removals of Ra-226 by ion exchange, adsorption, and membrane processes were expected to be equivalent to those of Ra-228. This assumption was checked many times during the study and was found to be true, within the limits of analytical error.

The gross radium alpha screening technique (USEPA method 900.1)[23] was used extensively in this study. It embodies $BaSO_4$ carrier precipitation, glass fiber filtration of the $BaRaSO_4$ precipitate, and alpha counting. It is a relatively simple, economical, and precise procedure and is highly suitable for field use. About 2,000 samples were analyzed using this method. A significant modification to the procedure, implemented in Lemont, was that a minimum waiting period of 20 days was used either to hold the water sample prior to precipitating $BaRaSO_4$ or to age the precipitated $BaRaSO_4$ prior to counting. (Holding the precipitate is preferred because it results in higher, more stable counts.) This waiting period is necessary to allow for the decay of Ra-224 (half life = 3.64 days) and its alpha-emitting progeny that interfere with the interpretation of the gross radium alpha counts, presumed, in method 900.1, to result solely from Ra-226 and its progeny.

Before it was realized that Ra-224 was present in the Lemont water, results from method 900.1 were higher than expected. In an attempt to explain the discrepancies and to provide outside lab verification of the radium analysis procedures, the Lemont water was analyzed by the USEPA in Cincinnati, Ohio, using standard USEPA methods[23] and by Argonne National Laboratories using a gamma counting technique to determine Ra-224, Ra-226, and Ra-228 activities.[24] The latter technique utilizes 200 mL of RSC to quantitatively adsorb the radium isotopes from a 20-L sample. The radium isotopes, concentrated on the resin, are counted by gamma spectroscopy. The procedure must be performed immediately after sampling if Ra-224 is to be determined. Samples checked by the USEPA were analyzed by USEPA method 903.1, the radon emanation technique, for Ra-226 determination and by method 904.0 for Ra-228 determination. Additional Ra-226 check samples were analyzed at Houston using the radon emanation technique. Some radium analyses from the early months of the study are compared in Table 1, which shows good agreement among the methods.

Equipment for the radon emanation technique (a centrifuge, radon de-emanation glassware, and a flask counter) was eventually purchased for field use in Lemont, where Ra-226 analyses by method 900.1 (with a minimum 20-day holding time for Ra-224 decay) were occasionally checked by the radon emanation technique during the later months of the study. After the study had been in progress for nine months, a USEPA analyst with expertise in radiological analysis techniques visited the field site for one week to instruct and work side by side with the field researchers and check the details of the radium analysis procedures in use. He concluded that the radium analysis methods in use, particularly method 900.1 with the 20-day holding time, were being properly executed and interpreted.

Chemical analysis. Samples also were analyzed for chemical contaminants at the field location. Specific conductance and pH were determined by appropriately calibrated meters. Total hardness was determined by EDTA titration (method 314B),[25] calcium by EDTA titration (method 311C),[25] magnesium by the difference between total hardness and calcium (method 318C),[25] and total dissolved solids (TDS or filterable residue) by method 209B.[25] Field test kits were used to determine chlorine (CN70) and iron (IR-24).[‡]

Some field support analyses were performed at the University of Houston. Inductively coupled argon plasma spectroscopy[§] was utilized to determine barium, strontium, iron, and magnesium[26] in filter backwash samples according to USEPA method 200.7. A complete analysis of the Lemont groundwater was performed by an outside laboratory prior to the start of the study (Table 2).

Results and discussion

Prefiltration. Resin bed fouling was observed during the first ion exchange and RSC runs. The medium tended to become plugged up with a dark brown residue that removed a significant amount (~30 percent) of the gross radium alpha activity and Ra-226 contamination. The small amount of iron in this water (0.22 mg/L) was probably oxidized by the chlorine added (the typical free chlorine residual was 1.0–1.5 mg/L) and was precipitated along with the other particulates present. The manganese content of the raw water was low (0.009 mg/L) and, therefore, did not contribute much to the resin fouling problem. It may have contributed significantly to the observed radium removal because MnO_2 is reported to be 40 times as good

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[*] Alcoa, New Kensington, Pa.

[†] Model H-82-C, Culligan International Corp., Northbrook, Ill.

[‡] Hach Co., Loveland, Colo.

[§] Model 5500, Perkin Elmer, Norwood, Conn.

as ferric hydroxide [Fe(OH)$_3$] for radium adsorption.[19] Following these initial runs, which required intermittent (and undesirable) backwashing, all subsequent runs used deep-bed-filtered raw water to avoid fouling of the packed beds and membranes. Table 3 presents the metals analysis results of two backwash samples, one from the softener and the other from the deep-bed filter, illustrating the predominance of iron and showing a small amount of manganese. The radium was in some way associated with the particulate iron (and probably the manganese) as shown in Figure 1, a plot of the iron and Ra-226 content of the deep-bed-filter backwash after about four days of operation. Manganese was not determined in the backwash samples, but probably followed the same elution pattern as iron.

Radium removal by RSC. The first RSC run was aborted because the unfiltered feedwater caused resin fouling and the need for frequent backwashing. Unfiltered raw water was unwittingly used during backwashing, and this led to premature leakage of radium that was presumed to have been deposited with the solids on the RSC beads near the outlet of the column. This leakage could have been avoided if treated water had been used for backwashing, but the more conservative solution of filtering the feedwater was chosen for future experiments. Another reason for choosing prefiltration was that pilot studies with the RSC[14] showed that prefiltration was often essential to produce long runs. Premature termination of runs had been forced by radium leakage that was always associated with the breakthrough of solids, particularly iron.

The Ra-226 breakthrough curve for the second run with RSC is shown in Figure 2. Up until about 18,000 BV, there was no measurable Ra-226 in the effluent. Excluding flow interruptions, the RSC was used for approximately 41 days (89,000 BV) before the adsorbent came to equilibrium with the feedwater. The run length to the radium MCL was approximately 38,000 BV. The MCL corresponded to 3.25 pCi/L of Ra-226 plus 1.75 pCi/L of Ra-228, using the assumption of a constant Ra-226-to-Ra-228 ratio of approximately 1.8, the ratio in the feedwater. The constant ratio assumption was checked many times during the Lemont study (Table 4).

Exhausted RSC samples from the top, middle, and bottom sections of the bed were analyzed for Ra-226 and Ra-228 by gamma spectroscopy[24] (Table 5). The fact that the radium loading was nearly uniform from top to bottom was a confirmation that the bed was exhausted. Assuming equal sizes for the three sections, the average radium capacity measured at exhaustion was 1.08 nCi/cm^3 (3.6 nCi/g). The maximum loading on the RSC, at the top of the bed, was 1.18 nCi/cm^3. This is much less than the advertised capacity range of 3–6 nCi/cm^3 (10–20 nCi/g)[18] but is, nevertheless, an excellent capacity.

The advertised capacity range is apparently based on the observed performance of the RSC for uranium mine water decontamination[12] in which the influent radium concentration was much higher than in Lemont. Table 6 compares the RSC capacity observed in Lemont with that in some mine water applications. It is clear that the radium adsorption capacity depends on the radium concentration in the feedwater and is nearly independent of TDS and total hardness in the ranges shown. For example, the Panel Mine water contained approximately five times the total hardness and TDS of the Lemont well water, and yet the observed RSC capacity when the Panel Mine water was treated was nearly 10 times that observed in Lemont. This strong dependence of maximum radium loading on initial radium concentration was not unexpected and leads to the suggestion that RSC be used to remove radium from spent ion exchange regenerant brines. Snoeyink et al[27] attempted to determine the radium capacity of RSC used to treat regenerant brine. Their conclusions were that the capacity was drastically reduced with 40,000 mg/L brine solution and that the presence of calcium in the brine tended to further reduce the RSC capacity for radium. In contrast to this are the results of Mangleson,[28] who successfully used a 4.1-cu ft (117-L) RSC column in an ion exchanger brine decontamination operation in Redhill, Colo.

In the Redhill study, the RSC column treated spent brine containing 41,000 mg TDS/L, 9,000 mg hardness/L, and 1,200 pCi radium/L. At the end of one year, the RSC was still removing 99 percent of the influent radium and showed no sign of exhaustion. The calculated concentration of radium on the RSC in Redhill at the end of one year was 2.7 nCi/cm^3 (9.1 nCi/g). This capacity, although not an exhaustion value, is shown in Table 6 for comparison purposes. Although the RSC in Redhill operated successfully to remove radium from brine, insufficient data exist to conclude that the capacity for radium is not seriously influenced by TDS and hardness. The calculated loading of 2.7 nCi/cm^3 for a 1,200 pCi/L feedwater is far below the expected loading based on the Panel Mine and Key Lake Mine data. Only further brine treatment to completely exhaust the Redhill RSC will establish its capacity. It does appear, however, that RSC and similar BaSO$_4$-loaded media are potentially excellent media for brine decontamination.

Meyers et al[8] suggested the use of BaSO$_4$ precipitation in spent CaCl$_2$ ion exchanger brines, and Jongeward[29] measured the radium capacity of BaSO$_4$ precipitates in brine and nonbrine solutions. She found only a 50 percent reduction in radium adsorption capacity in 40,000 mg/L brine compared with the nonbrine solution. If batch processes involving BaSO$_4$ precipitation in brines are feasible, then BaSO$_4$-loaded media in columns ought to improve results. This is because the BaSO$_4$ crystals in the columns will be in equilibrium with the high influent radium concentration, whereas in batch precipitations, the crystals will be in equilibrium with the lower effluent radium concentration.

Radon production and removal. Radon, notably Rn-222, the immediate daughter of Ra-226, with a half-life of 3.82 days, is present in the radium-contaminated groundwater. When in secular equilibrium (after 20 days or more), the Rn-222 and Ra-226 activities are equal. However, because of factors governing solubility in the subsurface, Rn-222 activity in fresh groundwater is usually much greater than Ra-226 activity. Therefore, Rn-222 will both enter the RSC column and be generated within it from the adsorbed Ra-226. (The radon daughter of Ra-228, Rn-220, is not important because its half-life is only 50 s.)

Radon, an uncharged, monatomic, nonhydrated, inert gas is not removed by RSC. Rozelle[13] reported nearly equal influent and effluent concentrations (15,000–20,000 pCi/L) from an RSC column during a full-scale test of the complexer in Bellville, Texas. There was no mention of the radon generated within the complexer, probably because the activity generated was not significant compared with the incoming radon. In that installation, GAC columns to remove the radon were added following the RSC columns. Initial removal was 64 percent, but this decreased to 20 percent after one month of continuous operation. Apparently the GAC columns were undersized, because properly sized GAC columns continuously and effectively remove radon from groundwater.[30]

The Rn-222 generated continuously within the column from the adsorbed Ra-226 is either washed out during continuous operation or accumulated during no-flow periods. The Rn-222 generation rate is directly proportional to the mass of Ra-226 accumulated on the media. If the RSC column flow is continuous, the Rn-222 generation problem is not serious. For example, the completely exhausted RSC in Lemont with 1.1 nCi/cm^3 total radium activity would add about 175 pCi Rn-222/L to the water passing through the column at an EBCT of 2 min. (This is a calculated value; radon was not measured in the Lemont study.) There is, however, some concern about radon generation in a standing RSC column. Extremely high Rn-222 levels, approach-

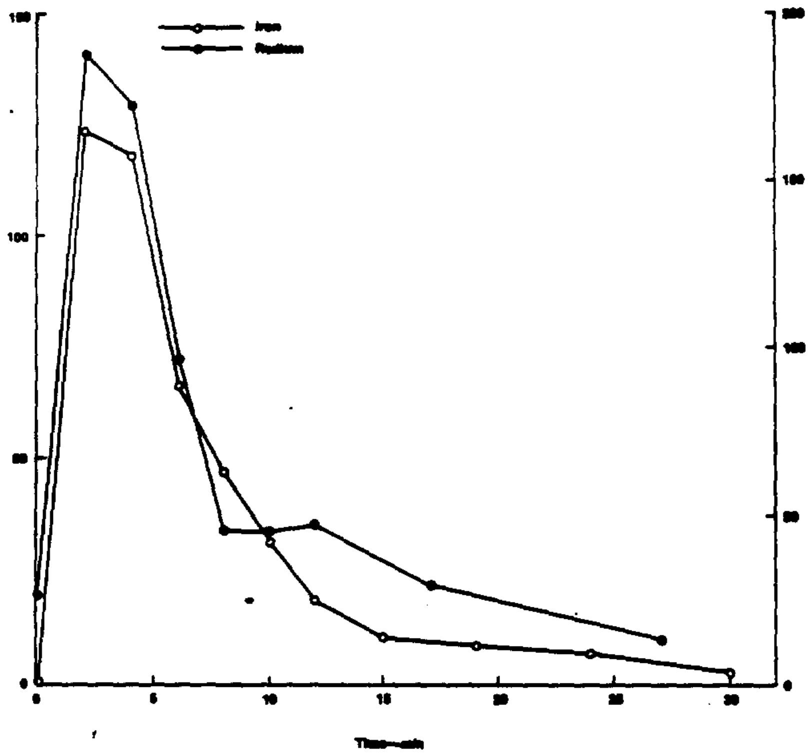

Figure 1. Iron and radium concentrations in backwash water from the deep-bed filter *(run—FB-1; filter backwash flow—5.1 gpm/sq ft; Ra-226 in raw water—11.8 pCi/L; Fe in raw water—0.2 mg/L)*

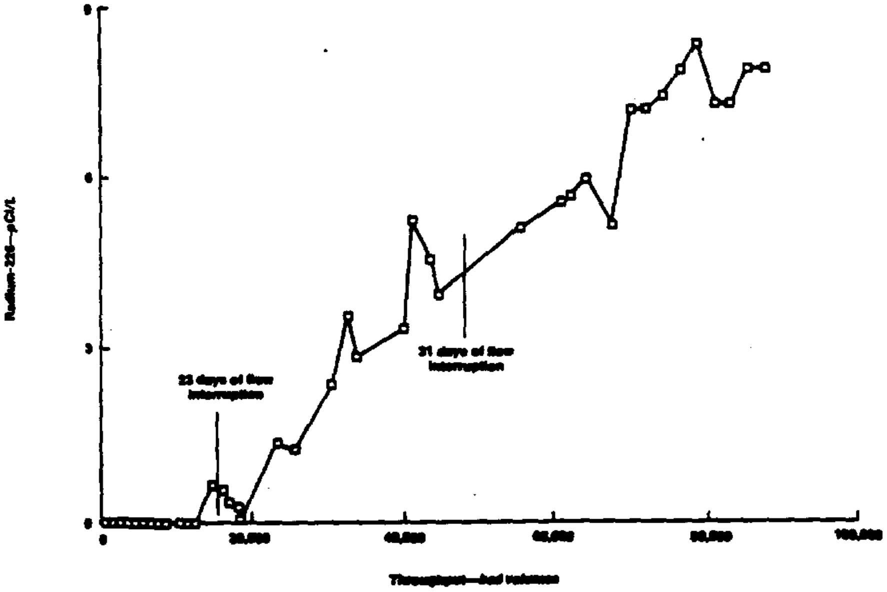

Figure 2. Radium breakthrough curve for RSC *(EBCT—0.64 min; bed volume—1.98 L; feed Ra-226—8.7 pCi/L)*

ing the adsorbed Ra-226 activity, are possible in the first few bed volumes following no-flow periods. Although radon measurements were not made in Lemont, the authors observed (using a radiation survey meter) some washout of radioactivity following the restarting of a partially exhausted RSC column. Similar observations were made upon restarting cation exchange columns following resting periods of up to 23 days.

The problem of radon washout upon restarting radium adsorbers after long rest periods should be quantified in future studies. Until more data are available, however, the first few bed volumes of effluent following long standing periods should be flushed to waste.

Disposal of spent RSC. The disposal options for radium-contaminated sludges and solids are not well-defined at this time. However, some regulations based on the radium levels in the sludges do exist. The situation in the state of Illinois regarding sludge disposal has recently been summarized in a state report.[31] This document points out that solids with radium contamination <50 pCi/g (on a dry weight basis) may be disposed of at existing Illinois EPA landfills. (This is such a low level that it is inapplicable to actual water treatment sludges.) Wastes with radium levels in the range 50 pCi/g to 10 nCi/g are currently undefined in Illinois. However, according to the report, "...it is useful for cost analyses to assume that waste containing ≥50 pCi/gram will have to be treated as low level radioactive wastes." Radium-contaminated solids in the range 10–100 nCi/g are considered as low level radioactive wastes, for which only one disposal site exists in the United States: US Ecology's Richland, Wash., site. At >100 nCi/g, the waste exceeds the limits permitted for near-surface disposal and must be suitably stored until disposal options are available.

As shown in this study, RSC is an excellent adsorbent for radium, but the disposal of spent media containing 3.5 nCi radium/g will be a costly proposition. It appears that the spent media must be stabilized, transported, and disposed of at the Richland site. In 1983,[13] the Dow Chemical Company reported the commercial availability of a "low level radioactive waste solidification process suitable for encapsulating the RSC into a stable, solid mass acceptable for disposal at sites approved by the Nuclear Regulatory Commission."[32] The company has now announced that RSC will no longer be commercially available, presumably because of low sales volume resulting from the inability to dispose of spent media at a reasonable cost. Whether the RSC medium will be available through another source is not known at this time. It is unfortunate that an excellent radium adsorbent should go out of production at a time when it is most needed.

Radium removal by activated alumina. The performance of plain alumina used to treat filtered Lemont water was unexpectedly good (Figure 3). Approximately 3,350 BV could be treated prior to Ra-226 breakthrough at 3.25 pCi/L, corresponding to 5 pCi/L Ra-226 plus Ra-228. At the unadjusted feedwater pH of 7.2, the alumina would have a slight positive charge[20] because the zero point of charge of F-1 alumina is approximately 8.2. Huang and Stumm[33] found significant adsorption of alkaline earth cations at elevated pH values (>8.0) on γ-alumina, but adsorption at pH 7 in dilute solutions was barely perceptible for the heavier alkaline earth elements strontium and barium. These researchers did not test radium, but an extrapolation of their data clearly suggests that it would be less adsorbable than barium. This good performance of plain γ-alumina for radium adsorption in the neutral pH range cannot be explained based on experiments done in Lemont. The phenomenon is under further study at the University of Houston where the role of $RaSO_4$ ion pairs[34] and complexation of radium by natural fulvic and humic acids are being studied in relation to radium adsorption onto γ-alumina.

$BaSO_4$-loaded alumina performed much better than plain alumina and nearly as well as RSC (Figure 4). Approximately 27,500 BV of filtered Lemont groundwater were treated prior to Ra-226 breakthrough at 3.24 pCi/L compared with 3,350 BV for the plain alumina and 38,000 BV for the RSC. Improved radium removal following periods of flow interruption (15 days and 7 days) suggests that intermittent operation of the alumina column would yield higher radium loading and lower radium concentration in the effluent. For example, resting the column for a day or two would result in improved radium removal upon restarting the flow. Stopping the flow to the column allows relaxation of the radium concentration gradient in the solid phase, which, upon restarting the flow, results in a higher liquid–solid concentration gradient and consequently a higher radium flux into the alumina granules. The phenomenon of improved performance resulting from intermittent operation of alumina beds is currently under study at the University of Houston. It has also been observed in studies of fluoride[21] and arsenic[35] removal by activated alumina.

The maximum concentration of radium on the exhausted $BaSO_4$-loaded alumina was measured by gamma spectroscopy. The results of analyses of spent alumina samples from the top, middle, and bottom sections of the bed are presented in Table 7. Even though the bed was run for 51,000 BV (106 days), it was still not completely exhausted, as evidenced by the lower loading of radium at the bottom of the bed. The top section of the bed, however, appeared to be exhausted or nearly exhausted. This high loading during the first experiment is an encouraging sign that an optimum $BaSO_4$-loaded alumina will be a viable alternative for removal of radium from groundwater in municipal-scale, point-of-entry (POE), and POU applications.

Barium in the effluent. The effluent from the RSC run shown in Figure 4 was not analyzed for barium. To determine barium leakage from the RSC, a separate run was conducted (Figure 5). The average amount of soluble barium added during passage of the water through the RSC was 0.17 mg/L for the first 3,500

TABLE 5

*Concentration of radium on exhausted RSC**

Location	Ra-226 *pCi/0.2 L*	Ra-228 *pCi/0.2L*	Ra *nCi/cm³*	Ra *nCi/g*	Ra-226 to Ra-228 Ratio
Top	30,031 ± 120	17,163 ± 168	1.18	3.9	1.75
Middle	27,782 ± 155	15,366 ± 317	1.08	3.6	1.81
Bottom	24,948 ± 106	13,734 ± 225	0.97	3.2	1.82

*All radium analyses were made by the gamma spectroscopy method.[24] The density of the exhausted RSC is approximately 0.3 g/cm³.

TABLE 6

Comparison of RSC loadings

Source Water	Ra in Feed *pCi/L*	Total Hardness *mg as $CaCO_3$/L*	TDS *mg/L*	Maximum Observed Radium Loading *nCi/cm³*	Maximum Observed Radium Loading *nCi/g*
Lemont	18.1	272	439	1.1	3.7
Panel Mine[11]	181.0	1,050	2,500	9.7	32.3
Key Lake Mine[11]	1,620.0	242	580	52.0	173.0
Redhill brine[27]	1,200.0	9.000	41,000	2.7*	9.1

*The radium loading for the RSC treating spent ion exchange brine in Redhill, Colo., is not a maximum value because the RSC was still removing 99 percent of the radium from the brine.[28]

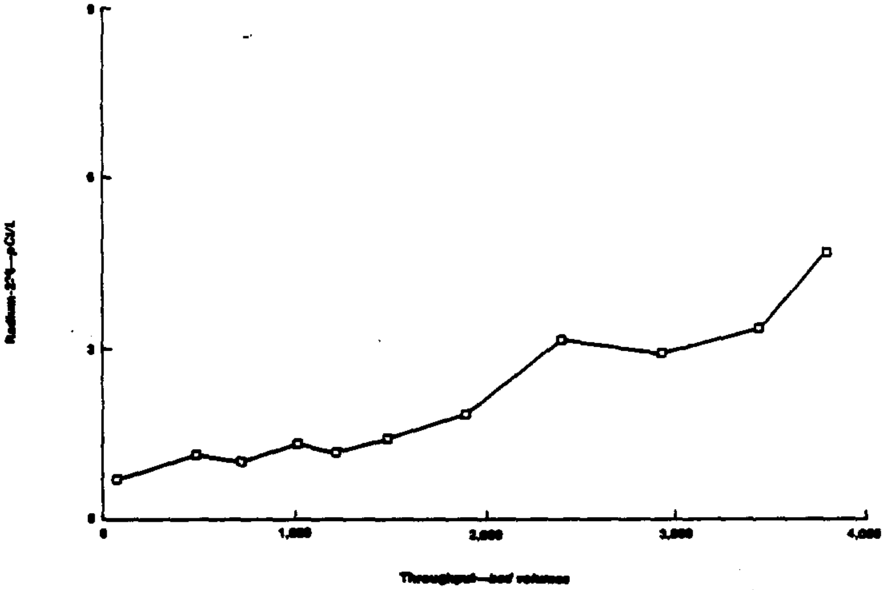

Figure 3. Radium breakthrough curve for plain activated alumina *(EBCT—3.0 min; bed volume—0.5 L; feed Ra-226—11.3 pCi/L)*

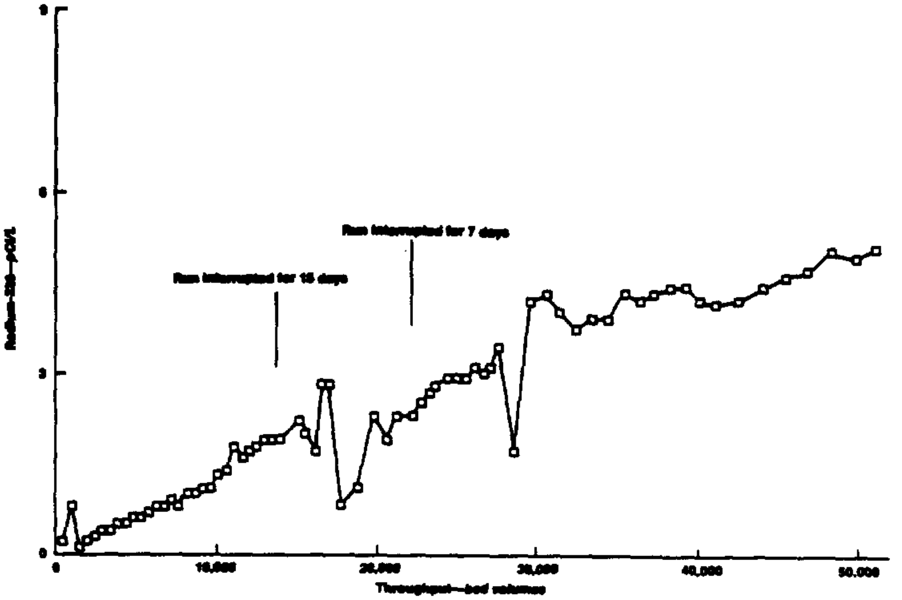

Figure 4. Radium breakthrough curve for $BaSO_4$-loaded activated alumina *(EBCT—3.0 min; bed volume—0.23 L; feed Ra-226—8.7 pCi/L)*

TABLE 7

*Radium loading on $BaSO_4$-impregnated alumina**

Location	Volume cm^3	Ra-226 *pCi*	Ra-228 *pCi*	Ra-226 to Ra-228 Ratio	Loading *nCi/cm³*	*nCi/g*
Top	80	33,512 ± 213	20,023 ± 218	1.67	0.67	0.77
Middle	100	36,731 ± 164	21,898 ± 383	1.68	0.59	0.68
Bottom	100	20,658 ± 197	12,176 ± 133	1.70	0.35	0.40

*Only the $BaSO_4$-loaded alumina at the top of the column is near exhaustion. The density of F-1 alumina is approximately 0.87 g/cm³.

TABLE 8

*Radium loading on various exhausted media**

Medium	EBCT *min*	Total BV	Run Time *days*	Total Radium Loading *nCi/cm³*	Bulk Medium Density *g.cm³*	Total Radium Loading *nCi/g*
Dow RSC (XSF 43230)	0.64	90,000	40	1.18	0.3	3.9
$BaSO_4$-loaded alumina	3.0	51,000	106.0	0.67	0.87	0.77
Isoclear CYC	1.5	5,500	5.7	0.00885	0.047	0.41
C-20 cation exchange resin	3.0	5,000	10.4	0.046	0.55	0.08

*Columns were run far beyond radium breakthrough to the MCL of 5 pCi/L. Radium loadings were actually measured using medium at the top (entrance) of the bed. Medium density is the approximate dry weight of exhausted medium/cm³ of wet exhausted medium. The bulk density of Isoclear CYC is an average; because of water swelling, the medium near the top of the columns was less dense (more swollen) and that at the *bottom more* dense (less swollen).

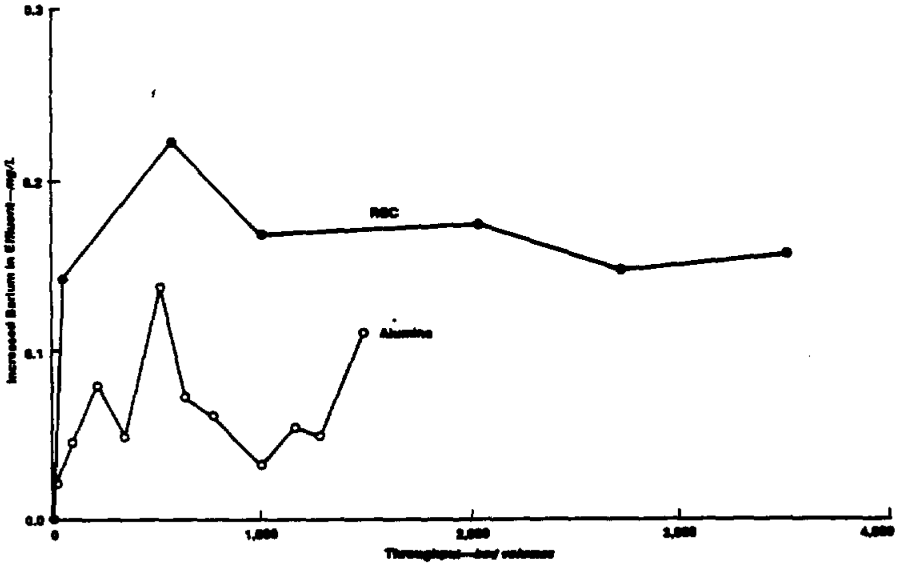

Figure 5. Increased barium concentrations in the effluent of the $BaSO_4$-loaded alumina and RSC columns

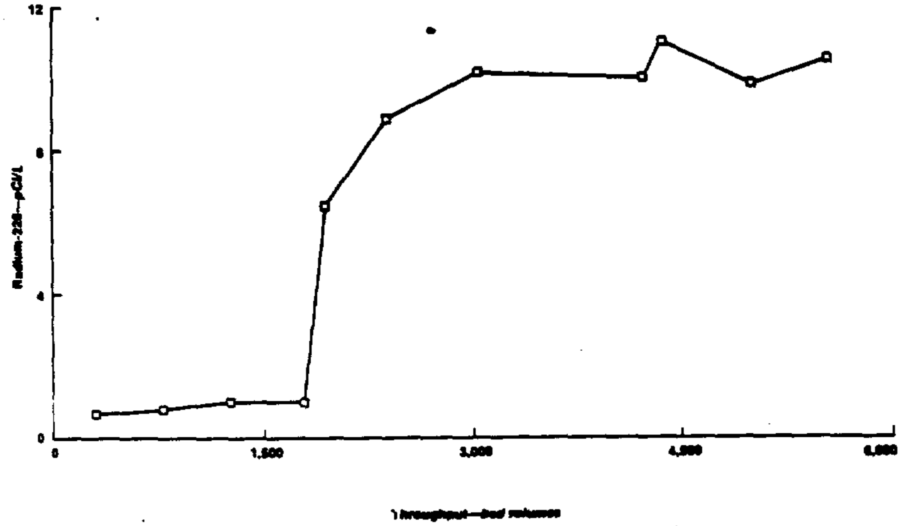

Figure 6. Radium breakthrough curve for CYC medium *(EBCT—1.5 min; bed volume—1.54 L; feed Ra-226—9.5 pCi/L)*

BV. This is far below the MCL of 1.0 mg/L and should be of no health concern. Significant barium leakage from the RSC was not observed in previous studies. In fact, a small amount of barium uptake by the resin was observed during exhaustion in one study in which the barium concentration in the feedwater was 0.05 mg/L.[12] Furthermore, RSC has been approved by the USEPA for use in the production of potable water, providing that a new bed is first rinsed to 10 BV of water.

$BaSO_4$-loaded alumina should show similar barium leakage to RSC because they both rely on $BaSO_4$-impregnated media for radium removal. During the preparation of the $BaSO_4$-loaded alumina, $BaSO_4$ must be rinsed from the alumina following its precipitation in the medium. The thoroughness of rinsing will determine the initial $BaSO_4$ particulate leakage, which can, if necessary, be controlled by postfiltration. Measurements of soluble barium in the effluent from a freshly prepared $BaSO_4$-loaded alumina column in Lemont are also presented in Figure 5. It is evident that with proper rinsing, soluble-barium leakage from the impregnated alumina is not significant: 0.06 mg/L during the first 1,500 BV.

Radium removal by other adsorbents. The first run using the RadiSorb beads was aborted because of the collapse and breakup of the medium early in the run under normal operating pressure—3–9 psig. This occurred even after extensive backwashing to remove the large amount of fines in the as-received sample. A second sample of RadiSorb was obtained and was operated both downflow and upflow with the same resulting breakup of the medium. This medium was too fragile for practical use even if it had possessed a high radium capacity. Some radium was removed, but the MCL was reached after only about 10 BV.

The CYC medium performed much better than RadiSorb with respect to radium removal but was nearly as fragile. During backwashing, the CYC medium tended to agglomerate, and during exhaustion the beads at the top of the column swelled while those at the bottom appeared seriously compressed. The swelling of the hydrated gel beads produced an overall bed expansion of 77 percent during the course of the run shown in Figure 6. The radium breakthrough at 1,800 BV was sharp and showed this fragile, presumably nonregenerable medium to be almost equivalent in radium capacity to a standard, regenerable SAC softening resin.

Table 8 compares the radium loadings on the various exhausted media; these are measured, not calculated, total radium (Ra-226 + Ra-228) loadings. They are the result of running the columns far beyond the MCL in an attempt to achieve complete exhaustion. Thus the loadings

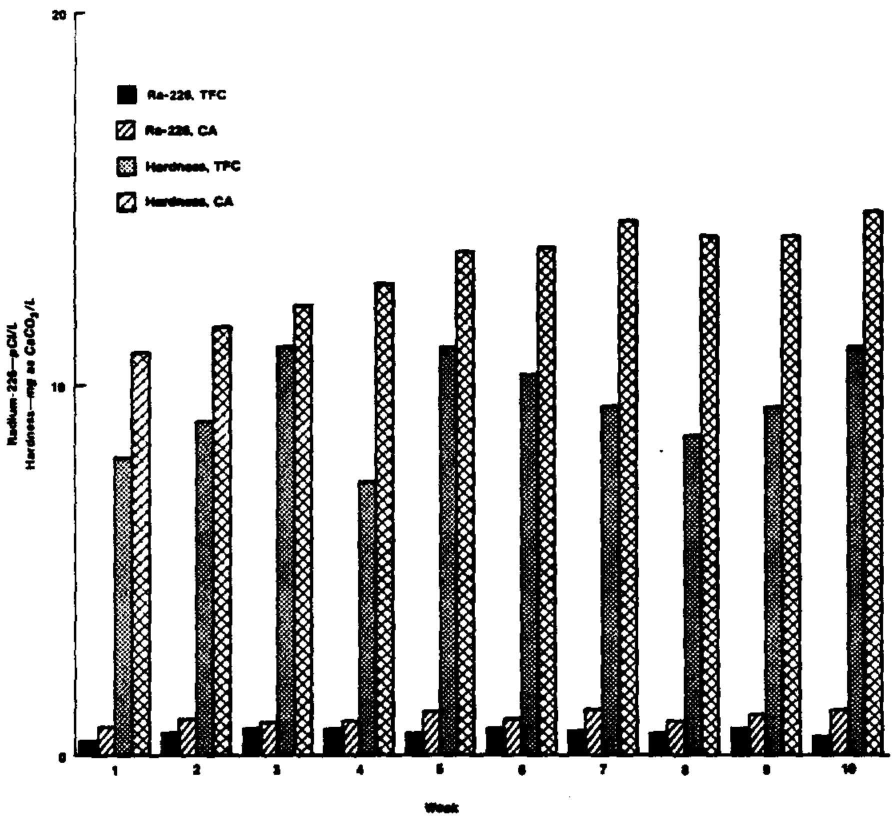

Figure 7. Radium and hardness in the effluent from the point-of-use system

TABLE 9

*Performance of reverse osmosis system**

Test Parameter	Type of Module Thin-Film Polyamide Hollow-Fiber	 Low-Pressure Composite Spiral-Wound	 Thin-Film-Composite
Designation	RO-1	RO-2	RO-3
Length of run—*days*	5	5	5
Water recovery—*percent*	50	15	25
Feed pressure—*psi*	350	125	70
Raw water pH	7.3–7.5	7.3–7.5	7.3–7.5
Adjusted feed pH	5.5–6.0	5.5–6.0	5.5–6.0
Conductivity			
Feed—*μs*	750	850	833
Product—*μs*	65	21	158
Brine—*μs*	1,450	1,000	979
Rejection—*percent*	91	97	81
Total dissolved solids			
Feed—*mg/L*	510	681	451
Product—*mg/L*	9	4	59.8
Brine—*mg/L*	1,022	838	561
Rejection—*percent*	98	99	86.8
Hardness			
Feed—*mg as $CaCO_3$/L*	253	249	247
Product—*mg as $CaCO_3$/L*	13	5	25.1
Brine—*mg as $CaCO_3$/L*	489	282	299
Rejection—*percent*	95	98	90
Ra-226			
Feed—*pCi/L*	11.6	13.9	13.0
Product—*pCi/L*	<0.1†	<0.1†	1.2
Brine—*pCi/L*	19.2	14.5	14.1
Rejection—*percent*	>99	>99	90.8

*Raw water characteristics: silt density index—3.2; free chlorine—1.0-1.5 mg/L; sodium sulfite was used for dechlorination at a ratio of 1.8 mg Na_2SO_3/mg Cl_2.

†The alpha counts of these samples could not be distinguished from those of radium-free blanks made from deionized RO product water.

are approximations of the maximum useful capacities expected in drinking water applications when the media are used for direct treatment. If the respective media are used for radium removal from spend regenerant brines, much higher loadings are expected.

The RSC capacity for radium (1.18 nCi/cm^3) is the highest of all, whereas the nonoptimized $BaSO_4$-loaded alumina is not far behind in radium capacity based on volume. Approximate densities of the media—grams of dry (105°C) medium per cubic centimetre of settled wet medium—are listed in Table 8. These densities were used to calculate the radium loadings on a weight basis—nanocuries of radium per gram of dry (105°C) adsorbent—for comparison with federal (10 CRF 60) and state sludge disposal regulations. The CYC has the lowest dry density, indicating that, when dried, most of its weight is lost. If drastic volume reduction is important, however, the possibility of combustion of the media must be considered. In this regard, the organic-based media—RSC, CYC, and cation resins—will yield greatly reduced volumes compared with activated alumina, which does not burn. Volume reduction is not always desirable because it can result in a radium loading >100 nCi/g, the maximum allowable for near-surface disposal of radioactive waste.[12]

RO systems. Table 9 summarizes the performance of the three different RO modules tested in Lemont. For the standard-pressure modules (RO-1 and RO-2), Ra-226 rejection exceeded 99 percent, whereas for the low-pressure (70-psig) module, it was 91 percent. In all three cases, radium rejection slightly exceeded hardness rejection, suggesting that hardness monitoring might be used as a surrogate for radium. This same trend of nearly equal radium and hardness removal can be observed in data reported by Sorg et al.[10] In their study of five hollow-fiber polyamide and three spiral-wound cellulose-acetate module systems, the average removal of Ra-226 and hardness was 96 percent.

Membrane technology is advancing at a rapid rate, and the newer standard SW TFC and CA membranes require much less feedwater pressure (125–400 psig) than the older standard units (400 psig). The experimental SW TFC module tested in Lemont required only 70 psig and yet produced 87 percent TDS rejection while reducing hardness and radium by 90–91 percent. The single-module recovery of the unit was 25 percent, i.e., higher than the manufacturer's recommendation and yet only half that of the HF module. (One major difference not always recognized between HF and SW modules, regardless of their membrane composition, is that HF modules typically recover 50 percent of the feedwater whereas SW modules recover only 12–18 percent. The signif-

TABLE 10
*Performance of point-of-use system**

Parameter	TFC Membrane	CA Membrane
Influent hardness—*mg as CaCo$_3$/L*	255	255
Effluent hardness—*mg as CaCO$_3$/L*	8.6	13.1
Hardness rejection—*percent*	96.6	94.8
Influent Ra-226—*pCi/L*	11.8	11.8
Effluent Ra-226—*pCi/L*	0.6	1.0
Radium rejection—*percent*	94.9	91.5
Conductivity rejection—*percent*	90	91

*The values are averages of 10 weekly grab samples taken from the product water storage tank during normal operation.

icance of this difference should not be overemphasized because it is a design problem readily solved by product staging of the SW modules for higher recovery.) At the usual RO system design value of 75 percent recovery, the brine will be concentrated by a factor of four, and brine disposal might be a problem in some states.

Although RO desalting remains an expensive technology for the removal of a single contaminant, the continuing advances in membrane technology are bringing it into a more competitive position. At a 70-psig operating pressure, household POE softening systems that would also remove radium are nearer to realization. A little recognized advantage of low recovery RO systems is that the brine concentration factor (CF) is low. In the case of single SW modules, as used in POE and POU systems, recovery is typically 15 percent and the brine CF is only 1.18. Of course, the disadvantage of such a low recovery system is that 85 percent of the feedwater is wasted unless a separate piping system is installed to store and reuse the reject water for toilet flushing, lawn watering, and the like. In POE systems, product water storage is also a major consideration because of the low production rates of RO systems compared with alternatives like softeners that operate on demand.

POU systems. The radium removal performance of the POU RO system with both the CA and TFC modules was similar to that of the full-scale modules tested in that the TFC module performed slightly better than the CA module for the rejection of radium, hardness, and TDS. (The CA module has an operational advantage, however, in that it is not attacked by the chlorine levels in drinking water; feed to a TFC module must be dechlorinated whereas CA feed need not be.) The average performance of both modules is summarized in Table 10, and the week-by-week effluent radium and hardness levels are presented in Figure 7. These POU results demonstrate that the use of either TDS or hardness rejection would be adequate as a surrogate for estimating radium removal.

In an independent test of Ra-226 removal conducted at the University of Houston with a POU RO system from another manufacturer, essentially the same rejections of Ra-226, hardness, and TDS were observed for a different CA membrane. Thus, it can be concluded that a properly designed and manufactured POU RO system containing either module would be adequate for radium removal from drinking water.

Conclusions

The Lemont groundwater contains a mixture of Ra-226 (11 pCi/L), Ra-228 (6 pCi/L), and Ra-224 (6 pCi/L). The presence of the Ra-224 interfered with the interpretation of the gross radium alpha test (USEPA method 900.1). However, allowing the raw sample or the precipitated $BaRaSO_4$ to age for 20 or more days eliminated the interference and made the test specific for Ra-226. Before implementation of the 20-day wait time, the gross radium alpha and gross alpha test results were highly variable and impossible to interpret.

The Ra-226 to Ra-228 ratio in the raw water was about 1.8. This ratio remained essentially unchanged for all aqueous and adsorbent samples analyzed and supported the assumption that the isotopes behaved in a physicochemically equivalent manner in spite of the presence and rapid decay of Ra-224. Thus, monitoring Ra-226 alone is sufficient to determine compliance with the MCL.

Filtration of the chlorinated Lemont groundwater resulted in 30–50 percent radium removal. The mechanism of removal is uncertain but likely involved adsorption onto $Fe(OH)_3$ or MnO_2 solids.

The radium-selective complexer was the best adsorbent tested for radium removal on the basis of BV to the MCL (38,000) or maximum capacity (1.18 nCi/cm^3). The RSC is a standard strong acid cation resin in the sodium form with $BaSO_4$ precipitated within the bead and is considered, although not proven, to be nonregenerable.

A nonoptimized sample of an experimental adsorbent, $BaSO_4$-loaded activated alumina, performed nearly as well as the RSC by attaining 27,500 BV to the radium MCL and a maximum loading of 0.67 nCi/cm^3. This adsorbent could prove to be a good substitute for RSC.

The radium capacity of RSC, and presumably of $BaSO_4$-loaded alumina, is highly dependent on the radium concentration in the feedwater and does not seriously deteriorate in the presence of brine or high levels of calcium as present in spent regenerant solutions.

Barium leakage from the RSC and $BaSO_4$-loaded alumina was measurable (0.05–0.2 mg/L) but not significant with respect to the 1.0-mg/L barium MCL.[33]

The disposal of spent radium-contaminated adsorbent media at a reasonable cost is an unsolved problem. It appears that spent media like RSC and $BaSO_4$-loaded alumina are now considered low-level radioactive wastes and may be disposed of only at one US location in the state of Washington.

Reverse osmosis hyperfiltration removes radium to a somewhat greater extent than it does hardness. For the standard-pressure (125–400-psig) modules, radium removal exceeded 99 percent. An experimental low-pressure TFC membrane module operated at 70 psig and removed 90 percent of the hardness and 91 percent of the radium.

Point-of-use treatment systems producing 8 gpd (30 L/d) of product water and utilizing TFC or CA SW membranes were very effective for radium removal. Approximately 30 percent of the radium was removed during the cartridge prefiltration step, whereas the overall removals were 95 percent and 92 percent, respectively, for the systems with TFC and CA membranes.

Recommendations

The federal government and individual states with radium problems must devise regulations and identify sites for the disposal of spent adsorbent media containing up to 10 nCi radium/g, i.e., the lower limit for low level radioactive wastes. If possible, these media should not be classified as low level radioactive wastes. Unless new regulations are promulgated and local disposal sites identified, there is no future for radium-selective adsorbents, either for the direct treatment of well water or for the decontamination of ion exchanger brines. Removing selective adsorption from the list of processes available to extract radium from drinking water will force communities that must treat to use processes like ion exchange softening that produce lower level radioactive wastes in much greater volumes. This will not diminish the amount of radium discharged to the environment but will, in fact, only distribute it more widely in the rivers and lakes into which the waste brines are eventually discharged by way of sewage plant effluents.

Laboratory and field studies should be carried out to determine the feasibility of using RSC and $BaSO_4$-loaded alumina to decontaminate spent ion exchanger brine prior to its reuse in processes that would selectively remove radium without softening. In such processes, sodium would be the initial exchangeable ion on the resin. After each regeneration, the brine would be decontaminated using a radium-selective adsorbent, and the brine would be reused. Eventually, the spent brine would primarily contain those ions in the feedwater, except for radium. A small amount of NaCl would have to be added as makeup during each cycle to replace brine lost during rinsing. Such processes would (1) introduce little sodium into the product water, (2) use only a small amount of salt for regeneration, and (3) produce a minimum amount of brine to be disposed of in the sanitary sewer. But the processes would produce a spent adsorbent medium for ultimate disposal. These experiments should be given the highest priority.

Activated alumina and $BaSO_4$-loaded alumina should be studied in bench-scale experiments to establish the optimum conditions for adsorbing radium vfrom waters of varying pH and background ionic composition. Experiments should be designed and executed to elucidate the mechanisms of radium adsorption, which may involve $RaSO_4$ ion pairing or complexation with natural organic matter.

The regeneration of spent RSC and $BaSO_4$-loaded alumina should be studied using the usual acid and base regenerants. The primary objectives should be to determine (1) the usefulness of the regenerated media, (2) the maximum possible concentrations of radium in spent regenerants, and (3) the possible means of final disposal of the radium-contaminated wastewaters.

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Commissioning the filter. Several steps are taken when a new filter unit is commissioned. With all outlet valves closed, clean (preferably) water is admitted from the bottom to flow upward through the drainage system, the gravel, and the sand bed until it reaches 10-15 cm above the sand bed. This method of charging ensures that air accumulated in the system, especially in the pores of the sand bed, is driven out. When no clean water is available for backfilling, raw water can be used.

Next, the inlet valve is gradually opened and water is allowed to flow into the shallow water layer on top of the bed. The rate of filling is initially low to prevent scouring of the sand around the inlet. With the increasing depth of the supernatant water, the rate of filling can be increased. When the normal working depth of the supernatant is reached, the outlet valve is opened and the effluent is run to waste at the design rate of the filter. The filter must now be run continuously for a few weeks to allow for ripening. The ripening period ends when bacteriological analysis indicates that the effluent quality has reached local water quality standards. Water can then be passed into supply. When adequate chlorination of the effluent can be guaranteed, the water can often be put into supply more quickly, provided that the turbidity of the effluent has become sufficiently low.

Daily operation. Daily operation is limited to checking and possibly adjusting the rate of filtration and to monitoring plant performance and effluent quality. More time is occasionally required for cleaning the filters.

Cleaning. To clean the filter it is necessary to first remove floating matter such as leaves or algae by raising the level of water in the unit to carry the floating matter over the weir. If this floating matter is not removed, it becomes a nuisance during cleaning. Subsequently, the water level in the bed is lowered to about 0.1-0.2 m below the top of the sand. This can be done by closing the inlet valve and opening both the supernatant water drain valve and the valve on the underdrains to quickly drain the filter to the required level.

The filter bed is then cleaned by scraping off the top 1-2 cm of the bed. The cleaning operation must be completed as quickly as possible in order to minimize interference with the life of microorganisms in the filter bed. When one unit is shut down, the others are run at a slightly higher rate to maintain the output of the plant.

After cleaning, the unit is refilled with water through the underdrains. In the absence of an overhead storage tank, this backfilling can be effected by using filtered water from an adjacent filter. This method may lead, however, to temporarily reduced plant output, which must be taken into account when the clear-water storage tank is designed because it must be assured that sufficient water is available for the users. An alternative method is to advise users to reduce water consumption for a short period of time.

When the filter is put back into service, a period of at least 24 hours is required to allow for reripening of the bed. After that period, the bacteriological flora has usually been sufficiently reestablished to be able to produce a safe effluent. In cooler areas this ripening period may have to be extended to a few days. With adequate chlorination, however, a period of one day will also be sufficient in these areas, provided the turbidity of the effluent is sufficiently low.

Resanding. Resanding becomes necessary when successive scrapings have reduced the thickness of the sand bed to 0.5-0.6 m. Fortunately, this rather lengthy operation has to be done only at two- or three-year intervals. Assuming, for example, an initial thickness of the filter medium of 80 cm and an average of six scrapings a year (or 6 × 2 = 12 cm total scraping a year), then resanding would be necessary only after two and a half years of operation. Before resanding, the filter bed must be cleaned and the water level in the sand must be lowered to the gravel layer. Then a layer of "new" sand (accumulated from earlier scrapings) is placed underneath the sand layer in the bed.

The method of handling the sand is as follows: The old sand that has to be replaced is moved to one side, the new sand is placed, and the old sand replaced on the top of the new. This "throwing over" process is carried out in strips (Figure 4). The removed material from the first strip is stacked to one side in a long ridge, the excavated trench is filled with washed or new sand, and the adjacent strip is excavated, throwing over the removed material from the second trench to cover the new sand in the first. When the whole of the bed has been resanded, then the material from the first trench is used to cover the new sand in the last strip. By doing so, the layer of old sand (which is rich in microbiological life) is replaced at the top of the filter bed, which will enable the resanded filter to become operational within a minimum reripening period.

Summary

Slow sand filters are used for water treatment in many parts of the world. This biological water treatment method is rapidly gaining importance for application by communities in developing countries as well as by utilities—particularly small systems—in other countries. When raw-water quality is high, a conservative design approach is used, and the plant is operated and maintained with care, slow sand filtration can produce water of good quality.

Acknowledgment

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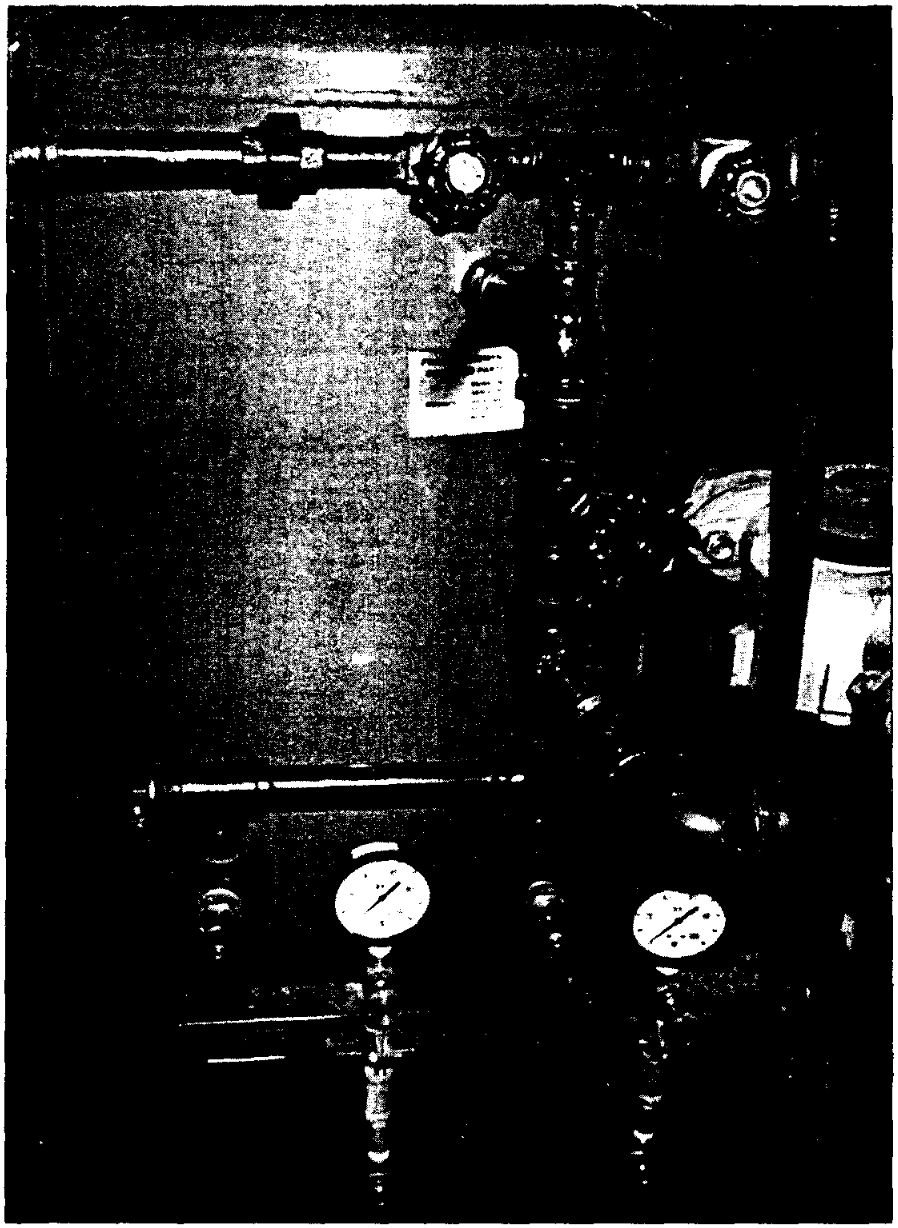

Regeneration water from the ion exchange process passes through the complexer tank before final disposal.

- The final disposal of smaller amounts of the radium-enriched RSC resin is simpler and less costly.

Plant processes

Raw water from two deep wells is pumped through a countercurrent-flow aeration tower located at the booster pump house (Figure 1). The purpose of the aeration process is to remove dissolved gases, specifically radon and carbon dioxide, from the raw water. The water is pumped to the treatment plant at a rate of about 90-100 gpm for further treatment to remove iron, radium, and hardness prior to chlorination and discharge to the water distribution system.

As the raw water enters the treatment plant, alum, potassium permanganate, and a polyelectrolyte are added to remove iron by chemical precipitation. The treatment unit is a prefabricated, self-contained unit that includes a mixing and flocculation chamber, tube settlers, and multimedia filtration. The effluent from the iron removal process is further treated to remove radium and hardness in an ion exchange system that utilizes a standard water-softening cation resin in the sodium form. The effluent from the ion exchange system is treated with chlorine and zinc hexametaphosphate to control corrosion and sequester any residual iron before being discharged to the treated-water storage tank. The radium removed in the ion exchange process is further removed from the regeneration brine by passing the brine through a separate treatment process in which the radium is permanently complexed on the RSC material. The wastewater from this process, along with the backwash wastewater from the iron removal process, is then pumped to the I-E pond for final disposal.

Disposal of wastewater and resin wastes

The original concept and design for the ultimate disposal of resin wastes and wastewater generated at the treatment plant was approved by the Colorado State Health Department.

Wastewater. All wastewater from the plant operation is discharged into the I-E pond to allow for rapid infiltration of plant wastewater into the Morrison geologic formation that dips steeply to the east from the pond. The Morrison formation is physically located beneath the Dakota geologic formation at the location of the raw-water supply wells. Raw water is pumped from the Dakota formation through deep wells to provide the raw-water supply for the Redhill Forest development.

Radium waste. Most of the radium removed from the raw water entering the treatment plant is eventually adsorbed and complexed on the RSC resin. As needed, the RSC resin is replaced, and the waste resin is transported to an approved hazardous-radiological waste disposal facility for final disposal.

Sampling and analyses

The project consisted of in-depth monitoring of the operation of the full-scale Redhill Forest water treatment plant over a 21-month period from October 1985 through June 1987. Concentrations of all water quality parameters were determined according to *Standard Methods*.[5]

Most of the water quality analyses were performed by a commercial lab* in Golden, Colo. Some work was performed by the USEPA laboratory in Cincinnati, Ohio, and some radon gas analyses were performed by a private firm.†

In-depth monitoring, including water quality sample collection and laboratory analyses, field measurements, and flow measurement, was performed to evaluate

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*Hazen Research Lab, Golden, Colo.

†Lowry Engineering, Unity, Maine

TABLE 1

Quality of influent and effluent of ion exchange process

Parameter	Influent	Effluent
Flow rate—*gpm*	90-100	90-100
Iron—*mg/L*	0.15-2.7	0.03-0.5
Manganese—*mg/L*	0.4-1.3	0.01-0.15
Sodium—*mg/L*	7.4-12.5	40-150
Hardness—*mg/L*	212-350	5-70
Radium-226—*pCi/L*	22-35	0-4

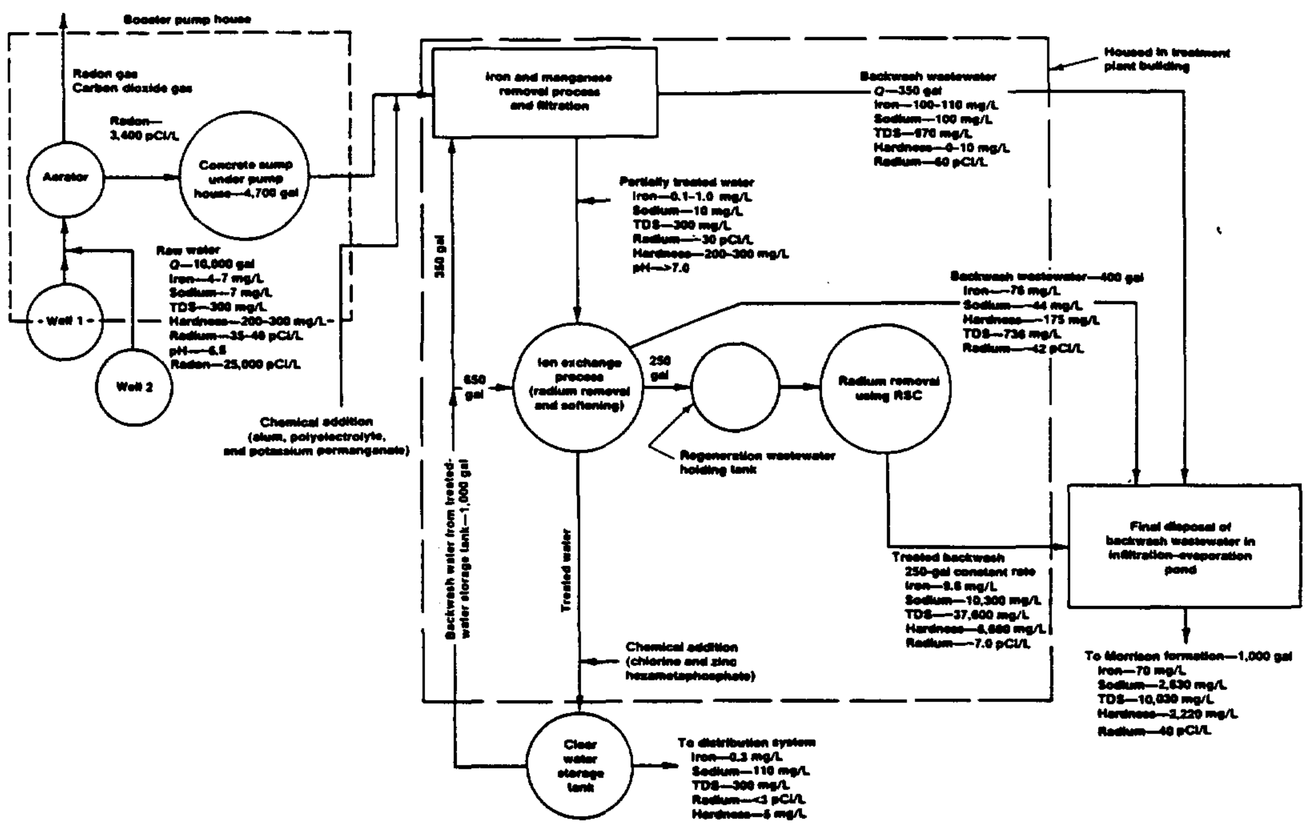

Figure 1. Flow diagram of treatment plant processes (*flows shown are averages for every 10,000 gal of raw water processed; ion exchange tanks are assumed to be backwashed after 40,000 gal of water have been treated*)

TABLE 2

*Water quality data for inflow and outflow of RSC resin**

Date	Accumulated Treated Volume *gal*	Bed† Volumes	Sample	Iron *mg/L*	Manganese *mg/L*	Sodium *mg/L*	Hardness *mg/L*	Total Solids *mg/L*	Total Radium *pCi/L*	Radium Removal *percent*
				Parameter						
7/10/86	0	0	Inflow	2.48	23.8	11,600	476	34,900	860 ± 30	
			Outflow	0.98	16.7	13,300	245	34,600	16 ± 11	98.1
7/30/86	2,400	77	Inflow	2.03	31.8	11,000	9,850	41,700	1,280 ± 40	
			Outflow	1.56	32.2	11,000	10,200	41,800	1.6 ± 3.2	99.9
8/31/86	9,460	305	Inflow	9.0	33.1	12,600	11,500	54,200	1,400 ± 40	
			Outflow	8.5	33.1	12,700	11,600	55,200	9.4 ± 3.5	99.3
9/29/86	14,600	471	Inflow	7.21	30.5	11,400	8,350	37,600	920 ± 30	
			Outflow	7.15	31.5	11,500	8,420	37,600	4.1 ± 2.4	99.6
10/30/86	22,600	729	Inflow	2.79	33.1	8,170	9,380	35,000	860 ± 50	
			Outflow	2.07	33.5	8,640	10,100	35,300	5.3 ± 2.8	99.4
11/26/86	27,700	894	Inflow	7.17	28.2	13,400	9,620	45,400	1,040 ± 30	
			Outflow	6.30	26.9	13,300	9,520	45,500	8.1 ± 3.3	99.1
1/14/87	39,550	1,276	Inflow	31.4	19.2	9,350	7,260	31,300	1,070 ± 60	
			Outflow	27.8	18.1	9,000	7,740	30,400	8.4 ± 2.3	99.2
2/21/87	49,700	1,603	Inflow	61.3	30.5	12,300	10,900	49,800	1,780 ± 80	
			Outflow	62.2	32.4	12,100	11,400	50,400	7.2 ± 7	99.6
3/18/87	57,700	1,861	Inflow	93.4	31.2	13,500	11,600	53,200	2,000 ± 80	
			Outflow	92.0	31.9	12,400	12,600	53,300	18 ± 9	99.1
6/10/87	71,700	2,313	Inflow	8.08	17.4	8,070	5,580	28,100	650 ± 20	
			Outflow	3.76	15.8	8,460	4,940	28,300	9.2 ± 2.4	98.6
Average			Inflow	19.8	29.9	10,850	8,890	40,590	1,181	
			Outflow	18.5	27.7	10,760	9,030	40,550	9.0	99.2

*From 7/10/86 to 6/10/87, i.e., 335 days, 71,700 gal of plant wastewater was treated in RSC tank. The following is the estimated amount of radium removed and deposited on the resin: radium removed = 71,700 (3.785) (1,181-9.0) = 318.1 × 10^6 pCi = 318.1 μCi or about 0.949 μCi/day; estimate for year = 347 μCi

†Resin bed volume—4.15 cu ft (31.0 gal)

the various components of the treatment plant operation.

Aeration system. Water samples were collected and analyzed for radon before and after aeration.

Treatment system. Samples were collected before and after the iron removal system and were analyzed for iron, manganese, gross alpha and gross beta radioactivities, and radium-226. The process wastewater from backwash operations was also analyzed on several occasions to determine the composition of the wastewater discharged to the I-E pond. Parameters of primary interest in the wastewater were total iron, manganese, total solids, and radium-226.

Ion exchange process. Water samples, collected before and after this unit process, were analyzed for iron, manganese, sodium, hardness, gross alpha radioactivity, gross beta radioactivity, and radium-226. Samples were collected on several occasions from the backwash, regeneration, and quick rinse water.

RSC process. This process was monitored frequently to determine the efficiency of radium removal from the ion exchange process wastewater and the buildup of radium in the complexer resin. Environmental radiation monitoring of the area outside the RSC tank surface was performed to determine potential exposure and to relate the exposure to radium buildup on the complexer resin.

I-E pond. Monitoring of radium present in the sand and soils at the bottom of the I-E pond was performed in order to determine the extent of radium buildup resulting from the disposal of plant wastewater that contained small amounts of radium.

General monitoring of plant flow rates, volumes of water processed, and wastewater volumes was performed for use along with water quality data in determining plant process efficiencies and plant operation and maintenance costs.

Limited measurements of radon gas were conducted on site using a radon-radon daughter detector unit.* Also, some samples were sent to a private firm† for additional radon gas analysis.

Results

Figure 1 shows the flow volume for each part of the total system operation for an assumed flow volume of 10,000 gal of raw water into the plant. The average water quality data for each process that makes up the treatment plant are also presented.

Aeration. The aeration system effectively removed radon and carbon dioxide from the raw water. Carbon dioxide is typically reduced from about 125 to 25 mg/L in the aeration system. The reduction of radon is about 85 percent, from about 23,000 pCi/L in the raw water to about 3,400 pCi/L. Additional measurements indicate that the radon gas concentration in the treated water from the main treatment plant is about 600 pCi/L. The iron removal process removed about 13 percent of the radium from the inflow to this process. This agrees with the results of Valentine,[6] who reported that <20 percent radium is removed with iron removal. When the iron removal system was backwashed, the radium removed was wasted in the I-E final disposal pond. Based on the results of the monitoring of the backwash water, the average concentration of radium in the wastewater was about 60 pCi/L.

Ion exchange. The ion exchange process is very effective in removing radium, hardness, and residual iron, and in polishing the effluent from the iron removal process, as long as the ion exchange capacity is not exceeded. The monitoring results generally indicate radium-226 levels of <3 pCi/L and iron levels of less than the recommended maximum level of 0.3 mg/L. Frequent monitoring of the system operation indicates that radium breakthrough occurs between 40,000 and 45,000 gal, i.e., 178 to 200 resin bed volumes. The quality of the influent and effluent of the ion exchange process was generally as given in Table 1.

RSC system. On July 10, 1986, new RSC resin was placed in the complexer tank and monitoring of the flow rate and the water quality of the inflow and outflow was initiated. Table 2 presents a summary of some of the results of the monitoring from July 10, 1986, through June 1987. The flow rate through the column is about 22 gpm, which is equivalent to a surface loading rate of about 10 gpm/sq ft. The RSC resin bed depth is 2 ft.

It can be seen in Table 2 that the RSC resin is highly radium-selective, with generally >99 percent removal of radium from the influent wastewater. The average inflow and outflow water quality data in Table 2 indicate that iron, sodium, hardness, and total solids were virtually unchanged in passing through the resin bed, whereas >99 percent of the radium in the influent was removed and concentrated on the RSC resin. The total quantity of radium removed and concentrated on the resin from July 10, 1986, to June 10, 1987, is also shown in Table 2. Based on the operation of the plant during this time, the rate of radium buildup on the RSC resin was about 347 μCi/year (347 × 10^6 pCi/year).

Further, it has been determined that the rate of radium removed from the raw water and permanently complexed on the RSC resin was about 9.6 μCi (9.6 × 10^6 pCi) per 100,000 gal of water treated at the plant. After a certain time of operation, the RSC resin containing the radium will be removed from the RSC tank and replaced with new resin, and the old resin will be disposed of at an acceptable low-level radioactive waste disposal site. It is anticipated that the RSC resin will be replaced when the radium on the

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*RDA-200, EDA Instruments Inc., Toronto, Canada

†Lowry Engineering, Unity, Maine

TABLE 3

*Results of environmental radiation monitoring of area around RSC tank**

Month	Average Environmental Radiation		Radium in RSC Resin at Mid-Month
	mrem/week	*mrem/h*	*pci*
November 1986	38.6	0.230	106.5×10^6
December 1986	13.0	0.077	132.1×10^6
January 1987	33.6	0.200	155.1×10^6
February 1987	40.4	0.240	185.1×10^6
March 1987	48.2	0.287	239.0×10^6
April 1987	67.9	0.404	300.5×10^6
May 1987	89.8	0.535	307.8×10^6
June 1987	62.0	0.369	315.1×10^6
July 1987	36.5	0.217	ND†
August 1987	46.3	0.276	ND
October 1987	41.3	0.245	ND
Nov.-Dec. 1987	95.2	0.567	ND
Jan.-Mar. 1988	83.3	0.496	ND
Apr.-Jun. 1988	89.0	0.530	ND
July-Sept. 1988	136.5	0.813	ND

*Location of radiation monitor is 3 in. from tank surface at mid-depth of RSC resin in tank.

†ND—not determined

TABLE 4

Treatment plant operating costs

Item	Cost of Water Treated—*$/1,000 gal*
Plant chemicals, e.g., alum, permanganate, chlorine	0.137
Salt	0.475
Energy costs	0.206
RSC resin disposal (includes new resin)	0.088
Total	0.906*

*Operator cost not included

complexer reaches about 3,080 μCi (3,080 $\times 10^6$ pCi). The 4 cu ft of RSC resin will be placed in a 55-gal drum, 3.35 cu ft of concrete will be added, and the entire drum will be transported to the disposal site for final and permanent disposal. This method of handling the radium waste ensures that the total radium content of the container to be buried will not exceed 10 nCi/g (10,000 pCi/g).

Environmental radiation. Beginning in November 1986, environmental radiation monitoring near the RSC tank was conducted to determine potential exposure levels. Table 3 presents the results of the monitoring from November 1986 through September 1988. The estimated radium concentrations in the RSC resin during each month are also presented in Table 3. The data in Table 3 are conservative in relation to the potential exposure to plant workers because the radiation monitor was located within 3 in. of the tank surface; it is expected, therefore, that the potential exposure to a plant worker would be less than is indicated in Table 3.

If it is assumed that a plant worker spends 3.5 hours per week for an entire year near the RSC tank (i.e., within 3 in. of the tank) when the average radiation is 0.813 mrem/h (i.e., average radiation for July–September 1988 from Table 3), then the total radiation received would approximate the average background radiation level of 150 mrem/year that the average person living in Colorado receives in one year.

Operating costs. Plant operating costs have been estimated (Table 4). These costs are for an annual production of 5.475 mil gal of potable water at the Redhill Forest treatment plant. The costs do not include the energy costs for transporting water from the deep wells to the treatment plant nor do they include operator costs. A more detailed description of costs is given by Mangelson et al.[1] The costs for a system that eliminates the ion exchange process completely and in which the radium is removed by directly passing the raw water through the RSC resin bed have not been estimated. This type of system would not have worked in treating the raw water available in this study because of the high concentrations of iron in the well water. For some well waters, this one-process approach might be feasible and economical.

Conclusions

Studies have shown that conventional cation exchange resins in the sodium form are effective in removing radium from drinking water supplies. Other recent studies have shown that RSC resin is very effective in removing radium from drinking water supplies and has a high adsorption capacity for radium. Based on the 21-month study of the operation of a full-scale radium removal system that uses both cation resins and the RSC resin, the following conclusions were made:

- Consistent removal of radium in well water can be accomplished at a reasonable cost by small systems by means of the ion exchange process using standard cation resins.
- The RSC resin is very efficient for removing radium from the regenerate brine from the conventional cation exchange process. Radium removal exceeds 99 percent for radium concentrations as high as 2,000 pCi/L in the brine wastewater. The average rate for radium removal exceeded 99 percent for the entire study period.
- The use of the RSC resin for ultimate radium adsorption provides a workable and reasonable-cost solution to the waste disposal problem.
- Gamma radiation buildup in the RSC does not appear to be a health concern.

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FOCUS ON: EMERGING TECHNOLOGIES

Radium removal using preformed hydrous manganese oxides

This article summarizes the results of a recent project conducted by Richard L. Valentine, associate professor of civil and environmental engineering at the University of Iowa. The project was supported by the Iowa State Water Resources Research Institute and the AWWA Research Foundation. Copies of the complete final report will be available in early 1991 from the AWWA Research Foundation, 6666 W. Quincy Ave., Denver, CO 80235; (303) 794-7711.

Radium in drinking water is of concern because it is a suspected carcinogen. Radium typically is found in groundwater in two isotopes—Ra 226 and Ra 228. The current federal regulatory standard limits total radium (Ra 226 and Ra 228) to 5 pCi/L in drinking water. The US Environmental Protection Agency (USEPA) is expected to propose new drinking water standards for radionuclides in January 1991; separate standards for Ra 226 and Ra 228 will be proposed along with standards for natural uranium, radon, gross alpha, and gross beta and photon emitters.

Most notable occurrences of radium in drinking water have been in midwestern states and in the Appalachian region. Systems with radium contamination have been faced with the need for either acquisition of alternative supplies or implementation of treatment systems to remove radium in order to comply with the standard. In most cases, removal efficiencies ranging from 50 to 75 percent are adequate to bring systems into compliance.[1]

Conventional technologies costly

Conventional radium-removal technologies include lime-soda softening, ion exchange, and reverse osmosis. However, such technologies can be unaffordable for many smaller communities, particularly with respect to initial capital outlays. Small water systems need a simple, inexpensive, and reliable method for radium removal.

Recent batch and pilot-scale studies have shown that addition of preformed hydrous manganese oxides (HMOs) at a dosage of 1.0 mg/L as Mn prior to sand filtration can significantly lower radium levels.[2,3] At this dosage, the cost of treatment has been estimated at $0.05/1,000 gal with little investment in equipment or plant modifications. All that is required is a conventional metering pump and a mixing tank.

Full-scale testing at two small water systems in Iowa showed that HMO addition achieved significant radium removals at costs estimated to be in the range of $0.01–0.05/1,000 gal. Because of the low cost and ease of implementing such a treatment process, its use should be evaluated whenever possible before more costly options are sought.

Study sites and methods

The two full-scale studies of HMO addition were conducted at the Iowa communities of West Liberty and Forest City, both of which rely on groundwater supplies. The West Liberty water system treats 0.3 mgd on average over a 12-hr period. Forest City's plant produces 0.5 mgd on average with a maximum observed flow of 1.0 mgd. Radium levels at West Liberty average about 3.0 pCi/L of Ra 226 and about 1.0 pCi/L of Ra 228. Radium levels at Forest City average 6 pCi/L of Ra 226 and less than 1.0 pCi/L of Ra 228.

Total hardness of West Liberty water was approximately 325 mg/L as calcium carbonate, and iron content was 0.5–1.0 mg/L. The temperature and pH of plant effluent averaged 19°C and 7.6, respectively.

West Liberty pumps raw water to a forced-draft aerator located on top of a large detention tank. Pressure filters comprise about 2 ft of sand. The plant was modified to allow dosing of HMOs either to the raw water prior to aeration or to the contents of the aerated detention tank immediately prior to filtration.

> *Small water systems need a simple, inexpensive, and reliable method for radium removal.*

The Forest City treatment plant uses a treatment train comprising pressure aeration of raw water followed immediately by sand filtration. For this study an 8-ft-diameter pressure filter was isolated and modified to allow HMO dosing; HMOs were pumped into the pressure-aerated water immediately ahead of the filter.

Total hardness of Forest City water was approximately 400 mg/L as calcium carbonate, and iron content was 0.5–1.0 mg/L. Water temperature and pH were 10°C and 7.2, respectively.

Influent and effluent samples were taken for radium, iron, and manganese analysis. Backwash samples were taken so that radium mass balances could be performed. Filter medium was analyzed for radium, iron, and manganese content. Only Ra 226 removal was studied because it behaves chemically in an identical manner as Ra 228.

Studies at West Liberty utilized HMO dosages of 0.2–1.0 mg/L as Mn whereas an HMO dosage of 1.0 mg/L was utilized at Forest City.

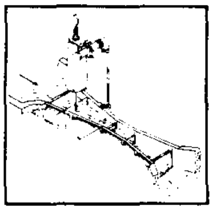
Effect of filter conditions

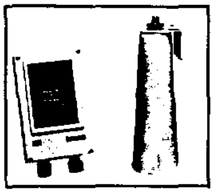
In the course of the study it became clear that the filter medium at Forest City was very old with a large amount of deposits. Although the plant was performing acceptably with respect to iron removal, approximately half the medium was much larger than original specifications and medium depth was about double what specifications called for. Chemical analysis showed that about half the medium weight was acid dissolvable and therefore not filter sand. Iron and manganese concentrations were found to be high, and it appeared that deposits also consisted of other materials such as calcium carbonate. The filter medium at West Liberty was normal, with only 1–2 percent being acid dissolvable.

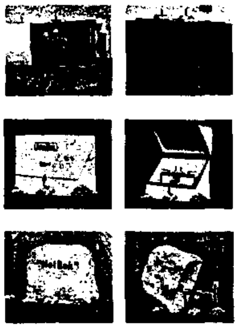
Radium content in Forest City's filter medium was estimated to be about 50–150 pCi/g, which was very high compared with West Liberty's reading of approximately 6 pCi/g. Forest City's high radium content in its filter medium is likely attributable to the poor condition of the medium. In contrast, HMO dosing at West Liberty reduced filter medium radium content to about 5 pCi/g.

Radium removal

At West Liberty, HMO doses of 0.2, 0.5, and 1.0 mg/L as Mn resulted in radium removals of 75, 80, and 85 percent, respectively. However, steady-state radium removals did not occur immediately following HMO dosing but continued to increase for four weeks as shown in Figure 1. Additionally, significant radium







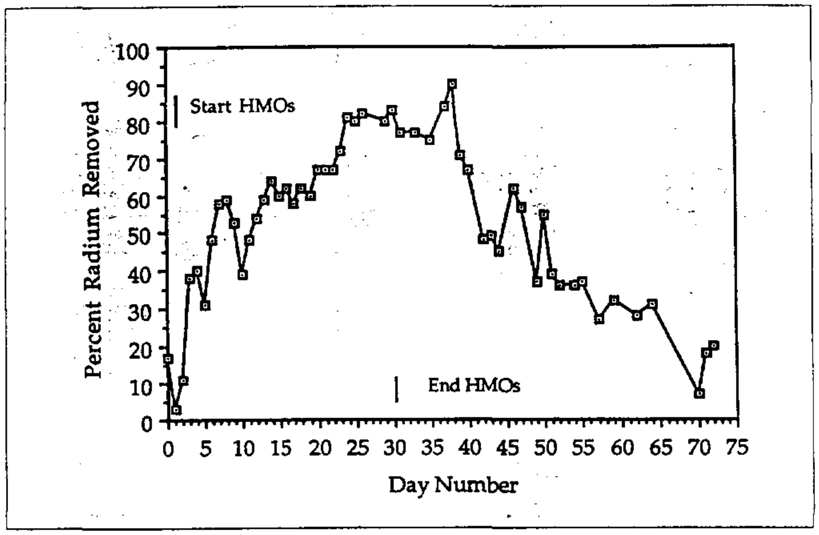


Figure 1. Percent removals of Ra 226 at West Liberty with HMO dosage of 0.5 mg/L.

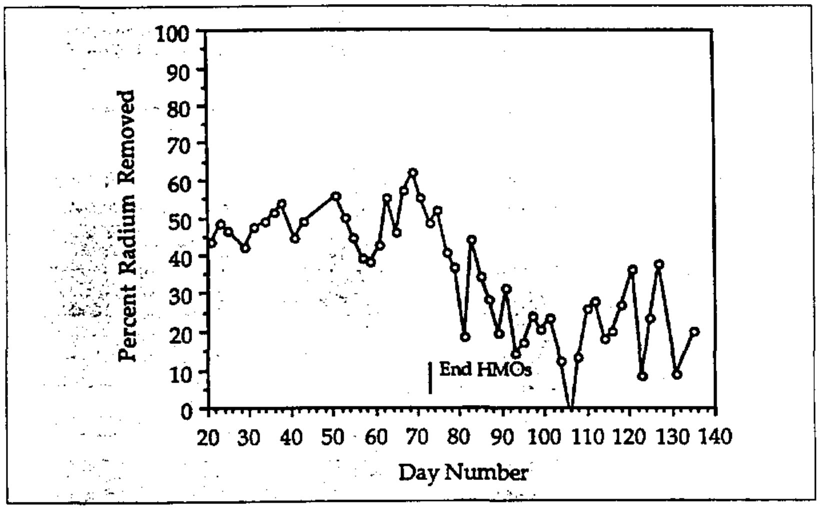


Figure 2. Percent removals of Ra 226 at Forest City with HMO dosage of 1.0 mg/L.

removals continued for four weeks after HMO dosing was stopped.

At Forest City, radium removal averaged 50 percent based on an HMO dosage of 1.0 mg/L as Mn as shown in Figure 2; this value is both less than the expected value of about 75 percent and lower than the radium removal at West Liberty. As at West Liberty, radium removal at Forest City continued long after HMO dosing ceased.

Evaluation of effluent manganese concentrations showed no significant increase at West Liberty, indicating that all added HMOs were being removed. However, effluent manganese was elevated at Forest City to approximately 0.25 mg/L as Mn when HMO dosing at 1.0 mg/L was practiced, a result that is likely attributable to the poor filter condition.

As expected, radium levels in backwash waters were elevated when influent water was dosed with HMOs; levels at both locations ranged between 150 and 200 pCi/L when HMO dosing was practiced.

At West Liberty, all removed radium appeared to be recovered in backwash water once steady-state removals were observed. On termination of HMO dosing, radium recoveries in backwash fell far below removals until the filter medium established equilibrium with the raw water, as indicated by lack of radium removal across the filter.

Radium recovered in Forest City backwash greatly exceeded the amount removed from the influent when HMO dosing was practiced. Excessive recoveries were also observed at West Liberty prior to achieving steady-state conditions.

Conclusions

Addition of HMOs to radium-contaminated water prior to sand filtration can achieve significant removals at a cost of between one and five cents per thousand gallons. Full-scale tests indicate that theoretical removals should be obtainable under optimum conditions. Because of the low costs and ease of implementation of the technology, it should be evaluated before more costly and complex treatment options.

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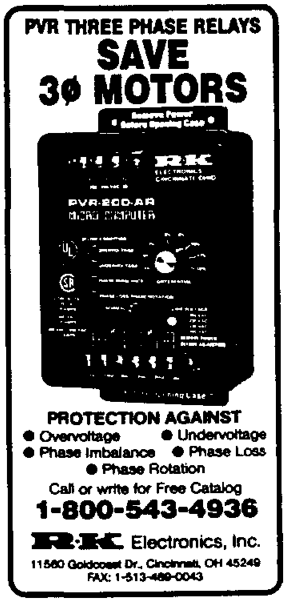
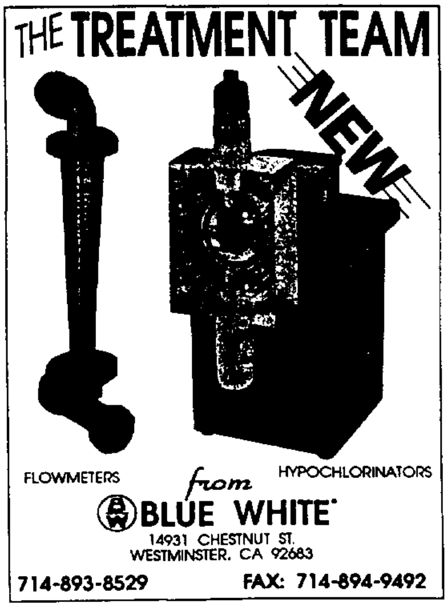
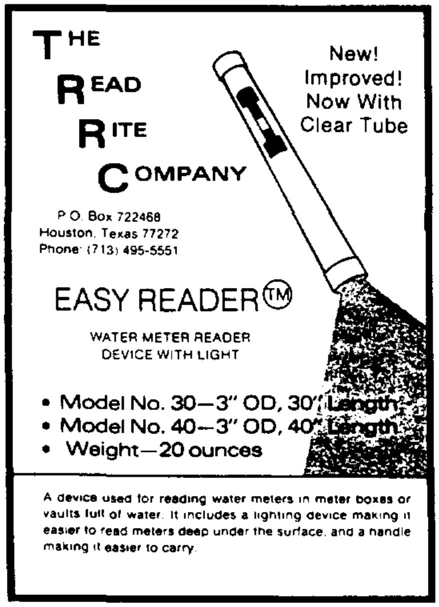






FOCUS ON: EMERGING TECHNOLOGIES

Low-cost technology initiative promises help for small water systems

By David Schnare, National Low-Cost Small Systems Technology Initiative coordinator, Office of Drinking Water, US Environmental Protection Agency (USEPA), Washington, D.C.

In 1988 a group of opinion leaders representing public and private sectors of the US drinking water industry met to discuss industry issues of the 1990s. These leaders believed there would be no meaningful implementation of the Safe Drinking Water Act amendments of 1986 without significantly increased state budgets for drinking water programs and the approval and use of low-cost water treatment technologies for small water systems.

A second group of industry and government leaders met in December 1988 to explore further the issue of low-cost technology; it was at this meeting that the National Low-Cost Small Systems Technology Initiative was born.

Committees established

By the end of January 1989 an overall steering committee and five program committees were established; mission statements and chairpersons were identified for each. The mission of the initiative was defined as identifying, marketing, approving, and applying simple, inexpensive drinking water treatment technologies for use by small systems to reach full compliance with state and federal drinking water regulations.

The steering committee was charged with identifying areas in which specific action agendas could be developed, promoting involvement of appropriate individuals and organizations, reviewing progress, and enhancing application of innovative technologies developed under the initiative. Program committees were established to evaluate the following five topics: technology development, technical plan review, market definition, financing, and alternative solutions.

Consensus to breach barriers

The public/private partnership convened again in March 1989 to review action plans developed by each program committee. From the meeting emerged a clear consensus that initiative participants would work to breach all barriers to approval and use of low-cost technologies for small systems.

Table 1. List of initial small systems demonstration sites selected to participate in the Low-Cost Small Systems Technology Initiative

Filtration

- Neihart, Mont.—This system uses a surface water supply and experiences recurring bacteria and protozoa contamination.
- Amsbury, Pa.—This system serves 25 residences; average flow is 6,000 gpd. Its source is an uncovered concrete reservoir filled by a mountain stream fed by three springs. The only treatment is with hypochlorite, and there is no finished water storage.
- Santa Rosa, Calif. (Freestone Water System)—This is a very small system that requires use of both surface water and groundwater sources. Two separate distribution systems serve 16 and 30 connections, respectively.

Radionuclides

- Livonia, Ga. (Brittany Harbor Subdivision)—This system currently serves about 100 homes with water from nine wells, one of which has radium levels ranging from 30 to 40 pCi/L.
- Leslie, Ark.—Serving 265 nonindustrial connections, this system has two well sources, each averaging 20 pCi/L of radium as well as fluoride levels exceeding 2 mg/L and serious sulfide contamination.
- Spicewood, Texas (Quail Creek Subdivision)—This system, with a single well source, currently serves 14 connections with the potential to serve 55. It cannot grow until radium contamination is controlled.

Fluoride

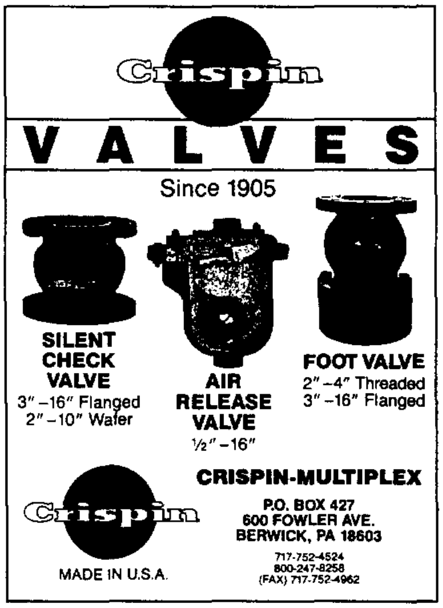
- Circle, Mont.—This groundwater system has fluoride levels exceeding regulatory standards.
- Valley View Water Association, Mont.—This system has fluoride problems and was selected because of its potential for application of point-of-use or point-of-entry technology. State approval of this demonstration site has been temporarily delayed.

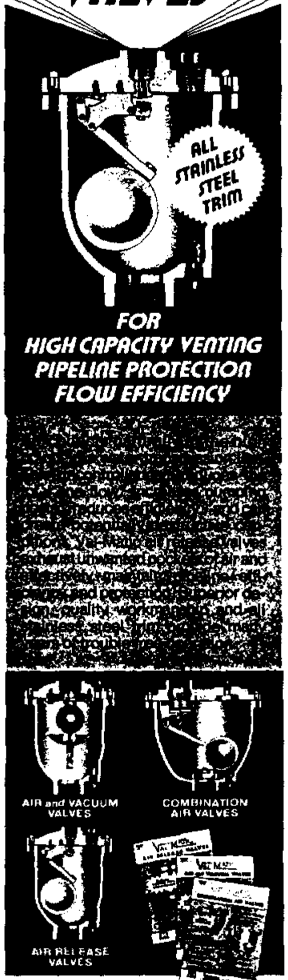
Nitrate

- South San Bernardino, Calif.—This groundwater system has both nitrate and DBCP contamination; the initial demonstration project will focus on a nitrate-laden well.
- Pleasant Valley, Ohio—This nitrate-contaminated system replaces West Milton, Ohio, as a demonstration site.

A major element of the meeting was the presentation by the Technology Development Subcommittee, chaired by Donald Mahlstedt, retired vice president of Culligan International Inc., of a plan to implement several full-scale, year-long demonstration projects of low-cost technologies. The intent of the plan was to allow selected small systems an opportunity to attempt to mitigate their specific water quality problems using innovative, low-cost treatment systems supplied at no cost to the utility by equipment manufacturers. Participating utilities would be responsible for systems operation and maintenance costs after completion of demonstration projects.

Targeted contaminants and compliance parameters for the demonstration projects included nitrates, volatile organic contaminants (VOCs), surface water treatment, radon, corrosion control, and disinfection. Treatment technologies for such projects were to cost





ASDWA, has distributed a questionnaire to state officials seeking data on how state review requirements can better accommodate more economical and efficient small systems.

Solution outlined

The work of the Low-Cost Small Systems Technology Initiative will be done when a small water system need make only one phone call to initiate a process that results in the installation of a reasonably priced water treatment system that ensures regulatory compliance. There are three principal objectives that must be achieved before this can happen.

First, states must adapt to changing times. Progressive states are now evaluating water treatment proposals in the context of engineering efficacy rather than through application of hard technical process criteria. These states are requiring their state engineers to seek performance agreements on technologies with which they are unfamiliar. When states give low-cost technologies a chance to work, the problem of small systems noncompliance will be one-third solved.

Secondly, only when there is a standing body of information on the applicability of low-cost small system technologies available to states, engineers, water system owners and managers, and the public will there be a basis for confidence in these technical solutions. Ease of access through human and electronic means that are available to all who need the information will ensure identification of technical solutions in a timely and effective manner. Developing such a data base and making such information readily available will solve another third of the problem.

Thirdly, the small systems compliance problem will be fully solved when there is a comprehensive network allowing sharing of resources and through which states, engineers, equipment manufacturers and distributors, water supply operators, and the public successfully cooperate to identify, acquire, and install technical and management solutions appropriate to and affordable by small systems. It can happen within five years.

Future considerations

Solving the small system noncompliance problem can be likened to hammering a large nail through hard maple—it takes many hammer blows, each of which advances the nail bit by bit. Several years ago the AWWA Research Foundation funded a study on the barriers to implementation of innovative technologies. At that time it was recognized that states would have to alter their institutions before routine approval of low-cost technologies would be realized.

More lately, the near-term regulatory pressure to approve filtration devices for small surface water systems is leading states such as New York to change significantly their approach to these technologies. Even with the completion of the study on states' current processes for reviewing and approving treatment plans, there remains a great distance to go before approval of established low-cost technologies for small systems will be seen.

There will likely have to be pilot projects in every state before all states change their routines. Implementing such pilot projects will improve understanding of which adjustments in state review processes will work best. After three to five years, enough information will be on hand to allow states to select alternatives that best meet their needs.

These pilot projects will have to show how cost efficiencies can result as well as which application forms, data-sharing activities, and accesses to data networks are most effective. Industry volunteers and equipment manufacturers and suppliers are needed to pioneer this new territory.

For example, the procurement practices of small systems need attention. The cost of a typical procurement is greater than the cost of the engineering solution under current practices. Privately held water companies have a great deal of experience in effective procurement, and the United States can learn from European and Asian procurement practices.

The data base of technological successes must grow as well. The initial low-cost technology demonstration projects will form only a small part of the eventual data base. These projects could result in dozens or perhaps hundreds of case studies documenting the applicability of these technologies. All such studies will have to be readily accessible and understandable. Development of storage and retrieval systems, with adequate protection of confidential business information, must be piloted immediately.

Approaches to ensuring appropriate or adequate quality control over data will have to be examined and tested. Alternative approaches using electronic bulletin boards, hard-copy files, facsimile transmission, and mail must be evaluated for efficacy, cost efficiency, and financial viability.

Finally, a vigorous, self-sustaining community partnership must be established. Data sharing, joint problem solving, testing of new technologies, and recording each success story is the only way the water supply community can realize a potential that is greater than the sum of its parts. Establishing such a cooperative enhances the marketplace's ability to provide solutions to water systems in need of them. ■

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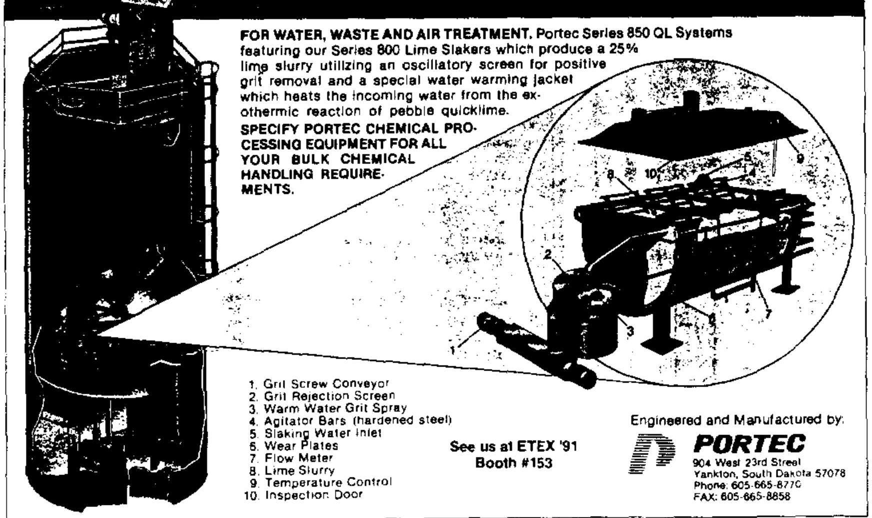

Waterworld News

1991 Editorial Calendar

March/April:
Drinking Water Additives

May/June:
Automatic Control and Instrumentation/
AWWA Annual Conference

July/August:
Industrial Water Systems

September/October:
Privatization of Wastewater Treatment/WPCF Annual Conference

November/December:
Groundwater Protection

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Radium-Removal Efficiencies in Water-Treatment Processes

William L. Brinck, R.J. Schliekelman, Dorothy L. Bennett, Charles R. Bell, and Ira M. Markwood

Studies of radium removal by various treatment processes indicate that the means do exist to meet the 5-pCi/l limit for radium in drinking water.

Naturally occuring radium-226 is found in numerous well waters. The EPA published interim regulations on Jul. 9, 1976, that limit the concentration of radionuclides in public water systems.[1] Methods for removing radium had to be identified so that drinking-water treatment plants could be designed to meet the limit of 5 pCi/l for combined radium-226 and radium-228.

Studies were performed by two state agencies, under EPA contracts, at fourteen cities in Iowa and Illinois to determine the radium-removal efficiency of four water-treatment processes—reverse osmosis, aeration and iron removal, sodium cation exchange, and lime-soda-ash softening.[2,3] This work was supplemented by a study at the water-treatment pilot plant located in the EPA's Cincinnati laboratory.

Description of Study

Water supplies were selected for the study on the basis of (1) high raw water gross alpha or radium-266 content (thirteen of the fourteen supplies had radium-266 concentrations greater than 5 pCi/l in raw water), (2) a variety of water-treatment processes, (3) availability for continuous operation during the study, and (4) a range of municipal populations served. A list of the cities included in the study and some pertinent characteristics are shown in Table 1.

Samples were collected from raw water supply wells and from points where treated water entered the distribution system. In addition, treatment systems in Iowa and at Herscher, Ill., were sampled at various points throughout the treatment processes to determine changes and removals of radium and other pertinent chemical parameters. Flows and other data were obtained to determine whether plants were meeting design rates and to provide data for determining a material balance of radium-226 removals.

A significant difference between the two states' efforts was that, in Iowa, each treatment system was sampled over the course of an operating cycle with numerous samples taken throughout the process, whereas in Illinois each system was sampled on each of three separate occasions at approximately one-week intervals. Thus, the results from Iowa give a detailed picture of system operation, while the results from Illinois give an indication of the variation of operating characteristics over a period of three weeks. Samples of well water generally were collected near the beginning of a pumping period and following longer pumping times to determine any time-related variability in radium, hardness, and other chemical parameters during pumping.

Radium-226 analyses were performed in Iowa at the Iowa State Hygienic Lab. using coprecipitation with mixed barium and lead sulfates in accordance with Standard Method ASTM D 2460-70. Illinois radium-226 analyses were performed by the Argonne Natl. Lab. using the radon-emanation method. Some samples were collected in duplicate for intralaboratory collaborative testing to check the accuracy of the radium-266 analysis. Typical overall precision of the radium analyses was approximately ± 10 per cent above 1.0 pCi/l and ± 0.1 pCi/l below 1.0 pCi/l for the Iowa analyses. The standard deviation of the Illinois analyses was less than 10 per cent at all concentrations.

The pilot plant in Cincinnati was used to investigate the removal of radium-226 under ideal conditions of lime softening at two different pH levels, 9.5 and 10.6. The raw water was brought to Cincinnati by truck from the Slade Ave. plant in Elgin, Ill. Collected were samples of raw water, treated water, and water at two points in the treatment process; these were analyzed using the radon-emanation procedure by EPA in Cincinnati.

Results

Reverse osmosis. In the reverse osmosis demineralizing process, high hardness water is pressurized and piped into a reverse osmosis unit where relatively pure water diffuses through the semipermeable membrane and becomes the product water, leaving a concentrated reject water. The much greater rejection of divalent ions, such as calcium, magnesium, radium, and sulfate, than of the monovalent ions, sodium and chlorine, is characteristic of essentially all reverse osmosis membranes.

The usual water supply for Greenfield, Iowa, is an impoundment of surface water. A reserve source of water from a deep well is used during periods of drought. The brackish well water, with a total solids content of more than 2200 mg/l, is treated in a reverse osmosis unit installed in 1971. A description of the plant and discussion of operating results was published in 1972.[4] Raw, product, and reject water samples were collected and analyzed for radium-226 and other chemical constituents; the results are shown in Table 2. The average radium-226 concentration in the reject water was 43 pCi/l, where 31 per cent of the influent water was rejected and 69 per cent was converted to low hardness product water.

Iron and manganese removal. The presence of iron and manganese in drinking water is objectionable primarily because of taste, and the precipitation of these metals turns the water a turbid yellow-brown color. The treatment processes employed in the removal or control of iron and manganese include:

1. Precipitation and filtration by aeration, detention (or sedimentation), and filtration or by oxidation by potassium permanganate, chlorine, or chlorine dioxide
2. Ion exchange with a continuously regenerated permanganate greensand filter

Iron and manganese removal is utilized in some form of pretreatment at five ion-exchange softening plants and as the only removal process at two other plants selected for study (Table 3).

A tabulation of radium removal, iron removal, manganese removal, and pH (Table 4) suggests that radium-226 is being removed on the manganese, just as manganese-impregnated fibers have been reported to remove radium.[5] There are radium-226 removals of 46–56 per cent when there is a significant manganese removal. Fair et al.[6] note that "Hydrous oxides of Fe (III) and Mn (IV) have high sorption capacities for bivalent metal ions.... Sorption capacities for Mn^{++} at pH 8 are on the order of 1.0 and 0.3 mole of Mn (II) sorbed per mole of MnO_2 and $Fe(OH)_3$, respectively." Although this may explain the removal of radium at these plants, further studies are required to confirm this theory. Radium concentrations in the iron filter-backwash water were as high as 250 pCi/l.

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TABLE 1

Cities Included in Radium-Removal Studies

City	Type of Treatment	Population Served	Plant Capacity m^3/day	Ra-226 Raw Water pCi/l	Remarks
Greenfield, Iowa	reverse osmosis	2 212	570	14	
Adair, Iowa	iron removal	750	380	13	aeration and continuously regenerated greensand filters
Stuart, Iowa	iron removal	1 354	530	16	aeration and pressure iron-removal filtration
Dwight Correctional Center, Ill.	ion exchange	235	110	3.3	
Eldon, Iowa	ion exchange	1 319	360	49	
Estherville, Iowa	ion exchange	8 108	3 300	5.7	
Grinnell, Iowa	ion exchange	8 402	3 790	6.7	
Herscher, Ill.	ion exchange	1 000	380	14.3	
Holstein, Iowa	ion exchange	1 445	590	13	
Lynwood, Ill.	ion exchange	4 000	600	14.7	
Elgin, Ill.	lime-soda-ash softening	25 000	18 000	5.6	
Peru, Ill.	lime-soda-ash softening	12 400	6 830	6.0	
Webster City, Iowa	lime-soda-ash softening	8 488	5 000	7.8	
West Des Moines, Iowa	lime-soda-ash softening	16 441	9 730	9.3	

TABLE 2

Radium-226 and Hardness Removals at Greenfield, Iowa, Reverse Osmosis Water-Treatment Plant

Sampling Point	Ra-226		Hardness	
	Amount in Sample pCi/l	Removal per cent	Amount in Sample mg/l as $CaCo_3$	Removal per cent
Well supply	14	–	610	–
Plant effluent	0.6	96	29	95

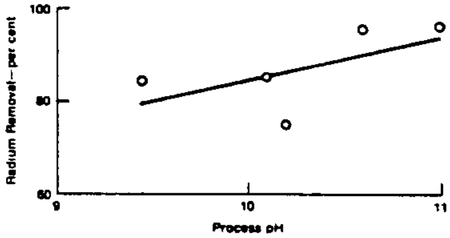
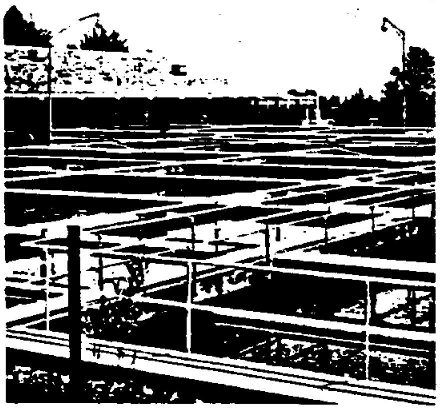


Fig. 1. Radium Removal vs Softening pH

TABLE 3

Radium-226 and Iron Removals at Water-Treatment Plants Using Iron- and Manganese-Removal Processes

City	Process	Filter Type	Sampling Point	Ra-226		Iron	
				Amount in Sample pCi/l	Removal per cent	Amount in Sample mg/l	Removal per cent
Adair, Iowa	iron and manganese removal	greensand	well	13	–	1.1	–
			filter effluent	8	38	0.02	98
Stuart, Iowa	iron and manganese removal	coal filter media	well	16	–	0.94	–
			filter effluent	12	25	0.03	97
Eldon, Iowa	ion-exchange pretreatment	coal filter media	well	49	–	2.0	–
			filter effluent	43	12	0.3	85
Estherville, Iowa	ion-exchange pretreatment	coal filter media	well	5.7	–	2.0	–
			filter effluent	5.1	11	0.67	66
Grinnell, Iowa	ion-exchange pretreatment	none	well	6.7	–	0.71	–
			detention effluent	5.7	15	0.41	42
Herscher, Ill.	ion-exchange pretreatment	coal filter media	well	14.4	–	0.1	–
			filter effluent	6.7	53	0.0	–
Holstein, Iowa	ion-exchange pretreatment	coal filter media	well	13	–	1.8	–
			filter effluent	7.2	45	0.05	97



Conventional treatment is suitable for radium removal.

TABLE 4

Radium-226, Iron, and Manganese Removals by Iron- and Manganese-Removal Processes

City	Raw Water pH	Ra-226			Iron			Manganese		
		Raw Water pCi/l	Treated water pCi/l	Removal per cent	Raw Water mg/l	Treated Water mg/l	Removal per cent	Raw Water mg/l	Treated Water mg/l	Removal per cent
Adair, Iowa	6.7–6.9	13	8	38	1.1	0.02	98	0.01	0.01	–
Eldon, Iowa	7.8	49	43	12	2.0	0.3	85	0.01	0.01	–
Estherville, Iowa	7.7	5.7	5.1	11	2.0	0.67	66	0.24	0.27	–
Grinnell, Iowa	7.6	6.7	5.7	15	0.7	0.41	42	0.01	0.01	–
Herscher, Ill.	7.6–8.3	14.9	6.6	56	0.2	0	–	0.47	0.02	96
	7.6–8.3	14.5	6.4	56	0.4	0	–	0.41	0.01	98
	7.6–8.3	14.9	6.9	54	0.1	0	–	0.48	0.01	98
	7.6–8.3	14.3	6.9	52	0.1	0	–	0.39	0	100
	7.6–8.3	14.0	6.9	51	0.1	0	–	0.45	0	100
	7.6–8.3	13.9	6.8	51	0.1	0	–	0.63	0	100
	7.6–8.3	13.9	7.3	47	0.2	0.1	–	0.44	0.13	70
	7.6–8.3	14.1	6.3	55	0.1	0	–	0.53	0.02	96
	7.6–8.3	14.3	6.5	55	0.1	0	–	0.50	0	100
Holstein, Iowa	7.4–7.6	13	7	46	1.8	0.09	95	0.15	0.01	93
Stuart, Iowa	7.6–7.9	16	12	25	0.94	0.03	97	0.01	0.01	–

TABLE 5

Radium-226 and Hardness Removals at Water-Treatment Plants Using Ion-Exchange Processes

City	Softener Type	Sampling Point	Ra-226		Hardness	
			Amount in Sample pCi/l	Removal per cent	Amount in Sample mg/l	Removal per cent
Dwight Correctional Center, Ill.	greensand	well	3.25	–	286	–
		softener effluent	0.36	89	43	85
		distribution system	0.65	–	67	–
Eldon, Iowa	zeolite	softener influent	43	–	360	–
		softener effluent	1.9	96	10	97
		distribution system	8.6	–	136	–
Estherville, Iowa	zeolite	softener influent	5.1	–	915	–
		softener effluent	0.3	94	46	95
		distribution system	0.4	–	76	–
Grinnell, Iowa	zeolite	softener influent	5.7	–	387	–
		softener effluent	0.2	97	11	97
		distribution system	1.4	–	120	–
Herscher, Ill.	zeolite	softener influent	6.7	–	404	–
		softener effluent	1.3	81	83	79
		distribution system	2.4	–	141	–
Holstein, Iowa	zeolite	softener influent	7.2	–	885	–
		softener effluent	0.5	93	18	98
		distribution system	0.8	–	346	–
Lynwood, Ill.	zeolite	well	14.7	–	848	–
		softener effluent	0.4	97	–	–
		distribution system	1.8	–	78	–

TABLE 6

Radium-226 and Hardness Removals at Water-Treatment Plants Using Lime-Soda-Ash Softening

City	Sampling Point	Ra-226		Hardness		pH of process
		Amount in Sample pCi/l	Removal per cent	Amount in Sample mg/l	Removal per cent	
Elgin, Ill.	well	5.6	–	237	–	–
	filter effluent	0.8	86	102	57	–
Peru, Ill.	well	5.8	–	326	–	–
	filter effluent	1.1	81	94	71	–
Webster City, Iowa (Aug. 1974)	well	6.1	–	507	–	–
	clarifier 1 effluent	1.9	–	333	–	10.0
	clarifier 2 effluent	2.6	–	282	–	10.1
	filter effluent	0.9	85	262	48	9.3
Webster City, Iowa (Feb. 1975)	well	7.8	–	482	–	–
	clarifier effluent	0.6	–	150	–	11.0
	filter effluent	0.3	96	106	78	9.9
West Des Moines, Iowa	well	9.3	–	376	–	–
	contact unit effluent	2.6	–	215	–	10.1–10.4
	filter effluent	2.4	75	190	50	9.4–9.5

It was noted at several supplies that there was significant reduction in radium-226 content after aeration and detention alone before the water was filtered. Further studies are required to fully understand this phenomenon.

Sodium cation exchange. Water softening by the sodium-cation-exchange (zeolite) process depends upon the ability of certain substances to exchange cations with other cations dissolved in water. When hard water is passed through a sodium cation exchanger, the calcium and magnesium in the hard water replace the sodium on the exchange medium. Because the reaction is reversible, after all of the readily replaceable sodium has been exchanged, the cation exchange medium can be regenerated with a solution of sodium chloride. In the regeneration process, the calcium and magnesium on the exhausted cation exchanger are replaced with a fresh supply of sodium from the regenerating brine solution. Then, after washing to free it from the calcium and magnesium cations and excess salt, the regenerated exchanger is ready to soften a new supply of hard water. The ion-exchange media studied included both naturally occurring greensand and synthetic polystyrene resins (zeolite).

Ion-exchange softening removes nearly 100 per cent of the hardness from the treated water. Consequently, unsoftened water is blended with the ion-exchange-softener effluent to provide sufficient calcium carbonate for deposition of a protective coating on the water mains and to reduce treatment costs. Thus, the water entering the distribution system usually has a greater radium concentration than that leaving the softener. The radium concentration in raw water could be the controlling factor in the amount of blending that would be permitted in order to meet the radium limit in drinking water.

The results of the measurements at ion-exchange-process water-treatment plants are shown in Table 5. The data shown from the plants in Illinois were the averages of nine separate data points. Ranges in the per cent reduction of radium-226 through softeners at the three cities were 70.7–98.3 at Dwight, 68.4–93.9 at Herscher and, 94.7–98.2 at Lynwood.

It was determined that the removals vary somewhat during a softener cycle between regenerations. Radium-226 removal usually continues for a short period after hardness breakthrough occurs. If the cycle continues for a longer period after hardness breakthrough, however, radium-226 removal drops rapidly.

Samples of softener brine and rinse effluent during regeneration were taken at various times during the regeneration cycle. Regeneration normally requires 1–2 hr. It was found that the major portion of the radium-226 leaves the ion-exchange media over a rather short period—10–30 min. Maximum radium-226 concentrations in the softener brine and rinse effluents ranged from 320 to 3500 pCi/l.

Lime-soda-ash softening. This process of softening depends on the use of lime and soda ash to change the soluble calcium and magnesium compounds into nearly insoluble compounds that are flocculated, settled, and filtered. Conditions for carrying out the precipitation of calcium and magnesium vary because different pH levels are needed for each—about pH 9.5 for maximum precipitation of calcium carbonate and pH 10.5 for maxium precipitation of magnesium hydroxide. Thus, if the magnesium concentration is low, treatment to a pH of 9.5 will be sufficient. If the magnesium concentration is high, excess lime, to produce a pH of 10.5, can be used. A more economical treatment is to raise the pH to 10.5 to precipitate the magnesium in a primary basin, then recarbonate with carbon dioxide to pH 9.5 to precipitate excess calcium in a secondary basin.

Normally, soda ash is added as needed to precipitate noncarbonate hardness, but because of a soda ash shortage, the West Des Moines plant was using only a small quantity. The Webster City plant was using lime only during the Aug. 1974 measurements but was using soda ash during the Feb. 1975 restudy.

Results of measurements obtained at water-treatment plants using lime-soda-ash softening are shown in Table 6. That the pH of the softening process is a parameter that controls radium removal, at least for water containing both

TABLE 7
Radium Measured in Lime Sludge and Filter Backwash in Four Water-Treatment Plants

City	Ra-226 In Lime Sludge	Ra-226 In Filter Backwash pCi/l
Elgin, Ill.	8.1 pCi/g	18.3
Peru, Ill.	9.0 pCi/g	36.9
Webster City, Iowa	980 pCi/l	80
West Des Moines, Iowa	2300 pCi/l	6.3

TABLE 8
Radium-226 Removal in EPA Lime-Softening Pilot Plant Using Elgin, Ill., Water Softened to pH 9.4–9.5

Step in Water-Treatment Process	Aug. 6, 1975		Aug. 7, 1975		Aug. 8, 1975		Average	
	Concentration pCi/l	Removal per cent	Concentration pCi/l	Removal per cent	Concentration pCi/l	Removal per cent	Concentration pCi/l	Removal per cent
Raw	4.19	–	3.95	–	4.87	–	4.34 ± 0.46	–
Settled	1.03	75.4	0.94	76.1	0.69	85.9	0.89 ± 0.16	79.1 ± 5.9
Dual media filter	0.79	81.1	0.65	83.6	0.61	87.4	0.68 ± 0.09	84.0 ± 3.2
Activated carbon filter	0.75	82.1	0.62	84.4	0.65	86.6	0.67 ± 0.07	84.3 ± 2.3

TABLE 9
Radium-226 Removal in EPA Lime-Softening Pilot Plant Using Elgin, Ill., Water Softened to pH 10.6, Recarbonated to pH 8.6

Step in Water-Treatment Process	Aug. 20, 1975		Aug. 21, 1975		Average	
	Concentration pCi/l	Removal per cent	Concentration pCi/l	Removal per cent	Concentration pCi/l	Removal per cent
Raw	4.78	–	4.86	–	4.82 ± 0.19	–
First settling basin	0.37	92.4	0.39	92.0	0.38 ± 0.02	92.2 ± 0.3
Recarbonation settling basin	0.33	93.1	0.36	92.5	0.35 ± 0.02	92.8 ± 0.4
Dual media filter	0.30	93.7	0.30	93.8	0.30 ± 0.00	93.8 ± 0.1
Activated carbon filter	0.18	96.2	0.28	94.3	0.23 ± 0.07	95.2 ± 1.3

TABLE 10
Radium-226 Removal Efficiencies in Water-Treatment Processes

Process	City	Removal-per cent
Reverse osmosis	Greenfield, Iowa	96
Iron removal	Adair, Iowa	38
	Eldon, Iowa	12
	Estherville, Iowa	11
	Grinnell, Iowa	15
	Herscher, Ill.	53
	Holstein, Iowa	45
	Stuart, Iowa	25
Ion Exchange	Dwight Correctional Center, Ill.	89
	Eldon, Iowa	96
	Estherville, Iowa	94
	Grinnell, Iowa	97
	Herscher, Ill.	81
	Holstein, Iowa	93
	Lynwood, Ill.	97
Lime-soda-ash softening	Elgin, Ill.	86
	Peru, Ill.	81
	Webster City, Iowa, pH 11.0	96
	West Des Moines, Iowa, pH 10.1	75
	Cincinnati Pilot Plant, pH 9.5	84
	Cincinnati Pilot Plant, pH 10.6	95

calcium and magnesium, is demonstrated in Table 6 and in Fig. 1. The least squares fit for a straight line through the data points of Fig. 1 indicates that the radium-removal percentages increased as the pH of the process increased.

The data shown from the plants in Illinois are the averages of three separate data points, taken at approximate one-week intervals. Ranges in the per cent reduction of radium-226 at the two cities were 86.0–89.9 in Elgin and 70.6–92.4 in Peru.

Samples of lime sludge and filter backwash water were collected also. The results are shown in Table 7.

Pilot plant. The EPA Munic. Envir. Res. Lab., Cincinnati, Ohio, operates a conventional coagulation water-treatment pilot plant.[7] The pilot plant, designed for flexibility in operation, is capable of treating in parallel two 7.6-l/min streams of water.

The two treatment systems each consist of two rapid mix tanks (in series), flocculation basin, sedimentation basin, and one or more filters (in parallel). The theoretical detention times for the mix tanks, flocculation basins, and sedimentation basins are 1 min, 60 min, and 6.5 hr respectively.

The water to be treated is pumped to the first rapid mix tank. Normally, the pH control chemical is added in the first mixing tank and coagulant introduced in the second tank. From the mixing tanks, the water flows by gravity through the flocculation and settling basins to the filters, which consist of 10.8-cm diameter clear plastic cylinders providing 93 cm² of media-surface area. A pump on the effluent side of each filter controls the flow rate through the filter with the excess water wasted through an overflow located about 1 m (several feet) above the filter media. For the radium-removal studies, two parallel filters were used. One was a dual media filter containing 50.8 cm of No. 1½ anthracite filter media* over 30.5 cm of 0.4-mm effective size sand.† The other filter consisted of 76.2 cm of granular activated carbon.‡ The filtration rate was controlled at 163 l/min/m².

The plant has instrumentation to record pH of the raw, flocculated, settled, and filtered water; turbidity and temperature of the raw, settled, and filtered water; head loss in the filters; and volumes of raw and filtered water pumped.

Two test runs in Aug. 1975 were made to determine the removal of radium-226 from Elgin, Ill., raw water by lime softening. The tests lasted about 100 hr each and were run at pH 9.5 and 10.5. The raw water was trucked from the Elgin Slade Ave. treatment plant to Cincinnati, Ohio.

For the first test run, lime at 220 mg/l was fed into the second rapid mix tank to increase the pH to 9.5. Commercial grade lime was used and fed as a 4 per cent slurry. The water then was flocculated, settled, and filtered.

Duplicate 1-l grab samples of the raw, settled, and filtered water were collected three times during the test period. The settled and filtered samples were collected about 7 hr after the raw sample, the approximate time required for the water to flow through the flocculation and settling basins. All samples were

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*Anthrafilt, Shamokin Filter Co., Shamokin, Pa.
†Muscatine sand, Northern Gravel Co., Muscatine, Iowa.
‡Filtersorb 200, Calgon Corp., Pittsburgh, Pa.

preserved with 1.5 ml of nitric acid.

The pilot plant was operated in a slightly different manner during the second test run. For this test, 270 mg/l of lime was added to increase the pH to 10.6. After settling, the treated water was pumped through the other treatment system where the water was recarbonated to lower the pH to 8.6, settled for a second time, and then filtered. Grab samples were collected also of the raw, settled, and filtered water. Water samples from the first settling basin were collected about 7 hr after the raw sample. Water samples from the second settling basin and filters were collected about 14 hr after the raw sample.

Results of the two tests indicated that pH affects the removal of radium-226 (Table 8,9). Lime softening at pH 9.5 resulted in removals of 79 per cent for settled water and 84 per cent for filtered water. Excess lime softening to pH 10.6, achieved 92–93 per cent removals in the settled water and 93–95 per cent in the filtered water. Little or no difference in removals were noted between the two types of filters—indicating that the carbon filter did not achieve any additional removal by adsorption.

Conclusions

A study to determine the efficiency of radium removal by conventional water-treatment processes indicates that reverse osmosis, ion-exchange, and lime-soda-ash softening are effective in removing the major portion of the radium from the water. High radium water was associated primarily with deep sandstone formations. The raw water radium concentrations ranged from 3 to 49 pCi/l at the water-treatment plants studied. The radium-removal efficiencies of four water-treatment processes are shown in Table 10.

Overall removal of radium at a reverse osmosis plant was 96 per cent as compared with a concurrent hardness removal of 95 per cent and a product water recovery of 69 per cent.

Radium-226 removals in the sodium-cation-exchange process were generally above 90 per cent, with the exception of Herscher, Ill., where an 81 per cent removal was noted. At the latter plant, about 53 per cent of the radium-226 was removed in the aeration, settling, and filtering preprocessing before the ion-exchange step. The data indicate that radium removal continues for a time after the hardness-removal capacity is exhausted; thus, a simple analysis for hardness may be used as an indicator when operating the plant for radium removal. In all of the ion-exchange-softener installations, 6–25 per cent unsoftened water was bypassed around the softener and blended with the finished water being pumped to the distribution system to provide sufficient calcium carbonate for deposition of a protective coating on the water mains of the system.

Overall removal of radium-226 by softening and filtering at lime-soda-ash softening plants can reach 95 per cent, dependent primarily on the pH of the process. Variations in radium removals also may depend on chemical dosage, magnesium removal, noncarbonate hardness removal, and filtration efficiency. Most of the removal efficiencies were found to fall within the range of 75–96 per cent.

Radium-226 extraction through iron-removal units varied from 11 per cent to 53 per cent using aeration, detention, and filtration. The manner of radium removal is possibly adsorption or catalytic action by the oxidation products [FE (III) and Mn (IV)] deposited on the filter media.

Relatively high concentrations of radium in wastewaters and sludges must be considered in determining the final disposal of these wastes. Currently, such wastes generally are discharged to watercourses. An exception is the discharge of lime sludge to evaporation lagoons in most instances. These practices may not be acceptable for the safe disposal of radioactive wastes. Additional research is needed to determine the most effective methods of waste disposal.

Acknowledgement

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A Waterborne Outbreak of Giardiasis in Camas, Wash.

J. C. Kirner, J. D. Littler, and L. A. Angelo

An account of the investigation launched in the wake of an outbreak of giardiasis reveals how and why the outbreak occurred and describes the steps taken to prevent a reoccurrence.

In late April and early May of 1976, local physicians in Camas, Wash., reported the occurrence of approximately 25 cases of giardiasis during a 2-3 week period. Subsequently, an investigation was launched by local, state, and federal health officials.

The results of the investigation revealed new areas of concern for the water-utility industry. The disease-causing organism was identified in the infected individuals and traced back through the water system to infected beavers active in the watershed. This was the first substantiated case of wild animals contaminating a human popula-

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tant because all of the impurities are removed in the filter, and the retention period ahead of the filter media is short. At a filter rate of 3.4 mm/s (5 gpm/sq ft) in a filter with a water depth 2.4 m (8 ft) over the media, the water is in the filter only 12 min before reaching the media. Depending on the coagulation time, the total time ahead of the media may only be 20-30 min.

In order to produce the maximum quantity of high quality filtered water with the minimum use of wash water, several factors must be considered. For filter production the hours of operation—multiplied by filtration rate and divided by the terminal head loss across the media—give a useful parameter for evaluating the operation of a particular media design. In order to achieve the optimum efficiency for a given plant, chemical application and mixing must be adjusted appropriately to accommodate variations in the raw water quality.

Where a major water supply is made up of more than one body of water in different watersheds or where different intakes exist, it is often necessary and always advantageous to continuously monitor the turbidity of the individual supplies, so that the treatment process can be designed for optimum efficiency. It is often possible to dilute the waters of a more turbid body of water with cleaner waters from other sources of supply.

Turbidity monitoring should be carried out at several points in the plant, with the raw water and filtrate being the normal minimum. Other points would be after coagulation, at the interface between the sand and coal of the filter, and at the point where water leaves the treated water storage and enters the distribution system.

Most physical and chemical analyses of the water can be carried out on a continuous basis. Free ammonia cannot be analyzed accurately and reliably except by laboratory samples. Where raw water supplies have free ammonia levels approaching 0.2 mg/L and where retention times are too short for laboratory analysis of samples, it may be necessary to superchlorinate above the breakpoint in order to ensure that a free residual is present.

Variables to be monitored. The raw water and in-plant monitoring needed will vary with the water quality and plant design. All plants should monitor turbidity, temperature, pH, and bacterial content of the raw water. Some will monitor algae, free ammonia, iron, manganese, and color depending on the source of supply. At the filter, turbidity monitoring would be carried out on the filtrate and probably at the sand-coal interface. Monitoring the filters provides a valuable indication of the condition of each and assists in maintaining a low filtered water turbidity. The latter can be very valuable in a direct filtration plant to evaluate the effects of the chemical treatment, to provide a warning of filter breakthrough, and to analyze the floc shear strength and consequent need for polymer or other coagulant. Residual aluminum would also be monitored on the filtrate, and the alum dosing would be adjusted or substituted by other coagulants should the level increase to an unacceptable level.

Methods of plant control. The operation and control of a filtration plant fall into two areas—hydraulic control and process control. Hydraulic control, including control of pumps, filter rate, and filter backwashing, can be and has been operated automatically for many years. None of these functions is very different in a direct filtration plant, as compared to a conventional plant with sedimentation, except possibly in the duration of the wash rate applied to the filters. Filter wash can be initiated by filtrate quality, high head loss across the media, or by time. These functions can be automated using analog systems or a process controller.

Most plants today still use manually adjusted chemical feeds from monitored information with analog control loops to sustain feed rates. Alarms will indicate substantial deviations. Pilot tests of filter media can be very valuable in predicting trends and in evaluating the effectiveness of chemical application.

The trend is toward more automation, particularly with direct filtration, because a process controller will respond quickly to variations in the monitored input data. Many plants have data logging, but few have true process controllers. A true process controller requires appropriate algorithms which take time to develop.

A fully automated plant can employ analog control of the chemical feed loops with process controller supervision of the analog loop settings. The plant may have direct digital control with all variables sustained directly by the process controller. The latter sometimes has analog backup in case of computer failure.

A report approved by the T & P Council, prepared by the Direct Filtration Subcommittee of the AWWA Filtration Committee. Subcommittee members are S. Bishop, T. F. Craft, D.R. Fisher, M. Ghosh, P.W. Prendiville, K.J. Roberts, S. Steimle, and J. Thompson.

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Treatment Technology to Meet the Interim Primary Drinking Water Regulations for Inorganics: Part 5

Thomas J. Sorg and Gary S. Logsdon

The fifth in a series summarizing existing treatment technology to meet the inorganic National Interim Primary Drinking Water Regulations, this report describes current methods for removing barium and radionuclides from drinking water.

Barium Removal From Water

Health effects. Because of its toxic effect on the heart, blood vessels, and nerves, barium was placed in the USPHS Drinking Water Standards in 1962.[1] Even prior to this action, concern for barium in drinking water existed because the 1946 USPHS Drinking Water Standards[2] stated that "salts of barium . . . shall not be added for water treatment purposes." With almost no new health effects data available on which to base a modification to the 1962 limit, EPA maintained the maximum contaminant level (MCL) at 1.0 mg/L.[3]

Barium is not believed to be an essential human element,[4] and both human and animal studies have shown that barium is a general muscle stimulant, particularly to the heart muscle.[3] Human symptoms following acute ingestion include salivation, vomiting, diarrhea, hypertension, twitching, and flaccid paralysis of the skeletal muscle. Severe poisoning leads to respiratory muscle paralysis and ventricular fibrillation that can result in death. The fatal dose of ingested barium chloride is reported to be 0.8–0.9 g (550–660 mg as barium).[3]

Barium is a bone-seeking element and possesses chemical and physiological properties that enable it to replace calcium. Barium replacement of calcium will affect the adrenal gland and the release of hormones by the gland.

Because of the lack of data on the effects of ingesting barium in drinking water, the MCL of 1 mg/L was originally established by the extrapolation of health data from industrial exposure, and on the assumption that 90 percent of the barium ingested is absorbed into the blood stream via the gastrointestinal tract.[4] Most recent animal studies show that absorption rates for barium decrease from 90 percent in young animals to less than 10 percent in adults.[6] These data suggest, therefore, that very young children will absorb more barium than adults.

Until 1978–1979, no epidemiological studies had been completed to assess the chronic effects on humans ingesting barium in drinking water. To expand the knowledge of the health effects of barium ingestion, an EPA-sponsored study was conducted in northern Illinois to determine whether health effects could be identified in communities exposed to barium in their drinking water.[7] The study revealed a higher death rate in communities exposed to barium at the 2-mg/L level as opposed to communities with no significant exposure to barium. The epidemiological data also showed significant differences in the blood pressure of the male population of communities having 7 mg/L in their drinking water, in comparison to communities with only 0.1 mg/L of barium. The investigators emphasized the need for caution in data interpretation because of the factors that could not be controlled in the study. Moreover, the study did not result in a definitive conclusion on a no-effect level of barium in drinking water that would permit the investigators to recommend a safe standard for barium.

Because of the higher death rate with barium concentrations greater than 2 mg/L and the elevation in blood pressure at 7 mg/L reported in this study, R.J. Garner, director of the Health Effects Research Laboratory (HERL), USEPA, advised that the current barium limit of 1 mg/L not be relaxed and that communities exceeding the MCL should proceed with plans to reduce barium exposure.[7] Both the investigators and Garner agreed that more data were needed to better identify a no-effect level of barium in drinking water.

Barium is not a serious problem in most drinking water supplies, although it is frequently detected in trace amounts in surface waters and drinking waters. Kropp and Kroner reported a barium detection rate of 99.4 percent of 1577 water samples from lakes and streams in the US collected from 1962 to 1967.[8] The minimum, maximum, and mean concentrations were 0.002, 0.34, and 0.043 mg/L, respectively.

The 1969 USPHS community water supply survey showed only one finished water sample from 967 water supplies (less than 1 percent) that exceeded the 1-mg/L limit.[9] Furthermore, the October 1973[10] and April 1975[11] reports on the chemical analysis of the interstate carrier water supply systems reported no violations of the barium limit. Finally, a very recent EPA study designed to develop relationships between inorganics in tap water and cardiovascular mortality rates reported a high frequency of detection of barium, but in low concentrations.[12] For example, barium was detected in 94.3 percent of the 35 areas surveyed; the survey included the analysis of 3834 individual household water samples. Using mean barium values for the 35 areas, the maximum "mean" was 0.35 mg/L; the minimum "mean" was 0.009 mg/L; and the mean "mean" was 0.067 mg/L.

Although barium is not commonly found in groundwater, a few areas of the US do have very significant levels of barium in their source water, the most noteworthy being in northern Illinois. The barium concentration in the groundwater in this area has been reported to be as high as 19 mg/L, with 2–7 mg/L being a common range.[13]

Barium chemistry overview. Barium is one of the alkaline earth metals, and thus it has chemical and physical properties similar to calcium, radium, magnesium, and strontium, the other elements in this group. In chemical behavior barium is more similar to radium, also included in the EPA drinking water regulations, than to the other three elements.

Barium occurs in only one major form as a divalent cation, Ba^{+2}. The sulfate and carbonate salts are relatively insoluble; barium sulfate is more insoluble than calcium sulfate. The solubility of barium sulfate is reported to be 2.2 mg/L in pure water at 25°C, and of barium carbonate to be 18 mg/L under the same conditions.[14] Barium sulfate (barite) is a common mineral, and because of its low solubility, waters containing sulfate should not contain any significant amounts of dissolved barium. The low soluble level of barium sulfate also suggests that alum and ferric sulfate coagulation should be effective methods for

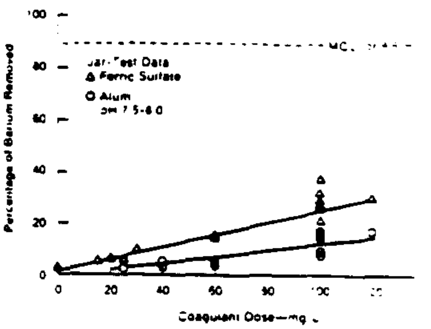

Fig. 1. Effect of Coagulant Dose on Bariu Removal From Well Water by Coagulatio Treatment

Barium Concentration—7.0–8.6 mg/L[13]

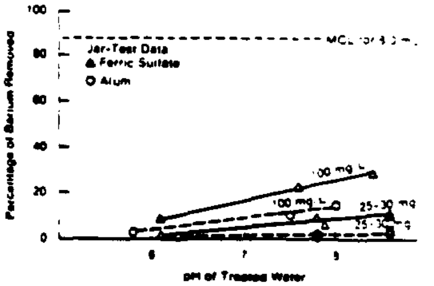

Fig. 2. Effect of pH on Barium Removal Fro Well Water by Coagulation Treatment

Barium Concentration—7.8–8.0 mg/L[13]

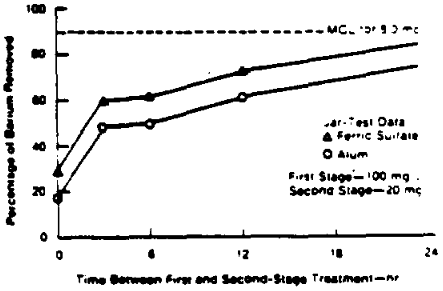

Fig. 3. Effect of Two-Stage Coagulation on Barium Removal From Well Water

Barium Concentration—7–8 mg/L[13]

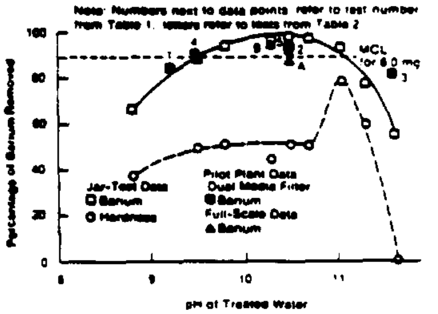

Fig. 4. Effect of pH on Barium Removal b Lime Softening

Barium Concentration—7.8 mg/L[13]

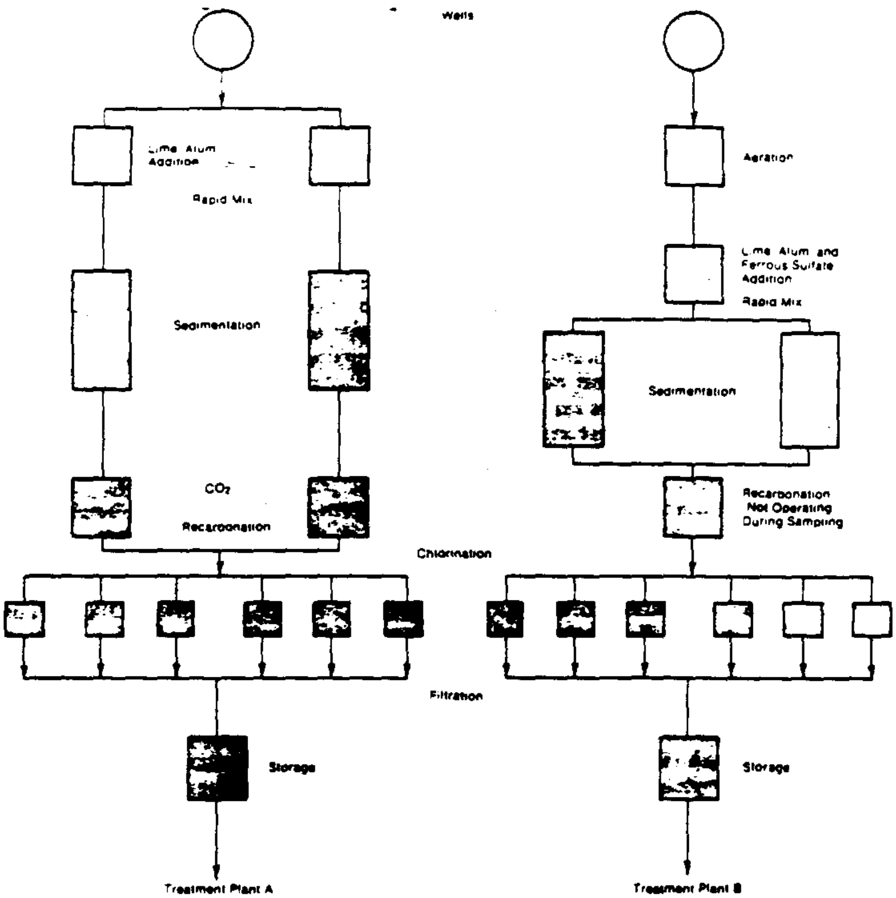

Fig. 5. Flow Diagram of Full-Scale Lime Softening Treatment Plants

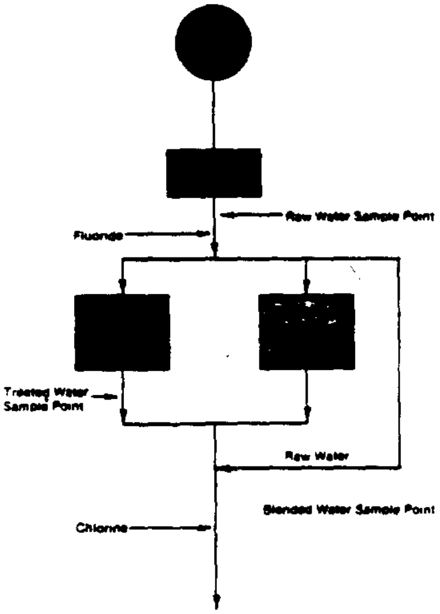

Fig. 6. Flow Diagram of Ion Exchange Softening Plant C

Constructed in 1964; Softening Cycle of 760 kL (200 000 gal)

barium removal. even though these ar not good techniques for calcium removal. Pilot plant tests (discussed in th following section) indicate that the formation of barium sulfate may be a very slow process. however. and thus not good mechanism for removal.

Many metals. such as cadmium. lead and chromium. form insoluble hydroxide complexes and may be precipitated from solution by pH adjustment. The formation of barium hydroxide. Ba(OH) is possible but the complex is very soluble (about twenty times more soluble than barium sulfate). and thus should no be a significant factor in removing barium by conventional coagulation treatment.[14] The chloride form is even more soluble than the hydroxide form. and waters containing chloride result in a higher solubility of barium sulfate.[14]

Because barium has chemical properties similar to calcium and magnesium. treatment methods effective for the removal of these elements should also be effective for barium. This suggests. therefore. that lime softening and ion exchange methods used to remove hardness should be good techniques for barium removal. The capacity of ion exchange resins for barium removal should be dependent on the concentration of calcium and magnesium and the selectivity of one or both of these elements over barium.

Treatment methods. Because barium is not a frequent problem in drinking water, very few studies have been conducted on its removal. Laboratory and pilot plant studies for barium removal by conventional coagulation and lime softening methods have been conducted by the Drinking Water Research Division (DWRD) of the USEPA at the Environmental Research Center in Cincinnati. Except for these data (reported here), very limited or no data exist on barium removal by other treatment methods, such as ion exchange, activated carbon, reverse osmosis, and electrodialysis.

Conventional coagulation. Laboratory jar-test studies for barium removal by alum and ferric sulfate coagulation treatment have been conducted by Logsdon et al.[15] These researchers expected to remove a very high percentage of the 7-8 mg/L of naturally occurring barium from an Illinois groundwater with both coagulants, by forming insoluble barium sulfate during the coagulation process. Their expectations were not achieved, however; removals did not exceed 30 percent even when the dose was as high as 120 mg/L (Fig. 1). Their data also revealed a small pH effect, with removals increasing slightly as the pH increased from about 6 to 8.5. Iron coagulation was slightly more effective than alum (Fig. 2).

The reason for the lack of high removals was suggested by the authors to be supersaturation of barium and inadequate time for the barium sulfate precipitate to form. A second series of tests were therefore conducted in which two-stage coagulation was carried out. The first stage consisted of treatment with 100 mg/L of either alum or ferric sulfate and 1 hr of sedimentation. The settled water was withdrawn and kept in separate jars for second-stage treatment at intervals of 3, 6, 12, and 24 hr. Second-stage treatment consisted of dosing the settled waters with 20 mg/L of the same coagulant, 1 hr of sedimentation, and centrifugation to represent filtration.

The results of two-stage treatment showed higher total removals, up to about 80 percent (Fig. 3). These data suggest that supersaturated barium sulfate solutions form the crystalline phase slowly, and thus coagulation treatment for removal of large amounts of barium would be a very costly solution.

Lime softening. Jar-test data, pilot plant tests, and limited full-scale treatment plant information indicate that barium can be effectively removed from water by lime softening, and that the efficiency is pH dependent. Logsdon et al[15] conducted jar-test studies at pH 9 to 11.5 using Illinois groundwater having 7–8 mg/L of naturally occurring barium. The results of these tests, plotted in Fig. 4, show that barium removal is pH dependent. Removals increased from 65 percent at pH 8.7 to a peak of 98 percent at pH 10.5, and then declined to 55 percent at pH 11.6. In order to achieve the barium MCL, 87.5 percent removal was required; this was attained from about pH 9–11.

Because barium is chemically similar to calcium and magnesium (hardness), the hardness removal results are also shown in Fig. 4 for comparison. The barium and hardness removal curves have similar characteristics, except in the pH range of 10.6 to 11.4 when the hardness removal curve develops a sharp increase. The reason for this sharp increase is suggested to be the removal of magnesium. The raw water had a magnesium concentration of 21 mg/L, and magnesium will precipitate in the hydroxide form at pH 10.6 (excess lime softening). As the pH is increased by the addition of more lime, an excess of calcium occurs and the percentage of hardness removed decreases.

Logsdon et al[15] suggested that the removal mechanism for barium is the precipitation of insoluble barium carbonate, the same mechanism as for calcium. The decline in barium removal above pH 11 was thought to be caused by the formation of the more soluble barium hydroxide. The formation of magnesium hydroxide at roughly the same pH level adds support to this suggestion.

To verify the jar-test results reported by Logsdon et al,[15] DWRD conducted five pilot plant lime softening tests using water from the same area of northern Illinois. The test water was trucked from two separate locations to the DWRD laboratory in Cincinnati, where the tests were run using the pilot plant system and test procedures described in part 2 of this series.* All tests were conducted for approximately 100 hr of continuous operation except for test 5, which was run for only 77 hr because of a power failure. The pilot plant test results are summarized in Table 1, and the dual media filter removals are plotted on Fig. 4 for comparison with the jar-test data.

Although the test water for the jar tests and pilot plant studies did not contain identical barium concentrations, the pilot plant results were comparable to the jar-test removal data and confirmed the dependency of removals on pH. The best removals were achieved at pH 10.5 (93 and 94 percent); lower removals resulted at the lower pH of 9.2 and 9.5 (84 and 91 percent) and at the higher pH of 11.6 (85 percent).

Several treatment plants in Illinois soften groundwater that contains natural barium. DWRD collected three sets of samples from two of these plants (flow diagrams shown in Fig. 5), and the results are given in Table 2. The data are also plotted in Fig. 4 for comparison with the laboratory and pilot plant test results. Although the plants were very old, operated beyond design capacity, and were undergoing changes in operation. barium removal results were comparar to the DWRD laboratory and pilot pla test data. Plant A achieved 88 perce removal with water having 7.5 mg L barium, and plant B achieved 95 perce removal with water containing 17.5 mg of barium.

Ion exchange. Cation exchange, usi natural greensand or synthetic resir has been utilized successfully for mai decades to soften hard waters. T cation exchange selectivity series shov these resins to have an even great affinity for barium than for either ca cium or magnesium. A typical selectivi series for divalent cations of strong-ac cation exchange resins is reported as ! > Pb > Sr > Ca > Zn > Fe > Mg This information suggests, therefore, th ion exchange softening should be mo efficient for barium removal than f calcium or magnesium removal.

Laboratory ion exchange studies r ported in the literature[16] confirm that t cation exchange process is an effecti method for barium removal. To veri this information, DWRD conducted short field study on two full-scale ic exchange softening plants in northe Illinois. This study consisted of collec ing raw, treated, and blended finish water samples through one treatme cycle of one ion exchange bed from ea plant to determine the amount of bariu and hardness removed. The flow di grams of the two treatment system

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*Jour. AWWA, 70:7:378 (Jul. 1978).

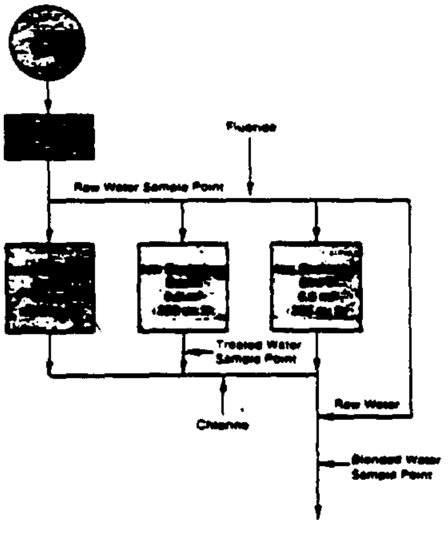

Fig. 7. Flow Diagram of Ion Exchange Softening Plant D

Constructed in 1964; Softening Cycle of 1.3 ML (350 000 gal)

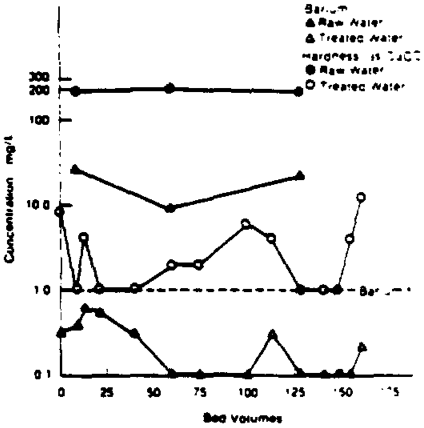

Fig. 8. Barium and Hardness Removal Fro Well Water by Full-Scale Ion Exchange Tre. ment Plant C

labeled plants C and D, are shown in Fig. 6 and 7, respectively.

Treatment plant C had two ion exchange beds and was treating raw water having a barium concentration of about 10 mg/L and a hardness of 218 mg/L (as $CaCO_3$). The second plant, D, consisted of three ion exchange beds and was treating well water having a barium level of 19 mg/L and a hardness of 230 mg/L (as $CaCO_3$).

The barium and hardness removal data collected from ion exchange bed 1 of treatment plant C are plotted in Fig. 8. These data show that the barium was reduced to less than 1 mg/L through 153 bed volumes [714 000 L (188 000 gal)]. This system was operated on a flow basis rather than on a hardness breakthrough basis, and thus the treatment cycle was terminated before either hardness or barium broke through the ion exchange bed. The percentage removals ranged from 94 to more than 99 percent for barium, and from 95 to more than 99 percent for hardness (Fig. 8). Eight finished water samples (blended water) showed barium to range from 2.4 to 4.5 mg/L.

The test data of one treatment cycle of ion exchange bed 2 of plant D are plotted in Fig. 9. This treatment cycle was run past both barium and hardness breakthrough; the barium concentration of the treated water was 5.8 mg/L, and the hardness concentration 142 mg/L (as $CaCO_3$) at shutdown. Barium breakthrough (1 mg/L) occurred around 100 bed volumes [787 kL (207 000 gal)]. The raw water for plant D had a barium concentration of 19 mg/L, almost twice the level in the raw water of plant C (10 mg/L). Therefore, barium breakthrough was expected to occur earlier. The data also show that hardness broke through (85 bed volumes) before barium.

On a percentage removal basis barium removal exceeded hardness removal very significantly, as shown in Fig. 10. At shutdown, 70 percent of the barium was being removed as compared to only 38 percent of the hardness.

Blending of raw water with treated water to increase the hardness, and thus stabilize the distribution water, was practiced at both plants. Five blended water samples collected from plant D during the test run had a barium concentration range of 1.5–4.0 mg/L. These data indicate that blending may not always be feasible for barium treatment. If the raw water has a high barium concentration, as experienced at both of these test locations, even a small amount of blending will raise the barium level above the MCL. If blending is not possible, restabilization of the very soft water will be necessary. Restabilization can be accomplished by several methods, such as blending the treated water with a barium-free water or adding lime.

To evaluate the potential disposal problem of the high barium wastewater, twelve water samples were collected during the regeneration cycle of plant D. A plot of the barium concentration with time is given in Fig. 11: the peak concentration was slightly above 6000 mg/L. The calculated amount of barium in the brine waste was about 18 000 g, or 85 percent of the amount calculated to be removed (22 100 g) during the treatment cycle. The average calculated barium concentration of the 35 kL (9300 gal) of wastewater was 534 mg/L.

Activated carbon. Because calcium and magnesium are not absorbed by either powdered activated carbon (PAC) or granular activated carbon (GAC), the potential for barium removal by PAC or GAC has always been assumed to be low. Data from a PAC study by Thiem and O'Connor[18] and from the DWRD pilot plant studies confirm this conclusion. Thiem and O'Connor reported on the effectiveness of three different materials, PAC 1,[*] PAC 2,[†] and PAC 3[‡] on the removal of 5 mg/L of barium at pH 7, 8, and 9. The effect of PAC doses up to 100 mg/L was also studied. The results of the laboratory tests showed that the highest removal achieved was only 7 percent with a 100-mg/L dose of PAC 1. Although the optimum pH for two of PAC materials was slightly different, the ranking of the three materials for barium removal at the three pH values remained the same: PAC 1, PAC 2, PAC 3. The optimum pH for PAC 1 was pH 9, and for PAC 2, pH 7.

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[*] Amoco PX-121, Amoco Research Corp., Chicago, Ill.
[†] Darco HDB, ICI Americas, Inc., Wilmington, Del.
[‡] Nuchar Aqua, Westvaco Corp., Covington, Va.

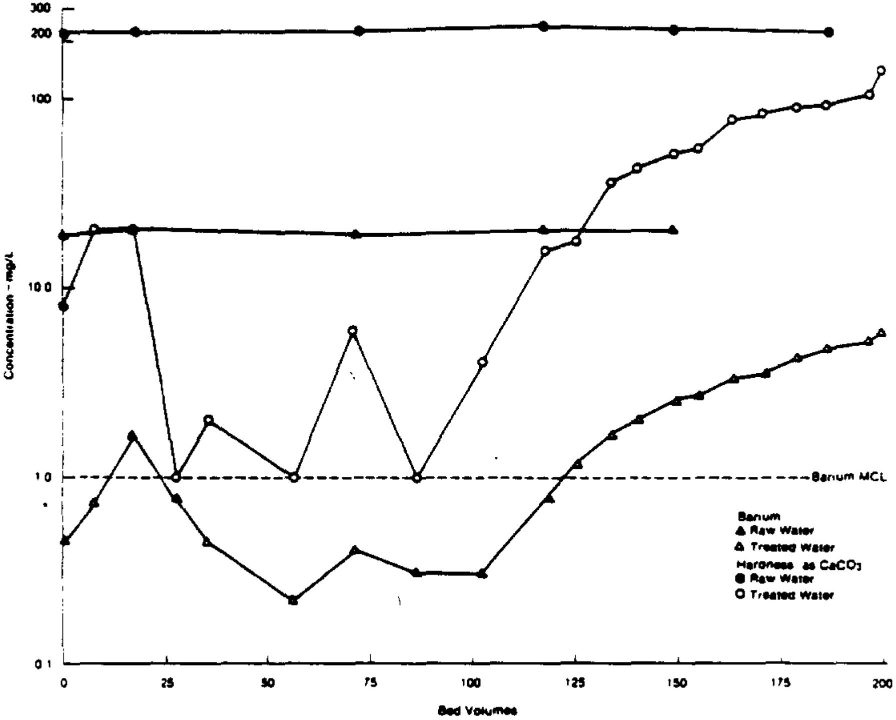

Fig. 9. Barium and Hardness Removal From Well Water by Ion Exchange Treatment Plant D

No change in removal was noted with pH for PAC 3: removals were 1 percent or less under all conditions.

The lime softening pilot plant data of Table 1 show no difference in removals between the dual media and GAC filters. These data indicate that the removal mechanism in the GAC filters is only filtration: no additional barium is being removed by GAC adsorption.

Reverse osmosis and electrodialysis. Reverse osmosis (RO) and electrodialysis (ED) are very effective treatment processes for hardness removal. and thus they should also be very efficient for barium removal. RO laboratory studies have been conducted for barium removal by Mixon[19] and by DWRD. and the test results support this conclusion. Mixon achieved 96-98 percent removal of 9.15 mg/L of barium in tap water by three different cellulose acetate (AC) membranes. A high removal of 98.6 percent was also achieved with tap water containing 7 mg/L of barium in combination with five other metal ions.

DWRD conducted two short RO laboratory experiments for the removal of naturally occurring barium (7 mg/L) in a northern Illinois groundwater. The tests consisted of treating 236-L and 200-L batches of water with a small RO system (AC membrane) operated at 1140 and 1240 kPa (165 and 180 psi), respectively. The test results were comparable to those achieved by Mixon: barium removal was 95-99 percent at 1140 kPa (165 psi) and 95-96 percent at 1240 kPa (180 psi).

A general literature review did not reveal any specific data on the removal of barium from drinking water by ED. Because of the similarities in treatment capabilities. however. this technique should be as effective as RO.

Barium treatment summary. Conventional coagulation treatment is not an effective method for barium removal. even when high doses of sulfate coagulants are used. Test data suggest that two-stage coagulation is required to achieve a barium removal of about 70 percent. This is expensive and thus impractical.

The two softening methods. ion exchange and lime precipitation. are very good methods for barium removal. Full-scale treatment data show that both methods are capable of lowering barium concentrations as high as 17-19 mg/L to below the MCL of 1 mg/L. One shortcoming of the ion exchange process is that the normal practice of blending raw and softened treated water is not always feasible with water having a high barium concentration. Stabilization of the very soft. treated water normally would be required to prevent any corrosion problems from developing in the distribution system.

Barium removal by lime softening is pH dependent: the optimum range (93-95 percent) is pH 10-10.5. Below and above this range. removals decrease with decreasing and increasing pH. respectively.

Powdered and granular activated carbon are ineffective methods for barium treatment. Laboratory data show that PAC achieved less than 7 percent removal under optimum conditions. and GAC essentially no removal. RO and ED are good methods for hardness removal. and thus are equally effective for barium removal (95-98 percent).

Because barium is not a widespre and frequent problem. and because s eral conventional water treatment me ods (ion exchange and lime softeni are good methods for barium treatme DWRD has no specific research proje devoted to barium treatment.

Radionuclide Removal From Wat

Health effects. Radioactivity. either n urally occurring or man-made. has be known for many years to produce de mental biological effects in man: dev opmental abnormalities. cancer. a death. The National Academy of Scier reports that radiation may conceival cause cancer of virtually any type or any site given the appropriate conditic of irradiation and host susceptibilit The primary basis for the EPA radior clide regulations[21] for drinking water the carcinogenic potential of this mate al. The carcinogenic potential of radio tivity was also the primary reason for t establishment of the 1962 USPHS regu tions for drinking water.[1]

The radionuclide regulations are mc complex than the MCL established the other inorganic contaminants. T MCL for the combination of radium- and 228 was set at 5 picocuries per li (pCi/L). This concentration is sligh above the 3 pCi/L standard establish for radium-226 alone by the USPHS 1962.[1] The USPHS Drinking Water St dards also had a 10-pCi/L standard strontium-90. whereas EPA incluc strontium in the general regulation beta particle and photon radioactivity

The second part of the EPA radior

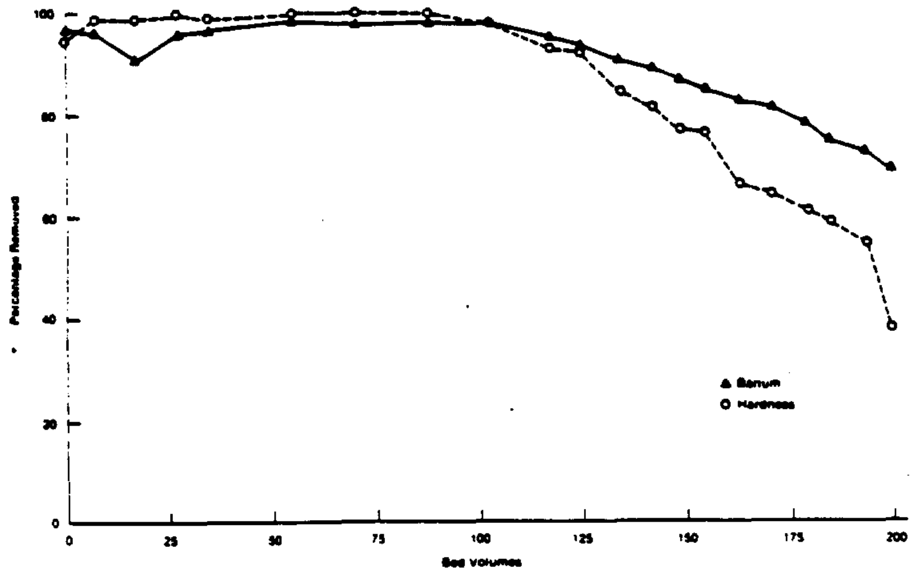

Fig. 10. Percentage of Barium and Hardness Removed From Well Water by Full-Scale Ion Exchange Treatment Plant D

clide regulations governs gross alpha activity, including radium-226 and 228. It states that the gross alpha activity shall not exceed 15 pCi/L. The third regulation that concerns man-made radionuclides states "the average annual concentration of beta particle and photon radioactivity from man-made radionuclides in drinking water shall not produce an annual dose equivalent to the total body or any internal organ greater than 4 millirem/year." The 1962 USPHS standards established gross beta concentrations of 1000 pCi/L as grounds for rejection of a water supply, except when more complete analysis indicated that concentrations of nuclides were not likely to cause exposure greater than the Radiation Protection Guides recommended by the Federal Radiation Council (FRC).[22]

A large body of information exists on the biological effects of radiation, and risk estimates for total body and specific body organ exposure have been published by a number of groups, including the National Academy of Sciences (NAS),[23] the United Nations Scientific Committee on the Effects of Radiation (UNSCEAR),[24] the International Commission on Radiological Protection (ICRP),[25] and the National Council on Radiation Protection and Measurements (NCRP).[26] Most of the low dose–effect relationships have been derived from studies of high radiation doses, based on the assumption that the effects are linearly proportional to the dose. In establishing the MCL for radioactivity in drinking water, EPA considered many factors, including the source of radioactivity and all the estimated risk factors for naturally occurring and man-made radionuclides.

Of the many radionuclides found in the environment, radium is considered the most significant because it is naturally occurring and frequently found in groundwaters in several sections of the US. Furthermore, radium is significant because it is similar to calcium in that it is a bone-seeking element: 80–85 percent of the radium retained in the body is deposited in bone.[27]

Bone cancer is not the only health risk from radium ingestion, but it is definitely the most significant factor. Using the risks estimates of the FRC and the NAS, a radium-226 concentration of 5 pCi/L in a drinking water system serving 1 million persons could result in an estimated health impact of 1.5 fatalities per year.[27] Although 1959 data suggest that radium-228 is not as toxic as radium-226, more recent data indicate that the toxicities are about the same.[24,25]

The potential of other alpha particle radionuclides also exists in drinking waters. The MCL of 15 pCi/L was established on gross alpha particle activity, rather than on each individual radionuclide, because of the impracticality and the expense of monitoring. The MCL for gross alpha activity was based on the radioactivity of other alpha particle-emitting contaminants relative to radium. The 15 pCi/L gross alpha particle limit (which includes radium) is based on the conservative assumption that if the radium concentration is 5 pCi/L and the balance of the alpha particle activity is caused by the next most radiotoxic alpha particle-emitting chain, starting with lead-210, the total dose-to-bone relationship would be equivalent to less than 6 pCi/L of radium-226.

When establishing the regulations for radioactivity in drinking water, EPA also recognized the potential contamination of surface water supplies by man-made radionuclides that decay by beta and gamma emission. Because of the multitude of man-made radioisotopes and the impossible task of establishing an individual limit for each element, EPA established an MCL based on a total body dose of radiation from the ingestion of these radionuclides. Using risk estimates, the 4-millirem annual total body dose regulation suggests a fatal cancer risk of 0.4 to 2 cancer deaths per 1 million people per year.[27]

Concentrations of most man-made radionuclides in drinking water causing 4-millirem/year exposure can be calculated using specific dose data from the National Bureau of Standards.[28] A list of these concentrations has been developed and published by EPA.[27] These data were calculated on the assumption that an individual consumes an average of 2 L of drinking water per day.

The most significant man-made radionuclides are probably those resulting from atmospheric nuclear weapon testing, strontium-90 and tritium. The annual average concentrations yielding 4 millirem/year for a 2-L intake rate are 8 pCi/L for strontium-90 and 20 000 pCi/L for tritium.[27]

Traces of radium have been found in many areas of the US. The results of radiological sampling from eight standard metropolitan statistical areas (SMSA) and the state of Vermont as part of the 1969 community water supply study showed radium-226 in 60 of 143 water samples analyzed.[9,28] However, only four of the 60 samples had radium-226 concentrations above the 5-pCi/L MCL (5.4, 5.4, 76.5, and 136 pCi/L). Although these data suggest that radium may not be a serious problem, other information indicates that radium will be a frequent problem in certain sections of the country, namely in northern Illinois, Iowa, central and western Florida, and in the uranium mining areas of the Rocky Mountain states.

In 1964 radium-226 was detected in 151 of 241 town water supplies studied in Iowa.[29] Nineteen of these supplies had concentrations at 5 pCi/L or above, and the maximum concentrations measured were between 15 and 20 pCi/L. Gilkeson[30] studied the natural occurrence of radium in Illinois and found that high concentrations of radiation in drinking water were restricted to the Cambrian–Ordovician aquifer in northern Illinois. Water samples from numerous wells showed that more than 300 wells exceeded 3 pCi/L of gross alpha, some having up to 15 pCi/L of radium-226.

In Florida radium-226 is usually associated with phosphate deposits. Concentrations as high as 21 pCi/L have been measured in drinking water systems.[31,32] Whether serious problems exist with other types of radionuclides in either ground or surface water is not known.

Radionuclide chemistry overview. Two of the three parts of the radioactivity regulations are general and therefore encompass at least 100 or more radioisotopes. Thus a discussion of the chemistry and treatment technology of radionuclides will be limited to radium, the radionuclide expected to be the most frequently occurring in drinking water.

Radium is an alkaline earth metal and thus it has chemical and physical properties similar to the other elements in this group—calcium, magnesium, barium, and strontium. Radium-226, an alpha emitter, is a member of the uranium series and has a half life of 1622 years. Radium-228, a beta particle emitter, is a member of the thorium series and has a half life of 5.75 years. Because the uranium series has been encountered more frequently in natural waters than the thorium series, radium-226 should be more commonly found in drinking water

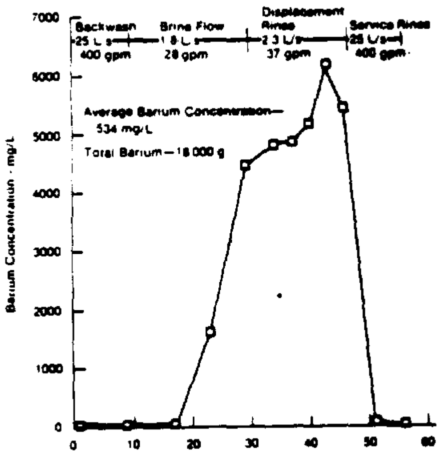

Fig. 11. Barium Concentration of Waste Water During Regeneration Cycle of Treatment Plant D

and thus have the most significance to the water utility industry.

Radium is a cation, Ra^{+2}, and was the last of the elements of the alkaline-earth group to be discovered (in 1896). Comparatively little information exists on the chemistry of radium and radium compounds because of the newness, scarcity, and lack of interest in this element. Of the five alkaline earth elements, barium and radium are more similar to each other than to any other elements.

Chemistry handbooks[14,33,34] report few data on the solubility of radium compounds. In some cases the data are even conflicting, particularly on radium sulfate. The solubility of radium sulfate is reported to be 2.1 mg/L and 0.021 mg/L, a hundredfold difference. The sulfate complex is the most significant, because its low solubility is the basis for the extraction of radium from mining ore. Other general information suggests that radium carbonate and chloride are less soluble than the barium salts. Radium solubility may have little significance in precipitation treatment because the MCL of 5 pCi/L is equivalent to 5×10^{-9} mg/L, a concentration many magnitudes below any of the reported solubilities of radium or barium compounds. Some radium chemistry information indicates, however, that radium solubility is significantly lower when it is found in combination with barium; the solubility is related to the total of the two elements in solution. Coprecipitation of radium with barium or other elements in the alkaline earth group may also occur. Because of the lack of information on radium chemistry, the predictive behavior of radium is based on the behavior of barium.

Treatment methods. Information on treatment will be confined primarily to radium, because it is beyond the scope of this report to present removal data on all the potential radionuclides included in the drinking water regulation. Furthermore, specific information on the effectiveness of water treatment processes for the removal of a wide variety of radioisotopes has been reported by Straub.[35]

Extensive treatment information for radium removal by softening techniques has been developed from several field studies in Iowa and Illinois.[28-30] Information also exists on the use of reverse osmosis.[31,32] Some data are available on conventional coagulation, but this method is considered ineffective and thus is not significant for radium treatment.

Conventional coagulation. The ineffectiveness of conventional coagulation treatment for calcium, magnesium, and barium suggests that this method should not be a good method for radium removal. A general literature review revealed no specific treatment studies on radium removal by conventional coagulation treatment using either alum or iron coagulants. Tsivoglou et al[37] reported that eight weeks of composite water samples from two full-scale conventional coagulation treatment plants in New Mexico showed radium (2–4 mg/L) to pass unchanged through the treatment plant, and thus not be removed by this process. Singley[38] suggests that chemical coagulation may remove up to about 25 percent of radium, but the results are variable and difficult to control.

Straub[35] reported that laboratory studies on the removal of strontium, another radionuclide and element of the alkaline earth group, have shown that conventional coagulation using either alum, ferric chloride, or ferric sulfate is not effective (0–15 percent) for this element of the group.[35] This same author also states that conventional coagulation with sedimentation and filtration removes 96–98 percent of phosphorus-32 and less than 1 percent of iodine-131.[35]

Because of the lack of success for the removal of all alkaline earth metals by coagulation treatment, DWRD did not conduct any laboratory studies by this method, but concentrated its efforts on studying the more promising methods of ion exchange and lime softening.

Lime softening. DWRD conducted two pilot plant tests for the removal of naturally occurring radium-226 from Illinois groundwater by lime softening at pH 9.6 and 10.6. These studies were conducted with the 0.1-L/s (2-gpm) conventionally designed pilot plant system and procedures described in part 2. The tests were run with the same waters and at the same time as the barium studies because the natural groundwaters used for testing contained both contaminants.

A summary of the two pilot plant tests is given in Table 3. Because of the complicated nature and the expense of the radium analysis, only three sets of samples were collected during the first test run and only two during the second run. The specific results have also been reported by Brinck et al.[36] The test data indicate that radium removal is pH dependent as was found with barium removal; at pH 10.6 dual media filter removals averaged 94 percent, decreasing to 84 percent at pH 9.50.

Radium removal information at full-scale lime soda ash softening plants in Iowa and Illinois was collected and reported by Brinck et al.[36] Radium reduction varied from 75 to 96 percent for waters having radium-226 concentrations of 5.6–9.3 pCi/L. Two Iowa systems reported pH measurements. A plant operated at pH 11.0 achieved a removal of 96 percent, while a plant operated at pH 10.1 removed only 75 percent.

Pilot plant and full-scale plant data suggest removals of 75–96 percent of radium-226, depending on the pH. At 75 percent the MCL of 5 pCi/L can be achieved by the process, providing the raw water concentration does not exceed
20 pCi/L. If 96 percent removal is att
able, the MCL can be met with a r
water concentration ≤125 pCi/L.

Lime and lime-soda ash soften
studies have been reported on the remc
al of strontium.[35] The removals follow
a pattern similar to calcium, barium, a
radium removal, increasing with incre
ing chemical dosages of lime and so
ash from 67 to 89 percent. As for bariu
and radium removal, the removal mec
anism suggested was coprecipitati
with calcium.

Lime-soda ash softening is al
reported to provide a 90 percent or bett
removal of barium-lanthanium-140, ca
ium-115, scandium-46, yttrium-91, a
zericonium-neobrium-95.[35] The proce
is reported to be ineffective for tungste
185 and iodine-131.[35]

Ion exchange. Cation exchange is a ve
good method of removing radium a
strontium from drinking water. Fu
scale ion exchange softening plants
Illinois and Iowa achieved avera
removals of 81–97 percent of radium-2
from water having initial concentratio
of 3.25–43 pCi/L.[36] A zeolite soften
plant achieved an average reduction
radium-226 from 43 to 1.9 pCi/L (
percent). Another plant was capable
lowering the radium-226 level from 5.7
0.2 pCi/L. In contrast to the proble
with barium treatment of not being ab
to blend treated and raw water, many
the plants were able to blend raw wi
treated water and to meet the radiu
MCL in the finished water. The stu
results also indicated that radium-2
was continuing to be removed for a sho
time after hardness breakthrough.

Samples of regeneration wastes, so
ener brine, and rinse waters were me
sured and found to have concentratio
of 320–3500 pCi/L. Additionally, th
major portion of the radium-226 wa
found to be removed from the ion
exchange beds during the first 10–30 min
of the regeneration cycles.

Cation exchange has been shown to
remove 95 percent or more of strontium-
89, cesium-137, cadmium-115, and bar
ium-lanthanium-140, elements present
solution as cations.[35] Strontium-90 w
also found to be removed by the catio
exchange resin after calcium brea
through, suggesting a greater affinity f
strontium than for calcium. Anio
exchange will achieve equal results
tungsten-185, an anion. Yttrium-91, sca
dium-46, and zirconium-niobium-95 a
removed partially by both cation a
anion exchange resins, which sugges
that they are in a colloidal form.[35]

Activated carbon. No specific inform
tion was found on radium removal
PAC. The lack of success of PAC f
calcium and barium removal sugges
however, that PAC would not be a goo
removal method for radium. The sam
reasoning also holds for GAC treatme

TABLE 1
*Results of Lime Softening Pilot Plant Tests for Barium Removal From Well Water**

Test Number	Date Started	Length of Tests hr	pH of Treated Water	Average Raw Water Concentration		Percentage Barium Removed				Percentage Hardness Removed	
						Settled Water		Filtered Water		Filtered Water	
				Barium mg/L	Hardness (as CaCO) mg/L	1st Stage	2nd Stage†	Dual Media	GAC‡	Dual Media	GAC‡
1	7/14/75	100	9.2	12.6	214	70		84	84	49	49
2	7/21/75	103	10.5	10.1	216	81	92	93	93	59	60
3	7/28/75	102	11.6	10.4	209	63	80	82	85	79	79
4	8/4/75	102	9.5	8.6	248	86		90	91	57	57
5	8/18/75	77	10.5	6.5	258	91	92	94	94	62	62

*Illinois groundwater containing naturally occurring barium. Two water sources: one source used for tests 1, 2, 3, and a second source for tests 4 and 5.
†Second settling stage followed recarbonation which lowered pH to 9.6
‡Filtrasorb 200, Calgon Corp., Pittsburgh, Pa.

TABLE 2
Barium Removal From Well Water by Full-Scale Lime Softening Water Treatment Plants

Treatment Plant	Treatment pH	Water Sample	Barium Concentration—mg/L				Hardness Concentration (as CaCO)—mg/L	
			Raw Water	Settled Water	Recarbonated Water	Filtered Water	Raw Water	Filtered Water
A	10.5	1	7.5	0.91	1.08	0.83	276	72
		2	7.5	0.96	1.15	1.00	268	68
		3	7.5	0.96	1.00	0.73		
Average concentration			7.5	0.94	1.08	0.85	272	70
(Average percent removed)				(88)	(86)	(89)		
B	10.3	1	17.3	1.60		0.90	240	72
		2	18.0	1.90		0.84	252	80
		3	17.0	2.06		0.68		
Average concentration			17.4	1.85		0.81	246	76
(Average percent removed)				(89.4)		(95.3)		(69)

TABLE 3.
*Results of Lime Softening Pilot Plant Tests for Radium-226 Removal From Well Water**

Test Number	Date Started	Length of Tests hr	pH of Treated Water	Average Raw Water Concentration		Percentage Barium Removed				Percentage Hardness Removed	
						Settled Water		Filtered Water		Filtered Water	
				Radium-226 pCi/L	Hardness (as $CaCO_3$) mg/L	1st Stage	2nd Stage†	Dual Media	GAC‡	Dual Media	GAC‡
1§	8/4/75	102	9.5	4.34	248	79		84	84	57	57
2**	8/18/75	77	10.5	4.82	256	82	93	94	96	62	62

*Illinois groundwater containing naturally occurring radium-226
†Second settling stage followed recarbonation that lowered pH to 9.6.
‡Filtrasorb 200, Calgon Corp., Pittsburgh, Pa.
§Samples collected only on 8/8, 8/9, and 8/10
**Samples collected only on 8/20 and 8/21

TABLE 4
*Radium-226 Removal by Full-Scale Reverse Osmosis Plants**

System Capacity		Radium-226 Concentration—pCi/L		Percentage Removed
m^3/d	10^3 gpd	Raw Water	Treated Water	
3780	1000	3.4	0.26	92
757	200	4.6	0.21	95
570	150	14	0.56	96
265	70	10.5	1.2	89
151	40	3.2	0.14	96
113	30	13.4	2.0	87
19	5	20.5	0.3	99
6.1	1.6	12.1	0.6	95
3	0.8	11.1	0.51	95

*Sources: T.J. Sorg et al[25] and W.L. Brinck et al[26] and the DWRD pilot plant data in Table 3 support this conclusion. The percentage removals for the dual media filters and the GAC filters were essentially the same, indicating that adsorption of radium-226 by the GAC material did not occur. Because removals by the GAC and dual media filter columns were identical, the GAC removal mechanism is attributed entirely to filtration.

Reverse osmosis and electrodialysis. Full-scale RO treatment system data have been developed for the removal of radium-226. Brinck et al[11] reported 96 percent removal of 14 pCi/L of radium-226 in Iowa. Sorg et al[12] studied the radium-226 removal by eight RO systems in Florida and reported removals of 87–98 percent of radium concentrations ranging from 3.2 to 20.5 pCi/L. A summary of these data are shown in Table 4.

A general literature review did not reveal any specific data on radium removal by ED, which should also achieve high removal because of its similarity in treatment capabilities to RO.

Both RO and ED are generally effective for the removal of most ionic species. Thus these two techniques should be good treatment methods for most radionuclides. Their most advantageous application is for the treatment of drinking water containing a mixture of radionuclides that would require the combination of several treatment methods to be effective.

Radionuclide treatment summary. The radionuclide that has the most significance in drinking water is radium because it is a naturally ocurring element frequently found in drinking water above the MCL. Radium has properties similar to calcium and barium, and thus the treatment methods effective for these two elements are the best methods for the removal of radium.

Conventional coagulation will remove only a small percentage (0–25) of radium from water. This treatment method is applied primarily to surface waters and would not normally be used for treating groundwaters containing radium.

The two most practical and economical treatment methods for radium are ion exchange and lime softening. Full-scale treatment plant information has been developed to show that ion exchange softening (cation exchange) can achieve removals in the 91–97 percent range. Treatment information also indicates that radium is more selective than calcium because calcium breakthrough occurs before radium breakthrough.

Lime and lime–soda ash softening have been demonstrated on a full-scale plant level to achieve 75–96 percent removal of radium. Pilot plant studies and full-scale plant data indicate removal efficiency to be pH dependent. For example, DWRD pilot plant studies showed 95 percent removal at pH 10.6 and 84 percent removal at pH 9.5. If 90 percent removal can be achieved by either softening method, the radium MCL of 5 pCi/L can be achieved with water containing 50 pCi/L or less of radium.

Activated carbon treatment, both PAC and GAC, is ineffective for radium removal. RO and ED are, however, good methods for radium; full-scale RO treatment data show removals in the 90-percent range to be easily achievable.

DWRD has conducted and spons several studies for radium remo Because these studies have shown ion exchange and lime softening are cient methods for radium, no furt research is now planned.

Treatment data exist to show t cation exchange resins will remo many cation radioisotopes and an exchange resins will remove anion rad isotopes. Mixed bed ion exchange resi RO, and ED are generally good treatme methods for the removal of mixtures radioisotopes. Specific treatment inf mation is generally lacking for m radioisotopes, however. Because of t lack of treatment data on man-ma radioisotopes, DWRD is develop plans to investigate methods to remo radioisotopes occurring most often drinking water. These studies will beg during fiscal year 1981.

Treatment Summary for Inorganic Contaminants

This report is the last of a five-p series that has expanded upon the su mary information presented in the EP document, Manual of Treatment Tec niques for Meeting the Interim Prima Drinking Water Regulations.[14] The ser has consisted of part 1 on fluoride a nitrate (*Jour. AWWA*, 70:2:105), part 2 arsenic and selenium (*Jour. AWW* 70:7:379), part 3 on cadmium, lead, a silver (*Jour. AWWA*, 70:12:680), part 4 chromium and mercury (*Jour. AWW* 71:8:455), and finally this last report, pa

TABLE 5

*Summary of Effectiveness of Water Treatment Processes for the Removal of Inorganic Contaminants**

	Conventional Coagulation			Ion Exchange			Activated Carbon			
Contaminant	Alum	Iron†	Lime Softening	Cation	Anion	Activated Alumina	PAC	GAC	Reverse Osmosis	Electrodialysis
Arsenic‡										
+III	P	P	P pH < 10.5 L-G pH > 10.5	P	L	P	P	P (bone char)§	G-F	G-F§
+V	G pH < 7.5 F pH > 7.5	E	F pH < 10 G pH 10-10.8 E pH > 10.8	P	E	E	P	E (bone char)	E	E§
Barium	P	P	E pH 9.5-10.8	E	P	P§	P	P	E	E
Cadmium	G > pH 6.5 F-P < pH 6.5	E pH > 8 F-L pH < 8	E	E	P	P§	L	L	E	E§
Chromium										
+III	E	E	E pH > 10.5 G pH < 10.5	E	P	P§	P	P	E	E§
+VI	P	P E (ferrous)	P	P	E	P§	P	P	E	E§
Fluoride	P	P	P	P	L	E	P	E (bone char)	E	E
Lead	E	E	E	**	P	P§	P	P	E	E§
Mercury										
Organic	F-P††	F-P††	P	ID	ID	P§	G-P	E	G-F§	G-L§
Inorganic	F-P††	F-G††	F	**	**	P§	G-P	E-G	G-F	G-L§
Nitrate	P	P	P	P	E	P	P	P	G	G
Selenium										
+IV	P	F pH < 7.5 L pH > 7.5	L-P	P	E	E-G	P	P	E	E§
+VI	P	P	P	P	E	L-G	P	P	E	E§
Silver	G pH < 8	G	G	E	P	P§	L	L	E	E§
Radium	P	P	E pH 9.5-10.8	E	P	P§	P	P	E	E§

*E—excellent, 90–100 percent; G—good, 70–90 percent; F—fair, 40–70 percent; L—low, 20–40 percent; P—poor, 0–20 percent; ID—insufficient data

†Results based upon ferric iron coagulant except as noted

‡Oxidation of As III to As V will result in As III removals similar to As V.

§Estimated

**Possible under controlled conditions, but not practical to water treatment

††Removal dependent on turbidity

5 on barium and radionuclides. The information presented indicates that no one treatment technology is ideal for all contaminants, although reverse osmosis and electrodialysis are generally effective for all contaminants. These two treatment methods are applied principally for desalting seawater and brackish groundwaters that are low in turbidity. They cannot be utilized on turbid surface water without extensive pretreatment, and thus would not be practical for removing contaminants from surface waters.

TABLE 6

Most Effective Treatment Methods for Removal of Inorganic Contaminants

Contaminant	Most Effective Treatment Methods
Arsenic	As V—iron coagulation, pH 6–8; alum coagulation, pH 6–7; excess lime softening; activated alumina, pH 5–6 As III—oxidation treatment of As III to As V; use same treatment list for As V
Barium	lime softening, pH-11; ion exchange softening
Cadmium	iron coagulation, above pH 8; lime softening; excess lime softening
Chromium	Cr III—iron coagulation, pH 6–9; alum coagulation, pH 7–9; excess lime softening Cr—ferrous sulfate coagulation, pH 7–9.5
Fluoride	ion exchange with activated alumina or bone char
Lead	iron coagulation, pH 6–9; alum coagulation, pH 6–9; lime or excess lime softening
Mercury	inorganic—ferric sulfate coagulation, pH 7–8; granular activated carbon organic—granular activated carbon
Nitrate	ion exchange with anion resin
Radium	lime softening; ion exchange with cation resin
Selenium	Se IV—ferric sulfate coagulation, pH 6–7; ion exchange with anion resin or activated alumina; reverse osmosis Se VI—ion exchange with anion resin or activated alumina; reverse osmosis
Silver	ferric sulfate coagulation, pH 7–9; alum coagulation, pH 6–8; lime or excess lime softening

TABLE 7

Most Probable Applications of Water Treatment Processes for Inorganic Contaminant Removal

Treatment Process	Principal Application for Water Treatment	Inorganic Contaminant Treatment Capability Effectiveness*			Most Probable Application for Inorganic Removal
		High	Moderate	Low	
Convention coagulation	Clarification of surface waters	Cd Cr III Cr VI As V Ag Pb	As III Se IV Hg(O) Hg(I)	Ba F NO_3 Ra Se VI	removal of Cd, Cr, As, Ag, or Pb from surface waters
Lime softening	removal of hardness from ground and surface water	Ba Ra Cd Cr III As V Pb	Se IV As III Hg(I) F	Cr VI NO_3 Se VI Hg(O)	removal of Ba or Ra from groundwaters; removal of Cd, Cr III, F, As V, or Pb from hard surface waters requiring softening
Cation exchange	removal of hardness from groundwaters	Ba Ra Cd Pb Cr III		As Se NO_3 F Cr VI	removal of Ba or Ra from groundwaters
Anion exchange	removal of nitrate from groundwaters	NO_3 Cr VI Se	Ba Ra Cd Pb Cr III		removal of NO_3 from groundwaters
Activated alumina	removal of fluoride from groundwaters	F As Se		Ba Ra Cd	removal of F, As, or Se from groundwaters
Powdered activated carbon	removal of taste and odors from surface waters		Hg(I) Hg(O) Cd	Ba Ra Cr III F NO_3 Ag	removal of Hg from surface waters during emergency spills
Granular activated carbon	removal of taste, odors, and organics	Hg(I) Hg(O)	Cd	Ba Ra Cr III F NO_3	removal of Hg from surface or groundwaters
Reverse osmosis and electrodialysis	desalting of sea water or brackish groundwaters	As V Ba Cr Pb Cd Se Ag F Ra Hg	NO_3 As III		removal of all inorganics from groundwaters

*High—greater than 80 percent; moderate—20 to 80 percent; low—less than 20 percent

The effectiveness of conventional water treatment processes, coagulation and filtration, lime softening, and ion exchange softening, varies in capability to remove the inorganic contaminants. These processes are very effective for some contaminants, and moderately or completely ineffective for others. The degree of treatment capability is generally dependent on several factors: contaminant form, valence, and concentration; pH of the treatment process; type (surface or ground) and quality (turbidity, TDS, and so forth) of water; type and amount of coagulant or other chemical additive; and the filtration, adsorption, or ion exchange media. Thus exact removal percentages cannot be assigned to a treatment process for any specific contaminant. However, the laboratory, pilot plant, and full-scale plant data and information presented in this series do provide a guide for selecting the best treatment method and operating condition, and for determining probable removal ranges for each contaminant. The treatment information has been summarized in matrix form for all contaminants and treatment processes in Table 5. In some cases when information was not available or was inadequate, removal estimates were developed by the authors using their best judgment. The matrix of Table 5 indicates that different methods are the most effective for the removal of each contaminant. For clarity, the most effective methods for the control of inorganic contaminants are summarized in Table 6.

Although a treatment process may be good for removing a contaminant, it may not be a practical or economical method. For example, neither ion exchange, reverse osmosis, nor electrodialysis are practical methods for treating surface waters and thus would not be used to remove a surface water contaminant even though the method may be effective. Consequently, certain treatment processes are considered to be more practical than others although both may be equally effective. Table 7 summarizes the treatment information according to treatment processes and the most practical or probable application. This table is also useful for evaluating the capability of an existing plant to solve a potential problem.

The major considerations, if not the most significant factors, in the selection of a treatment method are the economics—capital and operating costs. If modifications to or changes in the operation of an existing plant can solve the problem, only a slight increase in operating cost may occur, or possibly no cost increase at all. On the other hand, if no facility exists the costs will be the expense of constructing and operating a new treatment plant.

Because of the interest in the costs of

water treatment processes for the removal of the contaminants included in the NIPDWR, EPA contracted with Culp/Wesner/Culp, Consulting Engineers, Inc., for a detailed study of the construction and operation and maintenance costs for unit water treatment processes. The study was completed in 1979: four detailed reports were published by EPA in August 1979.[illegible] Construction and operation and maintenance cost curves were developed for approximately 100 unit processes: for each unit process, conceptual designs were formulated. All cost data are presented in January 1978 dollars, but a discussion on methods of cost updating is included.

The four reports show that the cost of treatment of the various unit processes is dependent on the design and size of the facility. A summary of the cost data is beyond the scope of this report, but copies of these detailed reports can be obtained from the Drinking Water Research Div., USEPA, Cincinnati, Ohio.

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Strong-Acid Ion Exchange for Removing Barium, Radium, and Hardness

Vernon L. Snoeyink, Carl Cairns-Chambers, and Julie L. Pfeffer

The research discussed in this article was conducted to investigate and document the performance of conventional sodium-form strong-acid resins for the combined removal of barium, radium, and hardness under controlled conditions. The reasons for the study were: (1) the premature leakage of barium from strong-acid ion exchange columns, (2) a lack of controlled laboratory studies to determine the factors affecting resin performance, and (3) a lack of documentation regarding the effect of the quantity of regenerant applied on effluent quality and on effluent quality as a function of the amount of water processed during a run.

Strong-acid ion exchange is a well established process commonly used to remove calcium (Ca^{2+}) and magnesium (Mg^{2+}) ions from water.[1,2] The process is often used for small supplies and in point-of-use treatment devices. The most commonly used resins have cross-linked polystyrene matrixes with sulfonate (SO_3^-) functional groups. They are used in either the sodium or the hydrogen form, although the sodium form is most common in drinking water treatment. The exhaustion reaction of the strong-acid resin involves a stoichiometric exchange of cations:

$$2RSO_3^-Na^+ + X^{2+} - (RSO_3^-)_2X^{2+} + 2Na^+ \quad (1)$$

in which R and X^{2+} represent the resin matrix and Ca^{2+} or Mg^{2+}, respectively. The regeneration reaction is the reverse of Eq 1,

$$(RSO_3^-)_2X^{2+} + 2NaCl - 2RSO_3^-Na^+ + X^{2+} \quad (2)$$

An excess of regenerant is required to force the regeneration reaction to take place. The capacity of the strong-acid resin depends not only on the number of functional groups on the resin, but also on the amount of regenerant applied. If low regenerant doses are used, a lower capacity will result because some sites are not converted back to the Na^+ form.[1]

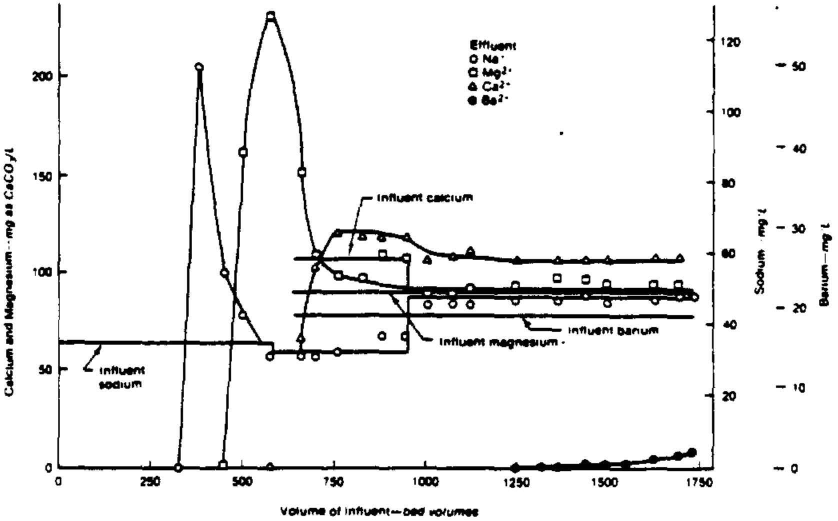

Figure 1. Breakthrough curves for Na^+, Mg^{2+}, Ca^{2+}, and Ba^{2+} for a column of virgin strong-acid resin in the hydrogen form

The addition of Na^+ to the product water (see Eq 1) is considered a disadvantage for some drinking water treatment applications.[3] If the addition of Na^+ is to be avoided, a weak-acid resin may be used in the H^+ form,[4] the strong-acid resin may be used in the H^+ form, or it may be necessary to use alternate sources of drinking water.

Radium (Ra^{2+}) and barium (Ba^{2+}) are alkaline earth metals found in water as divalent cations. Naturally occurring Ba^{2+} exceeds the maximum contaminant level (MCL) of 1 mg/L in some areas of northern Illinois, northeastern Iowa, and other locations.[5] In these same areas and in some parts of Florida, Texas, and South Carolina, the concentrations of ^{226}Ra plus ^{228}Ra exceed the MCL of 5 pCi/L.[5-7] The chemical behaviors of Ra^{2+} and Ba^{2+} are very similar and are much like those of Mg^{2+} and Ca^{2+}, the principal components of hardness in water. For example, these metals are removed well by softening, and there are some data that indicate that ion exchange is effective.[7-9]

Removal efficiency of Ba^{2+} at two full-scale ion exchange softening plants has been reported.[7] The first plant showed a reduction from 10 mg Ba^{2+}/L to less than 1 mg/L for 153 bed volumes (BV) of water processed, and the second showed a reduction from 19 mg Ba^{2+}/L to 1 mg/L or less for 100 bed volumes. The influent water at both plants contained 220–230 mg hardness as $CaCO_3$/L. But data obtained by Snoeyink et al[10] at Crystal

Lake, Ill., showed that the Ba^{2+} concentration in the ion exchange product water at one well site was sometimes in excess of the 1-mg/L MCL.

Brinck et al[9] sampled seven different ion exchange plants throughout Iowa and Illinois to determine their radium removal efficiency. The analyses showed average removals of ^{226}Ra to range between 81 and 97 percent for raw waters containing 3.25–45 pCi/L. Good radium removal was also observed to continue for a time after hardness had broken through. In other studies, Bennett[8] found 71–92 percent removal of ^{226}Ra at three plants in Illinois, and Singley et al[11] showed that the fraction of radium removed at operating plants was directly proportional to the fraction of total hardness removed.

Objectives and scope of research

The research discussed in this article was conducted to investigate and document the performance of conventional sodium-form strong-acid resins for the combined removal of barium, radium, and hardness under controlled, laboratory bench-scale operating conditions.

This work was undertaken for several reasons. Although others[7-9] have documented the effectiveness of several full-scale ion exchange softening facilities for removing barium and radium, premature leakage of barium into the softening column effluents has been reported.[12] The reason for this was not understood. A chemical company has also reported observing barium leakage from strong acid ion exchange columns.[13] Previous work on the use of conventional ion exchange softening resins in the sodium form for combined barium, radium, and hardness removal was based only on short studies of existing full-scale facilities. No controlled, laboratory bench-scale research work has been conducted to determine the factors that affect resin performance. The effect of the quantity of regenerant applied on effluent quality and on effluent quality as a function of the amount of water processed during a run also has not been documented.

Many of the public water supplies that exceed the MCLs for barium or radium are relatively small communities that use groundwater as the source of supply. Others[7,11] have shown conventional strong-acid sodium-form ion exchange softening to be a cost-effective way to remove barium and radium from groundwater, particularly for small facilities. Therefore, this research was conducted to determine how to apply strong-acid ion exchange resins more effectively for the removal of barium and radium to meet the MCLs and to help understand and solve problems similar to those likely to be encountered during full-scale operation.

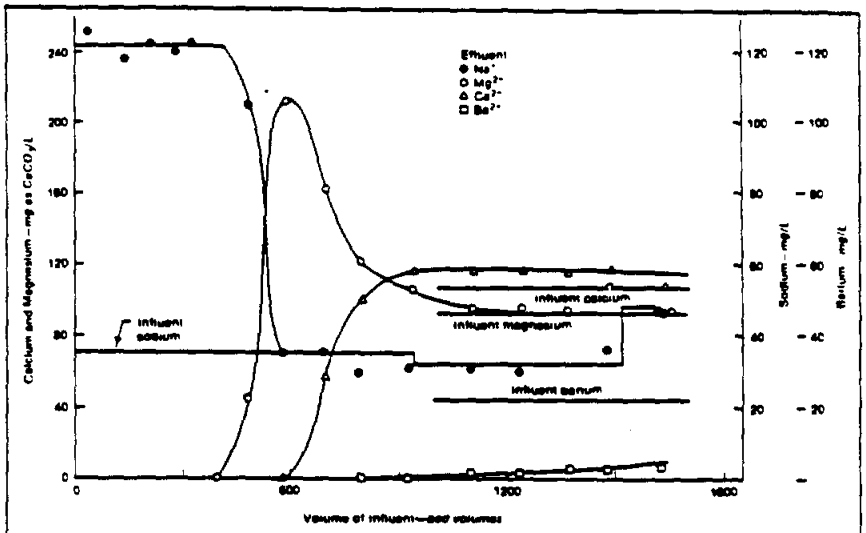

Figure 2. Breakthrough curves of Na^+, Mg^{2+}, Ca^{2+}, and Ba^{2+} for columns of virgin strong-acid resin in the sodium form

Materials and analytical methods

A strong-acid resin was used in all experiments.* The resin has a polystyrene-divinylbenzene cross-linked matrix with sulfonate functional groups. The capacity of this resin was measured to be 1.8 meq/wet mL (4.8 meq/dry g) by pH titration with base and also by isotherm batch tests.[10] The resin was shipped in the sodium form and was conditioned according to manufacturer's specifications. The mass of dry resin used in each experiment was determined by measuring the mass of centrifuged or filtered moist resin; the moisture content of a representative fraction was then determined and used to calculate the mass of dry resin. The test solutions contained 1.0 mM $CaCO_3$, 1.0 mM $MgCO_3$, 0.15 mM $BaCl_2$, 1.7 mM $NaHCO_3$, and 20 pCi ^{226}Ra/L. The solution thus contained 200 mg total hardness as $CaCO_3$/L, 285 mg total alkalinity as $CaCO_3$/L, and 20 mg Ba^{2+}/L.

An ion chromatograph† was used to measure the concentrations of magnesium, calcium, and barium. Atomic absorption and flame emission spectroscopy were used for some barium and sodium analyses. Alkalinity and some hardness measurements were made according to *Standard Methods*.[14]

Radium analyses were made by the Environmental Research Laboratory (ERL) at the University of Illinois at Urbana and the University Hygienic Laboratory (UHL) at the University of Iowa at Iowa City. The ERL used a radon emanation method;[15] the UHL, a technique involving coprecipitation of ^{226}Ra with $BaSO_4$ and alpha counting with an internal proportional counter.[16]

Experimental procedures

Glass columns 75 cm long with 2.5-cm-ID were used. The columns had stopcocks on each end and the 62-cm resin beds were supported by heat-resistant glass wool plugs. This bed depth is typical of the bed depth used in practice. Influent water was pumped‡ from a 230-gal (871-L) stainless-steel covered tank through vinyl tubing. The same pumps with PTFE tubing were used to pump regenerant solutions and deionized rinse water. The strong-acid resin was used in both the sodium and the hydrogen form.

Results and discussion

Column tests were made using resin in the hydrogen form and the sodium form to show effluent quality as a function of the amount of water processed. Influent was applied to the column with virgin hydrogen form resin at 4.31 BV/h (21.4 mL/min, 1.1 gpm/sq ft [5 × 10^{-4} mm/s]) for the first 263 BV and at 2.60 BV/h (12.9 mL/min, 0.65 gpm/sq ft [3 × 10^{-4} mm/s]) for the completion of the run. The influent was applied to the virgin sodium-form resin at 5.01 BV/h (24.8 mL/min, 1.2 gpm/sq ft [5.6 × 10^{-4} mm/s]). All resin volumes were measured with the resin wet and completely in the hydrogen form.

Breakthrough curves for the hydrogen-form resin column are shown in Figure 1, and those for the sodium-form resin column in Figure 2. Chromatographic displacement of the ions was observed in the reverse order of their position in the selectivity series.[2]

$$Ba^{2+} > Ca^{2+} > Mg^{2+} > Na^{+} > H^{+}$$

Sodium appeared first in the effluent of the hydrogen-form resin column and

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*Duolite C-20, Diamond Shamrock, Redwood City, Calif

†Autoion System 12 analyzer, Dionex Corp., Sunnyvale, Calif.

‡Fluid Metering Inc., Oyster Bay, N.Y.

its maximum concentration of 115 mg/L was more than three times the influent concentration. Magnesium was then eluted and reached a peak concentration of more than two times the influent value. Magnesium was followed by Ca^{2+}, and much later by Ba^{2+}. The exchange of ions in solution for the H^+ on the resin,

$$n\,R\text{-}SO_3^-H^+ + X^{n+} \rightarrow (R\text{-}SO_3^-)_n X^{n+} + nH^+ \quad (3)$$

in which X^{n+} represents Na^+, Ca^{2+}, Mg^{2+}, and Ba^{2+}, resulted in a solution pH that was less than 3.5 until the alkalinity broke through at about 300 BV. This H^+ would have to be neutralized and the CO_2 produced from neutralization of the alkalinity stripped if the column were used in this form. More details concerning the use of the H^+ form column are given elsewhere.[10] Similar results were observed for the sodium-form column, except that the Na^+ concentration instead of the H^+ was initially high (about four times the influent value).

Barium ion was also the most difficult to remove from the resin because of its selectivity. The hydrogen-form column was regenerated with 8.29 meq/mL (21.7 meq/g, 18.6 lb HCl/cu ft resin [8.4 kg HCl/m^3 resin], 3.66 BV) of an 8 percent HCl solution at 1.64 BV/h (8.2 mL/min, 0.4 gpm/sq ft [1.8×10^{-4} mm/s]). The sodium-form column was regenerated with 8.09 meq/mL (21.7 meq/g, 29.9 lb NaCl/cu ft resin [13.6 kg $NaCl/m^3$ resin], 3.23 BV) of a 13.5 percent NaCl solution at 1.64 BV/h (8.1 mL/min, 0.4 gpm/sq ft [1.8×10^{-4} mm/s]). The regenerant contact time was 2 h 14 min for the hydrogen-form column and 1 h 58 min for the sodium-form column. The regeneration curves for the hydrogen-form column and the sodium-form column are shown in Figures 3 and 4, respectively.

Calcium and Mg^{2+} are removed most effectively by the HCl during the first portion of the regeneration. At a regenerant dose equal to the number of equivalents of divalent cations removed by the resin (at line A), 50 percent of all ions were removed (based on the total amount eventually removed from the resin during regeneration); 75 percent of the Mg^{2+}, 51 percent of the Ca^{2+}, and only 15 percent of the Ba^{2+} were removed. After regenerant equal to three times the equivalents of divalent cations (Figure 3, line C) was applied, nearly all the Ca^{2+} and Mg^{2+} were removed from the resin, but more than half the Ba^{2+} remained.

The NaCl regeneration curves are similar to those obtained for the HCl; about 65 percent of all ions were removed with a NaCl dose equal to the number of equivalents of divalent ions removed by the resin (Figure 4, line A). When NaCl equal to three times the equivalents of divalent cations was applied, essentially all Ca^{2+} and Mg^{2+} was removed but more than 50 percent of the Ba^{2+} still remained.

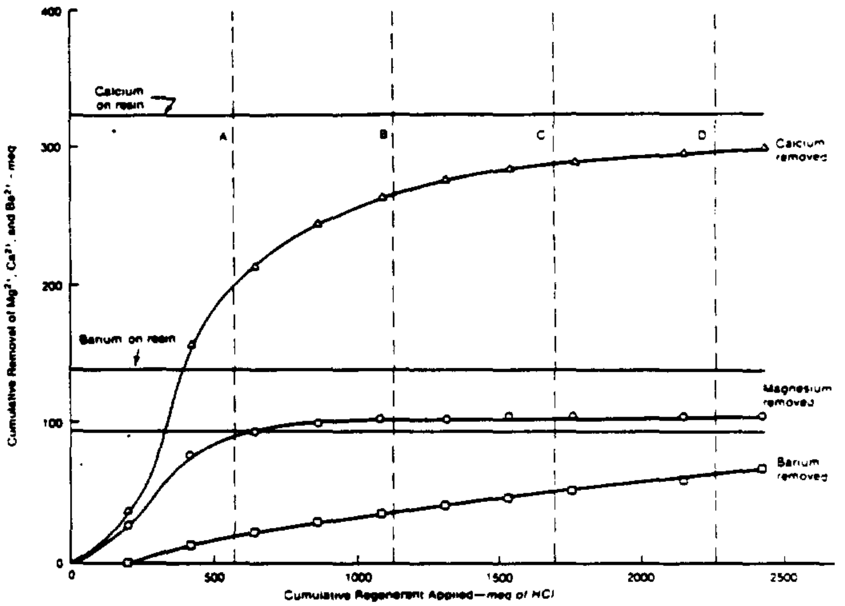


Figure 3. Acid regeneration of virgin strong-acid resin *(at A, B, C, and D, respectively, the amount of regenerant applied is 1, 2, 3, and 4 times the number of equivalents of divalent cations removed by the resin)*

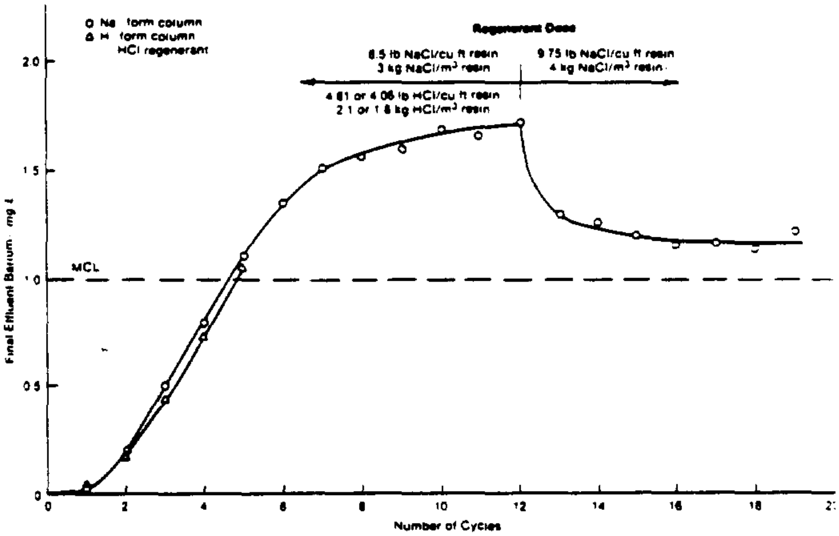
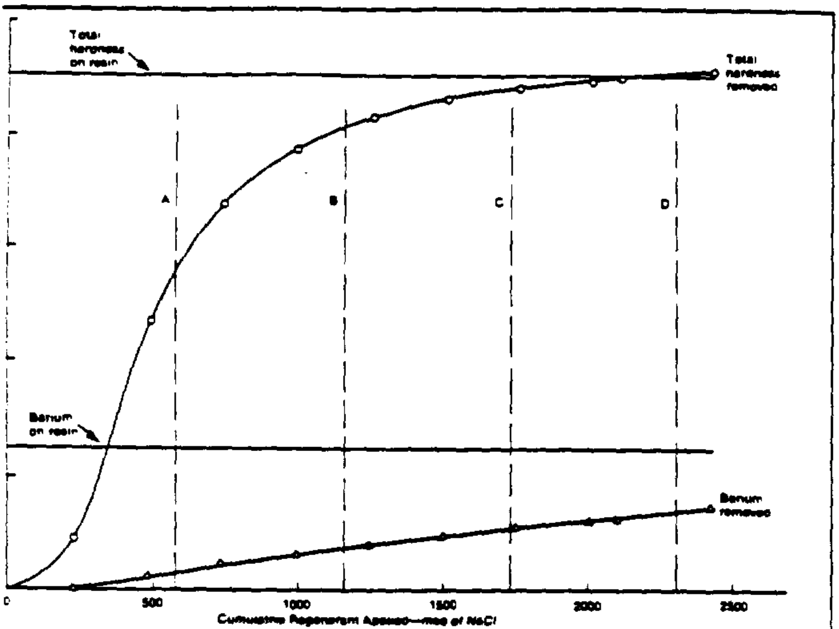


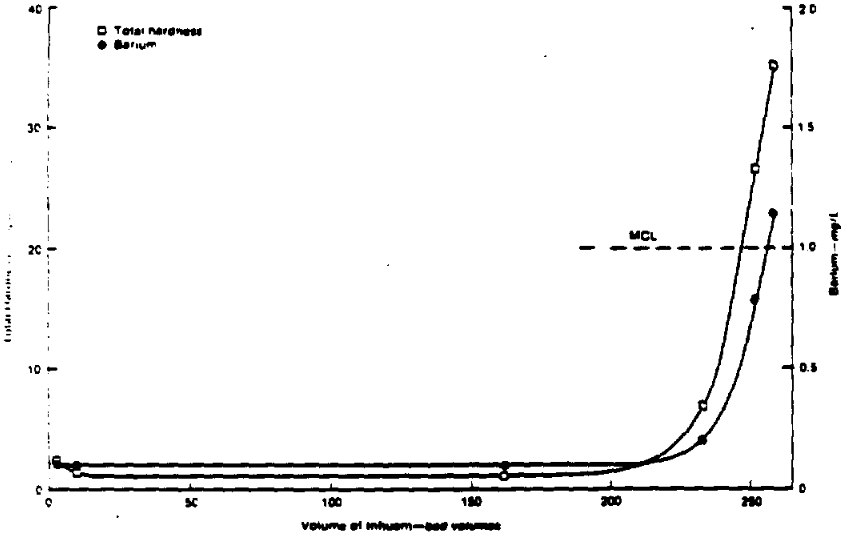
Figure 5. Barium effluent concentration from strong-acid resin exhaustion–regeneration cyclic studies *(barium values shown are for effluent hardness of 40 mg as $CaCO_3$/L)*

A large regenerant dose is required to regain the full capacity of this resin. In municipal softening operations, complete regeneration is not practiced. Instead, resin capacity is often sacrificed to achieve more efficient use of regenerant.[1] Municipalities often operate with a regenerant dose of approximately one times the capacity of the resin (or one times the equivalents of divalent ions on the resin) so that about 50–60 percent removal of the ions is obtained (according to the results shown in Figures 3 and 4). The capacity of the resin available in a subsequent production run is then limited by cations still present on the resin. The virgin resin results also indicate that with only partial regeneration of a resin, Ba^{2+} is likely to selectively accumulate on the resin over successive exhaustion–regeneration cycles. This condition will reduce the amount of water that can be processed before hardness and barium breakthrough.

Cyclic exhaustion–regeneration studies were conducted (1) to determine whether a quantity of regenerant in excess of that typically used would be required to hold the Ba^{2+} concentration at an acceptable level, (2) to document the extent of leakage during a column run, and (3) to determine whether the



ure 4. Sodium chloride regeneration of virgin strong-acid resin *(at A, B, C, and respectively, the amount of regenerant applied is 1, 2, 3, and 4 times the number of uivalents of divalent ions removed by the resin)*



igure 6. Breakthrough curves for the sixth exhaustion cycle of the strong-acid esin

bed volumes processed per cycle would change as the number of cycles increased. Both a hydrogen-form column with HCl regenerant and a sodium-form column with NaCl regenerant were used.

The influent feed was prepared in a 230-gal (871-L) stainless-steel tank. Effluent samples were taken throughout the exhaustion run for two cycles to determine the extent of leakage. For the other cycles, samples were taken near the end of the run to document breakthrough behavior. The column was backwashed prior to regeneration. Cocurrent downflow regeneration with 13.5 percent NaCl was used with an application rate of 1.7 BV/h (8.5 mL/min, 0.4 gpm/sq ft [1.8×10^{-4} mm/s]). Cycles 1-11 used 1.8 meq NaCl/mL resin (4.7 meq/g; 6.5 lb NaCl/cu ft resin [3 kg NaCl/m^3 resin]) and cycles 12-19 used 2.7 meq NaCl/mL resin (7.1 meq/g; 9.75 lb NaCl/cu ft resin [4 kg NaCl/m^3 resin]). The regenerant contact time for cycles 1-11 was 24.9 min, and for cycles 12-19 was 37.4 min.

Each exhaustion run was terminated when the effluent hardness reached 40 mg as CaCO$_3$/L. The effluent Ba^{2+} concentrations of both the sodium- and hydrogen-form resin columns are shown in Figure 5. The hydrogen-form column run was terminated after five cycles because performance was similar to the sodium-form column. The effluent Ba^{2+} concentration leveled off at 1.7-1.75 mg/L for the cycles in which 1.8 meq NaCl/mL resin (6.5 lb NaCl/cu ft resin [3 kg NaCl/m^3 resin]) were used to regenerate the sodium-form resin column. The effluent Ba^{2+} concentration dropped to about 1.2 mg/L at the end of each cycle after the regenerant dose was increased by 50 percent to 2.7 meq NaCl/mL resin (9.75 lb NaCl/cu ft resin [4 kg NaCl/m^3 resin]). These results show that Ba^{2+} does accumulate on the resin with successive exhaustion–regeneration cycles. In the tenth and eleventh cycles, only 58 percent of the maximum capacity of the resin was being used. The regeneration efficiency (defined as the number of equivalents of divalent ions removed per equivalent of NaCl applied times 100) of these two cycles was about 58 percent, and the average total dissolved solids (TDS) of the 2.4 BV of waste brine produced per 100 BV of product water was 16 400 mg/L. After increasing the regenerant dose to 2.7 meq NaCl/mL resin (9.75 lb NaCl/cu ft resin [4 kg NaCl/m^3 resin]), however, the regeneration efficiency decreased to 46 percent, the percent of maximum capacity employed increased to 68 percent, and the waste brine produced was 2.1 BV per 100 BV of product water with a TDS of 25 200 mg/L.

The hardness and Ba^{2+} breakthrough curves shown in Figure 6 were determined during the sixth exhaustion of the sodium-form resin column, with 1.8 meq NaCl/mL resin (6.5 lb NaCl/cu ft resin [3 kg NaCl/m^3 resin]) applied as regenerant. The leakage of Ba^{2+} during most of the cycle was approximately 0.10 mg/L, and low total hardness leakage also was observed. Barium and total hardness broke through together at approximately 225 BV, whereas breakthrough of Ba^{2+} from the virgin resin column occurred 600–800 BV after the hardness. The earlier breakthrough is consistent with Ba^{2+} accumulation on the resin with increasing numbers of cycles. When hardness broke through during the virgin resin column run, about 0.15 meq Ba^{2+}/mL (0.4 meq/dry g) was on the resin, but after several cycles with 1.8 meq NaCl/mL resin (6.5 lb NaCl/cu ft resin [3 kg NaCl/m^3 resin]) regenerant, this amount had increased to 0.38 meq/mL (1.0 meq/dry g). No further accumulation was indicated after several cycles because the concentration of Ba^{2+} at hardness breakthrough reached a constant value. At steady-state, only about 0.075 meq Ba^{2+}/mL (0.2 meq/dry g) was removed from the resin during regeneration with 1.8 meq NaCl/mL resin (6.5 lb NaCl/cu ft resin [3 kg NaCl/m^3 resin]). Effluent samples taken during the ninth, tenth, and eleventh cycles for this column continued to show

Ba^{2+} leakage of less than about 0.15 mg/L during most of each cycle. The breakthrough curves determined during the nineteenth cycle, when the regenerant dose was 2.7 meq NaCl/mL resin (9.75 lb NaCl/cu ft resin [4 kg NaCl/m^3 resin]), were very similar to those shown in Figure 6. The leakage of Ba^{2+} was between 0.10 mg/L and 0.20 mg/L for the entire run. Hardness and Ba^{2+} did break through somewhat later, at 260–270 BV, because a higher percentage of the resin capacity was being used.

Although Ba^{2+} clearly accumulated on the resin over several cycles, until a certain equilibrium with the regenerant dose and other operating conditions were achieved, the number of bed volumes to hardness breakthrough of 40 mg as $CaCO_3$/L remained constant throughout the cyclic studies for each regenerant dose (excluding the first cycle when the resin was in virgin form). Barium did not use up enough capacity on the resin to detrimentally affect hardness removal, although it did accumulate enough to affect Ba^{2+} removal efficiency.

Radium was also present in the influent solution for both the virgin resin column test and the cyclic run experiment. For the virgin resin, the influent ^{226}Ra averaged 18.5 pCi/L; Table 1 lists the effluent ^{226}Ra, total hardness, Ba^{2+}, and Na^+ concentrations. The data show that ^{226}Ra removal continued long after saturation of the column with hardness ions. The detection limit for the sample size used for this test was 1.0 pCi/L.

Larger samples were collected for the cyclic run studies to lower the ^{226}Ra detection limit to 0.1 pCi/L. All effluent values measured were near the detection limit as shown in Table 2. Not all the ^{226}Ra applied to the resin was removed during regeneration, but the accumulation per cycle decreased with each successive cycle (Table 3). After 11 cycles at a regenerant dose of 1.8 meq NaCl/mL resin (6.5 lb NaCl/cu ft resin [3 kg NaCl/m^3 resin]), almost 85 percent of the ^{226}Ra adsorbed during the eleventh exhaustion cycle was removed during regeneration. After eight additional cycles at a regenerant dose of 2.7 meq NaCl/mL resin (9.75 lb NaCl/cu ft resin [4 kg NaCl/m^3 resin]), close to 95 percent of the radium adsorbed during the last (nineteenth) cycle was removed by regeneration. The small accumulation of ^{226}Ra on the column will not affect resin capacity because only 1.026×10^{-12} g of radium will give a count of 1 pCi/L. Thus, less than 1 percent of the resin capacity would be occupied by ^{226}Ra even if all of the radium applied during the 19 cycles were retained on the resin. Because radium does accumulate, though, the radiation from resins in operating columns should be studied more closely to ensure no adverse impact on the health of those who handle and dispose of spent resins. Bennett[8] documented the buildup of radium on strong-acid ion exchange resins in several full-scale municipal water treatment plants in Illinois.

TABLE 1

Radium removal by virgin resin

Influent Applied BV	Effluent ^{226}Ra *pCi/L*	Effluent Na^+ *mg/L*	Effluent Hardness *mg as $CaCO_3$/L*	Effluent Ba^{2+} *mg/L*
25	1.4	0	0	0
380	2.1	112	0	0
881	2.3	36	214	0
1121	2.0	46	200	0
1622	2.7	47	201	1.0

TABLE 2

Radium removal during the cyclic test

Cycle	Influent Applied BV	^{226}Ra *pCi/L*	Total Hardness *mg as $CaCO_3$/L*
6			
Influent		17.7	199
Effluent	6	0.1	1.7
Effluent	164	0.3	1.2
Effluent	254	0.3	30
Effluent (composite)	259	0.1	1.6
11			
Influent		18.2	216
Effluent	27	<0.1	0.8
Effluent	182	<0.1	0.8
Effluent	248	3.0	23
Effluent (composite)	255	0.1	1.7
19			
Influent		18.4	200
Effluent	5	0.2	0.4
Effluent	216	<0.1	0.4
Effluent	261	0.1	4.6
Effluent	287	0.7	26
Effluent (composite)	295	<0.1	1.6

TABLE 3

Radium accumulation on the resin during the cyclic test

Cycle	Waste Brine ^{226}Ra *pCi/L*	Waste Brine Produced BV/100 BV Product Water	Percent ^{226}Ra Removed During Regeneration
4	274	2.30	
6	418	2.37	56.0
11	639	2.41	84.8
19	846	2.05	94.4

The experiments conducted as a part of this research did not examine how different concentrations of hardness in the influent water would affect the removal of barium and radium. It is expected that higher concentrations of hardness with respect to barium and radium would shorten the bed volumes of water processed to hardness breakthrough. Lower concentrations of hardness with respect to barium and radium than those used in these experiments would increase the bed volumes of water processed to hardness breakthrough and to barium and radium breakthrough.

Improved efficiency of countercurrent regeneration over cocurrent regeneration for hardness (Ca^{2+} and Mg^{2+}) removal has been documented.[2] Countercurrent regeneration is not widely practiced in municipal water softening because the equipment and operational control are not as simple as for cocurrent regeneration. Cocurrent regeneration was used in these experiments so that the investigations represent the most commonly used mode of ion exchange operation. The effect of countercurrent regeneration on barium and radium removal would need to be investigated, but it is not apparent that improved removal efficiency would result because the leakage immediately after regeneration did not exceed the MCLs.

Conventional strong-acid sodium ion exchange is used in the city of Crystal Lake, Ill., to remove hardness and Ba^{2+} from its groundwater supply. The community obtains its water from wells and treats this water at three sites. Samples from the three sites showed that Ba^{2+} and hardness removal were more of a problem at site 8 than at the other two sites. The effluent immediately after regeneration contained a hardness level of 21.7 mg as $CaCO_3$/L and a Ba^{2+} level of 0.4 mg/L. The water quality just prior to regeneration was considerably worse.

Acknowledgment

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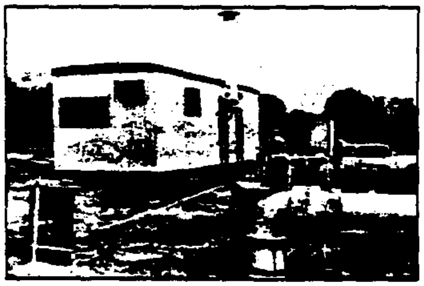
Applying Membrane Processes to Groundwater Sources for Trihalomethane Precursor Control

James S. Taylor, D.M. Thompson, and J. Keith Carswell

The use of membrane processes for controlling trihalomethanes (THMs) was investigated for Florida groundwater sources and one surface water source. All of the sources were used for public supply and had excessive THMs (>300 μg/L) in the finished water. The performance and projected cost of the membrane system using groundwater sources are reported in this article.



The pilot-scale trailer used in this study was designed to be self-contained and transportable.

The permeate water quality from any membrane process can be expressed mathematically by solvent and solute fluxes as shown in Table 1.[1,2] The equations indicate that water (solvent) flux through a membrane will increase as the feed pressure increases and decrease as the difference in osmotic pressure between the feed and the permeate increases. The solute flux of any dissolved material (total dissolved solids [TDS], trihalomethane [THM] precursors, dissolved organic carbon [DOC]) increases as the concentration gradient between the feed and the permeate increases. The recovery in a membrane process is the percentage of product relative to feed, as shown in Eq 3 of Table 1.[2] The product stream could be the permeate or the reject stream, depending on the application.

Separation of the feed stream into reject and permeate streams occurs continuously along the surface of the membrane. Because membranes tend to pass water and reject most dissolved solids, the membrane feed is continuously concentrated and permeate water quality is decreased as product recovery is increased. The osmotic pressure gradient also increases as recovery is increased, which reduces the water flux and further reduces permeate water quality. Historically, water plants utiliz-

United States
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Project Summary

Manganese Dioxide Coated Filters for Removing Radium from Drinking Water

Marc Y. Menetrez, David G. Anderson, and Edward P. Stahel

Research was performed using manganese dioxide (MnO_2) to demonstrate that above pH 3 cations are adsorbed from solution in the order of their affinity, and that the interaction is characterized by the pH dependence of the metal. The relationship of the zero point charge of pH and the solution ionic strength effects on interfacial surface potential and adsorption have been addressed. Characteristics of MnO_2 behavior, structure, and stability found in research investigation were reviewed.

Most of this study was on the use of MnO_2 coated filters for the removal of radium. A few comparison tests on radium removal with ion exchange were also made. Specifically, these tests have shown that acrylic fiber filters coated with MnO_2 will remove radium from water. For a high hardness water with pH = 7.4, total radium removal was 14,200 pCi/g MnO_2 before the MCL of 5 pCi/L was exceeded; and for a low hardness water with pH = 4.5, total radium removal was 5,000 pCi/g MnO_2 before the MCL of 5 pCi/L was exceeded. Hardness passed through the MnO_2 filters with little change; therefore, radium was highly preferred over hardness.

A step-by-step process for the preparation of acrylic fiber filters coated with MnO_2 is included in the full report.

This Project Summary was developed by EPA's Risk Reduction Engineering Laboratory, Cincinnati, OH, to announce key findings of the research project that is fully documented in a separate report of the same title (see Project Report ordering information at back).

Introduction

As of August 1, 1982, a total of 39 public water systems, or 1.9% of the systems tested in North Carolina, were found to be in violation of the 5 pCi/L combined radium limit. When a water system is found to have levels of combined radium activity that exceed the standard, the system owner is responsible for correcting the problem. If utilizing another water source is not realistic, then effective treatment to remove excess radium would be required. Ion exchange, lime-soda softening, and reverse osmosis have been demonstrated to be effective techniques for radium removal.

Adsorption onto MnO_2 coated filters is a treatment alternative that has been tested sparingly but never used to remove radium from drinking water. Extensive testing of MnO_2 filters in the laboratory, in the pilot plant, and in full-scale application is the focus of this report. The use of MnO_2 coated filters has been examined for metal removal and more extensively for radium removal in conjunction with efficiency, cost, safe use of application, and usefulness compared with other forms of treatment.

Flowthrough MnO_2 Filter Preparation System

A system to produce MnO_2 filters with a heavy MnO_2 loading was designed, constructed, and operated successfully. The flowthrough filter preparation system is described in detail in the full report. The system has been successfully used to generate MnO_2 fiber in both a prewoven filter element form as well as a loose-staple form. The acrylic-based MnO_2 fiber in both forms exceeded the previous limit of 10% to 13% MnO_2 (by weight). Loadings of 20% to 25% MnO_2 (by weight) were consistently produced with the flowthrough filter preparation system. A filter element 10 in. (25.4 cm) long weighing an average of 250 g with an average loading of 22.5% would consist of 56.25 g MnO_2 and 193.75 g acrylic-base fiber. The flowthrough filter preparation system was also used in washing the prepared fiber before packaging. Preparing MnO_2 fiber with the system took less than 1 day/batch to produce the filters. The cost of chemical was less than $2/10-in. filter element, or less than $8/kg of prepared fiber.

The examination of the prepared MnO_2 filter elements revealed that when the MnO_2 filter is installed and rinsed (with 3 to 5 filter volumes), no organic, soluble manganese, insoluble MnO_2, or fiber is released to the effluent stream. This indicates that MnO_2 fiber can be used to treat drinking water without contributing unwanted contaminants. The State of North Carolina, Department of Human Resources, Solid and Hazardous Waste Branch, examined the MnO_2 fiber for leaching under landfill conditions. The results of the (EPA) toxicity test revealed no leaching of any form of manganese or other inorganics at pH 5 conditions. This should allow the disposal of the MnO_2 filters in landfills in North Carolina, pending approval for radium.

Radium Analysis

A modified process for analyzing radium in water was devised and structured, and the results compared with EPA through a quarterly sample analysis and report program. The analysis procedure followed is described in Appendix B of the full report.

Results

Bleed Stream Testing

The bleed stream testing of the MnO_2 filter in the Highland Park water system and the Gateway Mobile Home Park water system revealed that radium was removed and that hardness was passed through the filter relatively unchanged.

The bleed stream results for the Highland Park water system (Table 1) show that total radium removal in excess of 86% can be expected for a treated volume of at least 22,653 L at a flowrate of 11.36 L/m per 25.4-cm filter element. At this level of radium removal, the influent concentration of 36.4 pCi/L total radium is reduced to a maximum effluent concentration of 5 pCi/L total radium, the U.S. EPA Regulation. The single, 25.4-cm filter element consisting of approximately 250 g of MnO_2 fiber removed 0.801 µCi total radium before decreasing to an efficiency of less than 86%. At an MnO_2-to-fiber loading of 22.5% (56.25 g of MnO_2), the total radium-to-MnO_2 ratio of 14,200 pCi/g MnO_2 was demonstrated prior to reaching the 5 pCi/L limit. The halfway point to breakthrough (or the point at which 50% efficiency was reached) occurred after approximately 57,000 L of water was treated, and the total breakthrough of radium occurred after approximately 160,000 L of water was treated.

Bench-top testing of a sample from the Highland Park water system was performed. The 202.5-L sample was pumped through 15 g of MnO_2 fiber containing approximately 3.35 g of MnO_2, at a rate of 0.75 L/min. The results indicated that the influent concentration of 36.4 pCi/L was reduced to zero in the effluent. A total of 7,370 pCi of total radium was removed, or 2,180 pCi of total radium per gram of MnO_2. The removal efficiency never changed from 100%, therefore, no conclusions can be made regarding total uptake of radium. However, a considerable volume of water was treated while maintaining total removal.

The bleed stream results for the Gateway Mobile Home Park water system (Table 2) show that total radium removal in excess of 66% can be expected for a treated volume of at least 28,000 L at a flowrate of 11.4 L/m per 25.4-cm filter element. At this level of radium removal, the influent concentration of 13.2 pCi/L total radium is reduced to a maximum effluent concentration of 5 pCi/L total radium, the U.S. EPA Regulation. The single, 25.4-cm filter element consisting of approximately 250 g of MnO_2 fiber removed 0.222 µCi total radium before decreasing to an efficiency of less than 66%. At a MnO_2-to-fiber loading of 22.5% (56.25 g of MnO_2), the total radium-to-MnO_2 ratio of 5,000 pCi/g MnO_2 was demonstrated prior to reaching the 5 pCi/L limit. The halfway point to breakthrough (or 50% efficiency) occurred after approximately 42,000 L of water was treated, and the total breakthrough of radium occurred after approximately 85,000 L of water was treated.

Table 1. *Highland Park Field Study; Bleed Stream Field Test – Total Radium Removal Efficiency and Uptake**

Effluent Radium (pCi/L)	*Total Flow† (ft^3)*	*Removal Efficiency*	*Total Uptake (nCi)*	*Uptake (nCi/g MnO_2)*
2.4	500	0.934	481.39	8.56
5.5	1,000	0.849	918.89	16.34
14.5	1,500	0.602	1,228.96	21.85
17	2,000	0.533	1,503.63	26.73
22	2,500	0.396	1,707.52	30.36
27.5	3,000	0.245	1,833.53	32.60
30	3,500	0.176	1,924.14	34.21
31.5	4,000	0.135	1,993.52	35.44
32.5	4,500	0.107	2,048.73	36.42
34.5	5,000	0.052	2,075.64	36.90

** For 25.4-cm long filter element, influent total radium concentration = 36.4 pCi/L, pH = 7.4, and total hardness = 227 mg/L as $CaCO_3$.*

† Multiply by 28.32 to convert ft^3 to liters.

Table 2. *Gateway Field Study; Total Radium Removal Efficiency and Uptake**

Effluent Radium (pCi/L)	Total Flow† (ft^3)	Removal Efficiency	Total Ra Uptake (nCi)	Uptake (nCi/g MnO_2)
2.0	500	0.848	158.58	2.82
4.5	1,000	0.659	281.75	5.01
7.0	1,500	0.470	369.54	6.57
9.0	2,000	0.318	429.00	7.63
12.5	2,500	0.053	438.91	7.80
13.2	3,000	0.000	438.91	7.80

* *For 25.4-cm long filter element, influent total radium concentration = 13.2 pCi/L, pH = 4.5, and total hardness = 23 mg/L as $CaCO_3$.*

† *Multiply by 28.32 to convert ft^3 to liters.*

In-Line Testing

In-line field testing utilized three standard water filtration housings. Each housing held twenty-one 25.4-cm filter elements. The three stainless steel housings were situated in series to extend filter life. The in-line field test results for the Highland Park water system indicated that the removal efficiency of total radium was less than that exhibited with the bleed stream field test. The point halfway to breakthrough of total radium was reached almost immediately for the three-filter housing system containing three canisters of twenty-one 25.4-cm MnO_2 fiber elements. This rapid decrease in filter efficiency was also recorded for the three filter housing systems containing one 20-µm, one 5-µm, and one MnO_2 filter element housing. The decreased capacity of the MnO_2 filters was believed to be caused by a high suspended solids loading on the filters. Large amounts of clay and silt were found coating the filter elements.

This fouling of the MnO_2 sites greatly decreased the ability of the filter to adsorb radium. Even after passing through a 20-µm and a 5-µm pleated paper filter element, significant amounts of fine suspended material were found building on the MnO_2 filter element. The reduced capacity of the in-line field test apparatus is believed to be caused by an improperly drilled and cased well. This problem was not encountered with the bleed stream testing and only became apparent when treating the entire well flow of 125 L/m and 75,700 L/day.

Conclusions

The successful removal of radium with MnO_2 coated filters appears conditional to the lack of fouling agents in the influent stream. Coating of the MnO_2 sites greatly reduces the interfacial attracting forces. Maximum adsorption efficiency is expected when only the target ion in solution is present. The bleed stream tests resulted in greater amounts of radium loading for the Highland Park water system and lower radium loading for the Gateway water system. This is believed to be caused by the different levels of pH of these two ground water sources. The pH 7.4 of the Highland Park water system allowed for a higher degree of adsorption than did the pH 4.5 of the Gateway water system.

In-lab, bench-top investigation on the removal of dissolved metals has indicated that radium ions as well as interfering ions in solution are adsorbed by MnO_2. The results of metal removal with MnO_2 fiber showed that low concentrations of cadmium, calcium, cobalt, cesium, iron, and manganese can also be removed from solution by adsorption onto MnO_2 fiber.

The ion exchange comparison column bleed stream results were inconclusive in that they lacked actual breakthrough information. A column of MnO_2 fiber 152.4-cm length and 5.08-cm diameter treated approximately 76,000 L of influent water containing 13.2 pCi/L total radium without any variation from an effluent level of 0 pCi/L total radium. A column of ion exchange resin 60.96-cm length and 5.08-cm diameter treated 76,000 L of water containing 13.2 pCi/L total radium, without any significant variation from 0 pCi/L total radium in the effluent. At this point the ion exchange column was regenerated with a 0.85 molar calcium chloride solution because the resin was mistakenly expected to be spent. An additional 78,000 L of water was treated with this column. Again, no variation from zero total radium discharge was observed. No conclusions could therefore be made regarding column life, efficiency, or comparisons of hydrogen ion exchange to calcium ion exchange.

The MnO_2 fiber can be used for treatment of drinking water for the removal of radium. Tests also showed that MnO_2 adsorbs other metals, specifically cadmium, calcium, cesium, cobalt, iron, and manganese. The bleed stream tests showed that radium was highly preferred over calcium and magnesium. A treatment application for radium involving an inability to regenerate resin because of the disposal of regenerant brine or backwash could eliminate the use of ion exchange resin. In these circumstances, MnO_2 coated filters could possibly be used for treatment and the filter disposed of in a sanitary landfill (pending approval by state authorities).

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Project Summary

A Study of Possible Economical Ways of Removing Radium from Drinking Water

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A study was undertaken to determine variables that control the incidental removal of radium observed to occur as a consequence of treatment to remove iron by oxidation-sand filtration. This study also evaluated the possibility of exploiting these factors to provide an inexpensive means of removing radium using existing or modified iron removal facilities. Emphasis was placed on the use of aeration to oxidize soluble iron. The initial objective was to determine how water chemistry influences ^{226}Ra sorption to iron oxides produced by aeration. Studies were also conducted to evaluate radium sorption to hydrous manganese oxides produced by permanganate oxidation. A latter objective was to evaluate the potential of exploiting sorption to filter sand as a novel removal technology. Batch and pilot plant studies were conducted in the laboratory and in the field at a city whose supply contains excessive radium.

Sorption of iron and manganese oxides and filter sand appears to be controlled primarily by the presence of calcium and magnesium, which are believed to compete for sorption sites. Excessive pH values would need to be used to obtain significant sorption to iron oxides at concentrations typical of natural waters. Removals obtained by freshly precipitated hydrous manganese oxides in batch studies were much greater than those obtained in systems containing only iron oxides or mixtures of iron and manganese oxides produced by the oxidation of ferrous iron by potassium permanganate. This suggests that sorption to manganese oxides could possibly be exploited to remove radium if iron did not interfere. The presence of iron may limit the utility of permanganate oxidation of iron as a means to reduce radium.

Filter sand has a potential capacity to sorb significant concentrations of radium at typical hardness concentrations if the capacity is maintained by periodically rinsing the sand with a dilute acid. Removal efficiencies of approximately 80% to 90% could be achieved in laboratory and field studies using a 61-cm (2-ft) deep sand bed at conventional loading rates when daily rinsing with a dilute acid was practiced.

This Project Summary was developed by EPA's Water Engineering Research Laboratory, Cincinnati, OH, to announce key findings of the research project that is fully documented in a separate report of the same title (see Project Report ordering information at back).

Introduction

Radium in drinking water is a concern because of suspected detrimental health effects, primarily the formation of cancers. Concentrations exceeding the current EPA-mandated maximum contaminant level (MCL) of 5 pCi/L total radium (^{226}Ra + ^{228}Ra) have been observed in several areas, the most notable occurrences in the United States being in Florida, North Carolina, Virginia, the New England states, the uranium mining areas of the mountain states, and in the midwestern states of Iowa, Illinois, Wisconsin, Missouri, and Minnesota.

It has been estimated that 500 municipal supplies may contain excessive radium. Lucas reported in 1985 approximately 1.4 million people in a total of 177 cities in the midwest alone are delivered a water containing ^{226}Ra in excess of 3 pCi/L (the 1962 U.S. Public Health Service drinking water standard and the value above which mandatory analysis for ^{228}Ra is required). Radium concentrations in the range of 5 to 20 pCi/L are typical of the majority of these supplies although higher levels (up to nearly 50 pCi/L) have been reported. In many cases removal efficiencies in the range of 50% to 75% would be adequate to put these waters into compliance.

Several conventional water treatment practices bring about the removal of radium. Sorg and Logsdon have pointed out that the two most efficient methods are sodium ion exchange and lime-soda softening, each generally removing about 85% to 95% of the influent radium. In those processes radium removal is only incidental to other changes in water quality that may not be needed or even desired. For example, sodium ion exchange causes an increase in sodium concentration, which has been associated with an increased risk of heart disease. In particular, conventional processes are not suited to the many small communities that have excessive radium. Other removal technologies have also been investigated but to date no processes to remove radium alone are in widespread use.

Many supplies having high radium also have unacceptably high concentrations of iron and sometimes manganese that must be removed, commonly through processes involving oxidation and sand filtration. It has been observed that some radium removal occurs during iron removal treatment, presumably by sorption/coprecipitation with the hydrous metal oxides. Radium, like other alkaline earth metals, probably exists in water as a divalent cation and has been shown to sorb to many types of materials. For example, sorption to glass is a common sampling problem that is overcome by acidification. An understanding of the factors that control radium removal in iron removal processes could possibly aid in the development of inexpensive radium removal methods based largely on the use of existing facilities.

The primary objective of this study was to determine what factors control the incidental removal of radium occurring in iron removal plants typical of those operating in the midwest utilizing aeration and sand filtration. The study also evaluated the possibility of exploiting these factors as an inexpensive means of removing radium from drinking water using existing iron removal facilities. The initial focus was on modification of the water chemistry expected to affect sorption to iron oxides produced by aeration. Studies were also conducted to evaluate radium sorption to hydrous manganese oxides produced by permanganate oxidation. A latter objective was to evaluate the potential of exploiting sorption to filter sand as a novel removal technology.

Tasks included (1) batch studies of the effect of variable water chemistry on sorption of ^{226}Ra to hydrous iron and manganese oxides, mixtures of iron and manganese oxides coproduced by permanganate oxidation of ferrous ion, and sorption to filter sand; (2) evaluation of radium removal occurring in a laboratory pilot plant simulating an aeration-sand filtration iron removal process under various operating conditions including regeneration of the filter sand using a dilute acid rinse to maintain radium sorption; and (3) field evaluation at Oxford, IA, of the use of a regenerable sand filter to sorb naturally occurring radium.

Materials and Methods

Waters Used

Batch studies were conducted using synthetic groundwaters and groundwaters obtained from Oxford, IA, and Eldon, IA. A hardness-free synthetic water was prepared by mixing 5 mM $NaHCO_3$, 1 mM Na_2SO_4, and 1 mM NaCl to deionized water. Calcium and/or magnesium sulfate were added to vary hardness. Finished waters were obtained from Oxford and Eldon and were filtered through a 0.45 μm filter prior to use. Oxford water had a hardness and alkalinity of approximately 1300 mg/L and 300 mg/L as $CaCO_3$, respectively. Eldon water had a total hardness and alkalinity of approximately 300 mg/L and 200 mg/L as $CaCO_3$, respectively. Oxford and Eldon water contained approximately 10 and 50 pCi/L of natural ^{226}Ra, respectively. Radium concentrations were generally increased by adding additional ^{226}Ra.

Laboratory pilot plant studies were conducted using the tap water obtained at the University of Iowa Hygienic Laboratory, which had a total hardness of approximately 150 mg/L as $CaCO_3$ and an alkalinity of approximately 300 mg/L as $CaCO_3$. Field studies at Oxford used water from the existing sand filter effluent (iron floc removed) or water from an existing aeration/flow equalization tank (containing iron floc).

Batch Studies

Batch studies were conducted using 2-L beakers thermostatted to 25°C and stirred with a Birds and Phipps* gang stirrer. Mixtures of varying carbon dioxide-air composition were bubbled through to provide pH control although in several experiments pH was adjusted by addition of concentrated sulfuric acid or sodium hydroxide.

Solutions containing pure iron oxides were generally prepared by first deaerating the water by purging with nitrogen gas, adjusting pH to approximately 6.5 by bubbling pure carbon dioxide, then adding ferrous sulfate and ^{226}Ra. The solutions were then aerated to approximately pH 8 with normal air to oxidize the iron, and the pH adjusted to the desired value by bubbling buffer gas (or acid/base addition) followed by aging for 1 hr. Solutions of pure manganese oxides were prepared similarly except using stoichiometric amounts of permanganate to oxidize added manganous ion. Mixtures of iron and manganese oxides were prepared by addition of enough permanganate to oxidize 90% of the initial added ferrous ion in deaerated solutions followed by aeration to oxidize the remaining iron and then aging for 1 hr at the desired pH. Sorption onto acid-washed and deionized-water-rinsed 0.5 mm filter sand (effective size) was studied using the same apparatus.

After aging, an aliquot was withdrawn and filtered through a 0.45 μm filter and the filtrate acidified for later analyses. In some cases the apparatus and beakers were rinsed with 0.1 N nitric acid and the rinses analyzed for radium to ensure that significant quantities of radium were not sorbing to these components.

Pilot Plant Studies

Laboratory pilot studies were conducted at the University of Iowa Hygienic Laboratory using an iron aeration-sand filtration pilot plant (Figure 1). The 10.2-cm diameter (4-in.) pressure filter was filled with 61 cm (2 ft) of filter sand and equipped for water backwashing and periodic rinsing with dilute acid. ^{226}Ra and ferrous iron could be added to the

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*Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

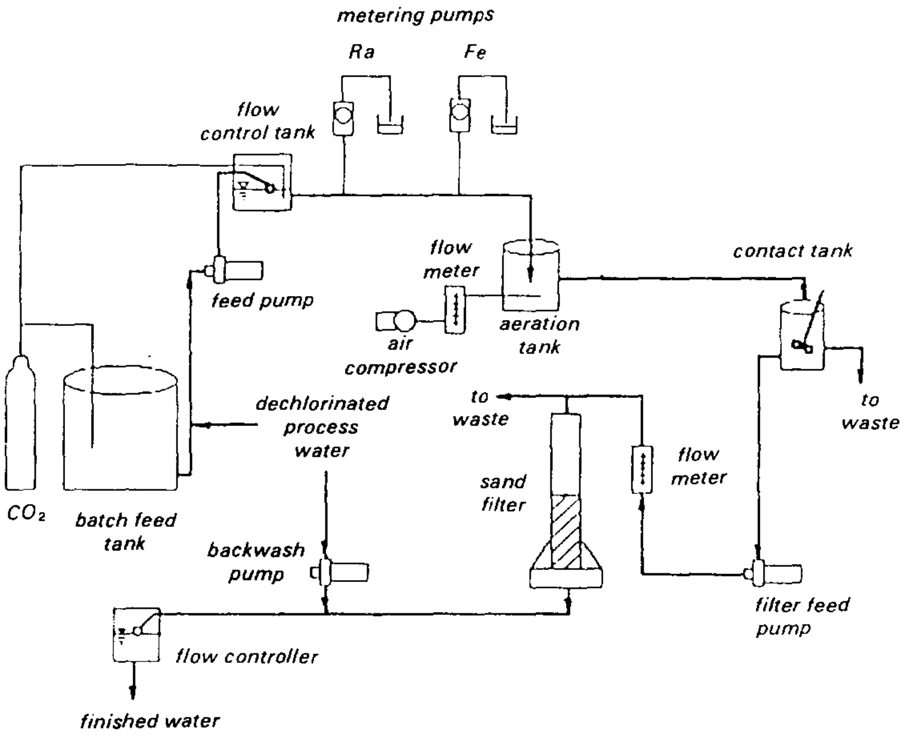

Figure 1. *Schematic diagram of pilot plant used in laboratory studies.*

water, which was maintained at pH 6.5 prior to aeration by bubbling carbon dioxide through it. Aeration to approximately pH 7.5 was accomplished in a tank having a 10-min hydraulic residence time. Filter influent was withdrawn from a small tank used to moderate flow. Flow rate through the filter was controlled using a small, constant-head tank equipped with a flow control float and valve. Automatic samplers were used to obtain discrete and composite samples. The water supply was usually 18°C ± 3°C. Field studies at Oxford utilized a similar pressure filter equipped for rinsing with a dilute acid in addition to water backwashing. Water temperature was usually 15°C ± 3°C.

Analyses

^{226}Ra concentration was determined using precipitation with $BaSO_4$ and counting of the radioactivity. Iron concentrations were routinely determined using Hach Chemical Kit ISR-18. Iron was also measured in several experiments by the phenanthroline colorimetric method. Hardness ions were determined by EDTA titration, and alkalinity by acidiometric titration.

Results and Discussion

Sorption to Hydrous Iron Oxides

Removals in a hardness-free synthetic groundwater by oxides produced by oxidation of 10 mg/L Fe increased with pH from approximately zero at a pH of about 5.5 to over 90% above pH 9.0 and 25°C (Figure 2). A linear isotherm,

$$q = k_d [^{226}Ra]$$

where q is the amount of radium sorbed per unit mass of iron, and k_d is a distribution coefficient, and $[^{226}Ra]$ is the concentration of radium remaining in solution, could be used to adequately describe sorption to iron oxides (Figure 3). As a consequence of a linear isotherm, percent removals do not depend on the initial radium concentration and can be equated to,

$$\text{percent removed} = 100\, k_d [Fe]/(1 + k_d [Fe])$$

where [Fe] is the total iron concentration.

Sorption was significantly reduced by the presence of calcium, magnesium, and barium and was a strong function of pH over the range of 5 to 9. Distribution coefficients characterizing sorption at 25°C and at approximately pH 8.0, decreased from about 0.3 L/mg obtained in the absence of hardness or barium to approximately 1/10 this value when the calcium was increased to approximately 3mM (300 mg/L as $CaCO_3$) or when 1 mg/L of barium was present in a synthetic groundwater. Above this hardness level little further reduction in the distribution coefficient was observed. This is consistent with the observation that removals were similar in both Oxford and Eldon water (approximately 10% to 20% in the presence of 10 mg/L Fe) despite a large difference in hardness. Based on measured distribution coefficients, radium reduction due to sorption to iron oxides is not expected to exceed 10% to 20% in typical groundwaters with iron concentrations of approximately 2 to 5 mg/L and hardness in excess of approximately 300 mg/L as $CaCO_3$. The pH would need to exceed at least 10 to obtain significantly better removals in natural waters at typical iron concentrations. Radium removals obtained in laboratory pilot studies designed to simulate an iron oxidation (aeration)-sand filtration plant generally supported the findings anticipated from batch studies.

Sorption to Hydrous Manganese and Iron/Manganese Mixtures

Sorption to freshly formed manganese oxides was significantly greater than that to iron oxides of comparable molar concentration and does not appear to be sensitive to either pH or hardness. Removals of approximately 80% were obtained in synthetic groundwater containing up to 3 mM Ca (300 mg/L as $CaCO_3$) and 1 mg/L of freshly formed MnO_2. Removals of 40% to 70% were obtained over the pH range of 5 to 9 by 1 mg/L MnO_2 in Oxford water and Oxford water diluted in half with deionized water (Figure 4).

Removals of ^{226}Ra in mixed manganese/iron oxides prepared by the oxidation of Fe^{+2} by $KMnO_4$ were much greater than by aeration produced iron oxides alone but were less than obtained by comparable concentrations of pure manganese oxides. Again, removals were less in Oxford water than in a hardness-free synthetic groundwater. In synthetic hardness-free groundwater, removals increased from approximately 75% at pH 7 to nearly 100% near pH 9 when 1 mg/L Fe was oxidized by $KMnO_4$

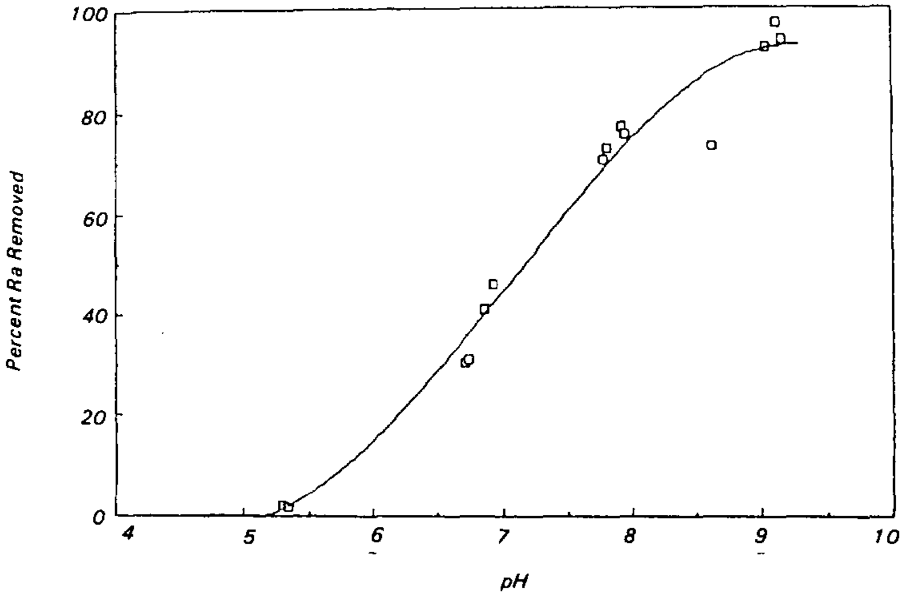

Figure 2. *Effect of pH on radium removal by sorption to iron oxides in hardness free synthetic groundwater. 10 mg/L Fe, ^{226}Ra = 48 pCi/L.*

addition. However, removals were less than 10% over pH 5.5 to 8.7 in a system prepared by permanganate oxidation of 2 mg/L Fe in Oxford water (equivalent MnO_2 concentration of 0.84 mg/L as MnO_2). A maximum removal of only approximately 40% at pH 9.0 was observed in Oxford water when 5 mg/L Fe was oxidized as described above (Figure 5). It appears that coproduced iron oxides produced by aeration after formation of manganese oxides may interfere with sorption to manganese oxides. This may limit the utility of permanganate oxidation of iron to increase radium removals.

Sorption to Filter Sand

Sorption to acid-rinsed filter sand was characterized by a linear sorption isotherm in studies conducted in synthetic and natural waters. Sorption generally increased with pH, but was not a strong function of pH in the range of 5 to 8. Magnesium and calcium reduced sorption in a similar manner when at identical molar concentrations. However, the distribution coefficient did not significantly decrease further with increasing hardness levels in excess of approximately 100 to 200 mg/L as $CaCO_3$. Distribution coefficients decreased from approximately 0.3 L/g in hardness-free synthetic groundwater to approximately 1/10 this value in the presence of 1 to 3 mM Ca over the pH range of 5 to 8. Isotherms obtained at approximately pH 7 for Oxford, Eldon, and a synthetic groundwater containing 6 mM Ca were nearly identical (Figure 6).

Theoretical calculations based on instantaneous equilibrium and a linear isotherm indicate that conventional filter sand beds may have a significant radium removal capacity if this capacity could be maintained by a regeneration step. Table 1 summarizes measured distribution coefficients, theoretical calculations of the ratio and bed volumes potentially treated to empty bed volume, and the breakthrough time, t_b, for a 1-m deep sand bed fed at a loading rate of 3.7 m/h (1.5 gpm/ft^2).

Laboratory and Field Pilot Plant Studies

The daily average amount of radium removed by the filter-fed UHL water (containing 2 mg/L of iron flocs and approximately 80 pCi/L added ^{226}Ra at a rate of 7.5 m/h (3 gpm/ft^2) increased from about 30% obtained with only a daily water backwashing, to approximately 85% to 90% when the bed was regenerated with 4 bed volumes of a pH 1 dilute HCl rinse (Figure 7). Performance did not decrease over a 4-day period in which daily regeneration was practices using 4 bed volumes of regenerant at a loading rate of 7.4 m/h (3 gpm/ft^2). Removals obtained using a daily pH 2 rinse at the same operating conditions were only slightly less (about 75% to 85%). Mass balances on the cumulative amount of radium removed by the sand filter and recovered during backwashing/regeneration showed that only approximately half of the radium removed prior to initiation of acid regeneration could be recovered in the backwash water, indicating that sorption to filter sand was occurring even when water-only backwashing was practiced. All radium removed by the sand filter was recovered with dilute acid regeneration using either pH 1 or pH 2 rinses.

Operation of the pilot facility at Oxford to treat the existing sand filter effluent using a 3.7 m/h (1.5 gpm/ft^2) influent loading rate and a daily pH 1 rinse (4 bed volumes, EBCT 5 min) for 4 days resulted in daily composite average removals of approximately 85% and the production of a water containing ^{226}Ra at concentrations less than 3 pCi/L (Figure 8). Upon termination of dilute acid regeneration, effluent radium concentrations increased greatly. The removals are a consequence of dilute acid regeneration and not sorption to iron flocs since radium is not removed by filtration in Oxford's existing filters nor was any significant amount found to be sorbed to the iron oxides as determined by filtration through a 0.45 μm filter and filtrate analysis. Removals were less when twice the water volume was treated using an influent loading rate of 7.4 m/h (3.0 gpm/ft^2) but still resulted in a product water containing approximately 3 pCi/L ^{226}Ra. In general, observed radium sorptive capacities were 3 to 4 times greater than those anticipated based on batch sorption studies. This may be due to lower temperatures used in the pilot studies or an unidentified mechanism.

Radium removals decreased significantly to approximately 25% to 30% when the pilot plant was used to simultaneously remove both radium and iron flocs (0.5 to 1.0 mg/L as Fe) from the aerator effluent. Figure 9 shows results being obtained using an influent loading rate of 3.7 m/h (1.5 gpm/ft^2) and a pH 1 regenerant of either dilute hydrochloric or sulfuric acid at rates of 3.7 and 7.4 m/h (1.5 and 3.0 gpm/ft^2). Sulfuric and hydrochloric acid appear equally effective at either of the regenerant loading rates.

The decrease in radium removal efficiency occurring as a result of a dual-use mode of operation was not expected based on the laboratory pilot plant

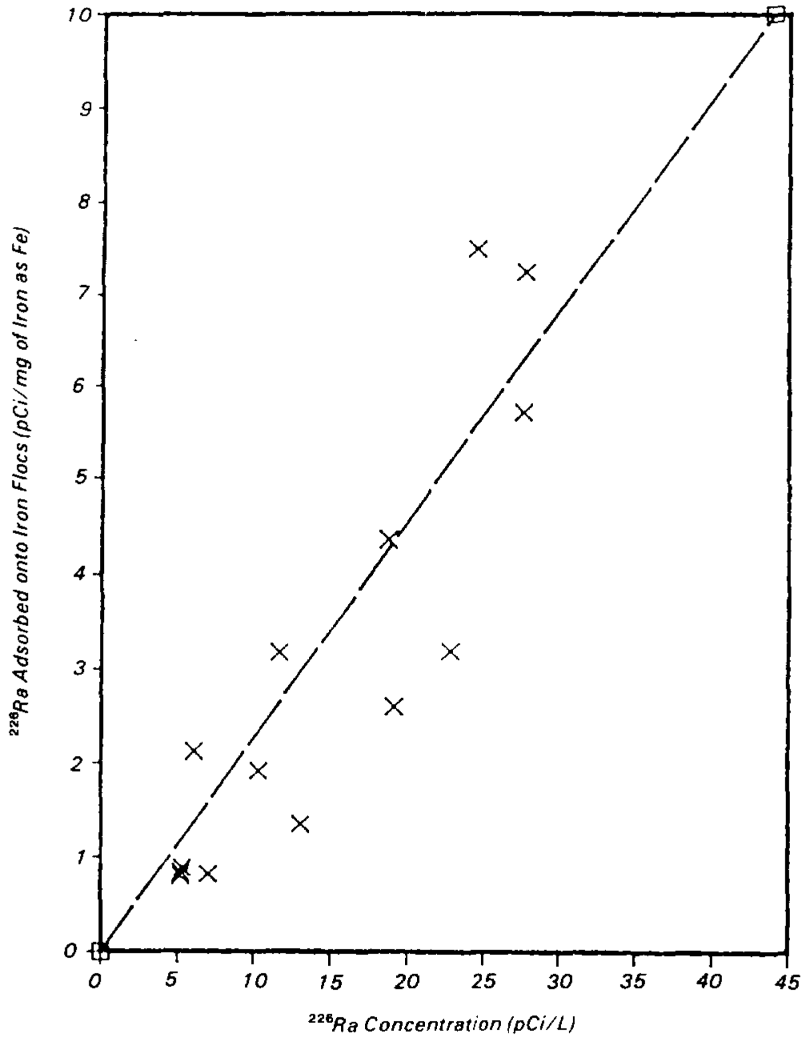

Figure 3. *Radium sorption isotherm obtained in hardness free synthetic groundwater at pH 8.1.*

results, which indicated little iron floc interference. While it may be supposed that the flocs coated the sand, thereby blocking radium sorption, it is believed that most of the iron flocs were removed in the first few inches of sand bed. Therefore, significant amounts of "clean" sand probably existed below the iron floc removal zone even after 1 day of operation. Poor removals may be attributable to the presence of significant concentrations of soluble iron.

Conclusions

Increasing sand filter influent pH to increase sorption to hydrous iron oxides produced by aeration does not appear to be a realistic strategy to increase radium removals. Differences in radium removals occurring as a result of iron removal processes involving aeration-sand filtration are probably due to differences in pH, iron concentration, and hardness. Regardless of these differences, little removal is expected.

The favorable results obtained using hydrous manganese oxide suggests that they could potentially be used to effectively remove radium if iron did not interfere. If a significant interference is caused by precipitation of iron oxides onto manganese oxides, then a two-stage oxidation process may be warranted when both iron and manganese are present. Iron could be oxidized by aeration in the first stage, followed by permanganate addition in the second to oxidize manganese. Filtration of the iron oxides prior to hydrous manganese oxide formation may also be desirable. Addition of manganous ion to waters containing low concentrations of naturally occurring soluble manganese might be useful to increase concentrations of hydrous manganese oxides. Direct addition of preformed hydrous manganese oxides may also be technically feasible and a relatively easily controlled approach to radium removal. In either strategy, filter runs are expected to decrease when a single filter is used to remove both iron and manganese precipitates. Length of filter run might, therefore, limit the utility of adding hydrous manganese oxides or manganous ion.

Radium removal using a regenerable sand filter has been demonstrated. While sorption to filter sand is believed an important mechanism, another is possibly involved. Significantly increased capacities are probably needed before use of a regenerable sand filter process could be generally recommended unless frequent rinsing with a dilute acid was acceptable. Further work is needed to better characterize the removal mechanisms, and to determine maximum possible removals and sorptive capacities. Studies should be conducted in a variety of waters and, in particular, its performance evaluated when operated to simultaneously remove both radium and iron flocs. Studies should also be conducted over much longer time periods than those used in this study. A tradeoff between dilute acid strength and volume is expected to lead to a least-cost regeneration scheme. However, even with a 4 bed volume pH 1 daily regeneration scheme, acid costs to treat Oxford water are estimated to be $0.12/1,000 gal (based on a cost of $7/100 lb of sulfuric acid). Neutralization of the spent regenerant will probably be required before discharge but since pH control is probably not critical, an inexpensive system utilitzing crushed limestone may be adequate. However, sludges may be produced that must be disposed of. Other anticipated costs are associated with possible modifications to make the sand filter resistant to damage by dilute acid.

The full report was submitted in fulfillment of Cooperative Agreement No. CR-810575-01 by the University of Iowa, under the sponsorship of the U.S. Environmental Protection Agency.

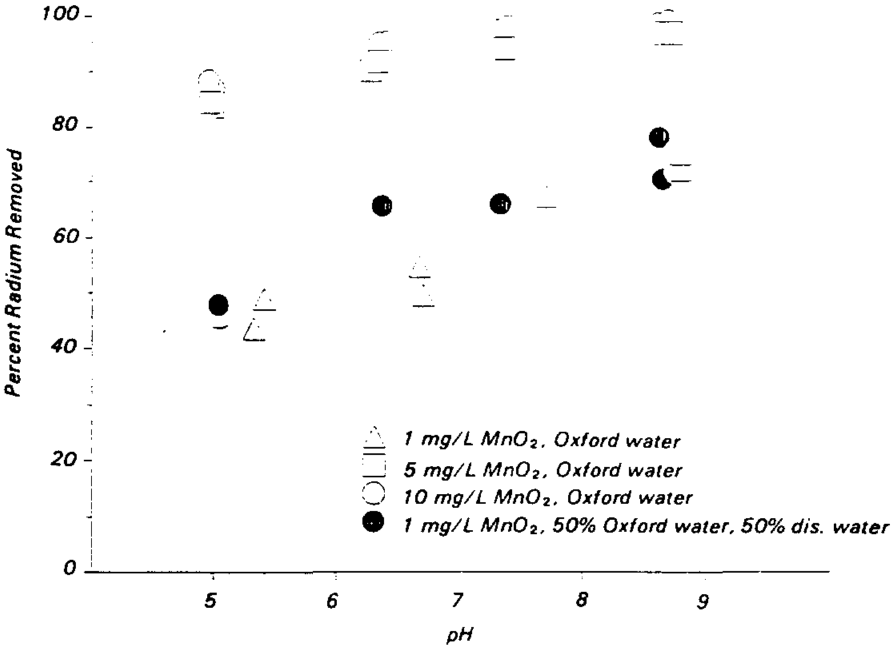

Figure 4. Radium removal by freshly precipitated MnO_2 in Oxford water and 50% Oxford water. Ra = 32 pCi/L.

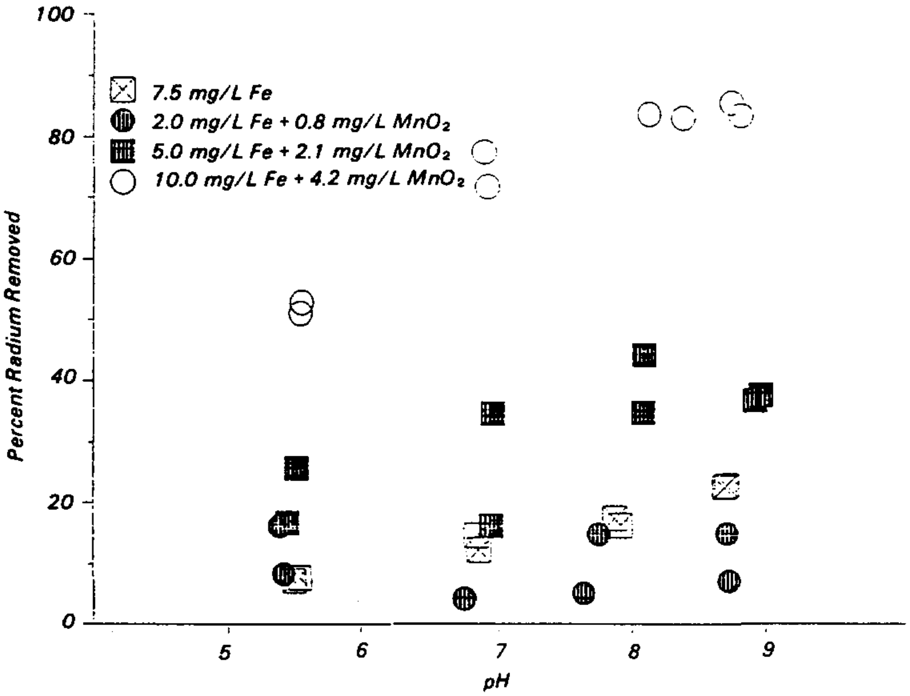

Figure 5. Radium removal by freshly precipitated (coproduced) iron and manganese oxides in Oxford water. Manganese oxides produced by addition of permanganate sufficient to oxidize 90% of the initial ferrous iron. Removal by 7.5 mg/L as Fe pure iron oxides shown for comparison. Ra = 32 pCi/L.

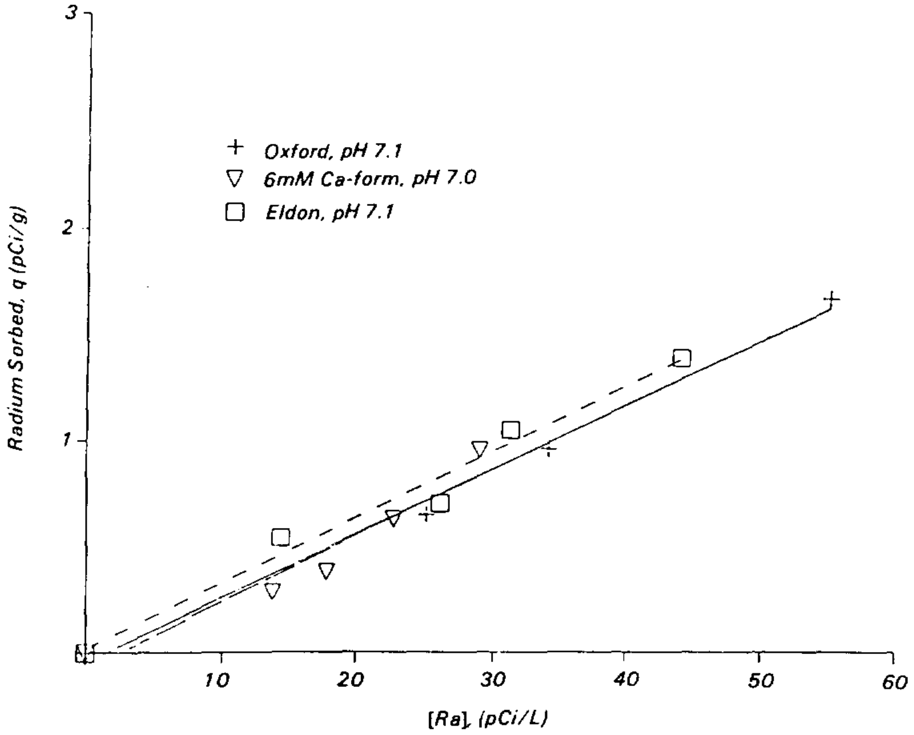

Figure 6. Comparison of ^{226}Ra sorption isotherms (filter sand) as developed for Eldon (pH 7.1), Oxford (pH 7.1), and synthetic groundwater (pH 7.0) containing 6 mM calcium.

Table 1. Summary of Distribution Coefficients, Bed Volumes Treated and Run Durations

Water Type	pH	K_d, L/g	V_t/V_b	t_b, hr
Na-Form	7.0	0.240	384	104
Oxford	6.3	0.037	59	16
Oxford	7.1	0.027	43	12
Oxford	8.7	0.027	43	12
Eldon	7.1	0.031	50	13
Eldon	8.6	0.046	74	20
6 mM Ca	7.0	0.032	51	14
6 mM Ca + 6 mM Mg	8.7	0.026	42	11

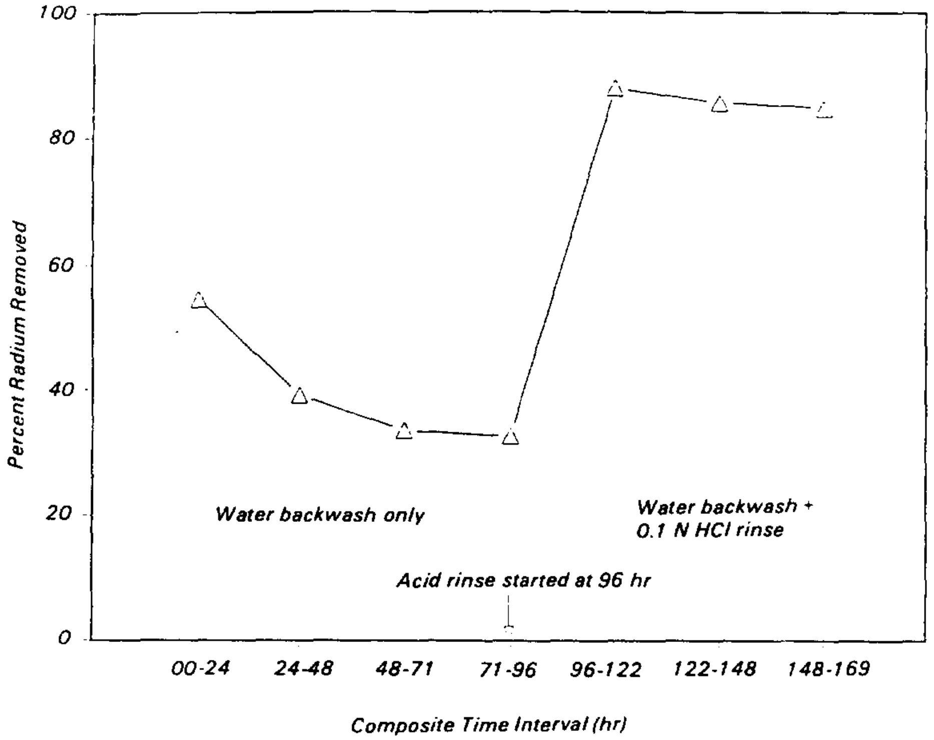

Figure 7. Percent of influent radium removed with and without daily pH 1 regeneration. Laboratory pilot plant experiment using an influent radium concentration of approximately 80 pCi/L. Fe = 2 mg/L. Total hardness ≈ 150 mg/L as $CaCO_3$. Filter influent pH 7.5. Dilute acid regeneration scheme started after 96 hr.

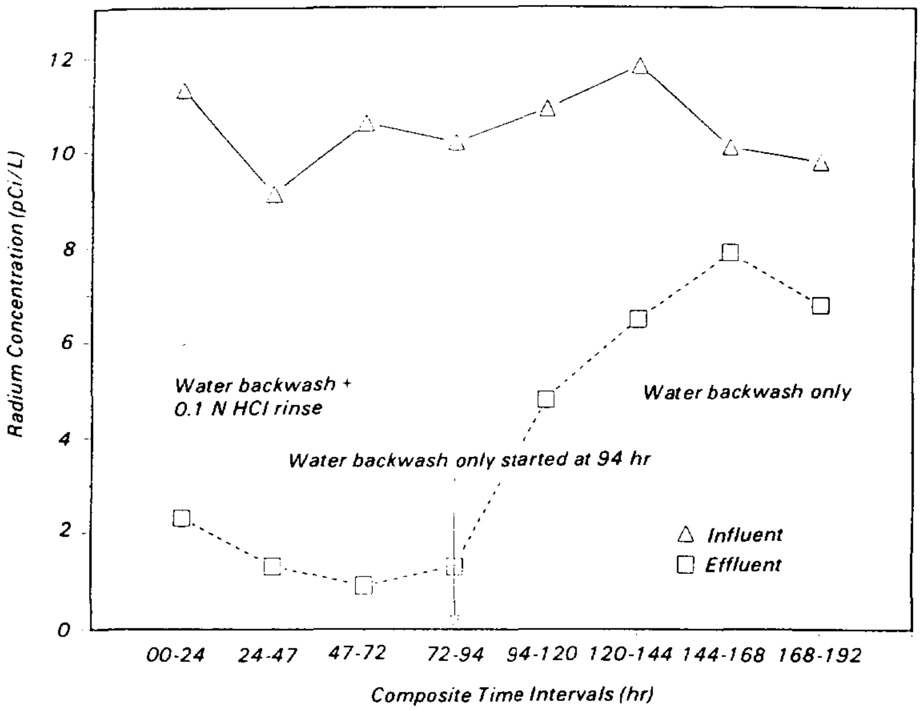

Figure 8. Influent and effluent ^{226}Ra concentrations from pilot studies at Oxford, IA. Existing filter effluent used as regenerable filter feed. Daily dilute acid regeneration terminated after 94 hr.

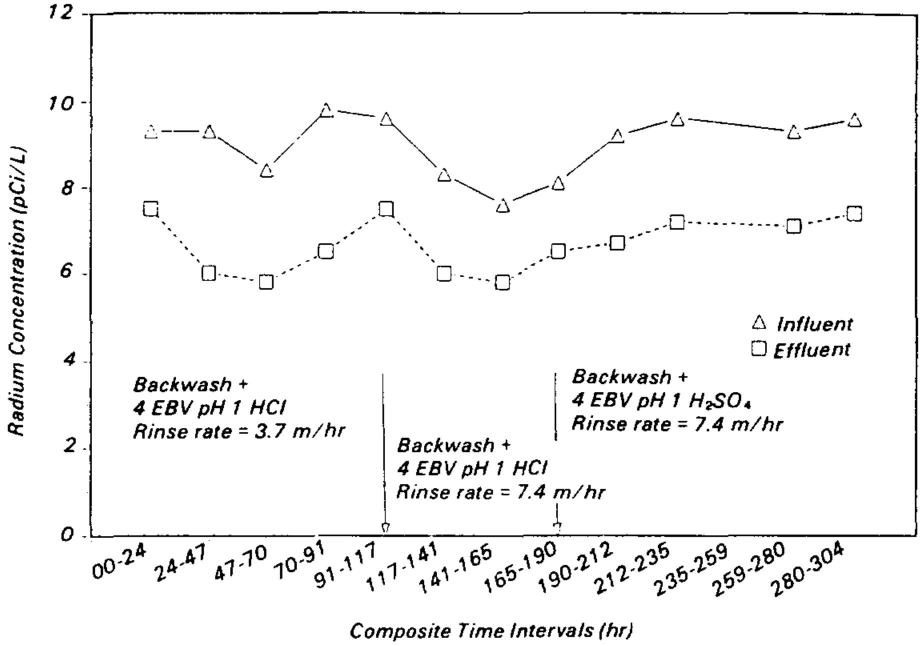

Figure 9. Influent and effluent ^{226}Ra concentrations from pilot studies at Oxford, IA. Simultaneous iron and radium removal from existing aerator effluent. Fe = 0.5-1.0 mg/L. Comparison of sulfuric and hydrochloric acid regeneration at two regenerant rinse rates.

United States
Environmental Protection
Agency

Municipal Environmental
Research Laboratory
Cincinnati OH 45268

Research and Development

EPA-600/S2-84-093 Sept. 1984

EPA

Project Summary

Barium and Radium Removal from Groundwater by Ion Exchange

Vernon L. Snoeyink, Julie L. Pfeffer, David W. Snyder, and Carl C. Chambers

A study was undertaken to develop technology that can be used by small water treatment plants to remove hardness, barium, and radium^{226}. Special emphasis was placed on finding an alternative to strong acid ion exchange (which is used in the Na^+ form) because this process adds large amounts to Na^+ to the treated water. The primary objective of this study was to determine the applicability of weak acid ion exchange resin for removal of hardness, barium, and radium from the types of groundwater encountered in northern Illinois. The capacity of the resin and the regeneration requirements were to be determined and compared with those of strong acid resins for the same application.

Additional tasks included (1) evaluating the performance of the strong acid resin now used at Crystal Lake, Illinois, to remove hardness and barium, (2) determining the barium and radium removal efficiencies of strong acid ion exchange softeners used in homes, and (3) modifying the surface of activated carbon to make it suitable for the selective removal of barium.

Both strong and weak acid resin systems were very effective in removing ^{226}Ra and Ba^{2+}. The weak acid system in the H^+ form does not add Na^+ to the water as does the strong acid system, but the weak acid system will cost more to use because of the need for acid-resistant materials and the CO_2 stripping.

This Project Summary was developed by EPA's Municipal Environmental Research Laboratory, Cincinnati, OH, to announce key findings of the research project that is fully documented in a separate report of the same title (see Project Report ordering information at back).

Introduction

Naturally occurring barium in drinking water exceeds the maximum contaminant level (MCL) in some areas of northern Illinois and northeastern Iowa. In these same areas and in some parts of Florida, the concentrations of radium^{226} exceed the MCL of 5 picoCuries (pCi)/L. Most of the contaminated supplies are used by small communities, many of which do not presently treat their water to reduce the concentrations of these substances. Both radium^{226} and barium are alkaline earth metals, and both are found in water as divalent cations. Their chemical behavior is very similar, and it is much like that of Ca^{2+} and Mg^{2+}, the principal components of hardness in water. Thus, processes used to soften water are very useful for removing these contaminants from drinking water.

A process particularly suited to small communities where hardness, barium, and/or radium are a problem is ion exchange. The objective of this research was to investigate the applicability of the ion exchange process to this problem in northern Illinois. Strong acid resins in the sodium form were to be evaluated, but because their use results in significant increases in the Na^+ content of the water, their performance was to be compared with that of weak acid resins in the hydrogen form. The latter can remove only the equivalents of divalent cations equal to the equivalents of alkalinity present, and ion exchange must be followed by CO_2 stripping and pH adjustment.

The resins were tested with an influent water containing hardness of approximately 200 mg/L as $CaCO_3$, total alkalinity of 250 mg/L as $CaCO_3$, 20 mg/L Ba^{2+}, and 20 pCi/L

^{226}Ra. The tests used a regenerant dose typical of softening operations.

In addition to investigating the applicability of ion exchange to northern Illinois groundwater, this project also (1) evaluated the performance of the strong acid resin now used at Crystal Lake, Illinois, to remove hardness and barium, (2) determined the barium and radium removal efficiencies of strong acid ion exchange softeners used in homes, and (3) attempted to modify the surface of activated carbon to make it suitable for the selective removal of barium.

Materials and Methods

The resins used in this study were Duolite C-20 and Duolite C-433 manufactured by Diamond Shamrock, Redwood City, CA.[*] C-20 is a strong acid resin with a polystyrene matrix and sulfonate functional groups. C-433 is a high-capacity weak acid resin with a polyacrylic matrix and carboxyl functional groups. Their capacities are 4.8 and 11.5 meq/g dry resin, respectively, and their properties are typical of resins made by a number of manufacturers. The weak acid resin can be used in solutions with pH > 5, and the strong acid resin can be used for pH > 0.

The solutions used for most of the column tests contained approximately 100 mg Mg^{2+}/L as $CaCO_3$, 100 mg Ca^{2+}/L as $CaCO_3$, 250 mg/L total alkalinity, 20 mg Ba^{2+}/L, and 20 pCi/L of ^{226}Ra. Solutions for selectivity determinations were prepared using reagent-grade chemicals to the specifications required by the test.

Ion chromatography was used for quantitative analysis of cations. Hardness and alkalinity were also determined in accordance with *Standard Methods for Analysis of Water and Wastewater* (15th edition, Amer. Public Health Assoc., 1980). Some samples were also analyzed by atomic absorption spectroscopy. The ^{226}Ra samples were analyzed at the Environmental Research Laboratory, University of Illinois, using the radon emanation method, and at the University Hygienic Laboratory, University of Iowa, by a technique involving coprecipitation of the ^{226}Ra with barium sulfate and by alpha counting with an internal proportional counter.

Experimental Results

Strong Acid Resins

Isotherms and column tests were used to determine the capacity of the strong acid resin with 4.8 meq/g dry resin (hydrogen form). The selectivity sequence was $Ba^{2+} > Ca^{2+} > Mg^{2+} > Na^+ > H^+$. Alkalinity had no effect on capacity, but the capacity for divalent cations decreased as the concentration of sodium increased.

Column tests were run using 2.5- x 62-cm columns with the resin in both the Na^+ and H^+ form. Application of the test solution at rates of 2.5 to 5 bed volumes (BV)/hr to virgin resin in the Na^+ form gave the breakthrough curves shown in Figure 1. The curves for the H^+ form of the resin were similar except for the Na^+ concentration in the effluent. In keeping with the selectivity sequence, Mg^{2+} is the first divalent cation to appear in the effluent, followed by Ca^{2+}, and much later by Ba^{2+}. Regeneration of the H^+ form of the resin with 8 percent HCl at 1.6 BV/hr gave the results that appear in Figure 2; results for regeneration of the Na^+ form

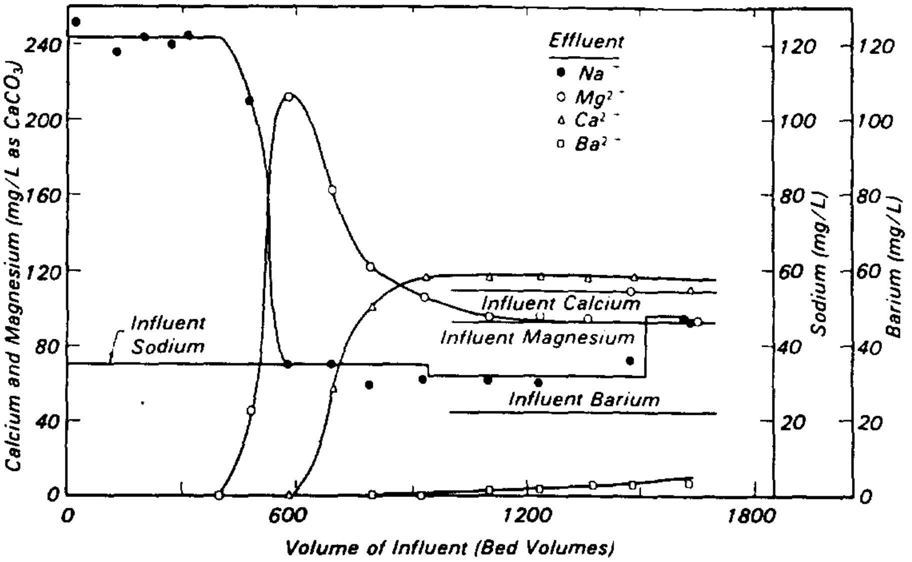

Figure 1. Breakthrough curves for virgin strong acid resin in the sodium form.

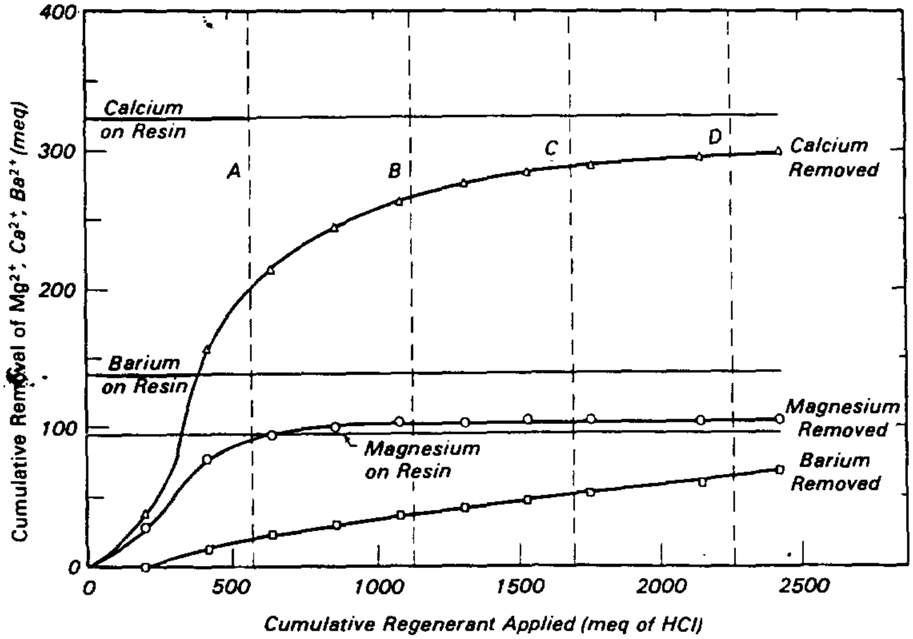

Figure 2. *Regeneration of the hydrogen form of the strong acid resin with HCl. At A, B, C, and D, the equivalents of HCl added per equivalent of divalent cation originally on the resin is 1, 2, 3, and 4, respectively.*

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[*] Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

with NaCl gave very similar results. At a regenerant dose equal to the equivalents of divalent cation on the resin, 55 percent of the ions were removed, including 80 percent of the Mg^{2+}, 67 percent of the Ca^{2+}, and 15 percent of the Ba^{2+}. A dose of three times the equivalents of the divalent cations on the resin was necessary to remove 50 percent of the Ba^{2+}.

Several exhaustion-regeneration cycles were then carried out to determine the behavior of the strong acid resin at steady-state conditions. The influent was applied at a rate of 17 BV/hr, and a typical NaCl regenerant dose for softening (6.5 lb/ft^3, or 4.7 meq/g) was applied co-current at 1.7 BV/hr, followed by a slow rinse at 1.7 BV/hr for 48 min and a fast rinse at 7 to 8 BV/hr for 26 min. The Ba^{2+} accumulated on the resin over successive cycles and thus caused the Ba^{2+} to break through much earlier than it did with virgin resin. The hardness and Ba^{2+} breakthrough curves that developed after several cycles appear in Figure 3. The curves show that approximately 225 BV of water were processed before hardness and Ba^{2+} broke through at about the same time. The regeneration efficiency (equivalents of divalent cations removed per equivalents of NaCl applied) was 58 to 59 percent. Column utilization (column capacity used/maximum capacity) was about 58 percent; a large portion of the remaining 42 percent was occupied by Ca^{2+} and Ba^{2+} that was not removed by the regenerant dose.

The data in Figure 4 show the Ba^{2+} concentration (when $CaCO_3$) effluent hardness was approximately 40 mg/L as $CaCO_3$ as a function of the number of cycles. The Ba^{2+} concentration leveled off at 1.7 mg/L for a regenerant dose of 6.5 lb NaCl/ft^3 (4.7 meq/g). An HCl regenerant dose of 4.06 lb HCl/ft^3 (4.7 meq/g) was tested on a second column for a few runs, and similar results were obtained. Increasing the regenerant dose by 50 percent to 9.75 lb NaCl/ft^3 reduced the Ba^{2+} to about 1.2 mg/L when the hardness was 40 mg/L as $CaCO_3$. The regeneration efficiency dropped to 46 percent; however, the column utilization increased to 68 percent. For the regenerant dose of 6.5 lb NaCl/ft^3, 2.4 BV of brine with total dissolved solids (TDS) of 16,400 mg/L were produced per 100 BV of product water.

The conclusion that Ba^{2+} can be removed to less than the MCL of 1 mg/L as long as the strong acid resin is not saturated with hardness was confirmed by observations at the Crystal Lake, Illinois, municipal ion exchange system. The influent and effluent of several home ion exchange softeners were also sampled, and these too showed Ba^{2+} removal below the MCL.

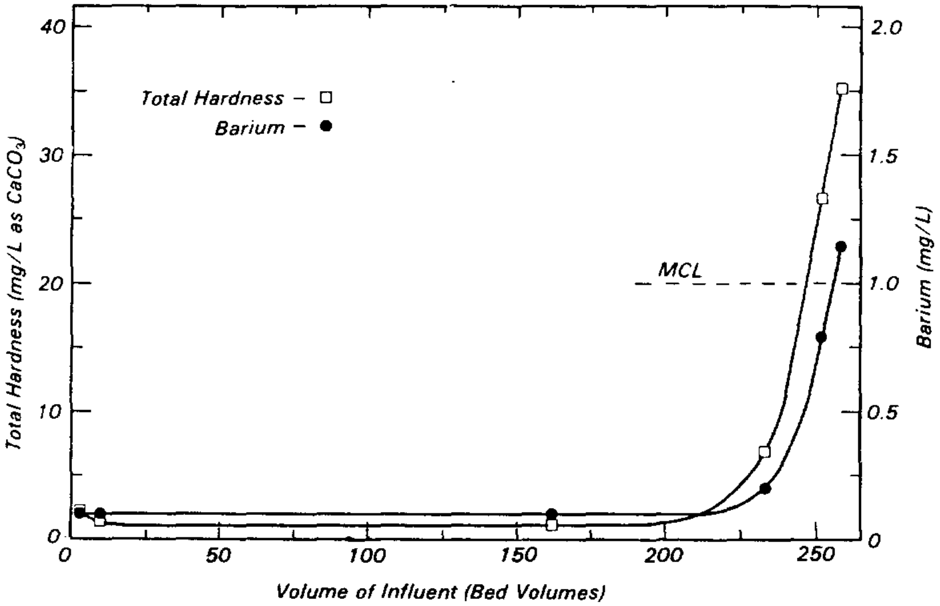


Figure 3. Breakthrough curves for hardness and barium on a strong acid resin in the sodium form after several exhaustion-regeneration cycles.

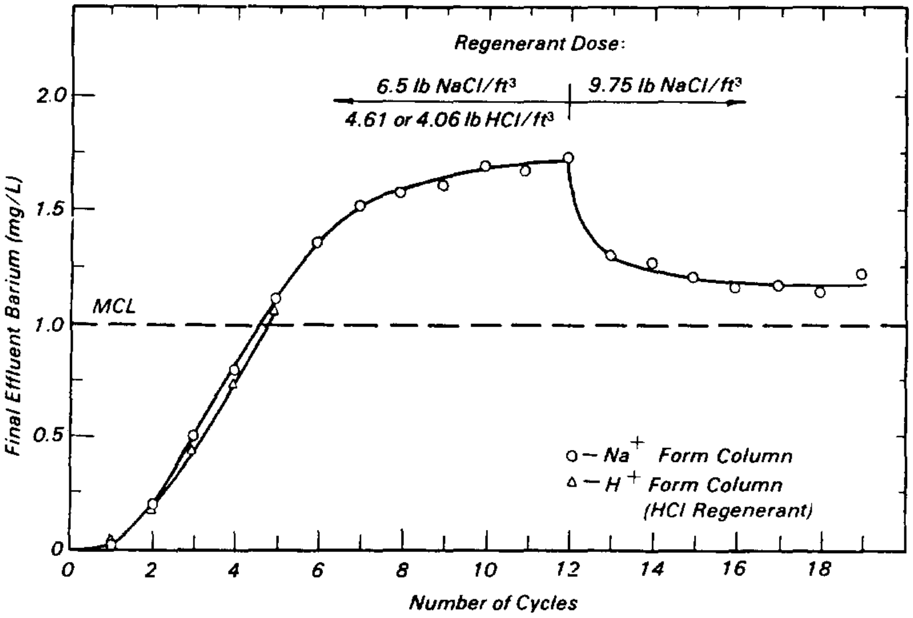


Figure 4. Barium effluent concentrations for the strong acid resin exhaustion-regeneration cyclic runs. The barium values shown are those when the effluent hardness concentration is 40 mg/L.

Weak Acid Resins

The maximum capacity of the weak acid resin was determined to be 11.5 meq/g dry resin, but the extent of swelling of the resin determined how much of the capacity could be used. For example, exposure of the resin in the H^+ form to a solution containing only $BaCO_3$ salt resulted in a capacity of only 7.5 meq/g, whereas carbonate salts of Ca^{2+} and Mg^{2+}, which cause the resin to swell more

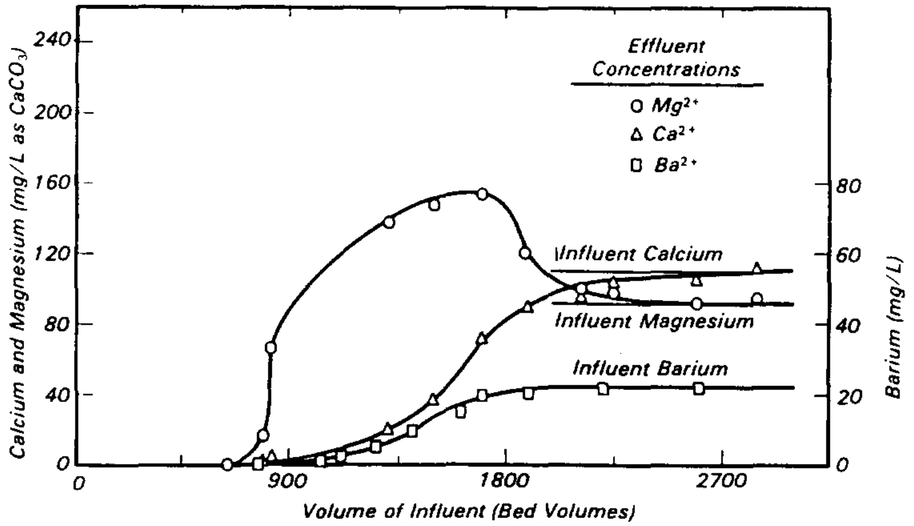


Figure 5. *Breakthrough curves for weak acid resin in the hydrogen form.*

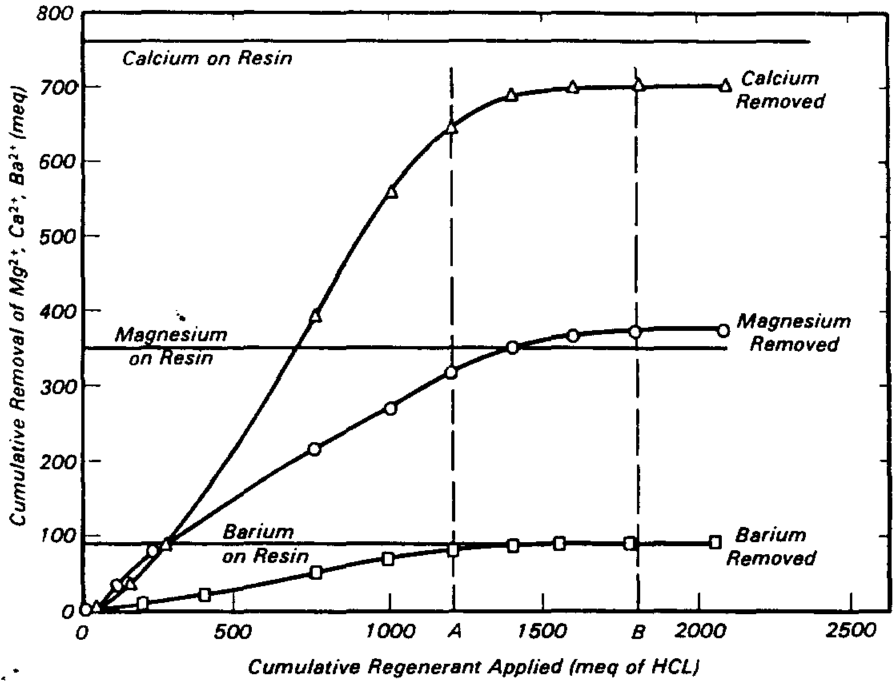


Figure 6. *Regeneration of weak acid resin with HCl. A and B represent 1.0 and 1.5 equivalents of HCl applied per equivalent divalent cation on the resin.*

because of their larger hydrated radii, used the full 11.5 meq/g. Addition of salts to the Ba^{2+} solution increased the swelling and allowed Ba^{2+} to use the full capacity.

Application of test solution to 2.5- x 62-cm columns of the virgin resin in the H^{+} form gave the breakthrough curves that appear in Figure 5. The equivalents of alkalinity per liter in the product water exceed the equivalents of divalent cations per liter, so the initial removal of Ca^{2+}, Mg^{2+}, and Ba^{2+} was complete. Their order of appearance in the effluent was in reverse order of the selectivity series, Ba^{2+} $\cong$ Ca^{2+} > Mg^{2+}, and the breakthrough curves were less steep than those for the strong acid resin, a result caused by a slower rate of exchange by weak acid resins.

A major advantage of using a weak acid resin is the ease with which it can be regenerated by strong acid. The data in Figure 6 show that application of one equivalent HCl per equivalent of divalent ion on the resin resulted in removal of 90 percent of the ions, and 1.5 equivalents HCl per equivalent of divalent cation gave essentially complete removal.

Several exhaustion-regeneration runs were then made to determine resin behavior under continuous operation. Influent was applied at about 17 BV/hr; three cycles were completed, each with a different regenerant dose. The breakthrough curve in Figure 7 resulted after steady-state behavior was obtained for the cycle using 8.5 meq HCl/g. More than 650 BV of product water were obtained before effluent hardness reached a level of 40 mg/L as CaCO$_3$. The Ba^{2+} concentration leveled off at 0.2 mg/L, as shown in Figure 8. For this regenerant dose, column utilization was about 70 percent, and regeneration efficiency was about 95 percent. The 30 percent of column capacity that was not used was attributable to slow exchange kinetics rather than to buildup of Ca^{2+} ions, as it was for the strong acid resin. Approximately 1.2 BV of spent brine was produced with a TDS of 19,900 mg/L per 100 BV of product water.

Application of less regenerant than the equivalents of divalent cations removed led to a higher Ba^{2+} concentration at hardness breakthrough and to a high leakage of Ba^{2+} and hardness at the start of the subsequent run.

Radium Removal

Radium^{226} removal efficiency for both the strong and weak acid resins was excellent. The concentration was reduced from the 20 pCi/L in the influent to much less than the MCL of 5 pCi/L, even after the resins were saturated with hardness and Ba^{2+}. Analysis of the ^{226}Ra in the spent regenerant showed that all ^{226}Ra removed during the exhaustion cycle was extracted from the resin during regeneration. Some ^{226}Ra accumulated on the strong acid resin during the first few exhaustion-regeneration cycles, but after several cycles, no additional accumulation was apparent, and the resin still performed satisfactorily.

Influent and effluent samples taken from home ion exchange softeners using NaCl regeneration also showed that they removed ^{226}Ra to below the MCL.

Cost

The cost of using strong acid ion exchange softening with NaCl as the regenerant and the type of water used in the laboratory study was estimated to be $1.36/1000 gal and $0.38/1000 gal for a 0.1 and 1 MGD facility, respectively, for the type

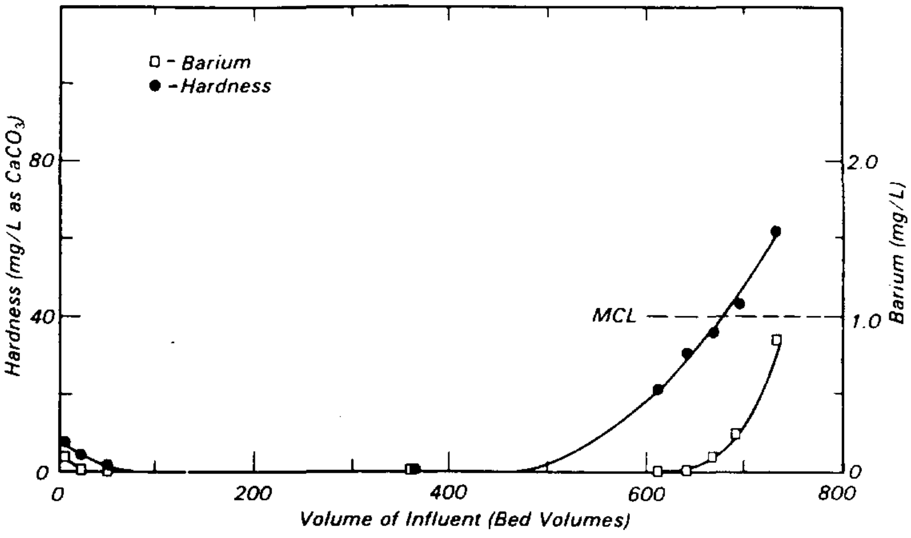


Figure 7. *Breakthrough curves for barium and hardness on a weak acid resin in the hydrogen form after several exhaustion-regeneration cycles.*

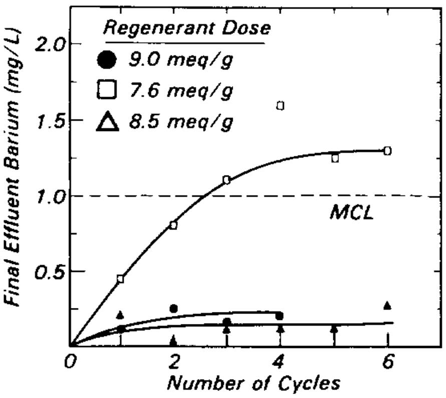


Figure 8. *Barium effluent concentration for the weak acid resin exhaustion-regeneration cyclic runs. The barium values shown are those that occur when the effluent hardness concentration is 40 mg/L as $CaCO_2$.*

of water used in the laboratory study. The additional cost for using a weak acid system with HCl regeneration will be \$0.15 and \$0.08/1000 gal, respectively.

The system designs were based on the performance observed when 4.7 meq NaCl/g and 8.5 meq HCl/g were used to regenerate the strong and weak acid resins, respectively. The flow scheme for the weak acid system included the resin columns, a CO_2 stripping column, pH adjustment, and chlorine addition. The strong acid resin system was similar except that the CO_2 stripping column was excluded. The costs were based on cost data supplied by the Illinois Water Treatment Company in Rockford, Illinois. The strong acid ion exchange units included the columns, the resin, regenerant day tanks, brine tanks, and interface piping. The weak acid unit included all of these plus a carbon dioxide stripping tower. A comparison of the two types of exchange systems was used to evaluate the additional costs required for weak acid exchange systems at these different treatment capacities. Complete treatment costs for strong acid ion exchange softening were determined from previously published data and appropriate price indices.

The costs do not include the costs of pumping raw, in-plant, or finished water. Brine disposal was also not included, since costs are very site specific. They do include the cost of adjusting product water pH, however, and this may not be necessary in all cases.

Conclusions

After several exhaustion-regeneration cycles at a regenerant dose typical of softening operations, Ba^{2+} and ^{226}Ra were effectively reduced well below their MCL's. For both types of resin, significant concentrations of Ba^{2+} appeared in the effluent at about the same time as did hardness, but ^{226}Ra continued to be removed even after saturation of the resins with hardness. These conclusions were consistent with the performance of an operating municipal ion exchange plant in northern Illinois and with the results obtained using home ion exchange softeners.

Though both strong and weak acid resin systems can effectively remove hardness, ^{226}Ra, and Ba^{2+}, the weak acid system in the H^+ form does not add Na^+ to the water as does the strong acid. The weak acid system will cost more to use, however, because of the need for acid-resistant materials and the CO_2 stripping. But this cost is offset as the plant size increases. The additional cost for the weak acid system declines from about \$0.15/1000 gal for a 0.1-MGD plant to \$0.08/1000 gal for a 1-MGD facility. In addition, the weak acid resin will remove only the divalent cations that are balanced by an equivalent amount of alkalinity. If a water contains divalent ions in excess of the alkalinity, the hardness left in the water may be desirable because it should make the water less corrosive during distribution. Whether the weak acid resin will remove Ba^{2+} from such a water must still be shown, however.

Conversion of an existing system that uses strong acid resin in the sodium form to one that uses weak acid resin in the hydrogen form must be done with care. In most cases, acid-resistant materials will have to be installed, and provision must be made to neutralize the spent acid before its discharge. Additional design considerations include that the capacities of the two resins are different and that the weak acid resin is subject to more swelling. Use of a weak acid resin will also require the installation of a CO_2-stripping process.

*USGPO: 1984-759-102-10663

United States
Environmental Protection
Agency

Water Engineering Research
Laboratory
Cincinnati OH 45268

Research and Development

EPA/600/S2-85/006 Mar. 1985

Project Summary

Barium and Radium in Water Treatment Plant Wastes

Vernon L. Snoeyink, Candy K. Jongeward, Anthony G. Myers, and Sharon K. Richter

Water Treatment plants at nine locations (10 plants) in Illinois and Iowa were studied to determine the characteristics and disposal practices for the sludge, brine, and backwash water containing radium (Ra) and/or barium (Ba). The treatment processes in these 10 plants include iron (Fe) and manganese (Mn) removal (3 plants), lime softening (4 plants), and ion exchange (3 plants). In the 10 plants, eight had concentrations of radium in their water treatment plants wastes and three had barium. The data are needed to determine whether special procedures are required to dispose of such wastes.

For the eight plants having radium in their wastes, the influent Ra^{226} concentrations ranged from 0.3 to 49 pCi/L. Radium removal averaged 8%, 75%, and 85% for Fe and Mn, lime softening, and ion exchange, respectively. Theoretically, the amount of radium removed estimates the quantity of radium in the waste. The data showed that backwash water from Fe and Mn removal plants contained average Ra^{226} concentrations that ranged from 21.2 to 106 pCi/L, and average Ra^{228} concentrations ranged from 5.7 to 20 pCi/L. Lime softening sludge, on a dry weight basis, ranged from <1.2 to 21.6 $pCiRa^{226}/g$ and from <2.4 to 11.7 $pCiRa^{228}/g$. Ion exchange plant brine contained peak Ra^{226} concentrations of 217 and 1,144 pCi/L.

Influent barium concentrations for the three plants studied ranged from 4.0 to 16.1 mg/L. Lime softening and ion exchange removed barium to concentrations that were below the maximum contaminant level (MCL) of 1.0 mg/L. Barium removal during lime softening is pH dependent. Theoretically, the amount of barium removed estimates the quantity of barium in the waste. Barium concentrations in the brine and rinsewater from two ion exchange plants that were tested averaged 328 and 1,297 mg/L. Peak concentrations of barium from the two brines were 1,197 mg/L and 5,161 mg/L.

Disposal processes used, for the plants that were studied, were lagooning, discharge to sanitary sewers, and discharge to a water course.

This Project Summary was developed by EPA's Water Engineering Research Laboratory, Cincinnati, OH, to announce key findings of the research project that is fully documented in a separate report of the same title (see Project Report ordering information at back).

Introduction

Naturally occurring barium (Ba) in drinking water exceeds the MCL of 1 mg/L in some areas of northern Illinois and northeastern Iowa. In these same areas and in some parts of Florida, the concentrations of $radium^{226}$ (Ra^{226}) and radium ^{228} (Ra^{228}) exceed the MCL, which is 5 pico-Curies (pCi)/L. Most of the contaminated supplies are used by small communities, many of which do not presently treat their water to reduce the concentrations of these substances. Both Ra and Ba are alkaline earth metals, and both are found in water as divalent cations. Their chemical behavior is very similar, and it is much like that of calcium (Ca^{2+}) and magnesium (Mg^{2+}), the principal components of hardness in water. The MCL's for Ba and Ra were developed to minimize the attack of these elements on human bones and the replacement of the Ca in bones. Their similarity to Ca, however, means that processes used to soften water are also very useful for removing these contaminants from drinking water.

Water treatment in small communities is often accomplished with iron and manganese (Fe and Mn) removal plants, lime softening plants, and ion exchange plants. The objective of this research was to characterize the backwash samples from all three types of plants, as well as the sludges from the lime softening plants and the brines from ion exchange plants. The concentration data were needed to determine whether special procedures are required to dispose of such wastes. The Ba and Ra removal efficiencies of the plants were also determined.

Materials and Methods

Samples were collected before and after the processes that were likely to cause a change in Ra or Ba concentration. Total alkalinity, total hardness, and Ca concentrations were determined according to *Standard Methods for the Examination of Water and Wastewater* (15th ed., American Public Health Association, Washington, D.C., 1980). Total dissolved solids were measured using a conductivity meter, and Ba was measured with an atomic absorption unit. Chloride was measured using an ion analyzer and an ion chromatograph. The sludge and backwash waters were analyzed for Ra^{226} and Ra^{228} at Argonne National Laboratory by monitoring gamma radiation; the other samples were analyzed by the University of Iowa Hygienic Laboratory by a technique involving co-precipitation of the Ra with $BaSO_4$ and by alpha counting the precipitate with an internal proportional counter.

Radium Removal in Fe and Mn Removal Plants

Fe and Mn removal plants often treat their waters by aeration, detention, and filtration. Ra removal during aeration and detention has been observed, perhaps as a result of Ra adsorption on the Mn oxides or Fe hydroxides. The hydrous oxides of Fe (III) and Mn (IV) have high sorption capacities for bivalent metal ions. Removal of Ra by sand, coal, or greensand media filters is likely attributable to removal of oxides containing adsorbed Ra, or to accumulation of these oxides in the filter followed by sorption of Ra on their surfaces.

Our data, combined with similar data of others, show removal efficiencies that range from a negative value (obtained from a sample taken at the end of a filter run) to more than 50%; but values in the range of 5% to 15% are most common.

Moore and co-workers reported $MnO_{2(s)}$ to have a high affinity for Ra. High removals ($\sim$50% of an influent concentration of 14 pCi/L) were reported for Hersher, Illinois, where 0.5 mg/L manganese was being removed as $MnO_{2(s)}$ on the filter. Periodic low removals then would be expected after filter backwashing when the $MnO_{2(s)}$ was removed from the filter. Sufficient data are not available from this study to show the expected relationship, however.

Ra^{226} in the backwash water was found to be 21 pCi/L at Adair, Iowa, and 106 pCi/L at Stuart, Iowa. The average Ra removal indicated by backwash concentration was 11%, which was comparable with the removal figure obtained by measurement of influent and effluent concentrations. Total solids levels in the backwash waters were approximately 2,000 mg/L. These solids could be settled in a lagoon, and the supernatant could be recycled to the plant.

Radium Removal in Lime Softening Plants

The sludges from the lime softening plants samples in this study were discharged to a sanitary sewer (Bushnell, Illinois) or to lagoons (Elgin and Colchester, Illinois, and West Des Moines, Iowa). Sanitary sewers have been noted as being somewhat unfavorable, since they simply transfer any sludge contaminants from the water treatment plant to the sludge of the wastewater treatment plant. Radioactive contaminants in lagooned sludges must be considered when choosing methods for reclaiming the land from permanent lagoon sites. For instance, home building on such sites is not recommended because of the potential hazards from the radon gas.

The softening process may yield Ba and Ra removal by direct precipitation, coprecipitation, or adsorption. Other ionic compounds present in the water and the pH may influence the mechanism and the extent of removal. Removal efficiencies (Table 1) ranged from 43% to 96%, with an average of 75%.

The relationship between hardness and Ra has also been examined and is expressed in the following equation:

$$y = 0.4566\frac{y}{x} + 0.2275$$

where y = the total hardness removal fraction, and x = the Ra^{226} removal fraction. This equation describes the data reasonably well, as shown in Figure 1.

A combination of the data collected in this study with data from the literature shows Ra^{226} concentrations in the sludge ranging from 1,000 to 11,000 pCi/L of sludge. The concentration per dry gram of solids was 10 to 20 pCi. Backwash water concentrations ranged from 6 to 50 pCi/L. Although the relative quantities of Ra in backwash and sludge was not determined, most of the Ra appears in the sludge. A good estimate of the Ra in the sludge can be calculated by multiplying the difference in concentration between influent and effluent by the volume of flow through the plant.

Radium Removal in Ion Exchange Plants

Two ion exchange columns are used in the water treatment plant at Eldon, Iowa. Both columns are usually regenerated with a 100%-saturated (26-37%) by weight) solution of NaCl, but a shortage of available salt resulted in the decision to regenerate one column with a 40%-saturated (10-15% by weight) NaCl solution. Eldon's spent brine is discharged into a storm sewer, which then discharges to a river.

The Ra^{226} removal obtained by ion exchange at Eldon was 60%. However, the regeneration was performed with a limited amount of 40%-saturated salt solution and the regeneration was incomplete. This removal can be compared with 65% to 85% removals found for other plants that were incompletely regenerating their exchange media. The literature indicates that 81% to 97% Ra removal can be achieved in well-operated plants, and that Ra removal is directly related to hardness removal (see Table 2). Laboratory studies cited in the literature using both strong and weak acid resins have indicated that good removal occurred long after hardness breakthrough in both types of resin, although operation through several service cycles with incomplete regeneration may lead to earlier Ra breakthrough.

An earlier study of seven water treatment plants reported that maximum Ra^{226} concentrations in softener brine and rinse effluent ranged from 320 to 500 pCi/L. Iowa ion exchange wastes were found to contain 7.8 to 98 pCi/L of Ra^{226} in the backwash water, and rinse wastewaters ranged from 114 to 1,960 pCi/L of Ra^{226}. Less than 5% of the Ra was

Table 1. Ra and Hardness Removal Efficiencies in Lime Softening Plants

Location and Date	Ra^{226} in Raw Water (pCi/L)	Ra^{226} Removed (%)	Total Hardness Raw Water (mg/L as $CaCO_3$)	Total Hardness Removed (%)
W. Des Moines, Iowa:				
8/1/74*	9.3	75	376	49
6/5/78†	1.9	43	NA	NA
8/2/83‡	6.9	78	389	61
Webster City, Iowa:				
8/13/74*	6.1	85	507	48
2/20/75*	7.8	96	482	78
7/26/78†	1.4	60	NA	NA
Peru, Illinois:				
2/20/75*	6.5	92	329	47
2/25/75*	5.5	70	278	35
3/4/75*	5.5	76	286	57
Elgin, Illinois:				
3/7/75*	7.5	90	246	60
3/14/75*	5.7	86	243	54
3/21/75*	3.5	80	242	61
6/27/83‡	0.3	(-)	253	67
Colchester, Illinois:				
8/83‡	12.1	74	698	57
Bushnell, Illinois:				
8/83‡	12.6	45	354	42

**Data from EPA-600/2-77-073.*
†Data from JAWWA, 61:541, 619, and 681.
‡Data from this study.

removed during backwash. Wastewater volumes generated were 2% to 10% of the product water. Wastewater characteristics varied greatly from one plant to another.

Barium Removal in Lime Softening Plants

Barium removal is pH dependent, with an optimum pH occurring in the 9 to 11 range. At Elgin, 90% of the influent barium (4 mg/L) was removed at pH 9.3.

Barium Removal in Ion Exchange Treatment Plants

Crystal Lake, Illinois, has an ion exchange system composed of three separate wells with two strong acid ion exchange columns for each. The spent brine is discharged to a sanitary sewer system, which eventually discharges to the wastewater treatment plant. Data indicate that ion exchange plants typically remove 92% to 99% of the incoming barium. Laboratory studies have shown that hardness and Ba will break through at the same time for a strong acid resin after operation through several exhaustion-regeneration cycles.

The regeneration of Wells 6 and 8 removed 84% and 153% of the exchanged Ba, respectively. At Well 6, 6.5 kg of Ba was contained in the 9,500 gal of wastewater generated from one regeneration cycle. Likewise, Well 8 produced 30 kg Ba in its 10,000 gal of wastewater. Less than 2% of the barium was found in the backwash water, and the remainder was in the spent brine and rinse water. Other researchers report 85% removal or 18 kg Ba in 9,250 gal of wastewater.

Peak hardness (33,000 mg/L) and Ba (1,200 mg/L) concentrations in the spent regenerant coincide at Well 6 (Figure 2), but the peak hardness (44,000 mg/L) occurs before the peak Ba (5,000 mg/L) concentration at Well 8. This phenomenon may be caused by the greater amount of Ba on the column before generation at Well 8 (78.4 g Ba/ft^3) compared with that at Well 6 (40.4 g Ba/ft^3). More Ba accumulates on the resin during the service cycle at Well 8 because the raw water Ba concentration is higher and because less regenerant is applied to the column per unit volume of water produced during the service cycle.

Conclusions

Ion exchange plants, lime softening plants, and Fe and Mn removal plants can remove Ra from water with varying degrees of success. The Fe and Mn treatment plants removed 0% to 54% of the influent Ra, which ranged in concentration from 6.0 to 49 pCi/L. The lower removals were more common. The average removal, including values taken from the literature, was 23%, but the mean for this study was only 8%. The total Ra content of the waters from a single backwash was 0.02 to 7.6 μCi, and concentrations were typically less than 100 pCi/L. The average Ra removal by the plant was 11%. Though a relationship between MnO_2 accumulation on the filters and Ra removal was expected, sufficient data were not available to show this relationship. Treatment methods such as lime softening or ion exchange seem better suited than Fe and Mn removal plants for removing Ra from water supplies.

Lime softening effectively removes Ra from waters. The influent concentrations were 0.3 to 24.2 pCi/L. Typical removal values ranged from 43% to 92%, with an average of 69% if data from the literature are included. The Ra removal at the plants sampled during the summer varied from 45% to 78%, with an average of 75%. Two correlations presented in the report relate hardness removal to Ra removal and can be used to predict Ra removal efficiencies. The Ra concentrations in the softening sludges ranged from <1.2 to 21.6 pCi/g dry solids for Ra^{226} and from <2.4 to 11.7 pCi/g dry solids for Ra^{228}.

Ion exchange plants also produce an effluent with low Ra concentrations. The plant at Eldon removed 60% of the influent Ra of 47 pCi/L, but sufficient salt was not applied to completely regenerate the resin. Common removals reported in the literature are 65% to 90%, and the average approaches 85%. Ra concentrations in the brines at Eldon reached 217 and 1,144 pCi/L for NaCl brines at 40% and 100% saturation with NaCl, respectively.

Ion exchange and lime softening plants were also analyzed for Ba removal. Ion exchange at Crystal Lake removed more than 90% of the influent 9.5 and 16.1 mg/L Ba. Barium concentrations in the brine at Wells 6 and 8 averaged 328 and 1,300 mg/L, with peak values of 1,200 and 5,200 mg/L, respectively. Lime softening at Elgin also removed 90% of the 4 mg/L Ba in the influent water.

The full report was submitted in fulfillment of Cooperative Agreement No. CR-808912 by the University of Illinois at Urbana-Champaign under the sponsorship of the U.S. Environmental Protection Agency.

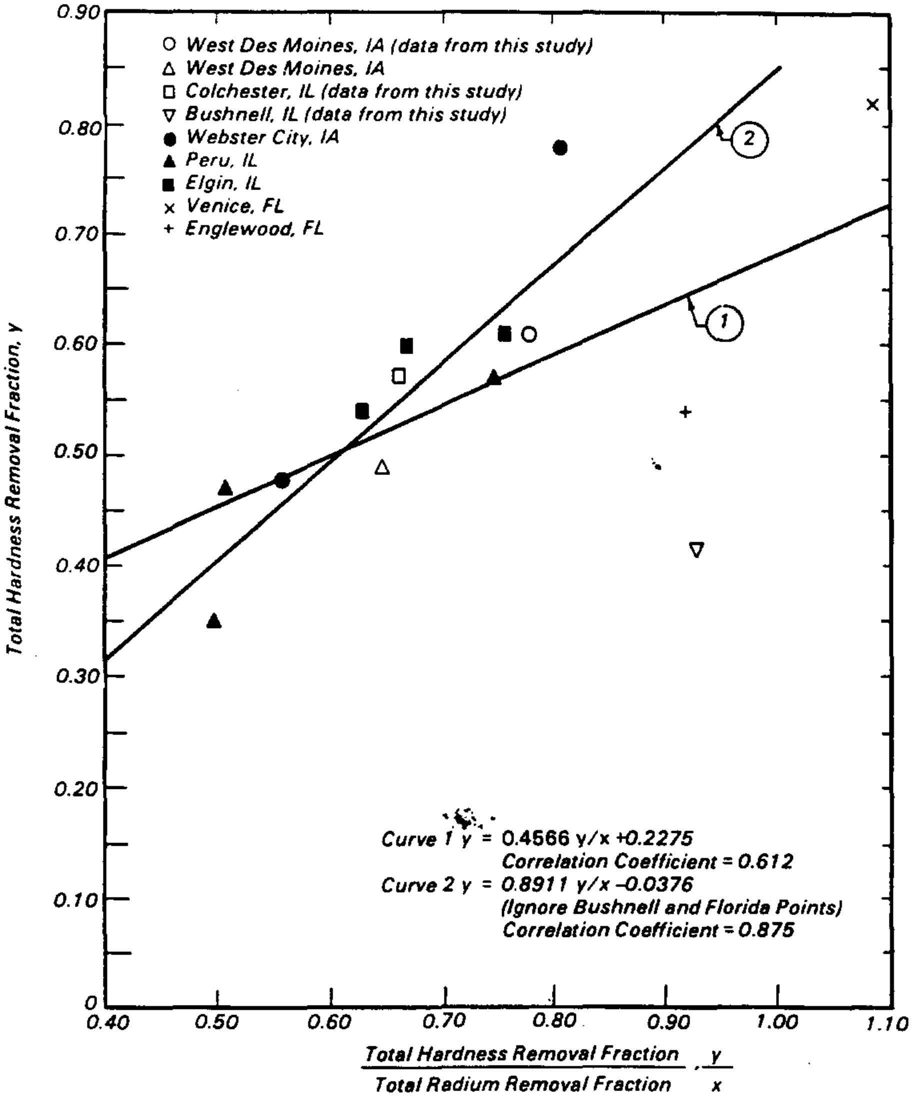


Figure 1. *Proposed correlation between Ra^{226} and total hardness removal fractions for lime softening plants.*

Table 2. Hardness and Ra^{226} Removals in Ion Exchange Plants

Location and Time of Removal	Ra^{226} in Influent (pCi/L)	Ra^{226} in Effluent (pCi/L)	Ra^{226} Removed (%)	Total Hardness in Raw Water (mg/L as $CaCO_3$)	Total Hardness in Product Water (mg/L as $CaCO_3$)	Total Hardness Removed (%)
Hersher, Illinois:*						
3/25/75:						
Just after regeneration	6.64	1.25	81.2	412	18	95.6
Midpoint	6.94	0.42	93.9	427	12	97.2
Near breakthrough	6.88	2.07	69.9	417	184	55.8
Dwight Correctional Center, Illinois:*						
2/13/75-2/14/75:						
Just after regeneration	3	0.4	88.0	286	16.0	94.3
Midpoint	3	0	92.5	284	4.1	98.6
Near breakthrough	3		70.7	279	131.0	53.7
Eldon, Iowa:†						
8/83:						
Just after regeneration‡	NA	14.2	66.5	NA	175	50
Midpoint	42.4	NA	NA	350	NA	NA
Near breakthrough	NA	20.1	52.6	NA	232	33.7

*Data from EPA-600/2-77-073.
†Data from this study.
‡40% saturated brine.

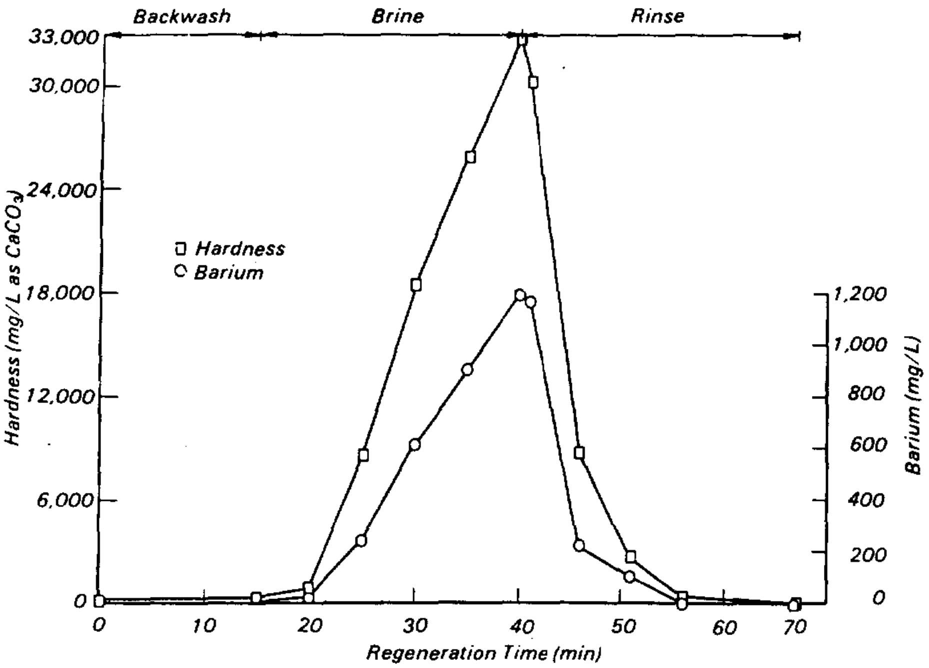


Figure 2. Barium and hardness concentrations in spent brine at Crystal Lake, IL Well #6.

1. Aeration for radon and carbon dioxide gas removal.
2. Chemical clarification, including settling and filtration for iron and manganese removal.
3. Ion exchange for radium and hardness removal.
4. Chlorination and water stabilization.
5. Removal of radium from ion exchange regeneration water by RSC resin.
6. Infiltration/evaporation (I/E) disposal pond for plant wastewaters.

The problem of radium in groundwater, which serves as the raw water supply for the development, is common for many communities in the United States. If the development of new water sources that do not have a radium problem is not possible or economically feasible, then a treatment process for radium removal needs to be considered. This report concerns itself with one treatment alternative and not with locating new raw water sources that are free of radium.

The treatment of well water for the removal of radium is not practiced to any great extent in the water treatment field. However, the ion exchange process using standard water softening type resins for radium removal is well documented. The Redhill Forest water treatment system incorporates a new process for concentrating the radium removed by the ion exchange process to simplify the final radium disposal problem. The regeneration water from the ion exchange process passes through a bed of RSC resin to remove the high levels of radium before the wastewater is discharged to the I/E pond for final disposal. There are no known water treatment systems like the Redhill system. The RSC resin has been used on a trial basis at several locations primarily in Texas and one site in Wyoming. In all these cases, raw water from the wells was passed directly through the RSC bed with radium levels up to about 100 Ci/L.

Experimental Procedures

Raw water from two wells is pumped through a countercurrent flow aeration tower located at the booster pump house. The purpose of the aeration process is to remove dissolved gases, specifically radon and carbon dioxide, from the raw water. The water is pumped to the treatment plant at a rate of about 90 to 100 gpm for further water treatment to remove iron, manganese, radium, and hardness prior to chlorination and discharge to the water distribution system.

As the raw water enters the treatment plant, alum, potassium permanganate, and a polyelectrolyte are added to remove iron and manganese by chemical precipitation. The treatment unit is a prefabricated self-contained unit that includes a mixing and flocculation chamber, tube settlers, and multi-media filtration. The effluent from the iron and manganese removal process is further treated to remove radium and hardness in a ion exchange system that uses a cation resin. The effluent from the ion exchange system is chlorinated and zinc hexame taphosphate added to control corrosion and sequester any residual iron before being pumped to the treated water storage tank. The radium removed from the water supply in the ion exchange process is removed from the regeneration brine by passing the brine through a separate treatment process in which the radium is permanently complexed on the RSC material. The wastewater from this process along with the backwash wastewater from the iron removal process is pumped to the final disposal I/E pond. Figure 1 is a schematic diagram of the processes presented above.

Ultimate Disposal of Wastewater and Radium Removal from Water Supply

The original concept and design approved by the Colorado State Health Department for ultimate disposal of waste generated at the treatment plant are as follows:

Plant Wastewater

All wastewater from the plant operation is discharged into an I/E pond. The main purpose of the pond is to allow for rapid infiltration of plant wastewater into a geologic formation, which dips steeply to the east and is located beneath the geologic formation in the area of the raw water supply wells. The deep wells obtain the raw water from this formation to supply the development.

Radium Waste

Most of the radium removed from the raw water entering the treatment plant is eventually complexed on the RSC resin. As needed, RSC resin is replaced and transported to an approved hazardous/radiological waste disposal facility for final disposal.

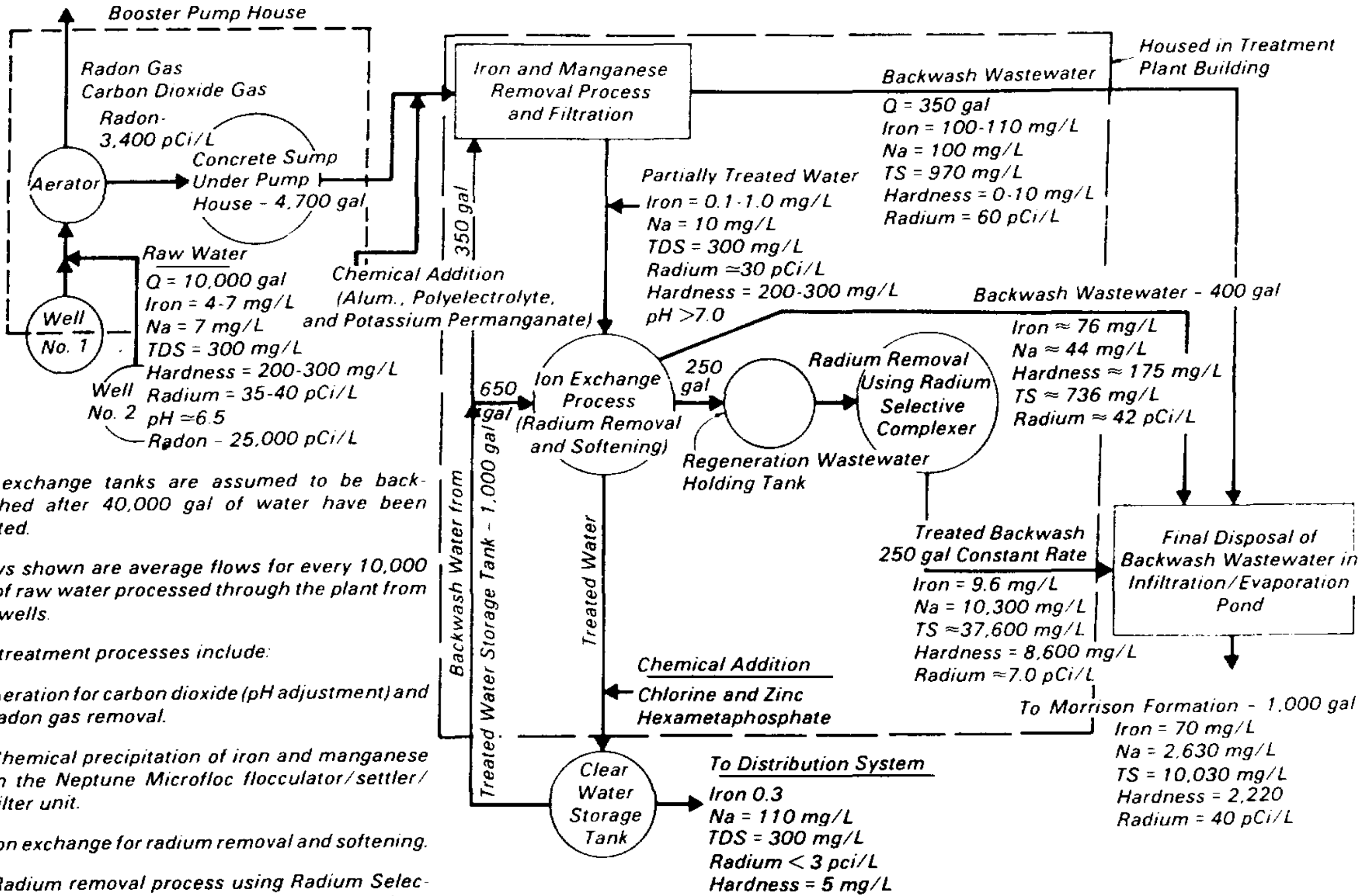
Sampling and Analyses

The project generally consisted of in-depth monitoring of the operation of the full-scale Redhill Forest water treatment plant over a 21-month period from October 1985 through June 1987. All water quality parameter concentrations were determined according to *Standard Methods for the Examination of Water and Wastewater (15th Edition).*

Most of the water quality analysis work was performed by Hazen Research Laboratory, a commercial lab in Golden, CO. Some analysis work was performed by the EPA Laboratory in Cincinnati, OH, and some radon gas analyses were performed by Lowry Engineering in Maine.

In-depth monitoring included water quality sample collection and laboratory analyses, field measurements, flow measurement, and detailed plant operation and was performed to evaluate the following components of the treatment plant operation:

1. Aeration system for radon removal. Water samples were collected and analyzed for radon concentration in the raw water before and after aeration.
2. Treatment system for iron and manganese removal. Samples were collected and analyzed on the raw water inflow to the process and the effluent from the system to assess the efficiency of operation. The water samples were typically analyzed for iron, manganese, gross alpha, gross beta, and radium 226. The process wastewater from backwash operations was also analyzed on several occasions to determine the composition of the wastewater discharged to the I/E pond for final disposal. Parameters of primary interest for the wastewater included total iron, manganese, solids, and radium 226.
3. Ion exchange process for radium and hardness removal. Water samples were collected for the inflow and outflow to the unit process. The samples typically were analyzed for iron, manganese, sodium, hardness, gross alpha, gross beta, and radium 226. Water samples were collected from the backwash, regeneration, and quick rinse water on several occasions.
4. Radium Selective Complexer process for radium removal. This process was monitored frequently to determine the efficiency of radium removal from the ion exchange process wastewater and the buildup of radium in the complexer resin. Environmental radiation monitoring of the area outside the RSC tank surface was done to determine the exposure and to relate the exposure to radium buildup on the complexer resin.



Notes

1. *Ion exchange tanks are assumed to be backwashed after 40,000 gal of water have been treated.*
2. *Flows shown are average flows for every 10,000 gal of raw water processed through the plant from the wells.*
3. *The treatment processes include:*
    a. *Aeration for carbon dioxide (pH adjustment) and radon gas removal.*
    b. *Chemical precipitation of iron and manganese in the Neptune Microfloc flocculator/settler/filter unit.*
    c. *Ion exchange for radium removal and softening.*
    d. *Radium removal process using Radium Selective Complexer (RSC). Removes radium from the ion exchange backwash wastewater and concentrates radium on complexer resin.*
4. *The normal treatment plant flowrate is about 100 gpm. The water from the ion exchange process for radium removal and softening is discharged into the wastewater holding tank from which the wastewater is pumped through the RSC at a constant rate for radium removal.*

Figure 1. *Flow diagram of water treatment plant processes.*

5. I/E pond monitoring of the sand and soils was done to determine the extent of radium buildup due to the disposal of plant wastewater containing small amounts of radium.

General plant monitoring of plant flow rates, volumes of water processed, wastewater volumes, etc., was performed for use along with water quality data in determining plant process efficiencies, plant operation and maintenance costs, etc.

Some radon gas measurements were conducted on site using a RDA-200 Radon/Radon Daughter Detector unit manufactured by EDA Instruments, Inc. Also, some samples were collected and sent to Lowry Engineering for additional radon gas analysis.

Results and Conclusions

Figure 1 shows the flow volumes for each part of the total system operation for an assumed raw water flow volume of 10,000 gal into the plant. Also presented are the average water quality data for each component that makes up the treatment plant.

The aeration system has been proven to effectively remove radon and carbon dioxide gases from the raw water supplied by the deep wells. Carbon dioxide gas has been typically reduced from about 125 to 25 mg/L in the aeration system. The reduction of radon gas has been about 85% from about 23,000 pCi/L in the raw water to about 3,400 pCi/L in the effluent from the aeration system. Additional measurements have indicated that the radon gas concentration in the treated water from the main treatment plant is about 600 pCi/L. The iron removed about 13% of the radium from the inflow to this process. When the iron removal system was backwashed, the radium removed was wasted in the I/E final disposal pond. Based upon the results of the monitoring of the backwash water, the average concentration of radium in the wastewater was about 60 pCi/L.

The ion exchange system removes radium, hardness, and residual iron and manganese through the use of a standard cation exchange resin. The process has been very effective in removing radium, hardness, and residual iron, and in polishing the effluent from the iron removal process as long as the ion exchange capacity is not exceeded. The monitoring results generally indicate radium 226 levels of less than 3 pCi/L and iron levels of less than the recommended maximum level of 0.3 mg/L. Frequent monitoring of the system operation has indicated that the radium breakthrough occurs between 40,000 and 45,000 gal (i.e., 178 to 200 resin bed volumes). The quality of the influent to and effluent from the ion exchange process has generally been as given in Table 1.

The RSC system is designed and operated to remove radium from the ion exchange process wastewater and to permanently concentrate the radium on the complexer resin. On July 10, 1986, new RSC resin was placed in the complexer tank and a detailed program of monitoring the flow rate and the water quality of the inflow and outflow was initiated. Table 2 presents a summary of some of the results of the monitoring from July 10, 1986, up through June 1987. It should be noted that the flow rate through the column has been about 22 gpm, which is equivalent to the surface loading rate of about 10 gpm/ft^2. The RSC resin bed depth is 2 ft.

It can be seen in Table 2 that the RSC resin is highly radium selective with generally over 99% removal of radium from the influent wastewater. Average data for the water quality parameters included in Table 2 are shown on the bottom of the Table. The average inflow and outflow water quality data indicate that iron, sodium, hardness, and total solids are virtually unchanged in passing through the resin whereas over 99% of the radium in the influent is removed and concentrated on the RSC resin. Also shown on the bottom of Table 2 is the total quantity of radium removed and concentrated on the resin from July 10, 1986, to June 10, 1987. Based upon the operation of the plant during this time, the rate of radium buildup on the RSC resin is about 347 μCi/yr (347 X 10^6 pCi/yr).

Further, it has been determined that the rate of radium removed from the raw water and permanently complexed on the RSC resin is about 9.6 μCi (9.6 X 10^6 pCi) per 100,000 gal of water treated at the plant. After some period of operation, the RSC resin containing radium will be removed from the RSC tank and replaced with new resin and the old resin will be disposed of at a Nevada waste disposal site. It is anticipated that the RSC resin will be replaced when the radium on the complexer reaches about 3,080 μCi (3,080 X 10^6 pCi). The 4 ft^3 of RSC resin will then be placed in a 55-gal drum, 3.35 ft^3 of concrete will be added, and the entire drum will be transported to Nevada for final disposal. This method of handling the radium waste will ensure that the total radium content of the container to be buried will not exceed 10 nCi/g (i.e., 10,000 pCi/g).

Finally, plant operating costs have been determined and estimated in Table 3.

The full report was submitted in fulfillment of Cooperative Agreement No. CR-812691-01-0 by the Redhill Forest Property Owners Association under the sponsorship of the U.S. Environmental Protection Agency.

Table 1. *Summary of Quality of Water to and from Ion Exchange Process*

Parameter	*Influent*	*Effluent*
Flow rate, gpm	*90 to 100*	*90 to 100*
Iron, mg/L	*0.15 to 2.7*	*0.03 to 0.5*
Manganese, mg/L	*0.4 to 1.3*	*0.01 to 0.15*
Sodium, mg/L	*7.4 to 12.5*	*40 to 150*
Hardness, mg/L as $CaCO_3$	*212 to 350*	*5 to 70*
Radium 226, pCi/L	*22 to 35*	*0 to 4*

Table 2. *Summary of Water Quality Data for Regeneration Wastewater from Ion Exchange Regeneration Through RSC Resin* (Effluent Discharged to I/E Pond)*

Date	Accumulated Volume Treated gal	Bed Volumes	Sample	Parameters					Total Radium 226 pCi/L	% Radium Removal
				Iron mg/L	Manganese mg/L	Sodium mg/L	Hardness mg/L	Total Solids mg/L		
7/10/86	0	0								
			Inflow	2.48	23.8	11,600	476	34,900	860 ± 30	
			Outflow	0.98	16.7	13,300	245	34,600	16 ± 11	98.1
7/30/86	2,400	77								
			Inflow	2.03	31.8	11,000	9,850	41,700	1280 ± 40	
			Outflow	1.56	32.2	11,000	10,200	41,800	1.6 ± 3.2	99.9
8/31/86	9,460	305								
			Inflow	9.0	33.1	12,600	11,500	54,200	1400 ± 40	
			Outflow	8.5	33.1	12,700	11,600	55,200	9.4 ± 3.5	99.3
9/29/86	14,600	471								
			Inflow	7.21	30.5	11,400	8,350	37,600	920 ± 30	
			Outflow	7.15	31.5	11,500	8,420	37,600	4.1 ± 2.4	99.6
10/30/86	22,600	729								
			Inflow	2.79	33.1	8,170	9,380	35,000	860 ± 50	
			Outflow	2.07	33.5	8,640	10,100	35,300	5.3 ± 2.8	99.4
11/26/86	27,700	894								
			Inflow	7.17	28.2	13,400	9,620	45,400	1040 ± 30	
			Outflow	6.30	26.9	13,300	9,520	45,500	8.1 ± 3.3	99.1
1/14/87	39,550	1,276								
			Inflow	31.4	19.2	9,350	7,260	31,300	1070 ± 60	
			Outflow	27.8	18.1	9,000	7,740	30,400	8.4 ± 2.3	99.2
2/21/87	49,700	1,603								
			Inflow	61.3	30.5	12,300	10,900	49,800	1780 ± 80	
			Outflow	62.2	32.4	12,100	11,400	50,400	7.2 ± 7	99.6
3/18/87	57,700	1,861								
			Inflow	93.4	31.2	13,500	11,600	53,200	2000 ± 80	
			Outflow	92.0	31.9	12,400	12,600	53,300	18 ± 9	99.1
6/10/87	71,700	2,313								
			Inflow	8.08	17.4	8,070	5,580	28,100	650 ± 20	
			Outflow	3.76	15.8	8,460	4,940	28,300	9.2 ± 2.4	98.6
Averages			Inflow	19.8	29.9	10,850	8,890	40,590	1181	
			Outflow	18.5	27.7	10,760	9,030	40,550	9.0	99.2

Note: From 7/10/86 to 6/10/87 (i.e., 355 days), 71,700 gal of plant wastewater was treated in RSC tank. The following is the amount of radium removed and deposited in the resin.

Radium removed = 71,700 gal (3.785 L/gal) (1181-9.0 pCi/L)
= 318.1 X 10^6 pCi
= 318.1 μCi about 0.949 μCi/day
Estimate for year = 347 μCi

*Resin bed volume = 4.15 ft^3 (31.0 gal)

Table 3. *Summary Treatment Plant Operating Costs*

	Item	*Cost/1,000 gal of Water Treated*
1.	*Plant Chemicals, Alum, Permangante, Chlorine, etc.*	*$0.137*
	Salt	*$0.475*
2.	*Energy Costs*	*$0.206*
3.	*RSC Resin Disposal (includes disposal and new resin)*	*$0.088*
	Total	*$0.906**

*Operator cost not included.

RADIUM-226 AND RADON-222 IN DOMESTIC WATER
OF HOUSTON-HARRIS COUNTY, TEXAS

a Cech, Mengistu Lemma and Howard M. Prichard,
University of Texas Health Science Center at Houston,
ol of Public Health, Houston, Texas

les W. Kreitler
University of Texas, Bureau of Economic Geology, Austin, Texas

RODUCTION

Ingestion or inhalation of radium (Ra) and products of its
y are considered a health risk. In the human body bones,
loid stem cells, and lungs are particularly sensitive to radia-
exposure. Epidemiologic studies have established a strong
ciation between certain occupational and therapeutic exposures
isotopes of Ra and bone cancer [1-5]. Radon (Rn)-222, an
orne progeny of decay of Ra-226, has been associated with lung
cer in miners [6,7], and recently has received national atten-
because of high concentrations observed in houses in some
ts of the United States [8,9]. Studies have also raised the
ue of cancer risk associated with Ra and Rn in domestic water
12].

Regulatory standards for radioactivity in water have been
blished for Ra but not Rn. The combined activity of the two
topes of Ra (226 and 228) should not exceed a maximum contami-
level (MCL) of 5 pCi/ℓ [13,14]. Rn in water is not yet
lated by Federal and State standards, although this is expected
change in the near future.

Ra regulation under the Safe Drinking Water Act of 1974 and
ated amendments went into effect in 1977, but because of the
er number of public water systems in a State as large as Texas,
assessment of Ra concentrations and compliance with EPA and
te regulations is a formidable task, not yet completed. In
ition, approximately 4 million people in Texas use private water
plies which are not covered by current regulations and for which
on Ra are not available. No regulatory agency now routinely
tors Rn, in either public or individually-owned supplies.

Deposits of uranium, the progenitor of Ra-226 and Rn-2 occur in Texas [15]. The major deposits occur in Eocene and youn formations and it is believed to be predominantly associated w volcanic ash in the Catahoula formation of the Miocene Age [16,1 In 1984, taking advantage of the cost-effective testing proced available at the University of Texas at Houston, School of Pub Health laboratory, a study was initiated by Cech co-investigators [18], in cooperation with several ground wa districts and river authorities in Texas, to assess the geograp distribution of Ra-226 and Rn-222 in domestic water in selec regions of this State. Anomalously high Ra and Rn concentrat were found in the Gulf Coast area including Harris Co (Figure 1); Ra in excess of 20 pCi/ℓ and 3000 pCi/ℓ for Rn observed in some public water supplies in this county.

Harris County, with a 1980 population of 2,684,000, has largest number of people among Texas counties and contains percent of the State population. The water supply and distribu system in Harris County is complex. The City of Houston a operates over 200 wells. A water well field may contain from 11 wells, and pump water from various depths into several coll ing plants. The eastern half of the City of Houston, roughl percent of the population of over one million, is supplied prin ly by surface water from Lake Houston (Figure 2); the western is supplied by ground water. The boundaries of areas serve each well, well field, or by surface water cannot be de accurately because systems are interconnected; these bound vary with variations in demand, pressure in water lines, se and even time of day.

Population growth and the rapid development of new sub subdivisions on the fringes of the City of Houston result numerous smaller water utilities. Approximately 300 such muni utility districts (MUDs) and 1700 community water systems s ground water in Harris County besides the City of Houston. Mo the wells with Ra concentrations that exceed MCL were from smaller utilities.

Approxmately 150,000 people in the rural parts of H County, and an estimated 10 percent of urban residents, are plied by water from individually-owned wells. Northwest, east, and southwest Harris County contain undeveloped areas. projections of urban growth indicate that these areas wi developed in the near future. Pockets of Ra-rich water fo MUD's in these parts of Harris County suggest that future su sions might encounter problems. However, no reliable guidan well location is presently available to well developers i County.

A more detailed sampling of ground and surface wat necessary in this area to assess patterns of distribution of Rn, so that some predictive tool could be developed to present and future water works. Thus, during 1985 and 19 intensive sampling for Ra-226 and Rn-222 was conduct Houston-Harris County.

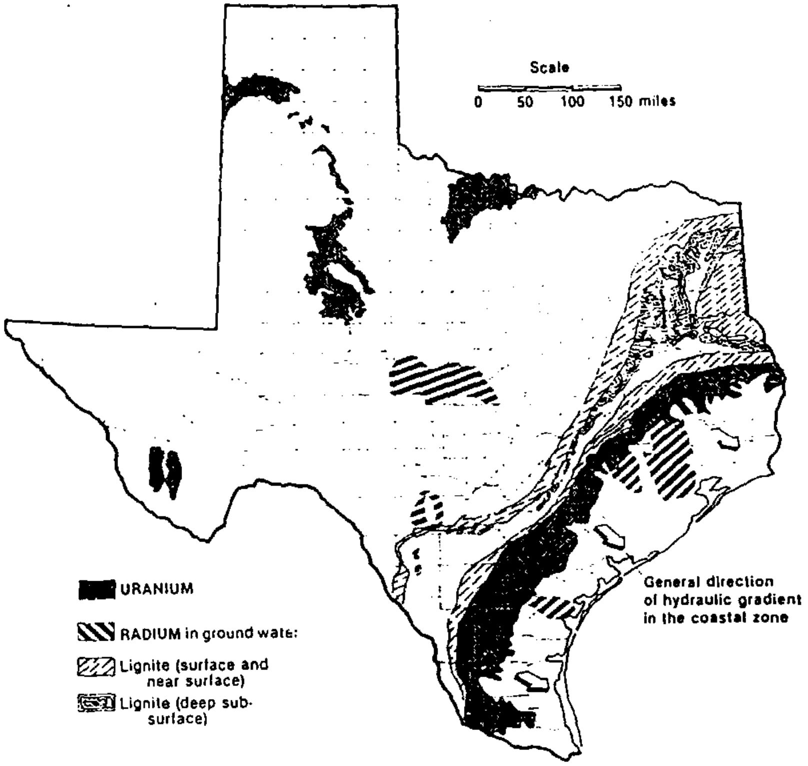


Figure 1. Radioactive deposits in Texas. Compiled by superim posing Ra-226 field data gathered by Cech et al [18] on a map of uranium deposits adapted from Kie et al. [15]. Additional sources were Texas Wate Development Board [19] and Brock [20].

METHOD

One hundred sixteen samples were collected, 77 of them fr distribution systems and 39 directly from wells. Of the sampl collected from distribution systems, 27 were taken from vario small municipal utilities districts in northwest Harris County. the 39 samples taken directly from wells, 26 were from southwe Houston. The rest of the samples were collected from public a individually owned wells throughout the County. Several sampl were collected in neighboring Fort Bend and Montgomery Counties estimate regional concentrations.

As a first step in this investigation, samples were gathe from residential and commercial dwellings. Testing of water at household tap involved visiting residences where permission

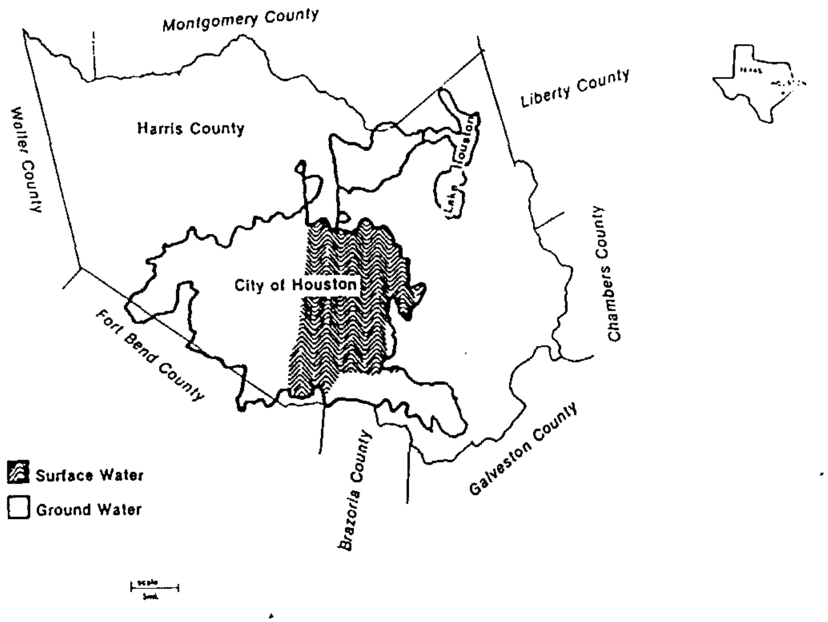


Figure 2. Greater Houston-Harris County, Texas: Approximate boundaries for surface water-ground water service areas, 1980.

secured from heads of households to collect one-liter water samples. Duplicate samples were taken to assure the reproducibility of the laboratory results.

Analyses for Ra-226 and Rn-222 were conducted with modifications of liquid scintillation techniques described in [21]. Ra-226 was extracted from the 0.5 to 1 liter water samples by passage through a 10 ml bed of cation resin (Dowex 50W-X8, 20-50 mesh). The resin beds were rinsed with distilled water, transferred to a 20 ml liquid scintillation vials and covered with distilled water. A 10 ml layer of toluene containing 5 g/l of 2,5-diphenyloxazole was added, and the vials were sealed and stored for at least a week prior to analysis. Three to twelve hours before the analysis, the vials were vigorously shaken to ensure adequate contact between the aqueous and the organic phases. The Rn content of the organic phase was then assayed using a liquid scintillation counter, with windows optimized for Rn in toluene. Because Rn originally present in the water sample is not retained in the rinsed resin, any Rn detected in the assay was due to ingrowth from Ra entrained in the resin.

Radon-222 concentrations were determined by drawing 15 ml samples directly from a source into a clean syringe. Care was taken to prevent aeration of the samples in the process. The samples were then injected beneath 5 ml layers of a mineral oil-based scintillation solution in 20 ml vials. The vials were vigorously shaken to promote phase contact, held for at least three hours to permit Rn daughter ingrowth, and assayed with a liquid scintillation counter. The results were corrected for the amount of Rn decay between sampling and assay.

The information obtained on this phase of the investigation was mapped using the computer technique SYMAP [22], as shown on Figures 3 and 4, and this helped to identify areas where further sampling, directly from wells, could provide additional, clarifying information. The subsequent step was to identify wells in the distribution network which caused Ra and Rn to appear in the residential water. This was a straightforward task when small single-source utilities were concerned. More difficult was to pinpoint Ra and Rn sources within interconnected multiple-well municipal systems.

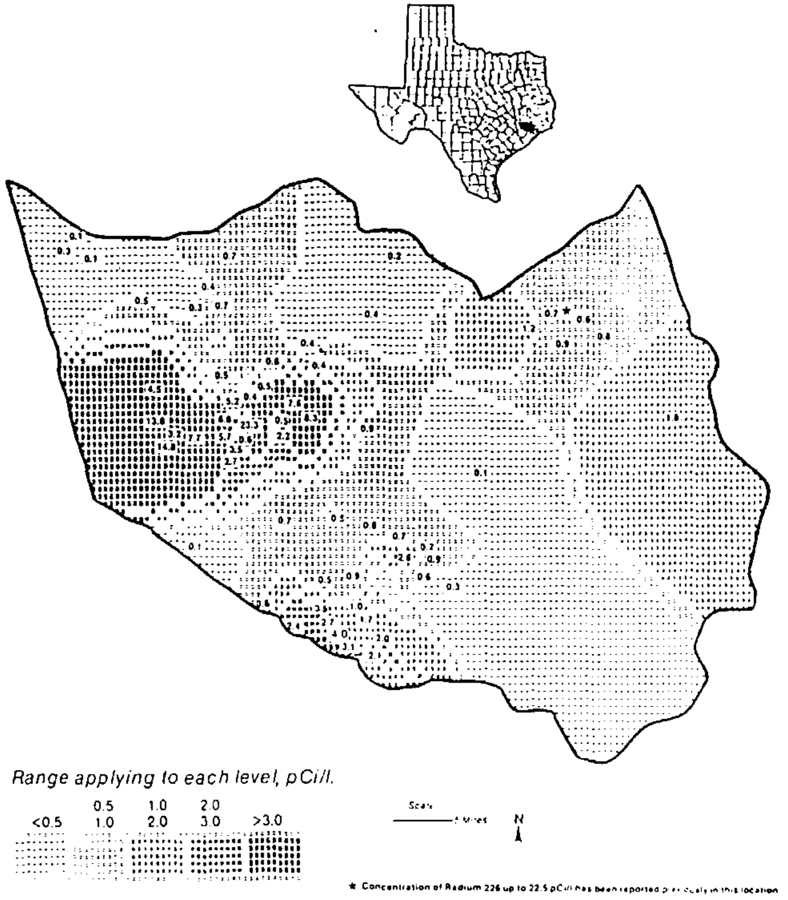


Figure 3. Concentrations of Ra-226 observed in tap water in Harris County, Texas, 1985-1986.

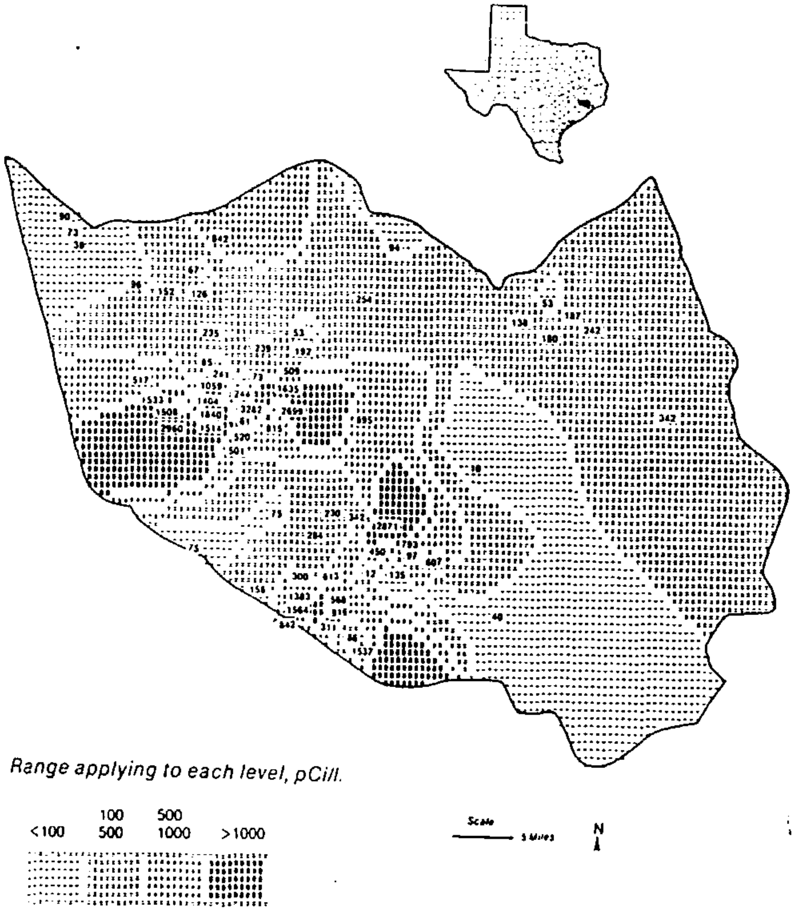


Figure 4. Concentrations of Rn-222 observed in tap water in Harris County, Texas, 1985-1986.

In order to assess variations in Ra and Rn as related to location and depth of wells, a subset of 64 points representing a diverse geographic distribution of water sources was formed and subjected to statistical analysis. This subset included 39 samples taken directly from wells and 25 other points taken from service areas of small utilities where the source-well definitely was known. The point-source locations are shown in Figures 5 and 6 (except for additional points to the north, in adjacent Montgomery County).

The general hypothesis investigated in this study was that Ra and Rn concentrations are a function of a) pumping depth, b) distance from uranium deposits in sandstone aquifers, and c) some modifying local structural features, in particular, the proximity

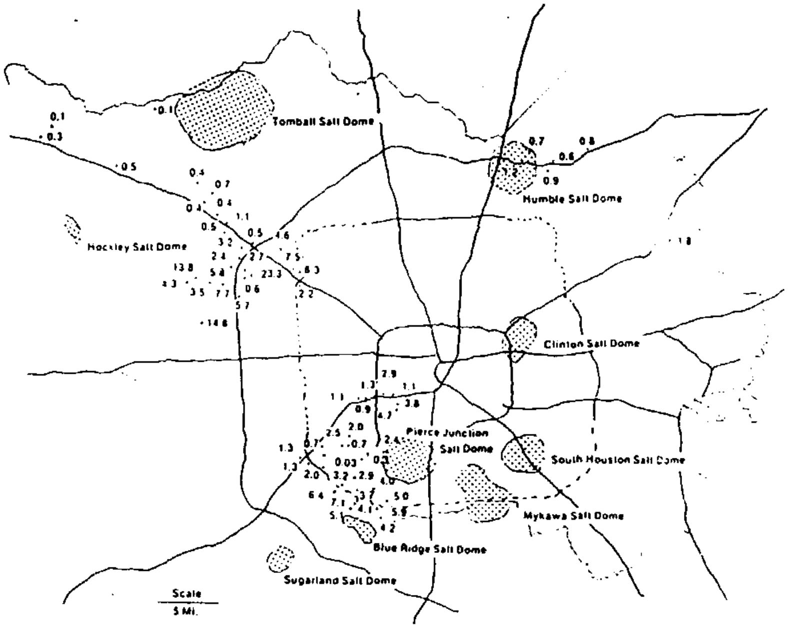


Figure 5. Point-source measurements of Ra-226, Harris County, Texas, 1985-1986. Samples were obtained either directly at well heads or from single-source municipal utility districts. Shown also are locations of salt domes.

of salt domes. The variables that might have influenced the distribution of radiochemicals were defined as follows:

Water-Bearing Formation and Pumping Depth

Depth to water sources varied from zero for surface water to about 610 meters (m) for wells. The data on depths of wells and water-producing formations were derived from the materials of the United States Geological Survey, [23]. Because many wells in the Gulf Coast region have multiple screens, the well depth in the statistical analyses was represented by upper and lower limits of pumping ranges.

Location

Well location was represented by the distance (downdip) from a reference line chosen approximately where known uranium deposits occur [15]. On the Gulf Coast, the uranium-bearing formations crop out parallel to the coast northwest of the study area and roughly

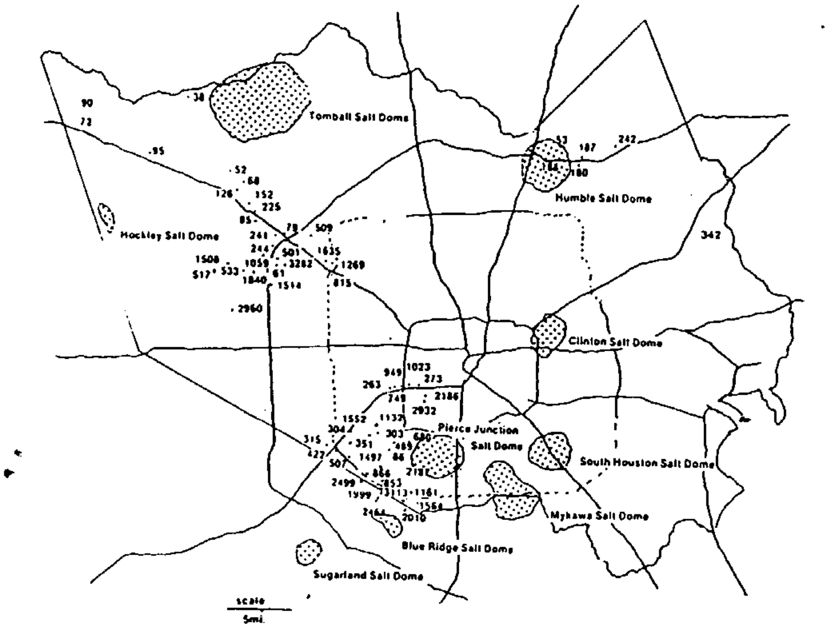


Figure 6. Point-source measurements of Rn-222, Harris County, Texas, 1985-1986.

coincide with the recharge zone of the Gulf Coast aquifers. The distance from the outcrop to the well was measured in the southeast direction to correspond with the general direction of hydraulic gradient [24-28].

Salt Domes

There are numerous salt domes in the study area (Figure 5). The proximity of salt deposits is a pertinent regional feature. Because the general trend in ground water movement may have been interrupted locally by pumping, the distance to salt domes was measured as the distance to the nearest dome, regardless of direction.

RESULTS

Figures 3-6 present Ra and Rn concentrations observed in the distribution systems and at the well heads in Greater Houston-Harris County in 1985-86. Concentrations of Ra and Rn were below limits of detection in the southeast part of the study area which is supplied by surface water, whereas detectable concentrations of these radioisotopes were present in virtually every sample of well water. In northwest and southwest Harris County ground waters exhibited concentrations greater than trace levels of Ra and Rn; several wells in these areas had concentrations in excess of the maximum allowable for Ra.

The major "hot" spot was located in northwest Harris County. Up to 23 pCi/ℓ of Ra-226 (more than four-fold in excess of the maximum allowable) and up to 3300 pCi/ℓ of Rn-222 were observed in an area geographically adjacent to Logenbaugh and White Oak Creeks, about 13 to 16 kilometers (km) south-southeast of the salt domes near Hockley and Tomball.

The second anomaly, with concentrations up to 7 pCi/ℓ of Ra-226 at the well heads and up to 4 pCi/ℓ in the distribution system, was found on the border between Harris and Fort Bend Counties (far southwest Houston). Of the 11 wells in this section, five had concentrations of Ra-226 greater than 5 pCi/ℓ (the maximum allowed for total Ra), and the lowest concentration (one well) was about 3 pCi/ℓ. Rn concentrations in this area were up to 3100 pCi/ℓ. These wells were developed near the piercement-type salt dome, Blue Ridge. Some of the wells were located in the adjacent Fort Bend County but belonged to the City of Houston (Figures 5 and 6).

In the southwest-central part of Harris County, Rn concentrations up to 2900 pCi/ℓ were observed near the salt dome at Pierce Junction. Ra-226 concentrations up to 4.7 pCi/ℓ were present at the well heads and about 3 pCi/ℓ was observed in the distribution system. In northeast part of Harris County, a concentration of 22 pCi/ℓ of Ra-226 was encountered earlier by others in a well developed near a piercement-type salt dome at Humble [29].

Figure 7 shows variations in the concentration of Rn-222 in well water as related to the concentration of its progenitor, Ra-226. The coefficient of correlation (R) between these two variables was 0.77 and the slope of linear regression was positive and significant (probability p of this correlation to be found by chance was less than 0.01). The empirical equation relating waterborne concentration of Rn to Ra was

$$\hat{Rn} = 307 + 187 \ Ra, \qquad /1/$$

where $\hat{Rn}$ is an estimate of Rn, in pCi/ℓ.

The proportion of variation in Rn concentrations which could be accounted for by Ra was 59 percent. This shows that Ra and Rn measured under field conditions correlated reasonably well, although not perfectly. The remaining scatter probably was due to differences in the underground retention opportunities for gas Rn and dissolved Ra. This, in turn, implies that our understanding of the distribution of Ra and Rn in water supplies may further profit from exploring the roles that pumping depth and well location play in this distribution.

The relationship of Ra and Rn concentrations to pumping depth was investigated first by fitting linear regression models to respective data sets. For Ra-226, the slope of regression was not significant, which indicated that the linear model was inappropriate, and this becomes immediately apparent when one inspects the scatter-plots on Figures 8a and 9a. On each of these figures, a

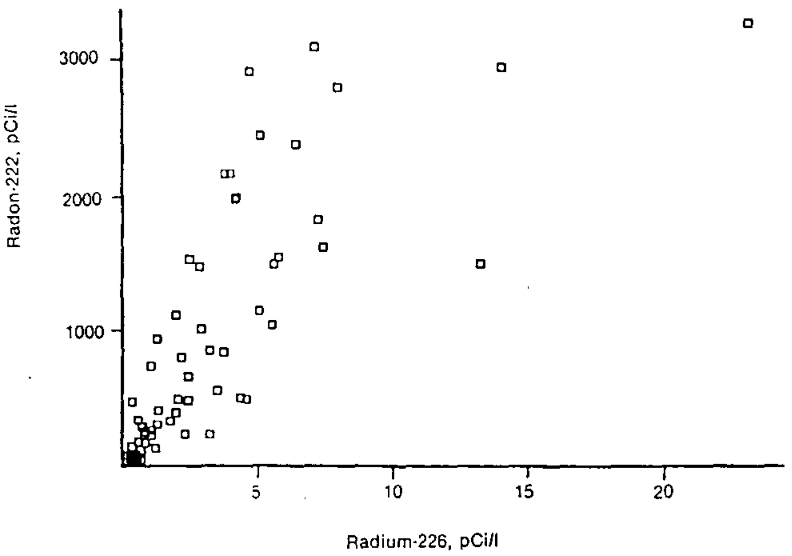


Figure 7. Relationship between concentrations of Ra-226 a Rn-222 in well water, Harris County, Texas, 1985-198

non-linear distribution of Ra with depth was evident. The conce trations were observed to peak in a zone between 180 m (Figure 8 and 320 m (Figure 9a), which corresponded to a fringe zone betwe Chicot and Evangeline aquifers (Figure 10a). Further increase depth caused concentrations of Ra to decrease, although not zero. This distribution suggests introducing a quadratic term in a previous model. If this is done, the resulting fit becom statistically significant ($p < 0.05$).

The greatest concentrations of Rn also were observed betwe 180 and 320 m below the surface (Figures 8b-10b) but, different from Ra, there was an overall underlying tendency for Rn conce trations to increase with depth. The linear model was appropria to describe the vertical concentration profile for Rn ($p < 0.0$ and the addition of a quadratic term to the linear model was not improvement.

The distance from the outcrop of known uranium deposits w not a statistically significant predictor for Ra, nor did t addition of a quadratic term to a linear model improve the fi Examination of the scatter-plot on Figure 11a indicates that distribution along the northwest to southeast axis downdip fr uranium deposits was bi-modal, with the first increase in conce trations observed at about 90 km southeast from a reference li and the second, after 115 km. The location of the first anoma coincided with the area of geological faulting stretching betwe the Hockley and Tomball salt domes, about 13 to 16 km southeast the domes [30]. The location of a second, comparatively le severe Ra anomaly coincided with the area of the fault syst associated with the Blue Ridge salt dome.

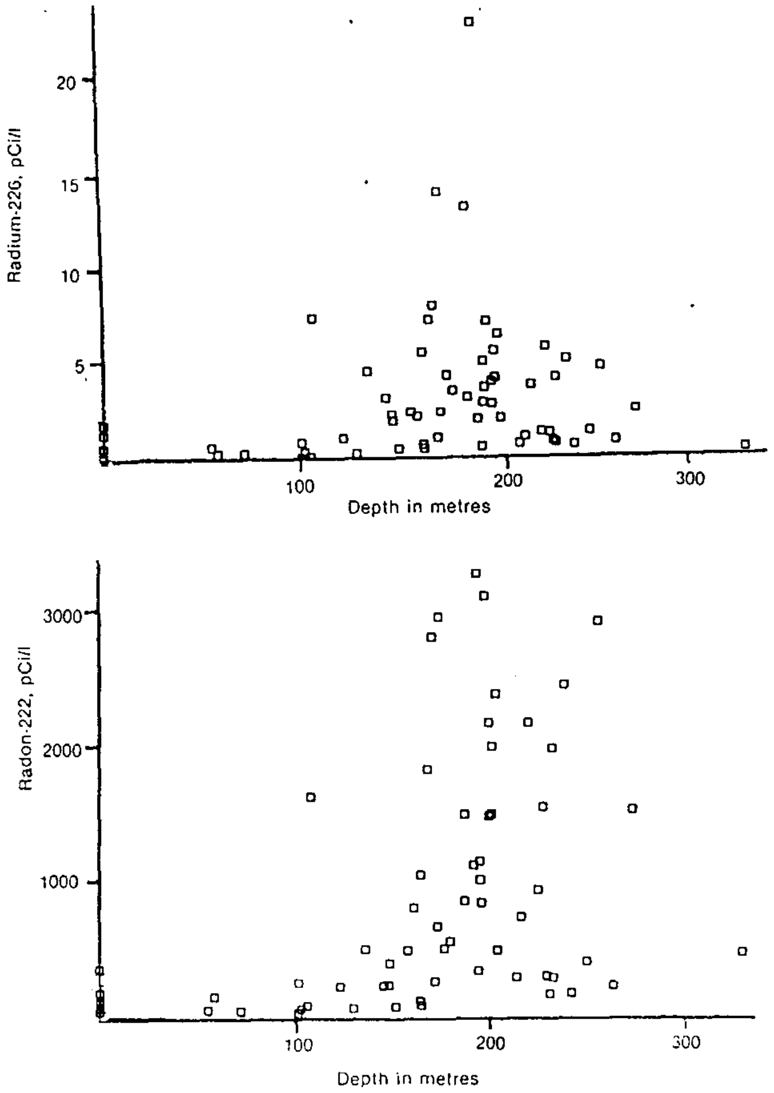


Figure 8. Distribution of (a) Ra-226 and (b) Rn-222 with depth; upper limit of pumping range.

These data indicate that Ra travels in a more complex path than could be expressed by a linear distance from known deposits of uranium and their Ra in Houston-Harris County water wells probably relates to local sources, instead of distant deposits in the outcrop.

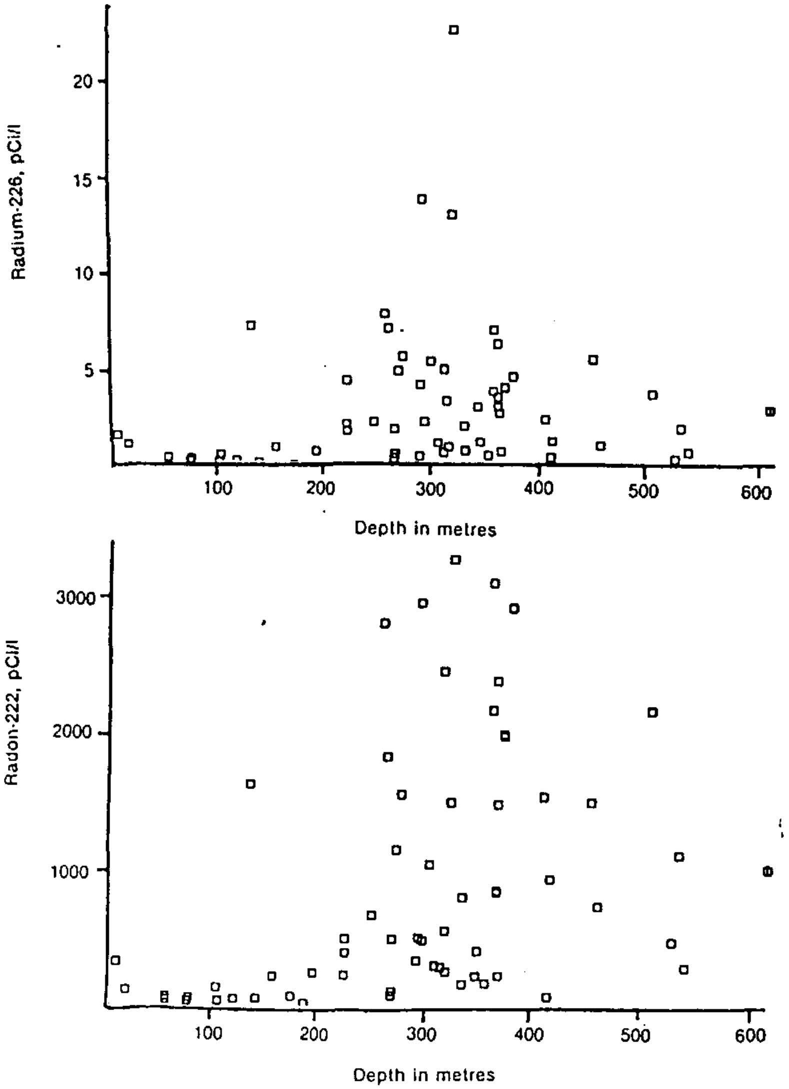


Figure 9. Distribution of (a) Ra-226 and (b) Rn-222 with depth; lower limit of pumping range.

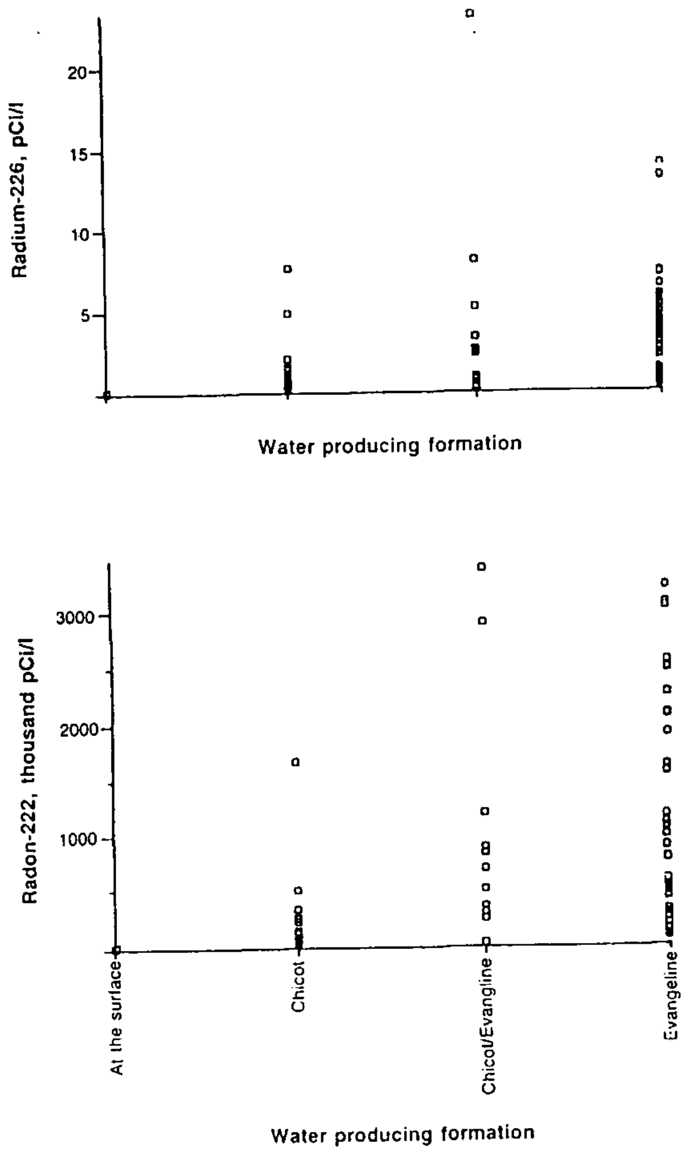


Figure 10. Distribution of (a) Ra-226 and (b) Rn-222 depending on water-producing formation.

The bi-modal distribution with distance was also noted for Rn (Figure 11b) but, again, differently from Ra, there was an underlying tendency for Rn concentrations to increase toward the coast. The linear model fitted to this distribution had a positive slope and was statistically significant ($p < 0.01$).

The proximity of a salt dome was an important predictor for the presence of both Ra and Rn in well water. The inverse relationship, the increase in concentrations as distances to the domes decreased, was particularly evident for wells developed near Blue Ridge and Pierce Junction salt domes (Figures 12a and b). Because of the rural location, no deep municipal wells were present in the vicinity of the Hockley and Tomball salt domes and this fact might

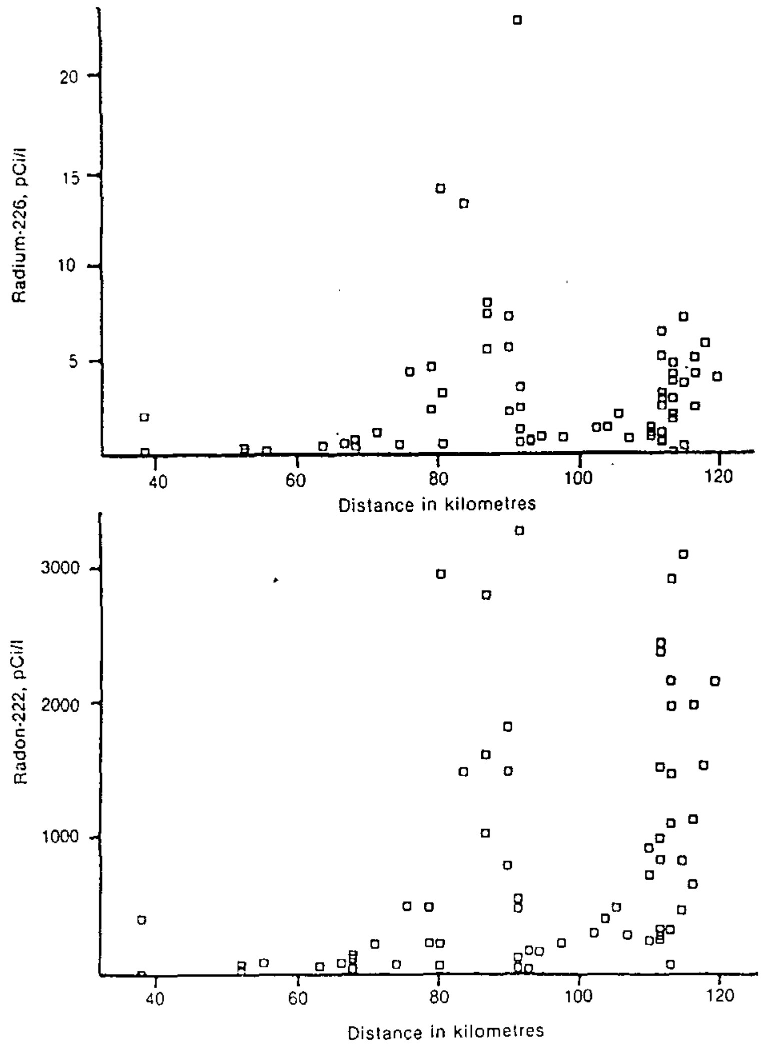


Figure 11. Distribution of (a) Ra-226 and (b) Rn-222 with the distance from the outcrop of known uranium deposits.

have obscured the statistical association. The correlation of Ra with other salt domes in Harris County could not be made because they were situated in a part of Harris County which was supplied by surface water.

Multivariate analyses showed that a combination of factors provided still a better fit. Where Ra is concerned, both pumping depth and the distance from the nearest salt dome were significant

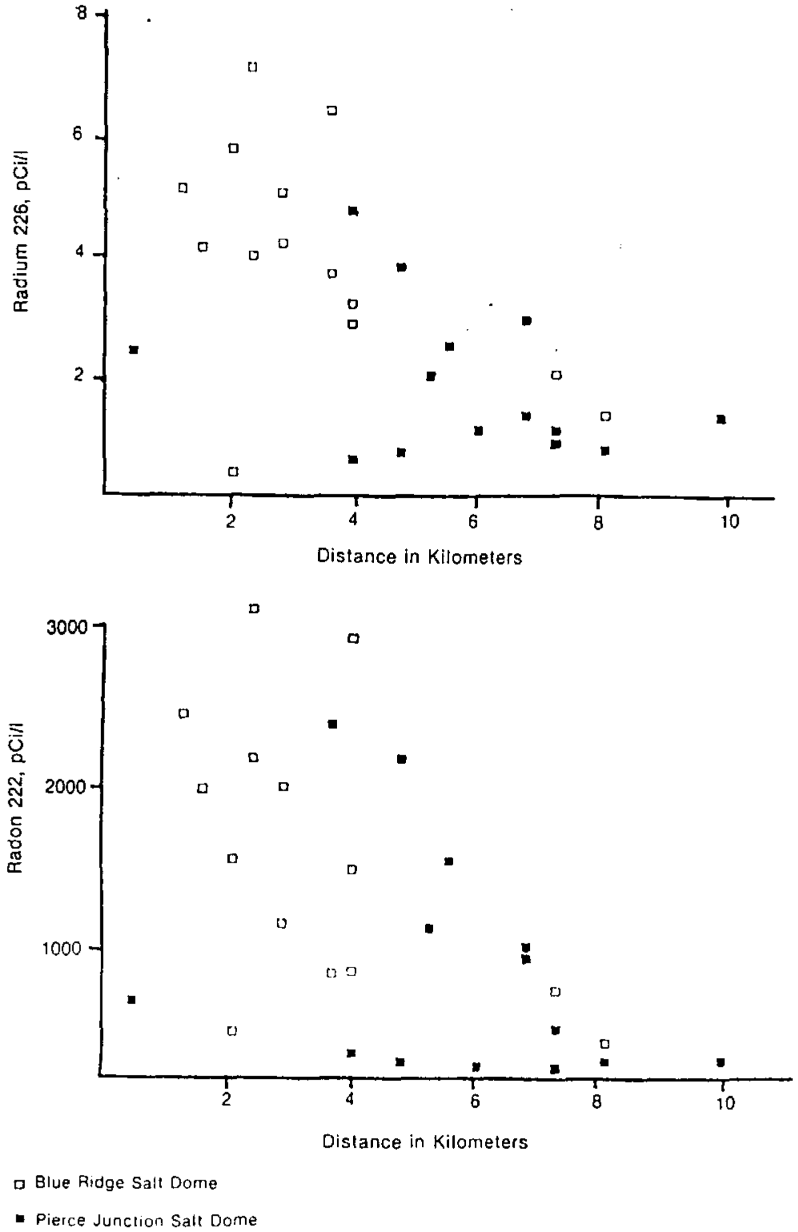


Figure 12. Distribution of (a) Ra-226 and (b) Rn-222 with the distance to Blue Ridge and Pierce Junction salt domes.

factors ($p < 0.01$) and the equation relating waterborne concentrations of Ra to these factors was

$$\hat{Ra} = 0.06 + 0.04(DEPTH) - 0.09\text{x}10^{-3}(DEPTH)^2 - 0.60(DNS) + 0.04(DNS)^2, \qquad /2/$$

where (DEPTH) stands for pumping depth (m) measured to the top screen of pumping range and (DNS) was the distance from the nearest salt dome (km). Because the proximity of salt domes was measured as the distance (km) from the dome nearest to each well and regardless of direction, the model in equation 2 includes quadratic terms for this variable as well as for the variable of depth. The statistical analysis supporting equation 2 is illustrated in Table 1.

Table 1. Multiple Regression Analysis Report for Ra-226 vs. Pumping Depth and Location.

		Sequential				Last	
Source	df	Sum-Sqr	R^2	F-Ratio	Tail Prob	Sum-Sqr	F-Ratio
Constant	1	632.33					
DEPTH	1	42.37	4.67	4.66	0.037	55.43	5.83
$(DEPTH)^2$	1	50.73	10.25	5.33	0.023	21.15	2.22
DNS	1	141.47	25.83	14.88	0.001	50.96	5.36
$(DNS)^2$	1	112.44	38.21	11.82	0.000	112.44	11.82
Model	4	347.0188	38.21	9.12			
Error	59	561.0715					
Total	63	908.0902					
Mean Square Regression		86.7569					
Mean Square Error		9.509686					

Parameter Estimation

Variable	Regression Coefficient	Standard Coefficient	Standard Error
Constant	5.98E-02	0	0
DEPTH	3.91E-02	.7475541	1.62E-02
$(DEPTH)^2$	8.66E-05	-.4710227	5.81E-05
DNS	-.62	-.9650517	.27
$(DNS)^2$	4.49E-02	1.42079	1.30E-02

DEPTH - Depth of pumping, top screen, m;

DNS - Distance from salt domes, km

The R^2 equal to 38 indicated that two basic factors, pumping depth and distance from salt domes, accounted for 38 percent of the variations in Ra.

An alternative expression, for depth measured to the bottom of the pumping range, was

$$\hat{Ra} = -0.13 + 0.02(DEPTH) - 0.03\text{x}10^{-3}(DEPTH)^2 - 0.61(DNS)^2 + 0.04(DNS)^2. \qquad /3/$$

For Rn, the best combination of variables was

$$\hat{Rn} = -413 + 4(DEPTH) + 11(DDR) - 197(DNS) + 12(DNS)^2 \qquad /4/$$

for the top of the pumping range and

$$\hat{Rn} = -190 + 2(DEPTH) + 10(DDR) - 216(DNS) + 12\ (DNS)^2 \qquad /5/$$

for the bottom of the pumping range.

The variable (DDR) in equations 4 and 5 represents distance (km) downdip from the recharge zone, as explained in the "Method" section. The statistical analysis that led to derivation of these equations is illustrated in Table 2. The combination of three basic variables describing pumping depth and location of water source accounted for 42 percent of the variations in Rn.

If data on Ra concentrations are available, the Rn concentrations can be estimated as

$$\hat{Rn} = -367 + 171(Ra) + 2(DEPTH) + 9(DDR) - 104(DNS) + 4(DNS)^2 \qquad /6/$$

for the top of the pumping range, and

$$\hat{Rn} = -266 + 173(Ra) + (DEPTH) + 8(DDR) - 112(DNS) + 5(DNS)^2 \qquad /7/$$

for the bottom of the pumping range. This combination of factors described 74 percent of variations in Ra (Table 3).

DISCUSSION

The Gulf Coast is underlain by productive ground water aquifers, an important factor which has facilitated the urban and industrial development of the Greater Houston-Harris County area. For years all of the water needs in the City of Houston and Harris County have been satisfied entirely by ground water. However, years of pumping large volumes of water have caused a decline in

Table 2. Multiple Regression Analysis Report for Rn-222 vs. Pumping Depth and Location

		Sequential				Last	
Source	df	Sum-Sqr	R^2	F-Ratio	Tail Prob	Sum-Sqr	F-Ratio
Constant	1	5.41E+07					
DEPTH	1	9408528	17.62	18.08	0.000	4851181	9.32
DDR	1	2793209	22.85	5.37	0.023	2443481	4.69
DNS	1	2863555	28.22	5.50	0.012	5003873	9.61
$(DNS)^2$	1	7618718	42.49	14.64	0.001	7618717	14.64
Model	4	2.27E+07	42.49	10.90	0.000		
Error	59	3.07E+07					
Total	63	5.34E+07					
Mean Square Regression		5671002					
Mean Square Error		520469.					

Parameter Estimation

Variable	Regression Coefficient	Standard Coefficient	Standard Error
Constant	-413.01	0	0
DEPTH	4.37	.345	1.43
DDR	10.72	.25	4.95
DNS	-197.16	-1.27	53.58
$(DNS)^2$	11.79	1.54	3.08

DEPTH - Depth of pumping, top screen, m;

DDR - Distance downdip from recharge km;

DNS - Distance from salt domes, km

the potentiometric pressure and subsidence of the land, especially in southeast Houston [31]. Since the 1950's the prevailing philosophy behind water development in Houston has been that a) reorientation of the water supply system is necessary from total dependence on ground water to a combination of surface and ground water sources; and that b) wells need to be redistributed to reduce the rate of subsidence, phasing out most of the wells in the southeast and developing new wells in northwest Harris County, closer to the recharge zone.

In 1954 the first surface water from the San Jacinto River via Lake Houston was provided to the east side of Houston. On the other hand, the development of ground water has been actively pursued in northwest, northeast, and southwest Harris County to accommodate urban expansion in these directions

Table 3. Multiple Regression Analysis Report for Rn-222 vs. Ra-226, Pumping Depth, and Location

		Sequential				Last	
Source	df	Sum-Sqr	R^2	F-Ratio	Tail Prob	Sum-Sqr	F-Ratio
Constant	1	5.41E+07					
RA	1	3.19E+07	59.76	134.10	0.000	169E+07	71.06
DEPTH	1	3578880	66.47	15.04	0.002	779210	3.23
DDR	1	2591628	71.23	10.89	0.002	1614991	6.79
DNS	1	568185	72.38	2.39	0.124	1307976	5.50
$(DNS)^2$	1	944674	74.14	3.97	0.048	944674	3.97
Model	5	3.96E+07	74.15	33.28	0.000		
Error	58	1.38E+07					
Total	63	5.34E+07					
Mean Square Regression		7918282					
Mean Square Error		237935.6					

Parameter Estimation

Variable	Regression Coefficient	Standard Coefficient	Standard Error
Constant	-366.87	0	0
Ra	170.83	.704	20.26
DEPTH	1.82	.14	1.01
DDR	-8.73	.20	3.35
DNS	-104.07	-.67	44.39
$(DNS)^2$	4.49	.59	2.26

Ra - Radium 226, pCi/ℓ

DEPTH - Depth of pumping, top screen, m;

DDR - Distance downdip from recharge km;

DNS - Distance from salt domes, km

Data gathered by Brock [20] in northwest Harris County indicated that at least 12 MUD's, those serving recently developed suburban subdivisions, violated standards with respect to Ra in the public drinking water.

Our findings confirmed excess Ra in the water served by these relatively small utility districts. In addition, water from five wells belonging to the City of Houston-proper (former MUDs which were acquired by the City to accommodate newly added subdivisions on the far southwest) contained Ra at concentrations greater than 5 pCi/ℓ.

Observed excess Ra was associated with ground water. Concentrations in water from surface sources were less than 0.5 pCi/ℓ and no surface-water violations for Ra were found.

Earlier attempts to correct contaminated wells in northwest Harris County involved plugging the lower-most screens, because of an assumption that Ra concentration increased progressively with depth. The present study, however, shows that this assumption is justified for Rn, but not for Ra. The linear model that was initially applied in an attempt to describe Ra variations with depth was found to be inadequate and a quadratic term for nonlinearity was required to achieve a better fit. Concentrations of Ra tended to peak between 180 and 320 m below the surface. This depth suggests that Ra might be associated with the upper Evangeline - lower Chicot aquifers. Under such conditions plugging the deepest screens in the multiple screen wells may cause Ra concentrations to become greater instead of smaller.

The increase of Rn with depth, unlike Ra, may be related to differential retention, accumulation, and diffusion properties of these radioisotopes. Rn is a short-lived, highly volatile radioactive gas whereas Ra is a dissolved cation. For an equal amount of paternal Ra, its gaseous progeny Rn may have a better chance to accumulate in the deepest strata, while it may dissipate more readily from progressively shallower formations. Water-bearing strata in the study area dip toward the coast and, hence, water wells in the southeasterly direction tend to be progressively deeper. The trend for Rn concentrations to increase in the direction of the Gulf may relate to this factor of increasing depth.

The Ra and Rn in ground water from the Evangeline and Chicot aquifers may indicate that waters have flowed around the flanks of salt domes that pierce the aquifers. Uranium deposits have been found in the rocks that flank or overlay Gulf Coast salt domes, such as the Palangana dome [32] and the Hockley dome [33], and may be the source for the Ra and Rn. The uranium presumably is precipitated in the reducing environment surrounding the dome. As ground water flows past the dome it entrains the soluble Ra but leaves the insoluble uranium.

Natural ground-water flow in the Gulf Coast aquifer is down the stratigraphic dip toward the coast at a rate of approximately 1 m per year [34,35]. However, the extremely heavy pumpage in the Houston area has created an extensive cone of depression [36] and reversed the direction of flow. The presence of high concentrations of Ra hydrologically updip from the Pierce Junction dome (Figure 5) further suggests this reversal. The dissolved Ra has been transported up to 4 km over 30 years (the approximate age of the cone of depression); the rate of 130 m per year is significantly greater than natural ground-water flow rates of 1 m per year.

An alternative hypothesis for the source of Ra and Rn is that they originated from uranium deposits in the deeper formations and migrated up the dome flanks and associated faults into shallower formations.

The major deposits of uranium in the Gulf Coast are associated with the Catahoula formation of the Miocene age. This formation, which in South Texas is mined commercially for uranium, extends along the entire coast from Mexico to Louisiana. The Catahoula Sandstone is also the deepest fresh-water bearing unit in the Gulf Coast aquifer complex. It crops out inland from the study area and dips toward the Gulf at a rate of about 17 m per km; near the coastline it is approximately 1.6 km below the land surface [28].

The formations which constitute the Gulf Coast aquifer range in age from the Oligocene and Miocene eras for the Catahoula sandstone to the Quarternary period for shallow alluvium. The actual production of well water involves strata of the Pliocene-Pleistocene age (Evangeline and Chicot aquifers) rather than the Miocene age. The presence of Ra and Rn in well water derived from the Evangeline and Chicot aquifers may be evidence of a cross-formational flow where deeper water with greater potentiometric head from Miocene strata leaks upward through confining strata [24;27;37;38]. Such seepage would be most prominent up the flanks of piercement-type salt domes or associated faults.

The vertical concentration profile exhibited by Rn supports the hypothesis of an upward flow from greater depths. However, the non-linear distribution with depth observed for Ra and a rather narrow range of pumping depths in which Ra concentrations occur suggest a source in the upper Evangeline aquifers. In addition, any Ra from a Catahoula source should have decayed to below detection levels considering the long flow paths downdip from a uranium deposit in the Catahoula to a dome, up the dome flanks, and then into the Evangeline and Chicot aquifers. Because the half-life of Ra-226 is 1600 years, Ra will decay to concentrations below detection limits in a ground water flow system if travel times for the Ra are more than a few thousand years. The areas of high Ra and Rn concentrations in the Evangeline aquifer are 80 to 120 km downdip from its outcrop. At a flow rate of 1 m per year, these waters are older than measurable by Ra concentrations. If a flow path through the Catahoula is envisioned, travel time will be even greater.

Ra as a ground water tracer may give conservative ground water flow velocities because of its chemically reactive nature. Dissolved Ra should function chemically similarly to dissolved calcium. Tanner [39] proposed that in saline waters the abundant cations compete with Ra for exchange sites and thus a greater portion of Ra ions remains in solution. Conversely, waters with low ionic strengths are favorable for Ra adsorption onto the aquifer matrix. Kraemer and Reid [40] studied Ra in deep saline formation waters and observed this direct correlation between salinity and Ra activity. In the Evangeline aquifer the migration of Ra may be retarded by ion exchange or conversely enhanced if dome dissolution increases the salinity around a dome.

In our study, the proximity of salt domes was found to be a strong predictor of Ra and Rn presence in well water, particularly in combination with a certain range of pumping depths. Whether Ra is related to uranium associated with salt domes or to brine leakage up the flanks of salt domes cannot be answered at this time. We advise, however, against developing domestic wells near salt domes, especially medium-to-deep wells (180 m and deeper).

The health significance of Ra and Rn-rich water in Harris County is not known. All the water wells in question were located in subdivisions that have existed for only about 10 to 15 years. In northwest Harris County the customers were notified by the respective MUD's that Ra levels in their water were in excess of MCL. In southwest Houston, where a blend of water from severa wells is distributed, the city, technically, is not under obligation to notify customers regarding Ra concentrations present in specific component-wells, as long as the levels in the distribution system are not in excess of the maximum allowable.

Blending water may not be the most reliable option, however, for dealing with Ra and Rn, since resultant concentrations vary depending on several, not entirely predictable factors, i.e., how many wells are in operation at any given time, the pumping rate of each contributing well, pressure in water lines, and demand. The boundaries of service areas for each particular component in the distribution system are not known. It would be advisable, instead, for the City of Houston to take steps toward replacing, re-working, or treating problem wells.

The EPA has published two booklets advising homeowners on methods to reduce indoor concentrations of Rn [8,9]. In these documents, users of private wells were informed that domestic water might contribute Rn to the indoor air. However, the EPA said this usually is not a problem when large community water supplies are concerned, since water supposedly releases most of its Rn before reaching individual houses. In our study in Houston-Harris County, parallel measurements of Rn made at municipal well heads and at consumers' taps indicated that the *en route* self-aeration of well water does not occur. There was no evidence of appreciable drop in Rn concentrations in municipal well water upon its arrival at household taps.

Water treatment for Ra and Rn is available. The technology of Rn and Ra removal does not differ much from other gases or alkaline earth elements, i.e., well-known and common technology for removal of calcium or magnesium hardness. The economics of rehabilitation by lime or lime-soda softening, weak acid ion exchange, or reverse osmosis have been reviewed recently by Snoeyink et al. [41] and Brock [20]). To date, some MUD's in Harris County have been successful in reducing levels of Ra and Rn by altering screening depths in their wells.

SUMMARY

This study showed that anomalous concentrations of Ra-226 and Rn-222 exist in parts of the upper Texas Gulf Coast. While all factors influencing the distribution and fate of pollutants in underground porous media might not be easy to measure or even identify, often the very expectation of this great complexity prevents one from observing patterns that otherwise might be quite helpful. The findings summarized in this communication present an encouraging picture for predicting depths and locations in the Gulf Coast where elevated Ra and Rn concentrations may be encountered. Two to four key variables accounted for the statistically significant variation (from 38 to 74 percent) in concentrations observed under field conditions.

ACKNOWLEDGEMENTS

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ENVIRONMENTAL RESEARCH BRIEF

Low-Cost/Low-Technology Aeration Techniques For Removing Radon From Drinking Water

N.E. Kinner, C.E. Lessard , G.S. Schell, and K.R. Fox

ABSTRACT

Simple low-cost/low-technology aeration techniques were investigated to determine their effectiveness in removing radon from drinking water. The techniques consisted of flow-through storage and minimal aeration in various configurations, and were found to be effective in varying degrees for the reduction of radon. These low-cost/low-technology aeration techniques may be easily applied in small communities.

INTRODUCTION

In an effort to assemble information concerning simple treatment techniques for the removal of radon from drinking water, the University of New Hampshire and the New Hampshire Department of Environmental Services (NHDES), through a U.S. EPA Cooperative Agreement, evaluated low-cost/low-technology aeration treatment techniques for radon removal. All the techniques involved storage and/or storage with minimal aeration. These tests included monitoring radon reduction in a distribution system, radon release from an open air storage tank with no mixing (still pool of water), radon reduction in a flow-through reservoir system with various influent control devices, and radon reduction in a flow-through reservoir system with minimal bubble aeration.

DISTRIBUTION SYSTEM

The site selected to evaluate radon loss in a distribution system was the Rolling Acres Trailer Park in Mont Vernon, NH. The trailer park consists of 33 mobile homes served by 2 wells that are currently being treated using granular activated carbon (GAC) to remove radon. The distribution system consists of 3.8 cm (1.5 in.) to 5.1 cm (2 in.) diameter pipe. Distances between sampling locations were measured with a surveyor's tape (Figure 1). Samples were taken in the evening from kitchen taps at 5 homes located at various distances from the pump house. The first two sets of samples were of GAC-treated water. During the next two sampling periods, raw water was pumped directly into the distribution system. The greatest reduction (18.8%) in radon concentration occurred at the sampling point furthest from the pump house (Table 1), but overall the reductions observed were very low (0% to 10%).

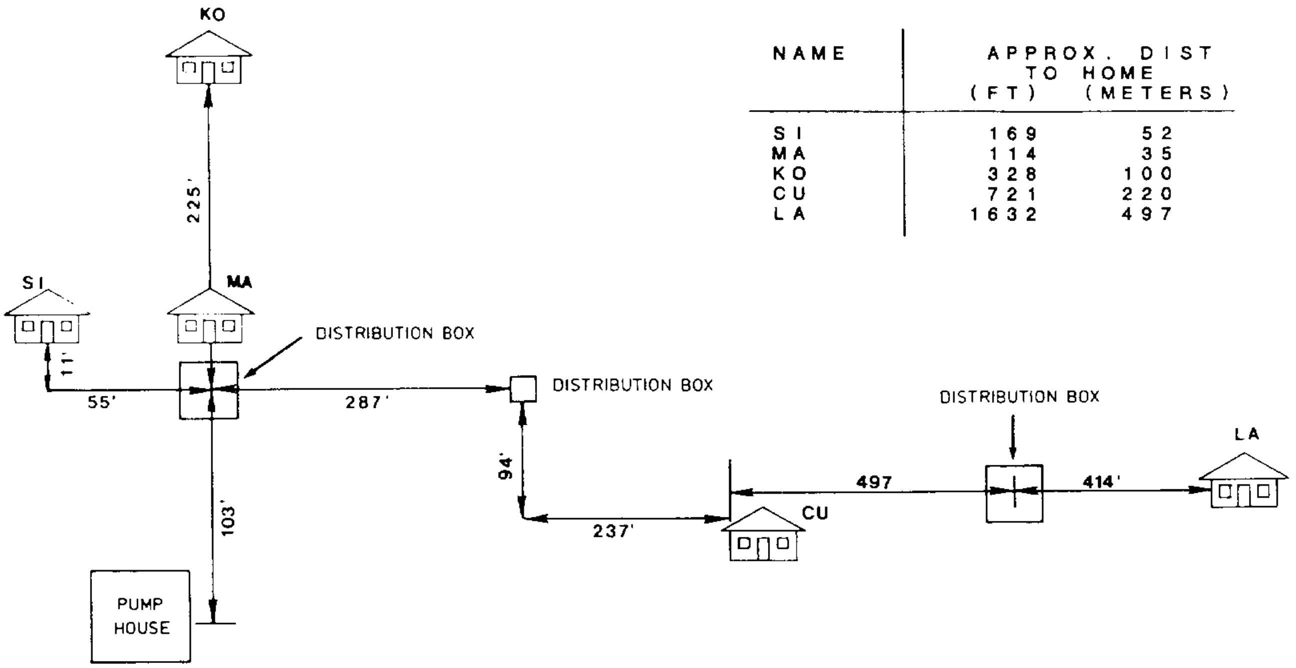
The actual reduction of radon in a distribution system would result from decay alone, so loss during distribution would not be significant unless the distribution system was extraordinarily long or the flow rate extremely slow. Small reductions are anticipated in short systems, but the removal rate would vary with water usage. Data were collected at Mont Vernon during periods of high flow and thus represented a worst case senario.

OPEN-AIR STORAGE

A laboratory study at the University of New Hampshire monitored radon reduction from a still pool of water. A 115-L (30-gal) plastic storage tank (Figure 2) was filled to a depth of 68 cm (27 in.) with water containing radon.During four individual runs, the radon concentration was monitored for 5 to 6 days (Figure 3). Samples were taken at various depths in the tank to determine if the radon concentration varied with depth. No significant difference in radon concentration was found within the tank.

High levels of radon removal (80% to 90%) were observed, but 5 to 6 days of storage were necessary to achieve these reductions. Theoretical reduction by radon decay alone would be 67% over 6 days, thus open air storage contributed to a greater reduction in radon removal than by decay alone. A 24- and 48-hr average reduction rate of

Figure 1. Mont Vernon Distribution System.



NAME	APPROX. DIST TO HOME (FT)	(METERS)
SI	169	52
MA	114	35
KO	328	100
CU	721	220
LA	1632	497

TABLE 1. Percent Radon Reduction in a Distribution System

	Approximate Distance from Pump House, m (ft)				
	35 (1114)	52 (169)	100 (328)	220 (721)	497 (1632)
Influent Radon (26,644 pCi/L GAC, Treated)					
Run 1	0	*	10.8	4.1	15.6
Run 2	0	*	7.3	9.2	*
Influent Radon (234,000 pCi/L, Untreated)					
Run 3	#	*	#	#	18.8
Run 4	2.1	†	#	11.8	7.5

* Radon concentration at sample point exceeded influent radon concentration.
\# Sample data not available.
† Homeowner not available during sampling.

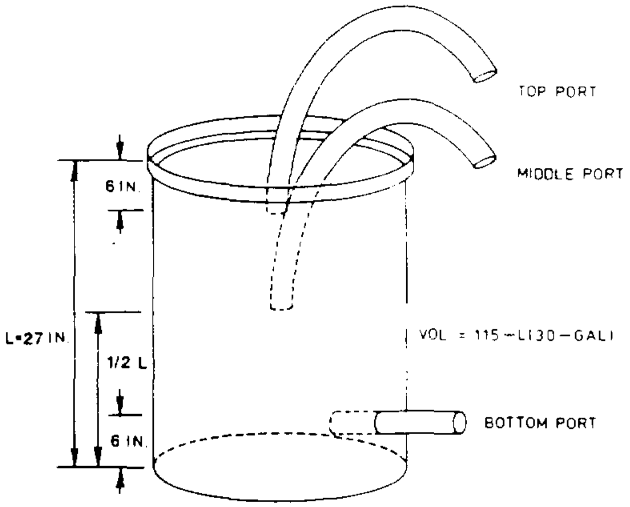
30% and 50%, respectively, was observed. A small community that could store the water at atmospheric pressure for several days could use this technique, although extended storage may be impractical in most cases.

FLOW-THROUGH RESERVOIR

A flow-through reservoir system (Figure 4) was constructed in Derry, NH, to treat water supplied by the Southern New Hampshire Water Supply Company. The storage tank was designed to contain water to a depth of 0.6 m (2 ft) [total volume of 1.64 m^3 (433 gal)], with variable influent entry. The modes of influent entry studied were: 1) entry at the bottom of the reservoir, 2) discharge 0.6 m (2 ft) above the reservoir level, 3) discharge 0.6 m (2 ft) above the reservoir level with a spray attachment, and 4) discharge 0.6 m (2 ft) above the reservoir level through a venturi apparatus to add air to the stream. In addition to the four tests with varied influent entry type, minimal bubble aeration was added to entry types 1 and 2.

In the tests where minimal bubble aeration was used, a laboratory-made bubbler was used. This bubbler was constructed using 0.6-cm (1/4-in.) I.D. plastic tubing with holes made at 10 cm (4 in.) intervals by puncturing the tubing with a thumbtack. Four radial arms of 81 cm (32 in.) length were placed in the bottom of the reservoir and air was provided by a laboratory air pump [maximum capacity 0.3 m^3/min (1.1 ft^3/min)]. The bubbles from the tubing appeared to be approximately 1.5 cm (3/16 in.) in diameter and were produced at an average rate of 2 to 3 per second from each hole.

Figure 2. **Laboratory Setup for Atmospheric Loss During Storage (No Flow).**



Flow controllers were installed in the influent stream to control the flow to 1.1 L/min (0.3 gpm), 2.2 L/min (0.6 gpm), and 3.2 L/min (0.9 gpm). This controlled flow resulted in theoretical detention times of 24, 12, and 8 hr. The nozzle attachment used a garden hose spray nozzle adjusted to give a spray of approximately 15 cm (6 in.) diameter at the water surface [0.6 m (2 ft) from the nozzle end]. At the low flow of 1.1 L/min (0.3 gpm), a fine spray could not be achieved, and thus data are not available for those conditions.

Good removals of radon were achieved in all test combinations (Figures 5, 6, and 7) except for the bottom entry tests. The bottom entry tests provided the minimum water disturbance of any of the tests and thus the lowest removal rate. In fact, the radon removals observed in the bottom entry tests were only slightly better than would be expected by radon decay alone.

In all cases where the water was allowed to fall (or was sprayed) to the reservoir surface, or where minimal bubble aeration was added, high radon removal rates were observed. A minimum removal of nearly 50% was seen with the shortest detention time and simple influent free fall. Higher removals (80% to 95%) resulted with longer detention times and supplemental aeration (Figures 5, 6, and 7). The data collected in this phase of the study showed that simple aeration can be very effective for radon reduction and might be easily applied in small communities.

A laboratory venturi was attached to the influent line of the storage reservoir (Figure 4). The venturi pulled ambient air into the water stream, and measurements were made to determine if the additional air would help to remove radon. The water was then allowed to free fall 0.6 m (2 ft) to the surface of the tank at the 3 flow rates of 0.9 L/min (0.24 gpm), 2.2 L/min (0.6 gpm), and 3.0 L/min (0.8 gpm). The venturi addition increased the radon removal over free fall alone by a range of 5% to 12%, except at the longest detention time [0.9 L/min (0.24 gpm)]. In this case, the radon removal with the venturi was 10% less than by free fall alone. The flow rates may have been too low to generate good venturi action (aeration), thus better removals were not observed in all the venturi runs.

SUMMARY

Simple low-technology low-cost aeration treatment techniques are capable of lowering the radon concentration in drinking water. Removal percentages of 60% to 87% can be achieved with only 9 hr of retention time and simple aeration. Better than 95% removal was observed with aeration applied during 30 hr of storage. Storage for 30 hr and the addition of minimal aeration should be within the operational capability of a small community. Larger scale testing of the simple aeration technique may be completed under the existing cooperative agreement with the NHDES. The data presented here are a qualitative description of some simple laboratory and field tests to remove radon. Further investigations are necessary to better understand the simple technologies and the options available to small communities.

THE AUTHORS

This low-cost/low-technology aeration study was conducted by Dr. Nancy Kinner, Ms. Carol Lessard, and Ms. Gretchen Schell of the University of New Hampshire, funded under Cooperative Agreement CR812602 with the New Hampshire Department of Environmental Services. The data shown here and the information presented here were compiled by Kinner, Lessard, and Schell (Department of Civil Engineering, University of New Hampshire, Durham, NH 03824) and will be finalized as a project report under U.S. EPA Cooperative Agreement CR812602. K.R. Fox, the EPA Project Officer, is with the Water Engineering Research Laboratory, Cincinnati, Ohio.

Figure 3. Radon Loss During Open-Air Storage.

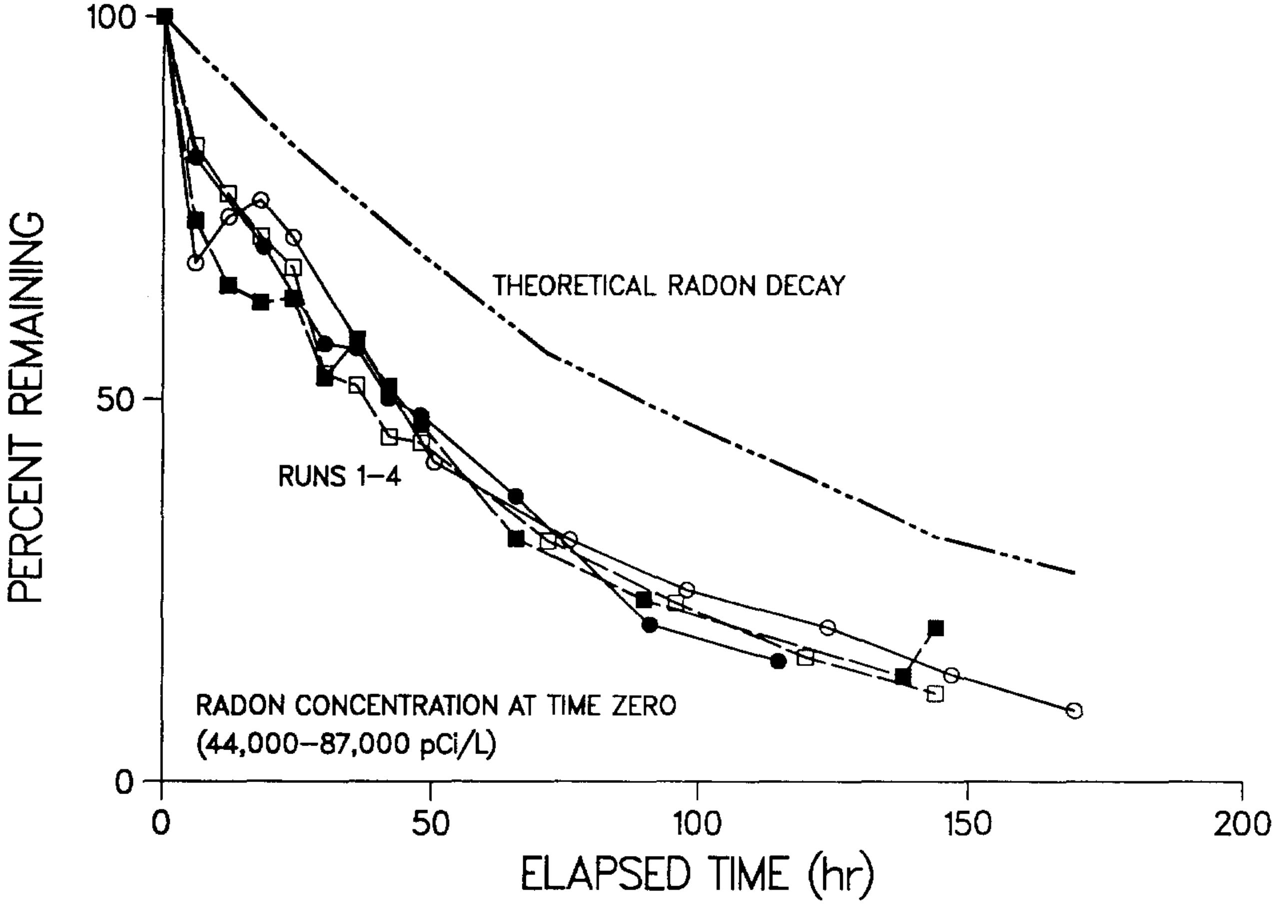


Figure 4. Pilot Scale Atmospheric Tank, Derry, NH.

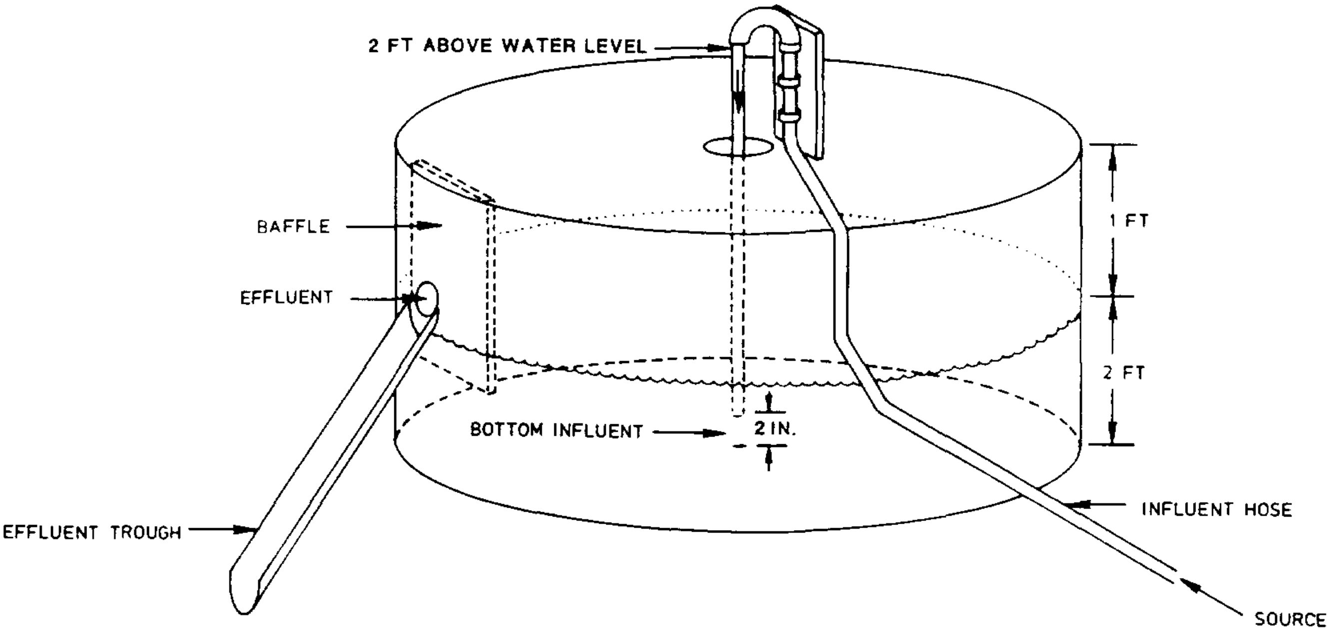


Figure 5. Radon Removal at Derry, NH (3.3 L/min (0.9 gpm) 8 hr Detention Time).

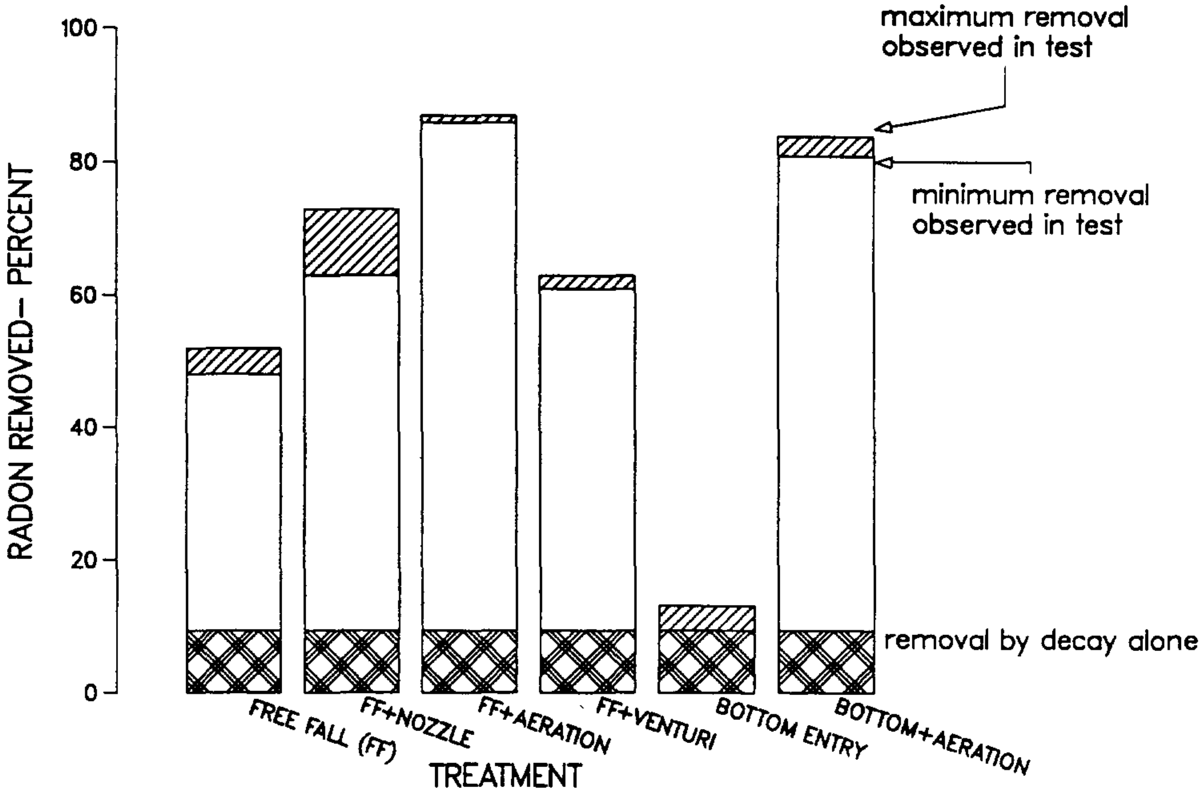


Figure 6. Radon Removal at Derry, NH (2.2 L/min (0.6 gpm) 12 hr Detention Time).

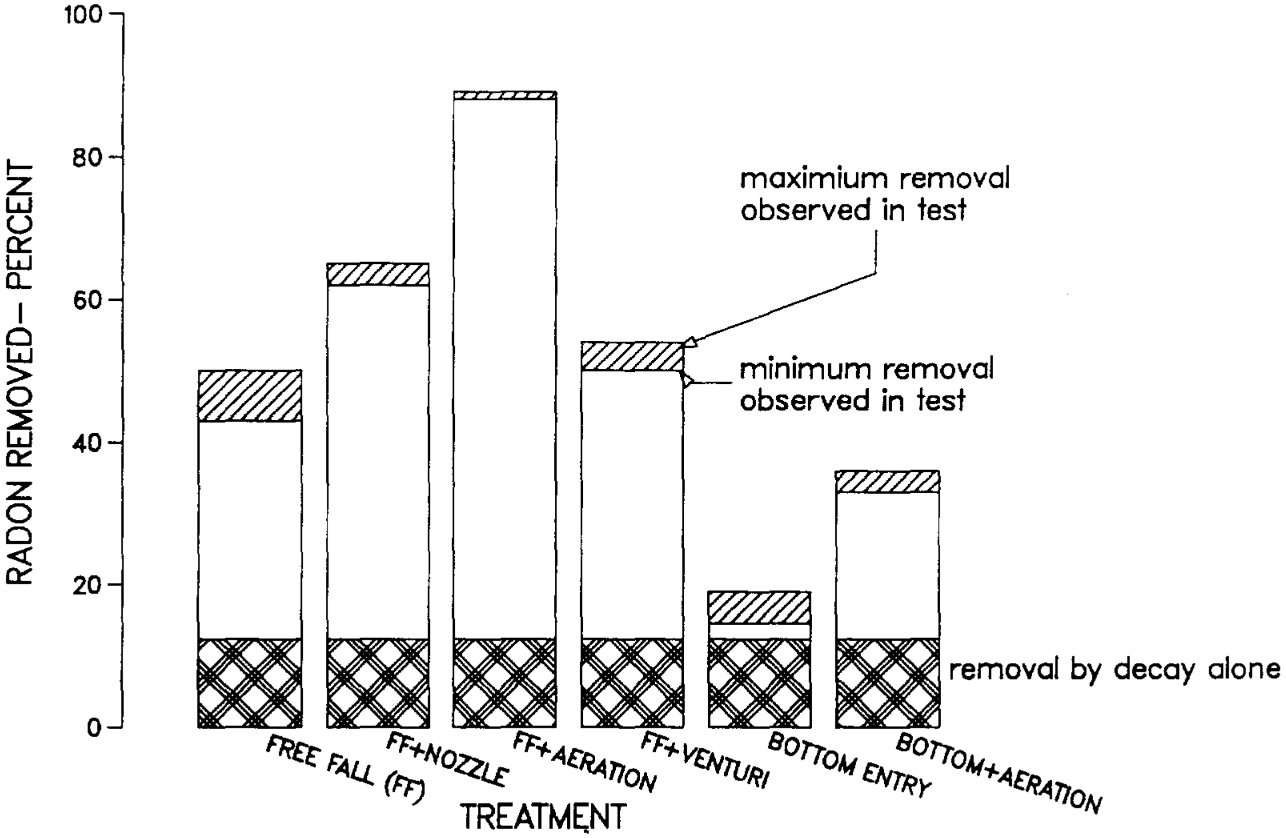
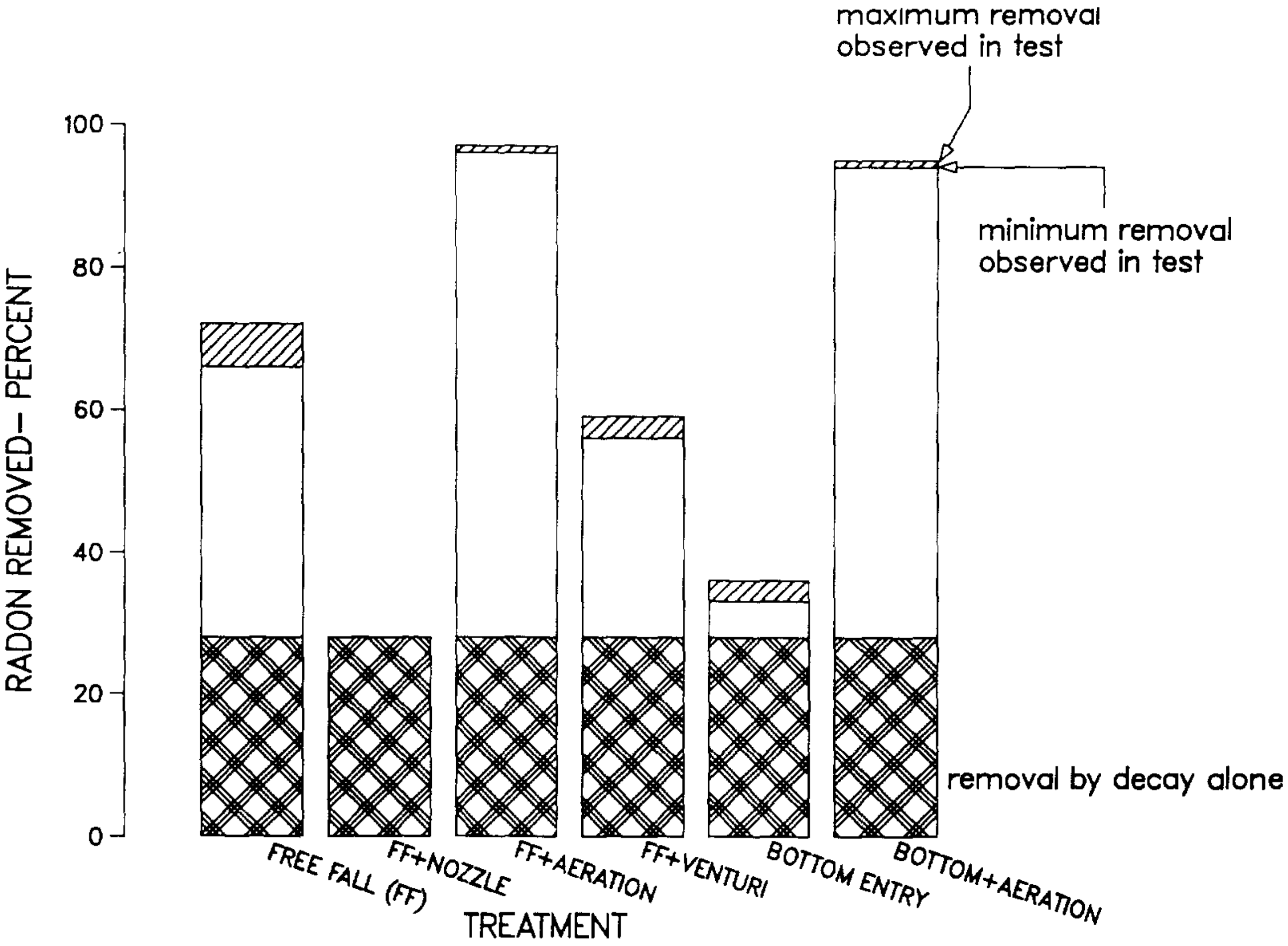


Figure 7. Radon Removal at Derry, NH (0.9 L/min (0.24 gpm) 30 hr Detention Time).



RADON REMOVAL TECHNIQUES FOR SMALL COMMUNITY PUBLIC WATER SUPPLIES

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DISCLAIMER

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FOREWORD

Today's rapidly developing and changing technologies and industrial products and practices frequently carry with them the increased generation of materials that, if improperly dealt with, can threaten both public health and the environment. The U.S. Environmental Protection Agency is charged by Congress with protecting the Nation's land, air, and water resources. Under a mandate of national environmental laws, the agency strives to formulate and implement actions leading to a compatible balance between human activities and the ability of natural systems to support and nurture life. These laws direct the EPA to perform research to define our environmental problems, measure the impacts, and search for solutions.

The Risk Reduction Engineering Laboratory is responsible for planning, implementing, and managing research, development, and demonstration programs to provide an authoritative, defensible engineering basis in support of the policies, programs, and regulations of the EPA with respect to drinking water, wastewater, pesticides, toxic substances, solid and hazardous wastes, and superfund-related activities. This publication is one of the products of that research and provides a vital communication link between the researcher and the user community.

This report presents the results of an evaluation, performed by the University of New Hampshire - Environmental Research Group (ERG), of radon removal in small community water supplies using full-scale granular activated carbon adsorption, diffused bubble aeration and packed tower aeration. Various low technology alternatives, such as loss in a distribution system and addition of coarse bubble aeration to a pilot-scale atmospheric storage tank were also evaluated. The report discusses each of the treatment alternatives with respect to their radon removal efficiency, potential problems (i.e., waste disposal, radiation exposure and intermedia pollution), and economics in small community applications. In addition, several sampling methods, storage times, scintillation cocktails and extraction procedures currently used in the liquid scintillation technique for analysis of radon in water were compared.

E. Timothy Oppelt, Director
Risk Reduction Engineering Laboratory

ABSTRACT

Radon is one of the naturally-occurring radionuclides which will be regulated as part of the Safe Drinking Water Act Amendments of 1986. The proposed rule must contain recommendations with respect to Best Available Technologies (BAT) and analytical methods. The purpose of this project was to evaluate the performance of full-scale granular activated carbon (GAC) and diffused bubble and packed tower aeration systems treating small community groundwater supplies containing radon. In addition, several low technology alternatives were evaluated including radon loss in a distribution system and pilot-scale tests of modifications to atmospheric storage tanks, such as coarse bubble aeration. Total production costs are provided for each treatment system evaluated, however, these are highly dependent on process equipment and percent removal required, which are site specific. The liquid scintillation analytical technique for radon was also evaluated with respect to sampling methods, storage times, extraction methods and cocktail selection.

GAC systems were operated at two small communities in New Hampshire. During continuous operation at the Mont Vernon site, the effluent radon activity averaged 24,653 ± 17,016 pCi/L (influent activity = 210,491 ± 41,384 pCi/L; flowrate = 36 ± 12 m^3/day). The GAC effluent activity at the Amherst site averaged 12,000 ± 5,900 pCi/L (influent activity = 41,800 ± 15,200 pCi/L; flowrate = 61 ± 5 m^3/day). The data indicated that the GAC units retained uranium, radium and lead. Concentration of these radionuclides appeared to be related to raw water quality and their retention could lead to classification of the GAC as a low level radioactive waste in some states. The GAC units also accumulated iron, manganese, particulates and microorganisms which may necessitate periodic backwashing or pretreatment to prevent significant headloss development. Gamma/beta emissions measured at the surface of the GAC units were substantially greater than background and shielding may be required to lower emissions to acceptable levels.

For the diffused bubble system operating at air:water (A:W) ratios $\geq$ 5:1 at high water flowrates (0.10 ± 0.0019 m^3/min; influent radon = 65,487 ± 5,657 pCi/L) and low water flowrates (0.047 ± 0.00053 m^3/day, influent radon = 78,385 ± 6,120 pCi/L), the overall radon removal efficiency ranged from 90.0 to >99.6%. The removal efficiencies for the packed tower ranged from 92.7 to 99.8% over the range of air flowrates tested in spite of variation in water flowrate, influent radon activity and packing type. Off gas radon activities were 10^4 to 10^5 times higher than ambient air levels for both aeration systems and the impact of the discharge on air quality should be considered. Precipitation of iron and manganese could result in operational problems in both systems and may necessitate pretreatment.

Radon loss during water transport in a distribution system (497 m, hydraulic retention = $\leq$ 17.9 min) was not effective in radon removal. In pilot-scale testing, atmospheric storage modified by free fall and spray nozzle entry and coarse bubble aeration achieved radon removals of 50 to 96%, but the system may have problems with off-gas emissions and/or iron and manganese precipitation.

When designing any system to remove radon from a small community water supply, it is imperative to have good data on water flowrates and influent radon activities. As observed in this study, variations in these parameters may be substantial and will, if underestimated, lead to inadequate system design and effluent radon activities which exceed the design goal. Since GAC adsorption-decay constants and aeration mass transfer coefficients are system specific, extrapolations of performance data obtained at one site should not be made to systems with (i) other design configurations, (ii) low influent radon activities and/or (iii) a requirement to meet a more stringent MCL. Pilot-scale studies should be conducted at each site to obtain the required design information.

Water samples collected using the direct syringe technique had higher radon activities than those collected in VOA bottles, but widespread use of this technique may be limited by costs, problems associated with distribution of syringes and the difficulty in collecting samples devoid of air bubbles. Sample storage in VOA bottles was affected by radioactive decay and leakage (losses up to 20% in 21 days). The OptiFluor O scintillation cocktail yielded significantly higher count rates than a toluene-based cocktail and was less expensive than the toluene and mineral oil-based cocktails. The extraction experiment data suggested that radon in a radium-226 standard is continuously transferred to the cocktail phase regardless of shaking and therefore, the extraction procedure should not be used to calculate the efficiency factor.

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LIST OF ABBREVIATIONS AND SYMBOLS

ABBREVIATIONS

ANOVA -- analysis of variance
A:W -- air to water ratio
CCI -- construction cost index
CFU -- colony forming units
DDI -- double deionized water
DI -- deionized water
DO -- dissolved oxygen
ENR -- Engineering News Record
EBCT -- empty bed contact time
FOB -- free on board
GAC -- granular activated carbon
GM -- Geiger Müeller
HTU -- height of transfer unit
LSC -- liquid scintillation counter
MCL -- maximum contaminant level
NTU -- nephelometric turbidity unit or number of transfer units
O&M -- operation and maintenance
PSNH -- Public Service Company of New Hampshire
SPC -- standard plate count
StT -- Student's t test
TDC -- total direct costs
VOA -- volatile organics analysis

SYMBOLS

α -- level of statistical significance
β -- contaminant diffusivity
C_A -- activity of radon in the air
C_{eff} -- effluent radon activity in water
C_f -- packing factor
C_G -- activity of radon in the gas phase
C_{inf} -- raw water (influent) radon activity
C_i -- activity of radon in the liquid phase
C_m -- density of the liquid
d_b -- bubble diameter
D_o -- oxygen diffusivity
D_w -- molecular diffusion coefficient

E -- efficiency factor
G -- air loading rate
H -- Henry's constant
H_L -- sidewall depth
K_La -- global mass transfer coefficient
K_{ss} -- steady state adsorption-decay constant
L -- liquid loading rate
mR -- milliRoentgen
mREM -- milliRoentgen Equivalent Man
P_t -- ambient pressure
ρ_w -- density of water
Q_g -- air flowrate
R -- stripping factor
$t_{1/2}$ -- radionuclide's half-life
μ_m -- viscosity of water
V_c -- liquid volume of diffused bubble tank
z -- packing height

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SECTION 1

INTRODUCTION

As part of the 1986 amendments to the Safe Drinking Water Act, the United States Environmental Protection Agency (EPA) will propose a rule for Maximum Contaminant Level Goals (MCLGs) and National Primary Drinking Water Regulations (NPDWR) including Maximum Contaminant Levels (MCLs) for radionuclides in drinking water (Federal Register, 1986). One of the radionuclides which will be regulated under the proposed rule is radon-222* (hereafter referenced as radon in the text). In a recent status report, EPA (1989) indicated it is considering setting the MCL for radon in the range 200 to 2,000 pCi/L. There have been several studies of the distribution of radon in groundwater supplies in the U.S. (Hess et al., 1985; Longtin, 1988; Dixon and Lee, 1988; Vitz, 1988) all of which indicate that there will be a large number of point-of-entry and public water supplies impacted by an MCL in that range. Many of the public water supplies affected will be those serving small communities (<76 m^3/day).

The rule will also contain recommendations with respect to Best Available Technologies (BATs) and analytical methods. Three conventional water treatment technologies - granular activated carbon adsorption (GAC), diffused bubble aeration, and packed tower aeration - have been used to remove radon from drinking water. Most of the studies in the U.S. on GAC have been performed by Lowry and associates (Lowry and Brandow, 1981, 1985; Lowry, 1985; Lowry et al., 1987, Lowry and Lowry, 1987; Lowry and Lowry, 1988; Lowry et al; 1988), though Dixon and Lee (1987) reported some radon removal in GAC units designed for organics treatment. The GAC process, which has been used to treat point-of-entry and small community water supplies, relies on the ability of radon to adsorb to the carbon. One unique aspect of the radon removal process is the fact that the bed is naturally regenerated as radon decays (Lowry and Brandow, 1985). As a result, the breakthrough/exhaustion profile typically seen when GAC is treating other conservative contaminants is not exhibited during radon removal.

The aeration methods are predicated on the fact that radon is a highly volatile gas with a relatively high Henry's constant (2.80 atm · $m^3 H_2O/m^3$ air at 10°C) and thus can be easily transferred from water to air. Aeration methods have been used in both point-of-entry and community public water

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* Though there are many radioactive isotopes of radon, radon-222 is the most common.

supplies (Perkins and Hewett, 1988; Cummins, 1987; Dixon and Lee, 1987; Lowry et. al., 1987; Hoather and Rackham, 1962; Castren, 1977; Lowry and Brandow, 1981).

In addition, Dixon and Lee (1987) have examined the potential of alternative technologies (for example, cascading tray aerators used for iron and manganese removal and loss in standpipes) which use simple devices often already in place and adapt them to radon treatment.

The analytical methods currently being examined by EPA (1989) for the proposed radon rule include liquid scintillation counting (Pritchard and Gesell, 1977; EPA, 1978), and the Lucas cell method (Lucas, 1957). These techniques have been used extensively and have been found to be equivalent in interlaboratory studies conducted for EPA (Whittaker et al., 1987).

RESEARCH OBJECTIVES

The purpose of this EPA Cooperative Agreement was to evaluate the performance of full-scale GAC and diffused bubble and packed tower aeration systems treating small community groundwater supplies containing radon. The water supplies contained 40,000 to 250,000 pCi/L of radon at flowrates ranging from 25 to 61 m^3/day. In addition, several low technology alternatives were evaluated including radon loss in a distribution system and pilot-scale tests of modifications to atmospheric storage tanks such as coarse bubble aeration. Each of these systems (GAC, diffused bubble aeration, packed tower aeration and the low technology alternatives) were evaluated with respect to three primary factors: radon removal efficiency, potential problems (i.e., waste disposal, radiation exposure, intermedia pollution) and economics. It was not the purpose of the study to develop design equations for these systems, but rather to evaluate their operation at small community facilities.

Another aspect of the study was the evaluation of the liquid scintillation counting technique used in radon analysis. This technique has been used with several scintillation cocktails, extraction techniques, sampling methods, and storage times. These aspects of the technique were studied to determine whether the various modifications available impact the results obtained with the method.

The specific objectives of the study were to:

- Evaluate full-scale GAC systems operating at two small communities in New Hampshire, monitoring them for changes in radon removal, radiation emissions, general water quality parameters (e.g., pH, iron, turbidity, microbial numbers);

- Conduct several special monitoring events of the GAC systems to assess the impact of diurnal variations in water flowrate and raw water quality, high water flowrate and backwashing on GAC performance;

- Core the GAC after several months of operation to determine if iron, manganese, microorganisms and/or radionuclides were accumulating in the units;
- Evaluate full-scale diffused bubble and packed tower aeration systems operating in small communities in New Hampshire, monitoring them for radon removal, general water quality parameters and off-gas emissions of radon;
- Operate the aeration systems over a range of volumetric air to water (A:W) ratios at two water flowrates to determine the effect of these parameters on radon removal;
- Evaluate three randomly packed plastic media with respect to radon removal efficiency in the tower aeration system;
- Evaluate the efficiency of radon removal in a small community distribution system;
- Evaluate the radon removal efficiency of several low technology modifications (e.g., free fall vs. bottom entry, spray nozzle entry, venturi entry, coarse bubble aeration retrofit) to an existing small community atmospheric storage tank;
- Assess the effect of sampling techniques (free fall, hose connector, direct syringe collection, VOA bottle collection), storage time, scintillation cocktail, and extraction via shaking on the liquid scintillation analytical technique for analysis of radon in water.

The report is divided into 8 sections including introduction (Section 1), conclusions and recommendations (Section 2), sampling and analytical methods (Section 3), separate sections discussing the results obtained in the evaluations of GAC (Section 4), diffused bubble aeration (Section 5), packed tower aeration (Section 6), low technology alternatives (Section 7), and the liquid scintillation analytical technique (Section 8). The treatment technology evaluation sections each contain: (i) an introduction; (ii) description of the units, experimental design and special sampling/analytical procedures; (iii) detailed presentation of the results and a discussion of their significance; and (iv) an economic analysis of the system operated.

FUNDAMENTALS OF RADIONUCLIDES

Radionuclides are different from the majority of drinking water contaminants regulated by EPA because of the instability of their nucleus and their ability to decay. This often leads to confusion resulting from the dissimilarity between the units used to describe radioactivity and those used for other water contaminants; as well as the physical meaning of these units. Therefore, a brief synopsis of some of the fundamental principles of radionuclides important to this project are discussed below.

Radon-222 is part of the uranium (4n+2) decay series (Table 1) which originates with uranium-238. Radon and its four short lived progeny (polonium-218; lead-214, bismuth-214 and polonuim-214) have some of the shortest half-lives (less than 27 min) of any of the radionuclides in the series. Radon is the only gas in the decay series and is a member of the Periodic Chart Group 0 (noble) gases including helium, argon and other chemically inert monatomic gases. Radionuclides are described in units of activity - curies - which indicate the rate at which a radioactive atom is undergoing decay (i.e., nuclear disintegrations). By convention a picocurie (pCi) means that 2.22 atoms of that radionuclide are decaying per min (disintegrations per min ≡ dpm). A radionuclide's half-life ($t_{1/2}$) is the time interval during which 1/2 of the atoms decay ($t_{1/2} = \ln 2/\lambda$ where λ = decay constant, 0.0075 hr^{-1} for radon). For radon and its progeny the half-lives are very short varying from 3.82 days (radon-222) to 0.00016 sec (polonium-214). The actual life of an individual atom, however, can range from 0 to infinity (i.e., decay occurs randomly).

Though the units specified for radionuclides in water are in pCi/L (i.e., activity per unit volume) and their range in groundwater is 10^0-10^6 pCi/L, the mass and number of atoms present is usually extremely small and can be determined using the equation: $A = N\lambda$ where A ≡ activity (pCi or dpm) and N = number of atoms. For example, 1,000 pCi of radon (2,220 dpm) in a liter of water is only 1.76×10^7 atoms of radon or a mass of 6.49×10^{-15}g (radon ≡ 222g/mole) or 6,490 pg/L of water.

Therefore, when designing treatment systems to remove radon from water, though the activities may be substantial (and have important public health significance), the numbers of atoms and mass to be removed is very small. However, radon is a highly volatile gas and its concentration in water (ex. 1,000 pCi/L ≡ 6.49×10^{-15} g/L) is still much greater than its concentration in air (0.1 pCi/L ≡ 6.49×10^{-19}g/L).

Detection capabilities for most drinking water contaminants are in the 10^{-8} to 10^{-3} g/L range. The instrumentation used in radon analysis detects light flashes created when the radioactive particles strike fluorescing compounds and therefore, is able to measure lower concentrations (32.5×10^{-16} g/L = <500 pCi/L). The instrument's detector records the number of counts (scintillations) per minute or cpm. The efficiency of the instrument is usually not 100%, so an efficiency factor must be determined using standards to relate the cpm obtained to the dpm or pCi actually present.

Atoms of the various radionuclides in a decay series are constantly being produced and are, in turn, decaying to form other unstable nuclides. For example, radon is created by the decay of radium-226 ($t_{1/2}$ = 1602 yr) and decays to polonium-218 (radon $t_{1/2}$ = 3.82 days). Because of this continuing pattern of formation and decay, it is important to understand the concept of

TABLE 1. ABBREVIATED[*] URANIUM (4n + 2) DECAY SERIES

Chemical Notation	Historical Name	Half-life	Radiation Emitted Upon Decay[+]
$^{238}_{92}U$ ↓	Uranium	4.5×10^{9} yrs.	α
$^{234}_{90}Th$ ↓	Thorium	24.1 days	β, γ
$^{234}_{91}Pa$ ↓	Protactinium	1.17 min	β, γ
$^{234}_{92}U$ ↓	Uranium	2.47×10^{5} yrs.	α, γ
$^{230}_{90}Th$ ↓	Thorium	8.0×10^{4} yrs.	α, γ
$^{226}_{88}Ra$ ↓	Radium	1602 yrs.	α, γ
$^{222}_{86}Rn$ ↓	Radon	3.82 days	α
$^{218}_{84}Po$ ↓	Polonium	3.05 min	α
$^{214}_{82}Pb$ ↓	Lead	26.8 min	β, γ
$^{214}_{83}Bi$ ↓	Bismuth	19.7 min	B, γ
$^{214}_{84}Po$ ↓	Polonium	164 µsec	α
$^{210}_{82}Pb$ ↓	Lead	21 yrs.	β, γ
$^{210}_{83}Bi$ ↓	Bismuth	5.01 days	β
$^{210}_{84}Po$ ↓	Polonium	138.4 days	α
$^{206}_{82}Pb$	Lead	-	Stable -

[*] Protactinium-234 (Uranium Z), Astatine-218, Thallium-210 and 206 are not shown. These represent radionuclides created in <0.2% of the decays of Protactinium-234 (Uranium Xg), Polonium-218, Bismuth-214 and Bismith-210, respectively.

[+] Only listed if % emission is >0.1%.

secular equilibrium. When a radionuclide is in secular equilibrium with another radionuclide or its progeny, there is a constant amount of that radionuclide present which is a function of the rate at which it is being created and the rate of its decay. The mathematical relationships are best explained by Evans (1969). Hence, at secular equilibrium between radon and its progeny or between radium and radon, the activities of the parent and its progeny are equal. For example, when 1,000 pCi/L of radium is in secular equilibrium with radon, the activity of radon will also be 1,000 pCi/L.

SECTION 2

CONCLUSIONS AND RECOMMENDATIONS

GRANULAR ACTIVATED CARBON

1. When designing a GAC system to remove radon from a small community water supply, it is imperative to have good data on water flowrates and influent radon activities at the site. These data are major inputs into the design model for determining the volume of GAC required. As observed in this study, for small community supplies, variations in flowrate and influent activity may be substantial and will, if underestimated, lead to inadequate system design and effluent radon activities which exceed the design goal.

2. The steady state adsorption-decay constant (K_{ss}), which is a critical component of the GAC design model varied over time, was site specific, and differed by 22% to 89% from the value previously reported in the literature for the carbon tested (Barnebey-Cheney Type 1002). The factors which will most likely influence K_{ss} for a given site are flowrate and radon concentration which affect mass transfer and adsorption kinetics, and raw water quality which affects adsorption capacity. Therefore, pilot scale testing should be performed at each site to determine the appropriate K_{ss} to be used in the design model.

3. The data indicated that the GAC units retained uranium 238 and 235, and radium-226 in quantities (pCi/kg) high enough to classify the carbon as a low level radioactive waste according to State of New Hampshire regulations. The units also retained lead-210 which is not currently regulated in the State of New Hampshire. Therefore, the GAC disposal costs could increase the production cost of a GAC system. Concentration of uranium, radium and lead by GAC systems appears to be related to raw water quality (e.g., pH and alkalinity). Changes in water quality (e.g., installation of a new well) should be monitored to determine if radionuclides are released to the water supply.

4. The GAC units accumulated iron, manganese and particulates (turbidity). In addition, there were significant numbers of microorganisms growing on the carbon. As a result, GAC systems may require periodic backwashing to prevent significant headloss development. In certain water supplies, concentrations of iron, manganese and particulates may be high enough to warrant frequent backwashing which would increase operating costs and reduce the hydraulic capacity of the system. In addition, previous studies have shown that frequent backwashing may decrease radon removal efficiency. Therefore, in these cases, pretreatment for iron, manganese and particulates is recommended to decrease the backwashing frequency.

5. Gamma/beta emissions measured at the surface of the GAC units were substantially greater than background measurements. Some form of shielding will be required to lower these emissions to acceptable levels.

6. The total production costs for the GAC system at Mont Vernon, NH were $0.95/1,000 gallons without pretreatment and GAC disposal, $2.45/1,000 gallons with pretreatment and $2.64/1,000 gallons with pretreatment and disposal. (Disposal costs were based upon an estimated 20 year design life for the GAC system). During continuous monitoring, the effluent radon activity averaged 24,653 ± 17,016 (influent activity = 210,491 ± 41,384 pCi/L; flowrate = 36 ± 12 m^3/day). For Amherst, these costs were $0.56/1,000 gallons, $2.06/1,000 gallons and $2.15/1,000 gallons, respectively. The effluent radon activity of the Amherst GAC system averaged 12,000 ± 5,900 pCi/L during continuous monitoring (influent activity = 41,800 ± 15,200 pCi/L; flowrate = 61 ± 5 m^3/day). The differences in total production costs between the sites reflect the greater volume of GAC at Mont Vernon (1.33 m^3 vs. 0.85 m^3) which was a function of the higher radon loading. Since production costs for a GAC system are highly dependent on the type, volume and disposal of the carbon which are site specific parameters, extrapolation of these cost estimates to other water supplies may be inappropriate.

7. Further studies should be conducted to examine the effects of backwashing and raw water quality parameters (e.g., organics and iron) on radon removal efficiency in GAC systems. In addition, the factors affecting the retention/release of radionuclides such

as uranium, radium and the radon progeny should be investigated.

DIFFUSED BUBBLE AERATION

1. When designing a diffused bubble aeration system to remove radon from a small community water supply, it is imperative to have good data on water flowrates and influent radon activities. These data are major inputs into the design model for determining hydraulic detention time, air flowrate and diffuser configuration. As observed in this study, for small community supplies, variations in water flowrate and influent activity may be substantial and will, if underestimated, lead to inadequate system design and effluent radon activities which exceed the design goal.

2. The percent radon removal in a diffused bubble system is directly proportional to the global liquid mass transfer coefficient (K_La), the detention time and the concentration gradient ($C_{liquid} - C_{gas}$). These findings are analogous to those reported for other more conventional contaminants. Since K_La, detention time and concentration gradient are system specific, extrapolations of performance data obtained at one site should not be made to systems with other configurations/diffusers, low influent activities and/or those required to meet a more stringent MCL. Therefore, pilot-scale studies may need to be conducted at each site to obtain the required design information.

3. The total production costs for the diffused bubble system at Derry, NH were $0.64/1,000 gallons without pretreatment and $2.14/1,000 gallons with pretreatment. When operating at A:W ratios $\geq$ 5:1, at both high water flowrate (0.10 ± 0.0019 m^3/min; influent radon activity = 65,487 ± 5,657 pCi/L) and low water flowrate (0.047 ± 0.00053 m^3/min; influent radon activity = 78,385 ± 6,120 pCi/L), the overall radon removal efficiency ranged from 90.0% to >99.6%. It is possible to determine the minimum air flowrate required to meet the design removal efficiency. However, for the range of water flowrates encountered in many small communities the cost effectiveness of a larger constant speed blower (i.e., capital and operating costs) outweighs the advantages of using a smaller blower capable of delivering the minimum air flowrate. Since production costs are highly dependent

on the process equipment and percent removal required, which are site specific, extrapolation of the cost estimates for Derry to other water supplies may be inappropriate.

4. Off-gas radon activities at the Derry site ranged from 3,361 to 18,356 pCi/L in the air exiting the diffused bubble system and were 10^4 to 10^5 times higher than ambient air levels. Therefore, the impact of the off-gas on ambient air quality should be considered.

5. Though precipitation of iron and manganese was not observed during the short-term diffused bubble testing, it is well documented that this can occur in aeration systems treating groundwater, resulting in operational problems. Therefore, raw water quality should be monitored to determine whether pretreatment is required.

6. In northern climates, diffused bubble systems, including the blower intake, are typically located inside the pumphouse. Since it has been shown in this study that the radon activity in the pumphouse air may be high (as at Mont Vernon, NH) use of this air may significantly affect the ability of the diffused bubble system to meet a stringent MCL due to mass transfer limitations. Therefore, the radon activity of the influent air should be considered in the design.

PACKED TOWER AERATION

1. When designing a packed tower aeration system to remove radon from a small community water supply, it is imperative to have good data on water flowrates and influent radon activities at the site. These data are major inputs into the model for determining tower design (e.g., packing height). As observed in this study, for small community supplies, variations in water flowrate and influent activity may be substantial and will, if underestimated, lead to inadequate system design and effluent radon activities which exceed the design goal.

2. The percent radon removal in a packed tower aeration system is directly proportional to the global liquid mass transfer coefficient (K_La), the hydraulic detention time (packing height) and the concentration gradient ($C_{liquid} - C_{gas}$). These findings are analogous to those reported for other more

conventional contaminants. Since the K_La, detention time and concentration gradient are system specific, extrapolations of performance data obtained at one site should not be made to estimate percent radon removals for systems with other tower configurations/packing media, low influent activities and/or those required to meet a more stringent MCL. Therefore, pilot-scale studies should be conducted at each site to obtain the required design information.

3. For some small communities which have high influent radon activities, the packing height required to meet a stringent MCL may be greater than the 4 to 12 m range used in standard practice, possibly eliminating packed tower aeration as a viable treatment alternative.

4. The total production costs for the packed tower system at Mont Vernon, NH were $0.60/1,000 gallons without pretreatment and $2.10/1,000 gallons with pretreatment. The radon removal efficiencies ranged from 92.7% to 99.8% in spite of variation in water flowrate (0.18 to 2.6 m^3/hr), influent radon activity (115,255 to 278,488 pCi/L) and packing type (Glitsch^R mini rings and saddles, and Koch^R pall rings) over the air flowrates tested (1.75 - 51 m^3/day). Since production costs are highly dependent upon process equipment and installation, which are site specific, extrapolation of these cost estimates to other water supplies may be inappropriate.

5. The off-gas radon activities ranged from 2,410 to 21,200 pCi/L in the air exiting the packed tower and were 10^4 to 10^5 times higher than ambient air levels. Therefore, the impact of the off-gas on ambient air quality should be considered.

6. Though precipitation of iron and manganese was not observed during the short-term packed tower testing, it is well documented that this can occur in aeration systems treating groundwater, resulting in operational problems. Therefore, raw water quality should be monitored to determine whether pretreatment is required.

7. At the Mont Vernon, NH site, the intake for the blower was located inside the pumphouse, which is a conventional practice used in northern climates to prevent freezing in packed towers. However, the radon

activity in the pumphouse air approached 100 pCi/L. In cases where this design approach is used, the concentration of radon in the influent air may significantly affect the ability of the packed tower to meet a stringent MCL because of mass transfer limitations. Therefore, the radon activity of the influent air should be considered in the design.

8. Because the conventional practice of locating the blower air intake indoors may not be practical for radon removal systems required to meet a stringent MCL, an outside air intake may be required. Insulation of the tower's plumbing may be critical in northern climates to prevent freezing resulting from intermittent operation.

LOW TECHNOLOGY ALTERNATIVES

1. Radon loss during water transport in a 497 m distribution system (hydraulic retention time $\leq$ 17.9 min) was not an effective and reliable means of radon removal.

2. In studies with the pilot-scale atmospheric storage tank at Derry, NH, free fall and spray nozzle entry, and coarse bubble aeration achieved radon removals ranging from 50 to 96%. Though the data show the potential of low technology alternatives, these retrofit designs are so site specific, results cannot be extrapolated to other water supplies, especially those required to meet a stringent MCL.

3. The total production cost of retrofitting the full-scale low technology coarse bubble aeration system used at Derry, NH was $0.15/1,000 gallons. This system achieved radon removals of 80% to 88%. These costs are presented as an example of the potential savings resulting from the use of low technology alternatives. However, the cost of these alternatives is very site specific and dependent on the radon removal required. Further, at some sites the requirement of sophisticated pretreatment prior to radon removal would defeat the purpose of using low technology alternatives.

4. Based on the off-gas results obtained from the diffused bubble and packed tower aeration systems, the impacts of the off-gas from the low technology alternatives on ambient air quality should be considered.

LIQUID SCINTILLATION ANALYTICAL TECHNIQUE

1. Water samples collected using the direct syringe technique had higher mean radon activities ($\leq$ 11%) than those collected in VOA bottles. However, the widespread use of this technique may be limited due to costs, safety problems associated with distribution of syringes and the difficulty in collecting samples devoid of air bubbles. Therefore, collection in VOA bottles is likely to be the preferred sampling technique. Based on the results of this research, it is recommended that a universal hose connector be used to fill the VOA bottles.

2. Sample storage in VOA bottles was primarily affected by radioactive decay, though leakage was found to account for losses up to 20% over 21 days. Therefore, maximum storage times should be established based on the MCL, the practical quantification level, radioactive decay and leakage.

3. A toluene-based scintillation cocktail yielded significantly lower count rate than mineral oil-based and Opti Fluor O cocktails. The Opti Fluor O is less expensive ($6.50/L) than the mineral oil ($31.00/L) or toluene-based ($12.00/L) cocktails and is therefore recommended.

4. The data from the extraction experiment suggest a continuous transfer of radon from the aqueous phase to the cocktail for the radium-226 standards regardless of shaking. This finding indicates that the extraction procedure should not be used to calculate the efficiency factor. However, samples should be shaken, especially those with low activities, to insure rapid transfers of radon to the cocktail.

SECTION 3

SAMPLING AND ANALYTICAL METHODS

This section outlines the procedures for sampling and the analytical methods used in all phases of the project. Techniques specific to a given part of the research are outlined within Sections 4, 5, 6, 7, and 8. A summary of all data from the quality control checks made throughout the project is provided in Appendix A.

SAMPLING MATERIALS

Table 2 shows the sampling materials, preservation, maximum holding time, and cleaning procedures for each type of sample collected. Double deionized water was transported into the field in a 50% HCl washed plastic jug and served as a water supply for field blanks which were collected for each analysis. Field spikes were only feasible for the iron and manganese analyses. All reagents used including washing acids were reagent grade unless otherwise specified.

ANALYTICAL METHODS

Temperature, pH, Dissolved Oxygen

Samples for temperature and pH were collected in 250 mL polyethylene bottles. When dissolved oxygen (DO) was measured polarographically (i.e., during GAC Phase I), analysis was performed in the 250 mL plastic bottles. For the iodometric dissolved oxygen procedure, 300 mL glass BOD bottles were used. All bottles for the three analyses were cleaned by soaking in soapy water, rinsing three times with deionized water, soaking in 50% HCl, rinsing three times in deionized water, rinsing once with double deionized water and air drying. Materials were soaked for a minimum of 60 min. Bottles and caps for temperature and pH were rinsed three times with the sample before collection.

Temperature was obtained using ASTM approved mercury thermometers (-20 to 150°C) which were calibrated monthly against an ASTM certified mercury thermometer (-8 to 32°C). Calibration was performed by obtaining measurements in an ice bath (0°C) and at room temperature (approximately 20°C) with all thermometers. The readings from the field thermometers were plotted against the readings from the certified thermometer and used as a calibration curve. Field measurements were corrected accordingly. Sample temperatures were taken by placing a thermometer in the 250 mL bottle immediately after collection. A reading was obtained after the thermometer equilibrated (~2-3 min).

TABLE 2. SAMPLING MATERIALS PREPARATION AND PRESERVATION

Analysis	Container Volume (mL)	Container Material	Preparation (Primary Cleaning Solution)	Preservation	Hold Time
Radon	20	G*	10% Extran^R (VWR Scientific)	None	4-12 hr+ (in growth)
pH	250	P**	50% HCl	None	24 hr
Temperature	250	P**	50% HCl	None	None
Dissolved Oxygen (Polarographic)	250	P**	50% HCl	None	None
Dissolved Oxygen (Titrimetric)	300	G*	50% HCl	$MnSO_4$ Alkali-iodide H_2SO_4	12 hr
Alkalinity and Turbidity	250	P**	50% HCl	4°C (iced)	48 hr
Iron and Manganese	125	P**	50% HNO_3	HNO_3 to pH<2	6 mo
Uranium Radium	3785	P**	50% HNO_3	HNO_3 to pH<2	6 mo
Microbial Numbers	60	P**	Sterilized Autoclaved 121°C @ 15 psig	4°C (iced)	24 hr

*G = Glass.

+After extraction via shaking.

**P = Polyethylene plastic.

Initially, the pH of the water samples was measured on-site with an Orion (Cambridge, Massachusetts) Model 211 portable pH meter standardized with pH 7 and pH 4 buffers. During Phase I, the weather became very cold and the field meter did not operate correctly. As a result, samples were transported to the laboratory on ice where the pH was determined as soon as possible using an Altex^R 71 pH meter (Beckman; San Ramon, CA) with a combination pH electrode. This procedure continued throughout the project. The pH meter was calibrated on each sampling day, as outlined in Standard Methods (1985) (Method 423), using pH 7 and pH 4 buffer solutions (VWR Scientific; Boston, MA). An initial reading was then taken of one of the standards. As a check on accuracy, an aliquot of that standard was analyzed every hour or every 20 samples, whichever came first. If the standard did not read within 0.10 units of the initial pH value, the meter was recalibrated.

The pH meter was also calibrated monthly using National Bureau of Standards (NBS) pH 4 and pH 7 standards to check the commercial buffer solutions. Quality control charts for accuracy and precision were made by running three standards in the order of high (10), low (4), and intermediate (7) pH values. This order was run seven times in duplicate on three separate days and the data was used to establish upper and lower control limits (EPA, 1979). Values were reported to the nearest 0.1 pH units. If the precision or accuracy limits established by these charts were exceeded during an analytical run, the meter was recalibrated, and samples since the last acceptable check on that day were reanalyzed. Periodically, samples were split and sent to the EPA Drinking Water Research Division (Cincinnati, OH) for pH analysis. Based on these samples, the project's pH analysis was within EPA (1983) limits for acceptable accuracy and precision.

Dissolved Oxygen

Dissolved oxygen (DO) was initially determined in the field using a Yellow Springs Instrument Co. (Yellow Springs, CO) YSI Model 57 portable oxygen meter. The instrument was air calibrated on site before each use as outlined by the manufacturer. The instrument's air calibration was checked monthly against a solution containing 0 mg O_2/L prepared with Na_2SO_3 and a trace of $CoCl_2$ (Standard Methods, 1985; Method 421F). Towards the end of Phase I of the GAC evaluations (November, 1986), the weather became very cold and the meter would not work properly. Thereafter, the iodometric titration method (Standard Methods, 1985; Method 421B) was used. Samples were fixed in the field with $MnSO_4$, alkali-iodide reagent and concentrated H_2SO_4. The 300 mL bottles were filled using the sampling tubes and were allowed to overflow for approximately 2-3 min. The fixed samples were transported at 4°C to the laboratory protected from the light. The samples were titrated against a solution of sodium thiosulfate (~0.025 N). The titrant's normality was checked daily against a potassium bi-iodate primary standard. One duplicate analysis was performed on each sampling day to monitor laboratory precision.

Alkalinity and Turbidity

Samples for alkalinity and turbidity were collected in 250 mL polyethylene bottles and preserved at 4°C for transport to the laboratory. Bottles and caps were rinsed three times with the sample before collection. They were cleaned by soaking in soapy water, rinsing three times with de-ionized water, soaking in 50% HCl, rinsing three times in deionized water, rinsing once with double deionized water and air drying. Bottles and caps were soaked for a minimum of 60 min.

Alkalinity was determined potentiometrically (Standard Methods, 1985; Method 403). After the initial pH of the sample was taken, the sample was titrated against 0.02N H_2SO_4 to a pH between 4.4 and 4.6, to determine the inflection point in the curve. Alkalinity was determined from the following equation:

$$\text{Alkalinity} = \frac{A \times N \times 50{,}000}{\text{mL sample}} \qquad \text{(eq. 1)}$$

where A is the volume (mL) of standard acid used in the titration and N is the exact normality of the standard acid. The normality of the acid was determined monthly with a Na_2CO_3 primary standard. Duplicate alkalinity titrations were analyzed every fifth sample to determine precision (which was acceptable in the range ±5 mg/L as $CaCO_3$). Periodically, samples were split and sent to the EPA Drinking Water Research Division (Cincinnati, OH) for analysis. Based on these results, the project's alkalinity measurements were within EPA (1983) limits for precision and accuracy.

Turbidity was analyzed on a Hach^R Model 2100A Turbidimeter following Standard Methods 214 (1985). The instrument was calibrated prior to use on the appropriate scale with the appropriate standard using sealed secondary standards manufactured by Advanced Polymer Systems, Inc. (Redwood City, CA). EPA approved primary turbidity standards supplied by Advanced Polymer Systems were used to check the secondary standards monthly. The meter was recalibrated at least every hour or every 20 samples, whichever came first. Duplicate analyses were performed every fifth sample to check precision. Precision quality control charts were constructed using the data from the first few months of the project (EPA, 1979). The precision for all of the duplicate sample analyses were combined and upper and lower control limits were calculated. Accuracy quality control charts were constructed in the same manner as the precision charts, only using the data from readback standards as the input. Each run, standards were checked and the accuracy was plotted on the accuracy control chart. If any accuracy or precision checks fell outside of the control limits, all the samples on a given day up to the last acceptable quality control check were reanalyzed. Values were reported to the nearest 0.05 NTU. Turbidity samples from the EPA (EMSL; Cincinnati, OH) water supply performance evaluation studies were analyzed in the laboratory during 1987 and 1988 as part of the quality assurance program for the project. Periodically, field samples were split and sent to the EPA Drinking Water Research Division (Cincinnati, OH) for analysis. Based on both the WS series and split sample results, the turbidity analyses for the project were within EPA (1983) limits for accuracy and precision.

Iron and Manganese

Iron and manganese samples were collected in 125 mL polyethylene bottles. The sampling materials were cleaned by soaking in soapy water, rinsing three times with deionized water, soaking in 50% HNO_3, rinsing three times with deionized water, rinsing once with double deionized water and air drying. The bottles and caps were soaked for a minimum of 60 min. Bottles and caps were rinsed three times with the sample before collection. The samples were acidified to $pH<2$ with concentrated HNO_3.

Iron and manganese samples were analyzed on a Perkin-Elmer® (Norwalk, CT) Model 2380 flame atomic adsorption spectrophotometer (AAS) using an air-acetylene flame and a 10 cm burner head according to Standard Methods 303A (1985). Standards were prepared daily from stock solutions of 1,000 ppm iron and 1,000 ppm manganese (EM Science; Cherry Hill, NJ). Although matrix problems were not anticipated, a $CaCO_3$ solution, outlined in Method 303A, was added to the samples and standards to combat ionization interference. Two standards, S1 and S2, for each metal were used to calibrate the AAS as described by the manufacturer. A standard curve was established for each metal using absorbances of 4 or 5 standards to monitor the linearity of the instrument based on S1 and S2 inputs. Readback standards were run after every 10 samples to check accuracy. Duplicates were analyzed every fifth sample. Concentration measurements were read directly off the AAS. Control charts were made for precision and accuracy. On each of three days, four standards of each metal were prepared and analyzed seven times as described in EPA (1979). Upper and lower control limits were calculated for accuracy using the percent recoveries and for precision using the difference between duplicates (EPA, 1979). The standards were also run at the end of each analytical run. Two samples at every site during every sampling period were randomly chosen and spiked with 25 µl each of 1,000 ppm iron and manganese. An aliquot of double deionized water brought into the field was also spiked with the same amount of the two metal stocks. Samples were periodically split and sent to the EPA Drinking Water Research Division (Cincinnati, OH) for analysis. Based on these results, the iron and manganese measurements were within EPA limits for precision and accuracy (EPA, 1983).

The GAC samples collected for metals' analysis during the coring experiment were dried to a constant weight in an oven at 105°C. Two subsamples approximately 2 g each were placed in tared 125 mL polyethylene bottles (cleaned as described above) and weighed on a Mettler AC-100 analytical balance. 50 mL of 10% HNO_3 were added to each bottle. The samples were digested according to the method outlined by Lessard (1987) for 24 hr in a 60°C water bath. The liquid was decanted, filtered through double deionized water washed GF/C filters and diluted to 100 mL in a volumetric flask. They were analyzed using the AAS according to the Standard Methods 303A (outlined above).

Microbial Numbers

Water samples to determine microbial numbers were collected in 60 mL sterile polyethylene bottles and preserved at 4°C during transport to the laboratory. The bottles were cleaned by soaking in soapy water, rinsing three times in deionized water and autoclaving at 120°C and 15 psig for 20 min. The bottles and caps were rinsed three times with the sample before collection.

Microbial populations were determined using the spread plate method. A 0.1 mL aliquot of sample was plated on tryptone glucose extract (standard plate count - SPC) agar (Difco, Detroit, MI) and incubated at 20°C for 7 days. The incubation period and temperature were determined in preliminary experiments with the water samples to yield the maximum number of organisms. The number of colony forming units (CFU) per plate between 20 and 200 were recorded using a Quebec colony counter. Aseptic techniques outlined in Standard Methods (1985, Method 907 and 907B) were followed. All samples were plated in duplicate. Normally, dilutions of 10^{-1} and 10^{-2} were adequate to achieve the correct number of colonies per plate. Sterile controls were run with every set of samples to check aseptic technique.

GAC samples for microbial analysis were collected in sterile polyethylene bags and stored at 4°C during transportation to the laboratory (~4 hr). Aliquots of approximately 5 g were placed in sterile, tared polyethylene bottles (125 mL) and weighed on a Mettler AC-100 balance. They were then mixed with 45 mL of 0.1% sodium pyrophosphate and sonicated for 30 min. Serial dilutions (10^{-1} to 10^{-8}) of the supernatant were plated in triplicate on SPC agar and subsequently counted as outlined above.

Uranium, Radium, and Lead Radionuclides

Aqueous samples for uranium and radium analyses were collected in 3.79 L polyethylene containers and acidified to pH <2 with concentrated HNO_3. Bottles and caps were rinsed three times before collection of the sample. The sampling materials were cleaned by soaking in soapy water, rinsing three times with deionized water, soaking in 50% HNO_3, rinsing three times with deionized water, rinsing once with double deionized water and air drying. The bottles and caps were soaked for a minimum of 60 min.

Samples were analyzed for total uranium and radium-226 by the State of New Hampshire Department of Environmental Services Laboratory (Concord, NH). Total uranium alpha particle activity was analyzed according to Method 908.0 (EPA, 1980) where the uranium species were coprecipitated with ferric hydroxide, redissolved in HCl and concentrated on an anion exchange column. Uranium eluant from the column was dried and counted for alpha particle activity using a Ludlum (Sweetwater, TX) 1000 scaler and a Ludlum 43-10 detector. Radium-226 was measured using the radon emanation technique (EPA, 1980). The radium was coprecipitated on barium sulfate, dissolved in EDTA and stored in a sealed chamber to allow ingrowth of radon-222. The radon gas was collected in a Lucas scintillation cell and counted using a Ludlum 1000 scaler and a Ludlum M-182 detector. A tracer spike of barium-133 was added to all radium samples.

Uranium 238 and 235, radium-226 and lead-210 activities of the GAC were determined from the coring experiment samples. After collection of approximately 200 mL of GAC per sample in 50% HNO_3 washed 227 g glass Mason jars, the materials were taken to the New Hampshire Division of Public Health Services Radiological Health Program's laboratory (Concord, NH). The contents of jars were dried to a constant weight at 105°C. The samples were then placed in the 105°C oven with a Mason jar lid on them and reheated. Immediately after removal from the oven, the screw cap lids were tightened and upon cooling the lids sealed as a vacuum was formed. This procedure was used to retain any radon subsequently produced. Samples were held at room temperature in the laboratory for 44 days before analysis.

Gamma spectroscopic analyses were conducted using a Nuclear Data (Schaumburg, IL) Model 6620 spectrometer fitted with a 13% Ge(Li) detector (10 hr counts/sample). The uranium-238 values represented the means of the activities of thorium-234 and protactinium-234, if both were found. The values of radium-226 were the means of the lead-214 and bismuth-214 activities. The GAC was assumed to have a bulk density of 428 kg/m^3 as noted by the manufacturer (Barnebey Cheney; Columbus, OH).

The samples were held in the laboratory for 1.5 years and subsequently transported to the University of New Hampshire. They were redried and counted on a Canberra high resolution gamma spectrometer with a coaxial Ge diode well detector (Model GCW 1022-7500). The system was coupled to a computer-based multi-channel analyzer (Model 35). A virgin GAC sample was counted for 15.4 hr. The GAC samples from the coring experiment were counted for 2-3.4 hr depending on their activities. Data was reduced according to the method outlined by Larson and Cutshall (1981) to determine the activity/kg of uranium 238 and 235, radium-226 and lead-210.

Gamma/Beta Emissions

Gamma/beta measurements were obtained by placing the unshielded detector of a Victoreen (Cleveland, OH) THYAC IV survey meter (Model 290) on the surface of the treatment units or holding it exposed to the air. A Geiger-Müeller probe (Model 491-40) was used and the meter set to record mR (milliRoentgen) dose rate per hr^{-1}. The meter was calibrated against a cesium-137 source as outlined by the manufacturer (Victoreen, 1985).

Radon

Aqueous samples for radon were drawn directly from the plastic sampling tubes or from inverted funnels filled overflowing with water using 10 mL Hamilton^R gastight glass syringes equipped with 13 gauge needles. The 13 gauge needles were used to minimize cavitation when the sample was drawn. The syringe was rinsed three times with the sample before collection. The 10 mL sample, devoid of air bubbles, was slowly injected into a 20 mL capacity low-absorbance liquid scintillation vial below the surface of 10 mL of scintillation cocktail [40.3 mL Liquifluor^R (Dupont, Wilmington, DE) per 1 L of scintillation grade toluene] already in the vial. The vials and syringes were cleaned before each use by soaking in soapy water, rinsing three times in deionized water, soaking in Extran 1000^R (VWR Scientific, Boston, MA), rinsing four times with deionized water and air drying. All materials were soaked for a minimum of 20 min.

The samples were analyzed once radon came to secular equilibrium, with its short-lived progeny (preliminary experiments indicated we should wait 4 hrs after collection) and before 12 hrs had elapsed between collection and analysis. Radon samples were analyzed using a Beckman (Fullerton, CA) LS 7000 liquid scintillation counter (LSC). Program 6 was used for all analysis (Appendix B). The counting time was 10 min per sample. Automatic quench compensation was used with a cesium-137 source. The percent error in channels 1 and 2 was two times the standard deviation. The instrument was calibrated with Beckman tritium, carbon-14 and background sealed standards. A radium-226 standard containing 260.85 disintegrations per minute (dpm) (approximately = 117.5 pCi/L) was used for daily standardization. The counts per minute (cpm) value of this standard, which contained a known amount of radon-222 in secular equilibrium with radium-226, was used to calculate a counting efficiency (E) for the instrument. The LSC provided results in counts per minute (cpm) for two windows: Channel 1 = 0 - 397 (corresponding to a full tritium channel) and a Channel 2 = 397 - 940 (a tritium - phosphorus-32 channel). The majority of the counts from radon-222 and its progeny were located in the channel 2 pulse heights. The total counts (Channel 1 and 2) for each sample were then converted to pCi/L using the equation:

$$A = \frac{(C_s - C_b) \times 1000 \text{ ml/L}}{(E)(V)(D)} \qquad \text{(eq. 2)}$$

where:
A = radon activity (pCi/L)
C_s = cpm of the sample
C_b = background cpm of field blank (cpm)
E = efficiency factor (cpm/pCi)
D = decay correction factor
V = sample volume (mL)

The decay correction was determined from:

$$D = e^{-0.693(T)/t_{1/2}}$$

where:
T = time from sampling to counting (days)
$t_{1/2}$ = half-life of radon (3.82 days)

Duplicate samples were analyzed every tenth sample to determine precision. The radium-226 standard and field blank were counted at the beginning of each analytical run.

The radium-226 standards were prepared from NBS or EPA (Environmental Monitoring Systems Laboratory; Las Vegas, NV) primary standards. The standards, in glass ampules, typically contained approximately 5 g of an HCl solution with radium-226 activities ranging from 4 to 5 nCi/g. The contents of the ampule were emptied into a 50% HNO_3 washed and air dried glass beaker to facilitate weighing in a Mettler AC-100 analytical balance. The contents of the beaker were then transferred to a 200 mL volumetric flask (cleaned using 10% Extran^R, 50% HNO_3 and double deionized water) and the beaker was rinsed several times with 0.05M HNO_3 to insure all radium-226 was transferred. The stock radium-226 solution was prepared in a balance of 0.05M HNO_3. It was stored in the wax-sealed 200 mL volumetric flask at 4°C. The stock solution usually contained 117.5 pCi/mL (260.85 dpm/mL) of radium-226. 5 mL of the stock solution was added using a volumetric pipette to a clean scintillation vial and diluted with 5 mL of 0.05M HNO_3. The contents of the vial were purged for 20 min with fine bubbles of laboratory grade nitrogen gas. Subsequently, 10 mL of the toluene-based scintillation cocktail were slowly added to the standard. The scintillation vial was sealed and counted daily on the LS 7000 to determine when the radium-226 was in secular equilibrium with radon-222 (~20-25 days). In most cases, standards were used for a period of 2-3 months before replacement. Once the standard was in secular equilibrium it was used. If the cpm of the radium standards began to change before that time, they were no longer used. Samples were sent regularly to the EPA Environmental Monitoring and Support Laboratory (Cincinnati, OH) for interlaboratory checks until it closed. Duplicates were analyzed every 10 samples to determine precision. The detection limit for the UNH radon analysis was determined according to the procedures outlined in EPA (1984) to be 139 pCi/L for a 10 min counting time.

The activity of radon in the off-gas was monitored for the diffused bubble and packed tower aeration systems using a Pylon^R AB-5 portable radiation monitor (Ontario, Canada). The monitor was programmed to pump air (0.5 L/min) for 10 min through a hose into a Model 300 Lucas Cell. After 10 min, the cell was sealed and the radon allowed to come to secular equilibrium with its progeny (~3 hrs). The radiation monitor came equipped with a 3368 dpm radium-226 standard calibration cell. The efficiency of the machine was determined daily using this standard. The efficiency was used to convert sample cpm to pCi/L. A machine efficiency of approximately 73% was maintained according to the manufacturer's specifications. The procedures used were outlined in the manufacturer's guide (Pylon, 1987).

Efficiencies (E) were calculated in percent by:

$$E = \frac{\text{standard cell cpm}}{\text{standard cell dpm}} \times 100\% \qquad \text{(eq. 3)}$$

Sample radon activities were found using the average stable cpm after approximately 3 hr of ingrowth in the equation:

$$\mathrm{pCi/L} = \left[\frac{C_s - C_b}{3 \times 2.22 \times 0.285 \times E}\right] \times 100 \qquad \text{(eq. 4)}$$

where:

C_s = cpm found for sample
C_b = cell background cpm
3 = number of alpha particles emitted during radon's decay
2.22 = conversion factor (dpm/pCi)
0.285 = volume of Model 300 sample cell (L)

The monitor had a maximum count rate of 10^6 per second.

Percent Moisture Content

GAC samples were collected during the coring experiment and placed in sealable polyethylene bags. They were transported to the laboratory where approximately 30-40 g aliquots were placed in tared aluminum dishes. They were subsequently weighed on a Mettler AC-100 analytical balance and placed in an oven at 105°C until a constant weight was attained. The samples were cooled in a dessicator and then reweighed on the balance. The change in weight was used to calculate the percent moisture content of the samples. These data were used to correct all coring analyses to kg of GAC (dry weight).

SECTION 4

GRANULAR ACTIVATED CARBON

INTRODUCTION

The ability of granular activated carbon (GAC) to remove radon has been demonstrated by Lowry (1985) who reported removals of 80-99.9% for waters containing 40,000-750,000 pCi/L using point-of-entry systems with 0.028 to 0.071 m^3 of GAC. However, less information is available on the efficacy of using GAC for radon removal in small community systems (Lowry and Lowry, 1988; Reed, 1989). In this phase of the research, downflow GAC systems installed at two mobile home parks located in Amherst and Mont Vernon, NH were studied. The Mont Vernon GAC system had been operating for approximately 1 yr before the study began. For this research project, the existing carbon was removed and replaced with virgin GAC.

The water system at Amherst serves 56 homes at an average daily flow of 59 ± 4 m^3/day. The water is obtained from one well containing an average radon activity of 49,500 ± 11,200 pCi/L. The water system at Mont Vernon supplies 40 homes at an average flow of 31 ± 11 m^3/day. Water is obtained from two wells with an average radon activity of 222,000 ± 52,000 pCi/L. The well water in both systems was pumped to atmospheric storage tanks and subsequently, pumped through the GAC systems upon demand from the community. A schematic of both systems is provided in Figures 1 and 2.

The systems were evaluated based on radon removal efficiency, retention of radionuclides, gamma/beta emissions, and economics. Radon and several general water quality parameters (alkalinity, turbidity, dissolved oxygen, temperature, pH, iron, manganese and bacterial numbers) were monitored at each site. Uranium and radium were also monitored in the water supply. The GAC was cored and analyzed for accumulation of uranium-238, uranium-235, radium-226, lead-210, iron, manganese and microbial numbers. Gamma radiation measurements were taken at the surface of the units, and at locations inside and outside of the pumphouses to determine whether exposure presented significant health and safety problems.

An economic analysis of the GAC systems was performed to determine the costs of using GAC for radon removal from small community drinking water supplies. The analysis included capital, operation and maintenance, and disposal costs. In addition, the total annual cost and the production cost for each GAC system was estimated.

There are several advantages of using GAC systems for radon removal in small community water supplies: (i) GAC units can be installed in-line so no additional pumping is required (systems are already under pressure), (ii)

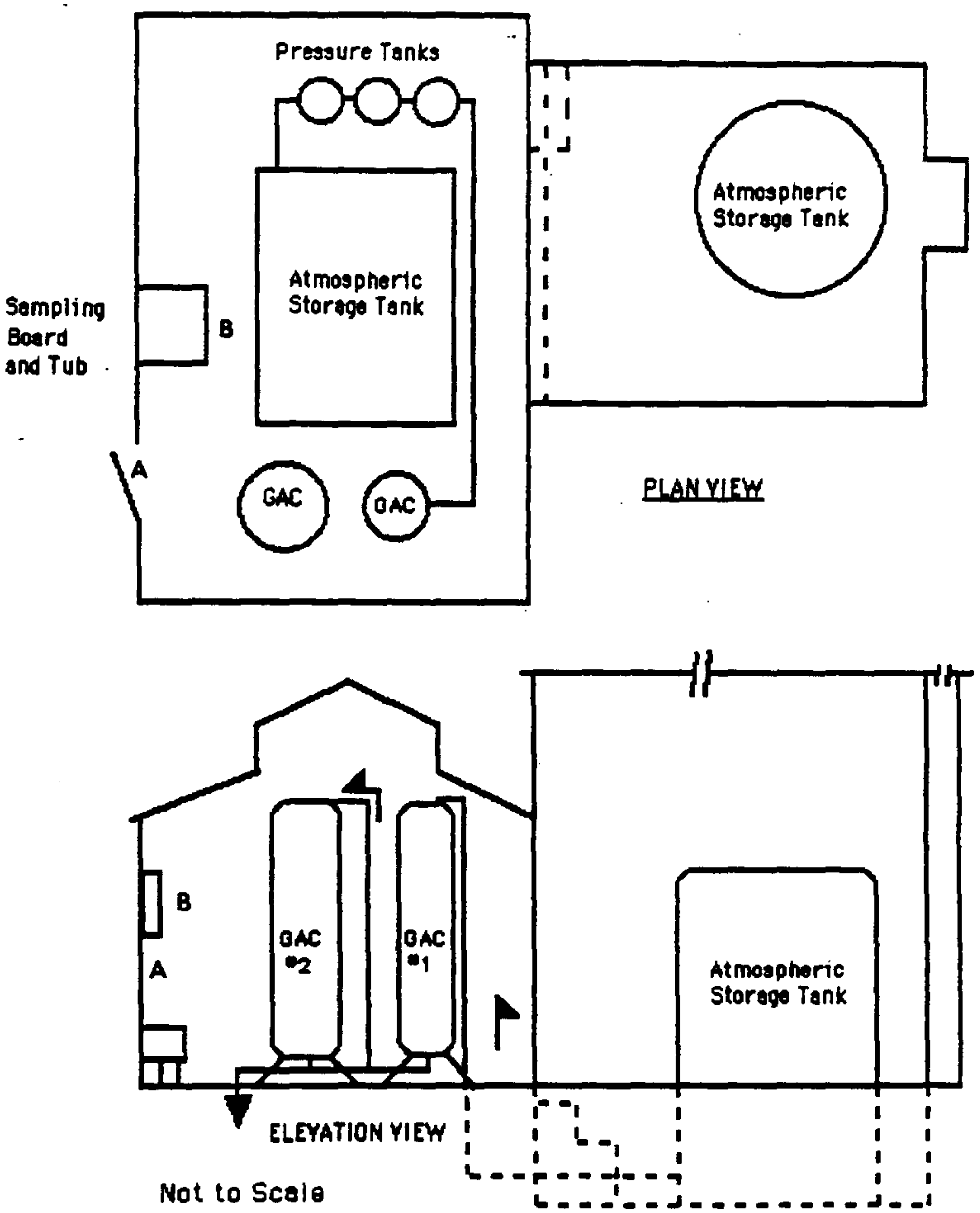


Figure 1. Schematic of the Mont Vernon, NH water supply and GAC system (not to scale). A and B are locations of gamma/beta emission measurements.

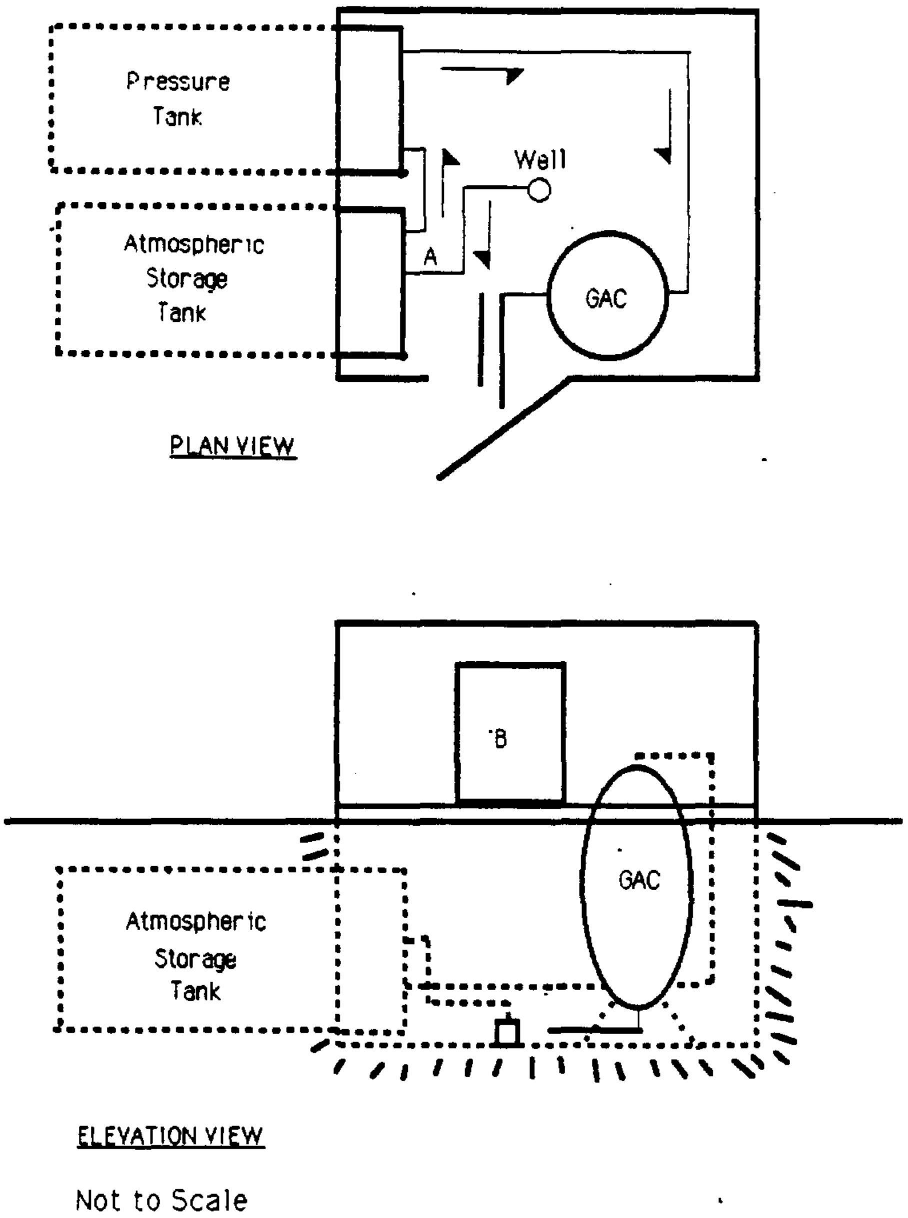


Figure 2. Schematic of the Amherst, NH water supply and GAC system (not to scale). A and B locations for gamma/beta emission measurement.

radon is contained, not released into the air, and (iii) GAC has lower maintenance requirements than aeration systems and has no moving parts. The disadvantages of GAC are: (i) the potential for substantial gamma emissions from radon progeny, (ii) the potential for accumulation of long-lived radionuclides (i.e., uranium 238 and 235, radium-226 and lead-210), (iii) the long empty bed contact time (EBCT) required, and (iv) the potential for fouling of the GAC by oxidized metals, organics, particulates, and/or microorganisms, requiring frequent backwashing.

EXPERIMENTAL PROCEDURES

Description of Units

The GAC systems units were designed by Dr. Jerry Lowry (Unity, ME) and started on October 8, 1986. The GAC used for the research project, Barnebey Cheney 1002, was chosen based on previous studies by Lowry (1985) and placed in the fiberglass tanks installed in the pumphouses. After the GAC was in place, the units were backwashed to remove fines. At Amherst, the system consisted of one 91.44 cm diameter contactor containing 0.85 m^3 of GAC. At Mont Vernon, the system consisted of two contactors in series; a 76.2 cm diameter contactor (GAC #1) containing 0.57 m^3 of GAC followed by a 91.44 cm diameter contactor (GAC #2) containing 0.76 m^3 of GAC for a total of 1.33 m^3 of GAC. The system at Mont Vernon was designed with two units because of the high influent radon activity.

Sampling taps for water samples were installed on the influent and effluent lines and at intervals within each unit. The sampling taps before and after the units were constructed of Schedule 40 PVC tees and brass valves. Taps within each unit were constructed of 1.91 cm diameter Schedule 80 PVC pipe. The PVC pipe was slotted laterally with 0.0305 cm slots and extended 22.86 cm into the GAC bed. The pipe was supported by a 1.91 cm PVC bulkhead installed in the contactor wall. Six millimeter plastic tubing was connected to the bulkhead which extended to a sampling board containing brass valves. The sampling tap locations for the Amherst and Mont Vernon systems are shown in Figures 3 and 4, respectively. The sampling tap locations corresponded to the GAC volumes shown in Table 3.

Experimental Design

The sampling frequency was varied to meet the needs of each experimental phase. The systems were monitored daily for 3 to 4 days and then every 2 to 5 days for approximately one month during the initial operating period (Phase I). Thereafter, during Phase II, they were monitored weekly, biweekly and then monthly. The system at Amherst was monitored for 122 days, while the system at Mont Vernon was monitored for 478 days. Several special sampling events were performed on the GAC systems, including coring the GAC material, diurnal observations, exposure to high flow and backwashing.

Coring

All GAC units were cored on July 29, 1987. Solid GAC samples were obtained by forcing a coring apparatus through the material as the bed was

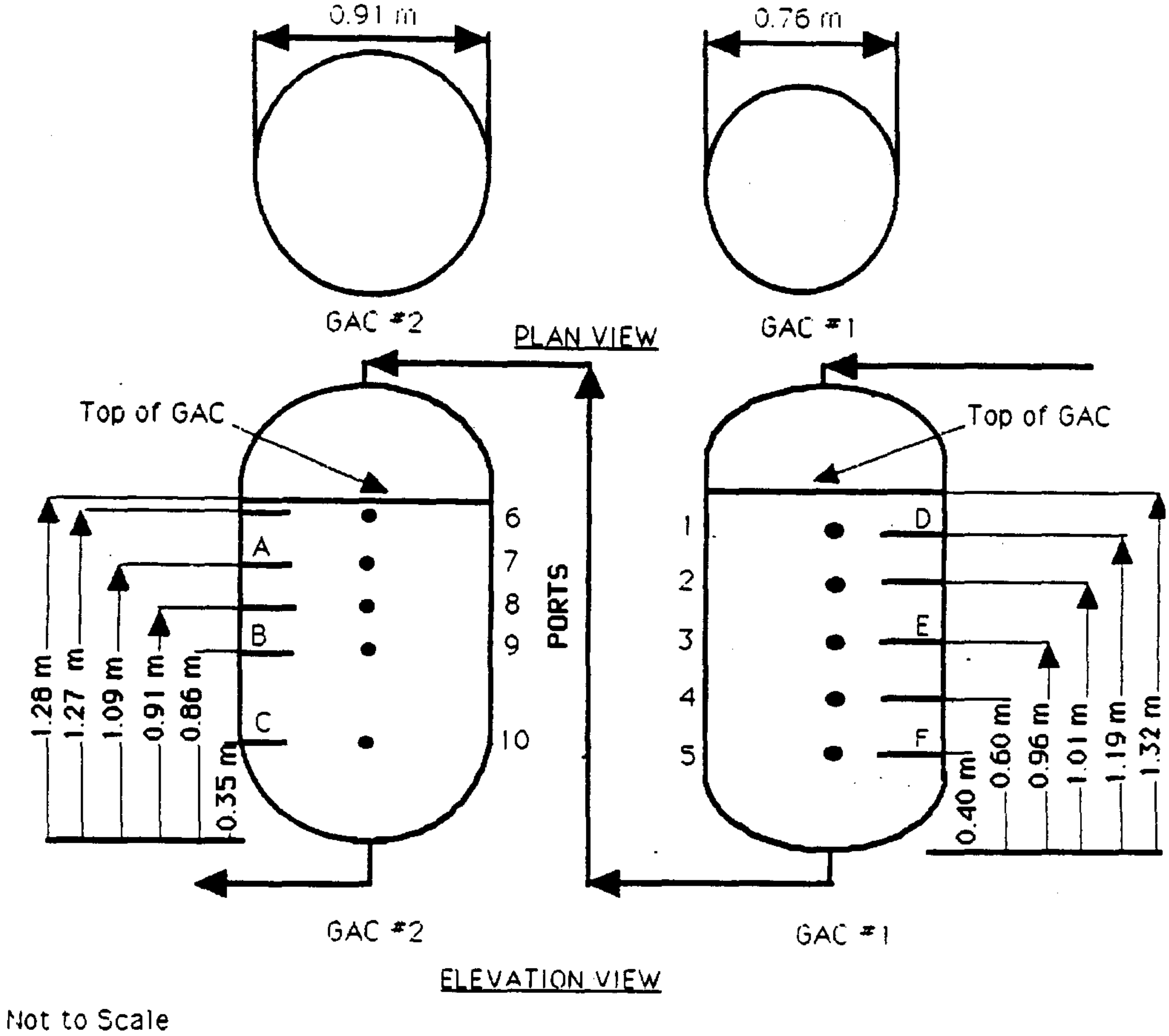


Figure 3. Schematic of the Mont Vernon, NH GAC system. A,B,C,D,E, and F are locations of gamma/beta emissions measurements.

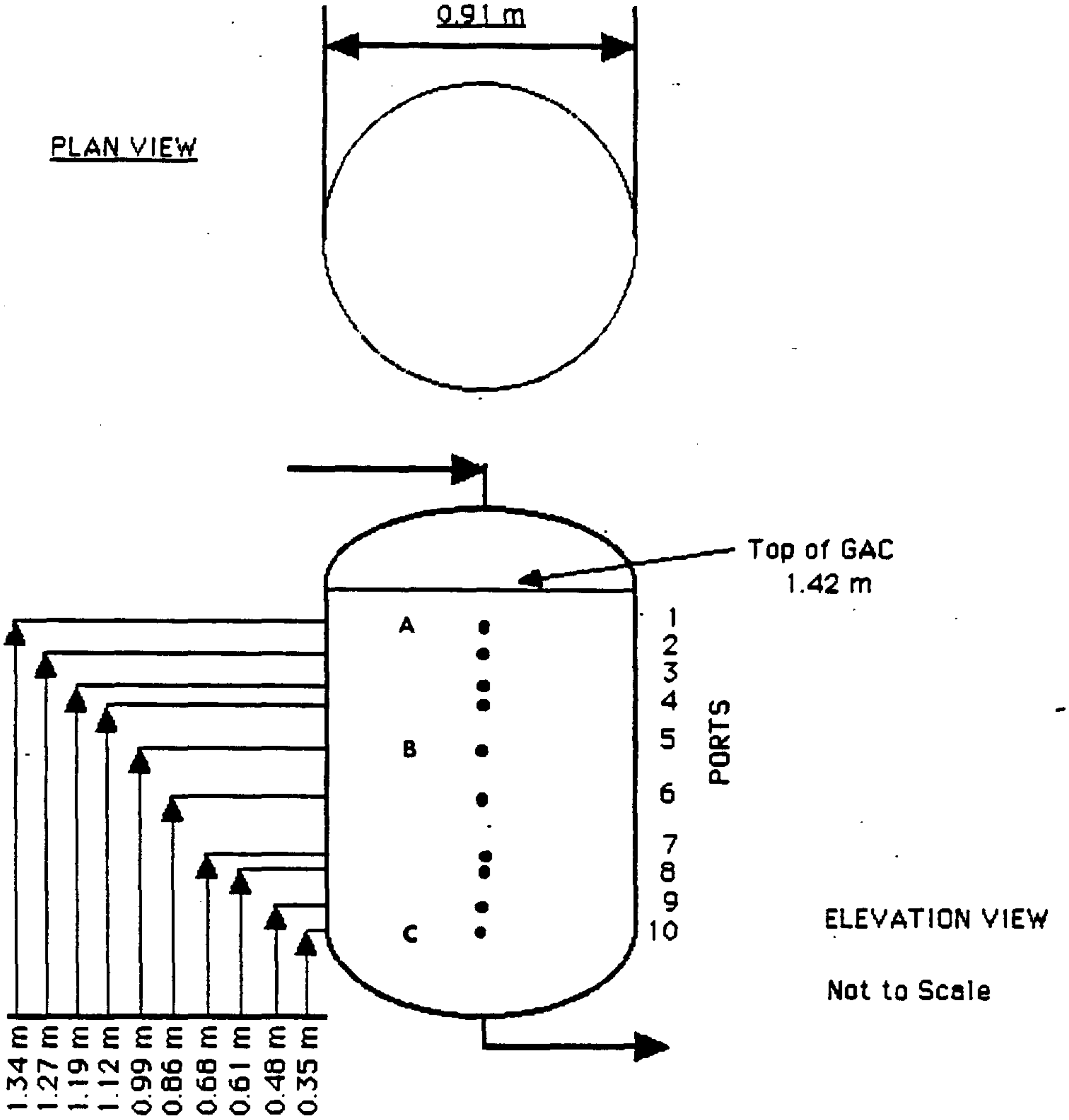


Figure 4. Schematic of the Amherst, NH GAC system. A, B and C are locations of gamma/beta emission measurements.

TABLE 3. RELATIONSHIP BETWEEN SAMPLING PORTS AND GAC VOLUMES

	Mont Vernon		Amherst	
Port Number	GAC Volume Between Ports (m^3)	Total GAC Volume (m^3)	GAC Volume Between Ports (m^3)	Total GAC Volume (m^3)
Influent				
1	0.058	0.058	0.050	0.050
2	0.081	0.14	0.050	0.10
3	0.081	0.22	0.050	0.15
4	0.081	0.30	0.050	0.20
5	0.12	0.42	0.084	0.28
6	0.16	0.58	0.084	0.37
7	0.12	0.69	0.084	0.45
8	0.12	0.81	0.084	0.53
9	0.12	0.93	0.084	0.62
10	0.25	1.17	0.084	0.70
Effluent	0.16	1.33	0.150	0.85

slightly expanded with backwash water (raw water from the storage tank). The coring device was a 2.54 cm diameter (I.D.) plexiglass tube 4 cm O.D. with 7.62 cm, sampling compartments centered every 10.16 cm. Samples of the bottom 36 cm, middle 36 cm and remaining material on top were composited. The samples were placed in sterile polyethylene bags and half-pint glass Mason jars.

The GAC samples in the jars were analyzed for uranium-238, uranium-235, and radium-226 by the State of the New Hampshire Radiological Laboratory and the Institute of Earth, Oceans, and Space at the University of New Hampshire (UNH). The UNH facility also analyzed the samples for lead-210. The GAC samples were also analyzed for percent moisture content, microbial numbers, iron and manganese.

Diurnal Variations

The units at Mont Vernon were sampled on July 1 and 2, 1987 to investigate the effect of diurnal variations in the water supply on the performance of the system. Samples were collected from the influent and effluent ports every hour for 12 hr (13 sampling times). The samples were analyzed for all parameters except microbial numbers, gamma/beta emissions, uranium and radium.

High Flow

On July 16, 1987, the units at Mont Vernon were sampled to observe the effects of high flow on the performance of the system. A tap installed after GAC #2 allowed the sampling team to waste 33 m^3/day of water in addition to the community demand, creating an artificially high flow through the GAC system. This represented a 91% increase in demand above the average flowrate. Samples were collected over a 3 hr period until the wells could not sustain the demand. Radon samples were collected from all ports. Samples for all other analyses were collected from the influent and effluent ports only. Uranium and radium analyses were not performed during this event.

Backwashing

The units at Mont Vernon were backwashed on January 28, 1988. The system was designed to be backwashed at 12.2 m^3/m^2-hr. This flowrate was too high, creating excessive loss of GAC. The backwash rate was lowered immediately to 55 m^3/day or 5.0 m^3/m^2-hr for GAC #1 and 3.4 m^3/m^2-hr for GAC #2. The units were backwashed separately for 10 min using raw well water from the storage tank.

Samples were collected prior to backwashing, as well as 1.3, 2.6, 3.9 and 24 hr after backwashing. Radon samples were collected from all ports. Samples for all other analyses, except uranium and radium, were collected from the influent, effluent and ports 1, 3, 5, 6, 8, and 10. Samples of the backwash water effluent were also collected at time = 0 and time = 10 min. These samples were analyzed for radon, iron, manganese, uranium and radium.

Sampling and Analysis

Water samples were collected directly from the plastic tubes connected to the sampling ports and flow was controlled using brass valves. The tubes were routed to a sampling board located some distance from the units to minimize the gamma radiation exposure to sampling personnel. Tubing was allowed to flush for 3 min before sampling commenced. Gamma/beta emissions were measured on the outside surface of the GAC contactors using a Geiger Müeller (GM) survey meter placed at locations shown in Table 4 and Figures 1 and 2. Flow was monitored using a flow totalizer downstream of each GAC system.

RESULTS AND DISCUSSION

Phase I - Initial Operation

Water samples were collected from the system at Mont Vernon daily for 4 consecutive days and every 2 to 5 days for approximately 32 days thereafter. Water samples were collected from the Amherst GAC system daily for 3 consecutive days and every 2 to 5 days thereafter. On October 12, 1986 (Day 4), the system at Amherst was shutdown because of problems with the high and low flow electrodes in the atmospheric storage tank. The system was started again on October 16.

By the end of Phase I, sufficient data on each system's operating characteristics were collected to allow sample frequency to be decreased to once per week.

pH and Alkalinity--

Mont Vernon site--The pH and alkalinity of the water changed dramatically through the GAC units during the first days of operation. The influent pH remained constant at 6.5 ± 0.20, while the effluent pH decreased from 9.9 to 6.5. There was a substantial increase in pH from Day 1 to Day 15 as the water passed through the units (Figure 5). The effluent pH remained significantly higher ($\alpha = 0.05$ and 0.01, Student's t test [StT]) than the influent throughout Phase I.

A similar increase through the unit was observed for alkalinity from Day 1 to 29 (Figure 5). The alkalinity increased markedly as the water passed through the units during the first days of operation. During the latter part of Phase I, the increase in alkalinity through the GAC system was less substantial, but still statistically significant ($\alpha = 0.05$ and 0.01, StT).

Amherst site--As with the Mont Vernon system, the pH of the water in Amherst was affected during Phase I. The influent pH averaged 7.9 ± 0.26, while the effluent pH decreased from 9.5 to 7.8. Figure 6 shows the marked increase in pH as the water passed through the unit from Day 1 until Day 13. During the remainder of Phase I, there was no significant difference between influent and effluent pH ($\alpha = 0.05$ and 0.10, StT). The influent pH of the Amherst water was significantly ($\alpha = 0.05$ and 0.01, StT) higher than the Mont Vernon water's pH (6.5 ± 0.20).

TABLE 4. LOCATIONS OF GAMMA/BETA EMISSIONS MEASUREMENTS* ON GAC UNITS

Mont Vernon Port Number	Mont Vernon Total+ GAC Volume (m³)	Amherst Port Number	Amherst Total+ GAC Volume (m³)
1	0.058	1	0.05
3	0.22	5	0.28
5	0.42	10	0.70
6	0.58		
8	0.81		
10	1.17		

*Measurements were made on the outside surface of the GAC contactors.

+Based on distance from top of GAC in contactors .

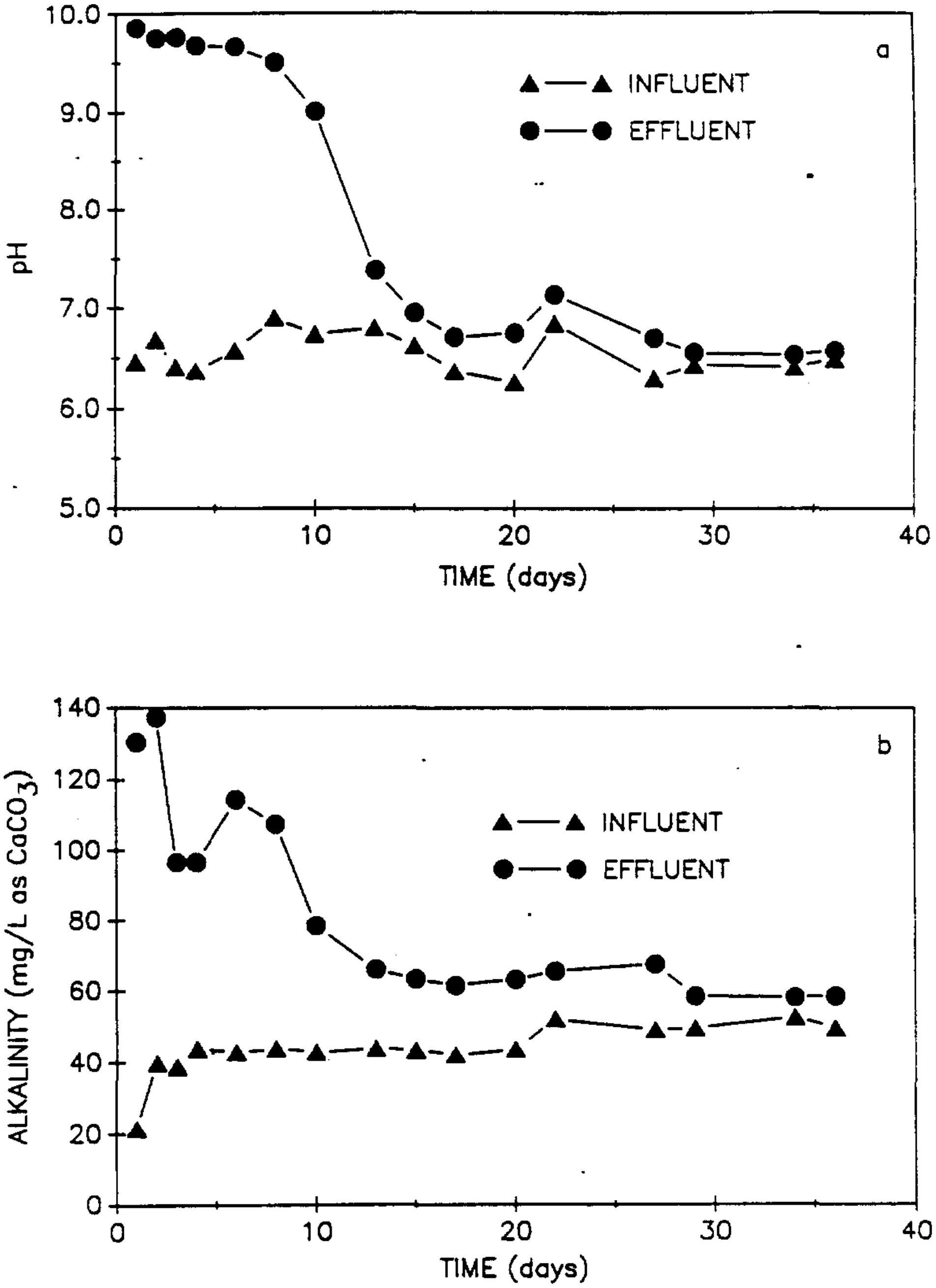


Figure 5. Phase I - Mont Vernon, NH GAC system. pH (a) and alkalinity (b) data for first 36 days operation.

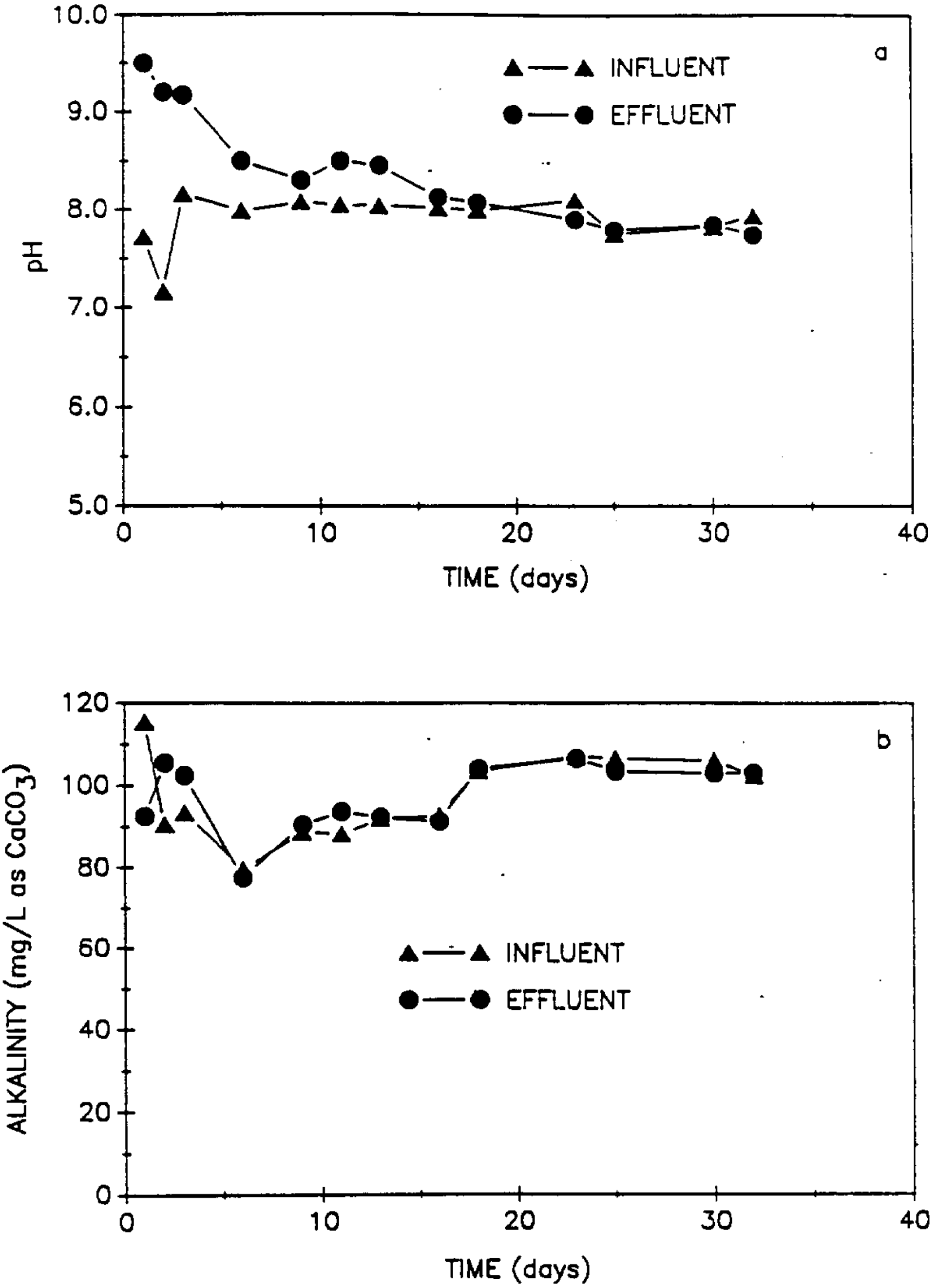


Figure 6. Phase I - Amherst, NH GAC system. pH (a) and alkalinity (b) data for first 32 days of operation.

The influent (97 ± 10.1 mg/L as $CaCO_3$) and effluent (97 ± 8.5 mg/L as $CaCO_3$) alkalinities at Amherst were not significantly different ($\alpha = 0.05$ and 0.01, StT) during Phase I (Figure 6). Unlike the Mont Vernon system, there was no increase in alkalinity through the unit.

The pH and alkalinity data obtained at Mont Vernon and Amherst during their start up periods is best explained by the influent conditions and the chemical composition of the GAC. It has been shown that coconut shell-based carbons can contain significant quantities of $CaCO_3(s)$ (from the base material) and lime ($Ca(OH)_2$ or CaO) from the activation process (S.J. Shields; Barnebey and Sutcliffe, Corp.; personal communication; 1989). Therefore, the initial rise in pH and alkalinity observed at Mont Vernon can be explained by the dissolution of $CaCO_3(s)$ and/or lime present in the GAC. Further, the theoretical solubility for $CaCO_3(s)$ (Stumm and Morgan, 1981) for the water quality conditions at Mont Vernon and Amherst would predict that rapid dissolution should occur below pH 8 (i.e., at the Mont Vernon and Amherst influent pH's) and that the equilibrium pH of the $CaCO_3(s)$ system following dissolution would be between pH 9.5 and 10.5 which corresponds to the peak effluent pH values observed at both sites.

The differences observed at Mont Vernon and Amherst are likely the result of several factors: (i) influent pH and alkalinity at Amherst were much higher than at Mont Vernon, (ii) the Amherst system contained only 0.85 m^3 of GAC whereas the Mont Vernon system contained 1.33 m^3, and (iii) the average daily flow at Amherst (59 m^3/day) was almost double that of Mont Vernon (31 m^3/day). These factors serve to explain why the pH and alkalinity changes (influent vs. effluent) observed at Amherst where reduced and occurred for a shorter period. Subsequent data for Mont Vernon shows that by Day 50 there were no significant differences between influent and effluent pH or alkalinity. This confirms that the trend observed for Amherst also occurred at Mont Vernon, but due to the greater volume of GAC and lower flowrate, a longer operating period was required to flush the system of the $CaCO_3$ and/or lime.

Temperature and Dissolved Oxygen--

Mont Vernon site--The temperature of the water was not affected by the treatment units, never changing by more than 1.0°C from influent to effluent (Figure 7). The average temperature of the effluent water was 10.9 ± 0.9°C and did not differ significantly ($\alpha = 0.05$ and 0.01, StT) from the influent (11.1 ± 0.7°C). This was expected, in spite of the detention times (0.4 - 1.7 hr) because the pumphouse was not heated. The startup period occurred in October and November when ambient air temperatures in New England are cool.

Dissolved oxygen (DO) in the influent water averaged 10.8 ± 0.73 mg O_2/L during Phase I. A substantial amount of DO was removed on Day 1 (influent = 9.8 mg O_2/L, effluent = 1.5 mg O_2/L) (Figure 7). There was an increase in effluent DO from 1.5 mg O_2/L on Day 1 to 4.5 mg O_2/L on Day 15. However, by

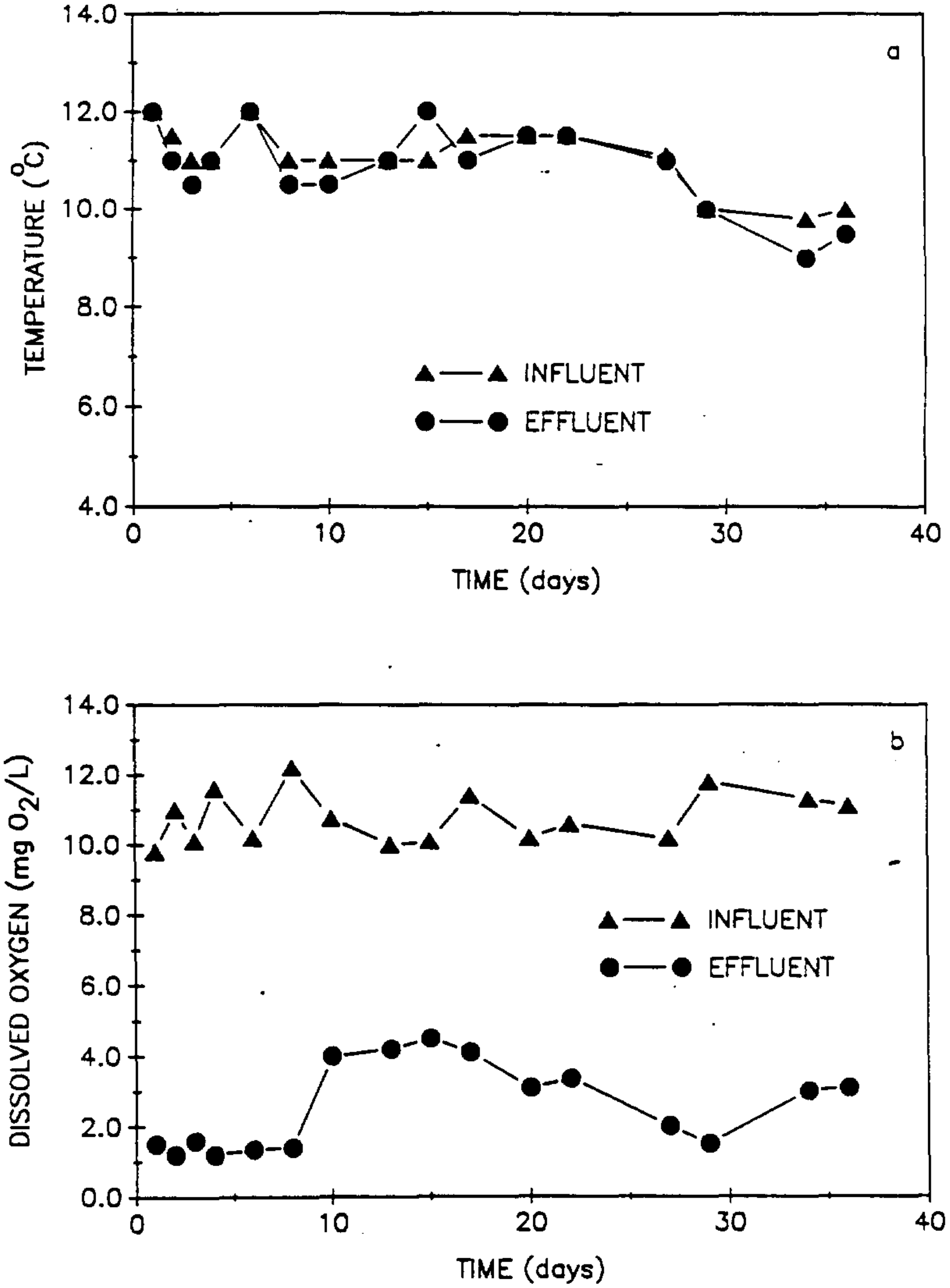


Figure 7. Phase I - Mont Vernon, NH GAC system. Temperature (a) and dissolved oxygen (b) for the first 36 days of operation.

Day 27 the effluent DO concentration decreased to 2.0 mg O_2/L and remained substantially lower than the influent throughout Phase I. Overall, the effluent DO concentration averaged 2.4 ± 1.2 mg O_2/L for the 36 day period of Phase I.

Amherst site--The temperature of the water was not affected by the treatment unit, never changing by more than 1.0°C from influent to effluent (Figure 8). The average temperature of the effluent water (11.7 ± 1.1°C) did not differ significantly (α = 0.05 and 0.01, StT) from the influent water temperature of 11.9 ± 1.0°C. The temperature of the water was not expected to change because of the short detention time in the GAC unit ranging from 0.28 to 0.37 hr and the cool ambient air temperatures in the pumphouse during the start-up period (October and November).

Dissolved oxygen in the influent water averaged 4.9 ± 0.67 mg O_2/L during Phase I. During this 32 day period, DO was consistently removed by the units (Figure 8), and the effluent DO concentration averaged 2.8 ± 0.84 mg O_2/L.

Analysis of the temperature and DO data collected during Phase I indicates that both systems were performing as expected. The minimal change in temperature is consistent with results observed for numerous applications of GAC in groundwater treatment. Snoeyink (1983) reported that the effect of temperature on GAC adsorption from groundwater would be minimal compared to other factors.

The DO data can be evaluated by computing an average mass of DO adsorbed per mass of GAC for each system based on average flow, average influent DO and effluent DO concentration and the time of operation. The resulting calculations indicate that 13.9 mg O_2/g GAC were adsorbed during the 36 day period of Phase I at Mont Vernon and 9.1 mg O_2/g GAC were adsorbed during Phase I (32 days) at Amherst. These values compare well to those reported by Prober et al. (1975), who found DO adsorption averaged 4.8 to 21 mg O_2/g GAC for a variety of GAC's tested up to a period of 1,700 hrs (70.8 days). They also reported that GAC exhaustion was not reached during the 70.8 day study. This finding is consistent with results observed at Mont Vernon and Amherst, during Phase I, since neither system demonstrated complete GAC exhaustion with respect to DO (i.e., influent DO = effluent DO).

The differences observed in DO adsorption at Mont Vernon and Amherst are likely the result of the influent DO concentrations and the volume of GAC at each site. Since the driving force (i.e., concentration gradient) of DO adsorption at Mont Vernon was higher than at Amherst, the higher non-equilibrium adsorption of DO observed at Mont Vernon would be expected. Further, Prober et al. (1975) concluded that up to 95% of the DO adsorbed by GAC results from surface reactions with GAC forming carboxylic and other acidic surface groups. Therefore, the fact that Mont Vernon had 1.33 m^3 of GAC (Amherst = 0.85 m^3) may also explain the higher DO adsorption at Mont Vernon.

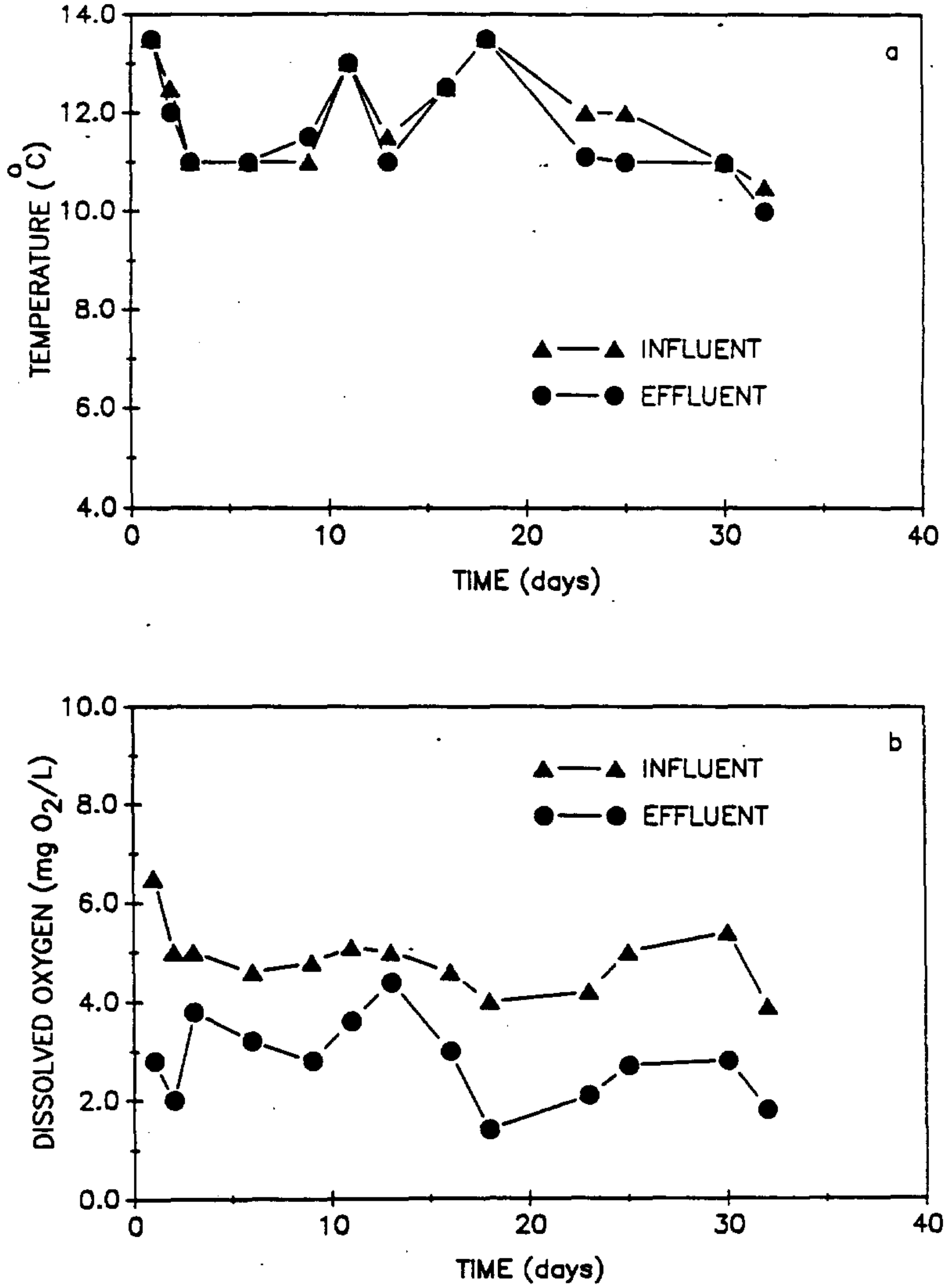


Figure 8. Phase I - Amherst, NH GAC system. Temperature (a) and dissolved oxygen (b) for first 32 days of operation.

Turbidity--

Mont Vernon site--During Phase I, the average influent turbidity was 0.17 ± 0.05 NTU (Figure 9). The effluent turbidity during this period (0.12 ± 0.04 NTU) was significantly less than the influent (α = 0.05 and 0.01, StT). In a survey of water treatment plants in the U.S. that used GAC, influent turbidities of 1.1 to 7.0 NTU were decreased to 0.1 to 0.5 NTU through the units (Graese, et al. 1987). Although a slight reduction in turbidity by the Mont Vernon system was observed, the low influent turbidity, typical of many groundwaters, made hydraulic headloss through the GAC bed negligible during Phase I.

Amherst site--The average Phase I influent turbidity was 1.75 ± 0.98 NTU (Figure 10). This is significantly higher (α = 0.05 and 0.01, StT) than Mont Vernon's. Some of this difference may have been a function of the iron precipitates observed entering the Amherst GAC system from the atmospheric storage tank. The effluent turbidity during this period was 0.35 ± 0.11 NTU, which was significantly less than the influent turbidity (α = 0.05 and 0.01, StT). Graese et al. (1987) surveyed a GAC water treatment plant in the United Kingdom (GAC type and volume unknown) whose influent turbidity of 1.5 NTU decreased to 0.32 NTU through the unit, similar to the trend observed in Amherst. Despite fluctuations in influent turbidity, the GAC unit produced a relatively constant effluent. The effluent turbidity at Amherst may have been higher than Mont Vernon's because the GAC volume was lower and raw water turbidity and flow were higher. Despite the higher turbidity reduction at Amherst, headloss was not considered significant during Phase I.

Flowrate--

Variations in water flowrate are significant in radon removal, because removal is controlled by the empty bed contact time (EBCT) which is a function of contactor volume and flowrate (Lowry and Brandow, 1985). The average flowrate was determined from the total volume of water passed through the GAC systems over periods between sampling. During Phase I, measurements were taken daily or every few days, which was considered adequate for this study. However, in small systems where rapid changes in flow can significantly affect the data, flows should be taken immediately before and after each sampling event for a more accurate determination of mass radon removals.

Mont Vernon site--The average flow during Phase I at Mont Vernon was 24 ± 4 m^3/day (Figure 11) yielding an average EBCT of 78.5 min. The GAC system was designed for an average flow of 25 m^3/day (EBCT=77.9 min).

Amherst site--The average flow at the Amherst facility during Phase I was 59 ± 3.2 m^3/day (Figure 12) which corresponds to an average EBCT of 20.6 min. Flow data was not collected prior to Day 6 because there was a delay in installation of the monitoring equipment. This flowrate was much higher than that monitored at Mont Vernon and higher than the anticipated design flow of 34 m^3/day (EBCT=35.5 min). The flowrate was fairly constant over the period, except when the GAC unit was shutdown by the operator (Days 4 to 8).

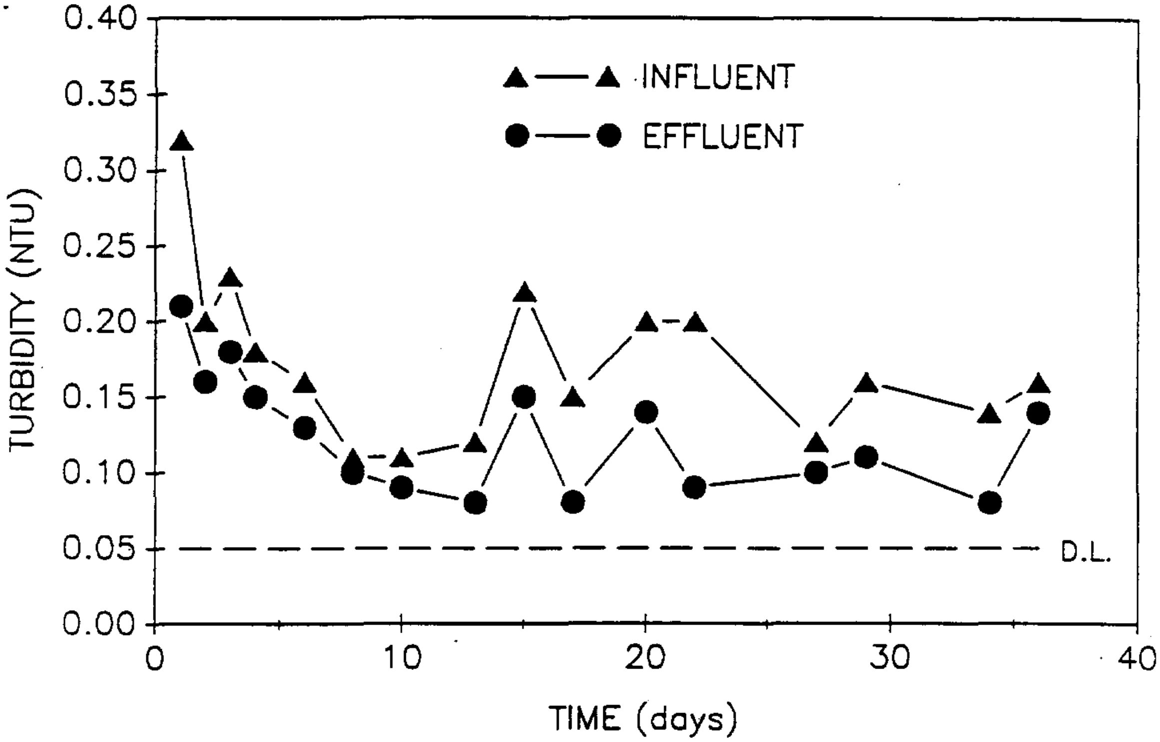


Figure 9. Phase I - Mont Vernon, NH GAC system. Turbidity during first 36 days of operation (detection limit = 0.05 NTU).

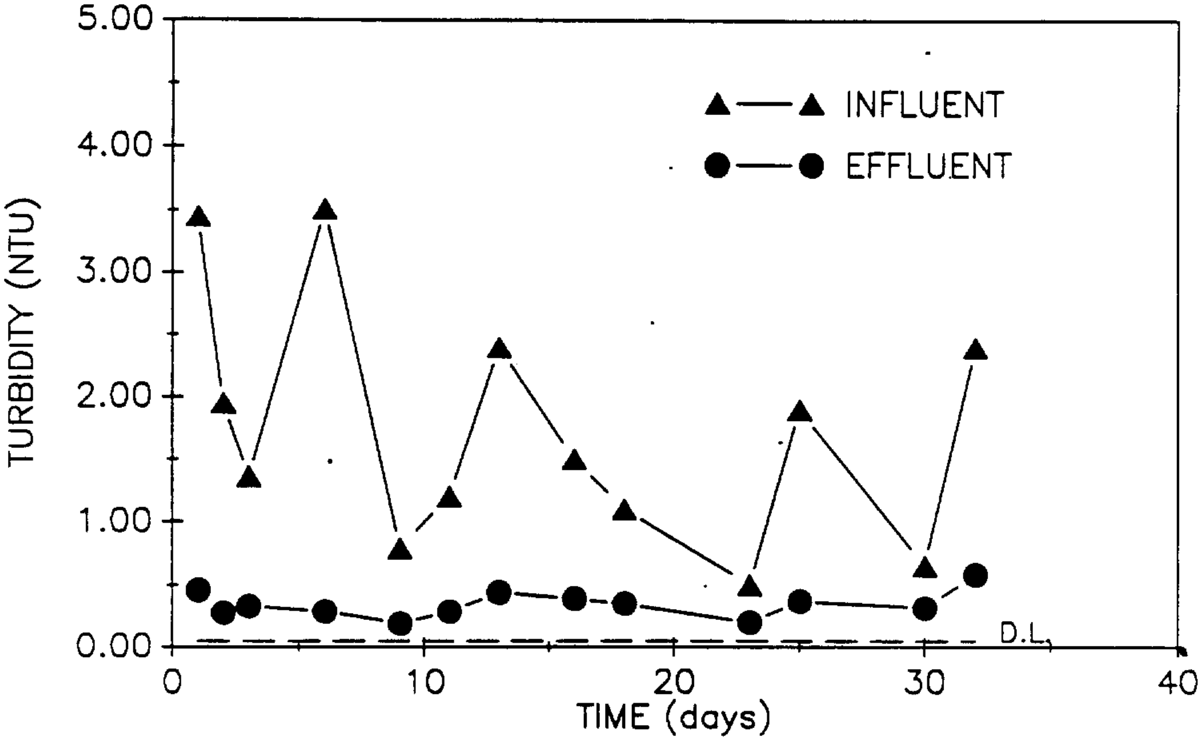


Figure 10. Phase I - Amherst, NH GAC system. Turbidity during first 32 days of operation (detection limit = 0.05 NTU).

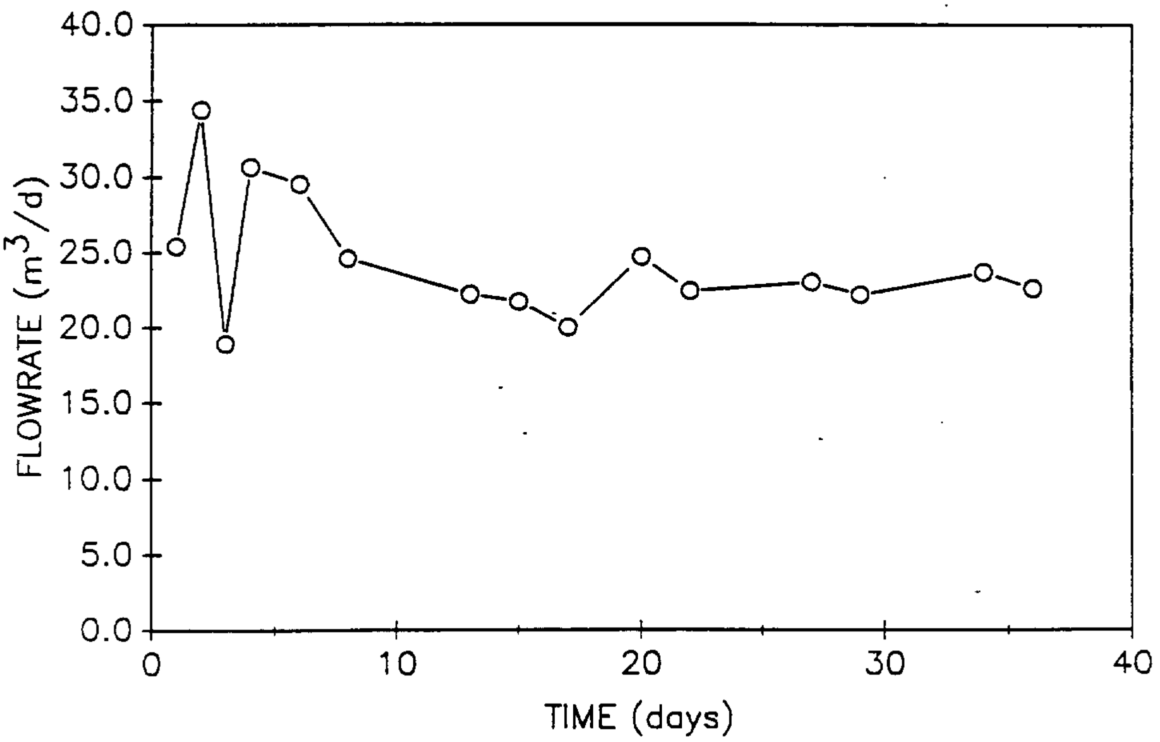


Figure 11. Phase I - Mont Vernon, NH GAC system. Water flowrate through GAC during first 36 days of operation.

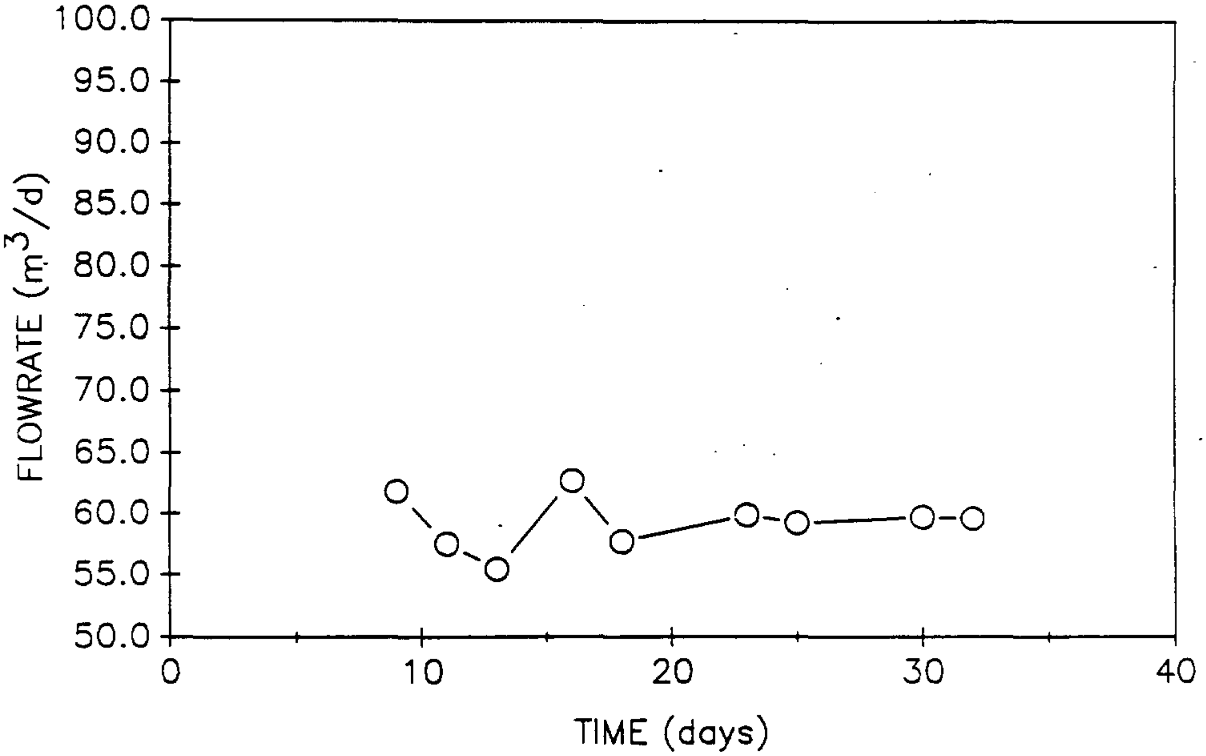


Figure 12. Phase I - Amherst, NH GAC system. Water flowrate from Day 6 to Day 32 (metering equipment not installed until Day 5).

Radon--

Mont Vernon site--The GAC system at Mont Vernon was designed to treat an influent radon activity of 155,000 pCi/L and to achieve a 98% radon removal efficiency once steady-state operation was achieved (predicted effluent activity = 2,860 pCi/L). The average influent radon activity during Phase I was 249,900 ± 45,800 pCi/L.

Figure 13 presents radon data collected from each port during Phase I. All of the radon applied was removed in the first unit (GAC #1) during the first few days of operation. The radon removal wave front moved through the units over time, eventually breaking through into the effluent by Day 25. Prior to that the radon activity was below the detection limit (138 pCi/L) in the effluent from GAC #2. By Day 36, the effluent activity from GAC #2 was 2530 pCi/L.

Based on the adsorption data for GAC #2, this unit was not at steady state with respect to radon removal by the end of Phase I. The data from GAC #1 indicated that it was near steady state conditions within approximately 20 days, considering the varying water flowrate and influent radon activity inherent in a small community water supply. Using a radon adsorption model developed for GAC by Lowry (1985), the influent and effluent radon activity and the averaged flowrate over the period, the steady state radon removal rate constant (K_{ss}) was determined for comparative purposes. The K_{ss} for GAC #1 averaged 3.3 hr^{-1} (ranging from 2.7 hr^{-1} in the top 0.14 m^3 of GAC to 4.9 hr^{-1} in the bottom 0.15 m^3) once secular equilibrium was achieved between radon and its progeny. The design K_{ss} used for this carbon at water temperatures in the range 6-10°C was 3.02 hr^{-1} (Lowry and Lowry, 1987), which was similar to the average value observed. It is well documented for conventional pollutants (Weber, 1972; Snoeyink, 1983; Crittenden, 1987) that GAC adsorption is affected by column hydraulics and site specific raw water quality conditions. Therefore, variations in K_{ss} would be expected as column hydraulics and raw water quality change. As predicted by Lowry (1985), the GAC unit dampened changes in loading once it was at steady state.

Amherst site--The GAC system at Amherst was designed to treat an average influent of 39,750 pCi/L and provide a steady-state effluent of 6,280 pCi/L (84% radon removal) at a flowrate of 34 m^3/day. The average influent radon activity during Phase I was 56,600 ± 11,100 pCi/L (Figure 14).

As observed at Mont Vernon in GAC #1 (0.57 m^3), the GAC system at Amherst (0.85 m^3) came to a steady state condition fairly rapidly (~15 days). This is similar to observations ranging from 10-22 days for other GAC units (Lowry et al., 1987; Lowry and Brandow, 1985; Reed, 1989). The radon appeared in the effluent within a few days of startup and increased to a steady state average of 7,971 ± 738 pCi/L after 15 days of operation. Fluctuations in influent loading were also dampened in the Amherst system, however, to a lesser extent than at Mont Vernon where there was a greater volume of GAC.

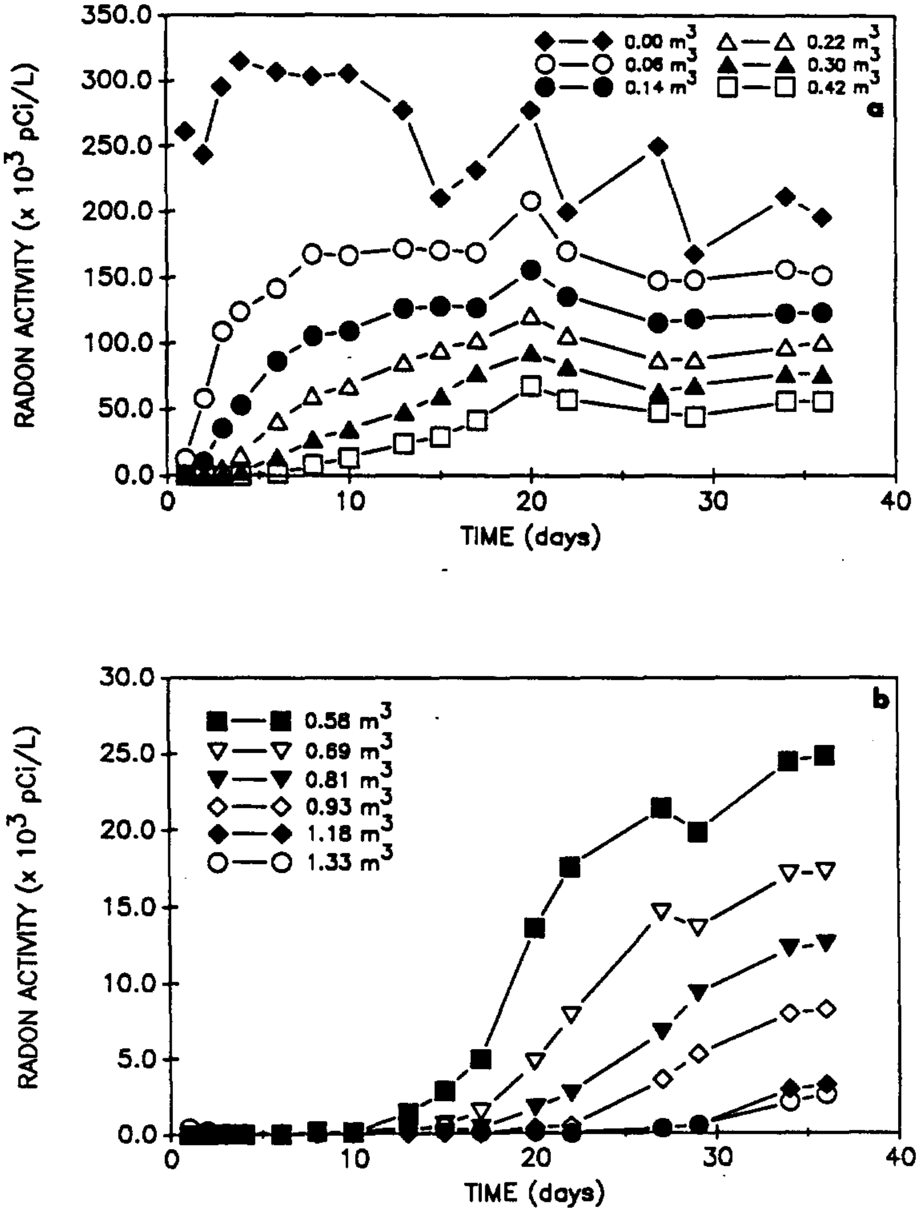


Figure 13. Phase I - Mont Vernon, NH GAC system. Radon activity during first 36 days of operation for GAC #1 (a) and GAC #2 (b). Note difference in scales on ordinate in (a) and (b).

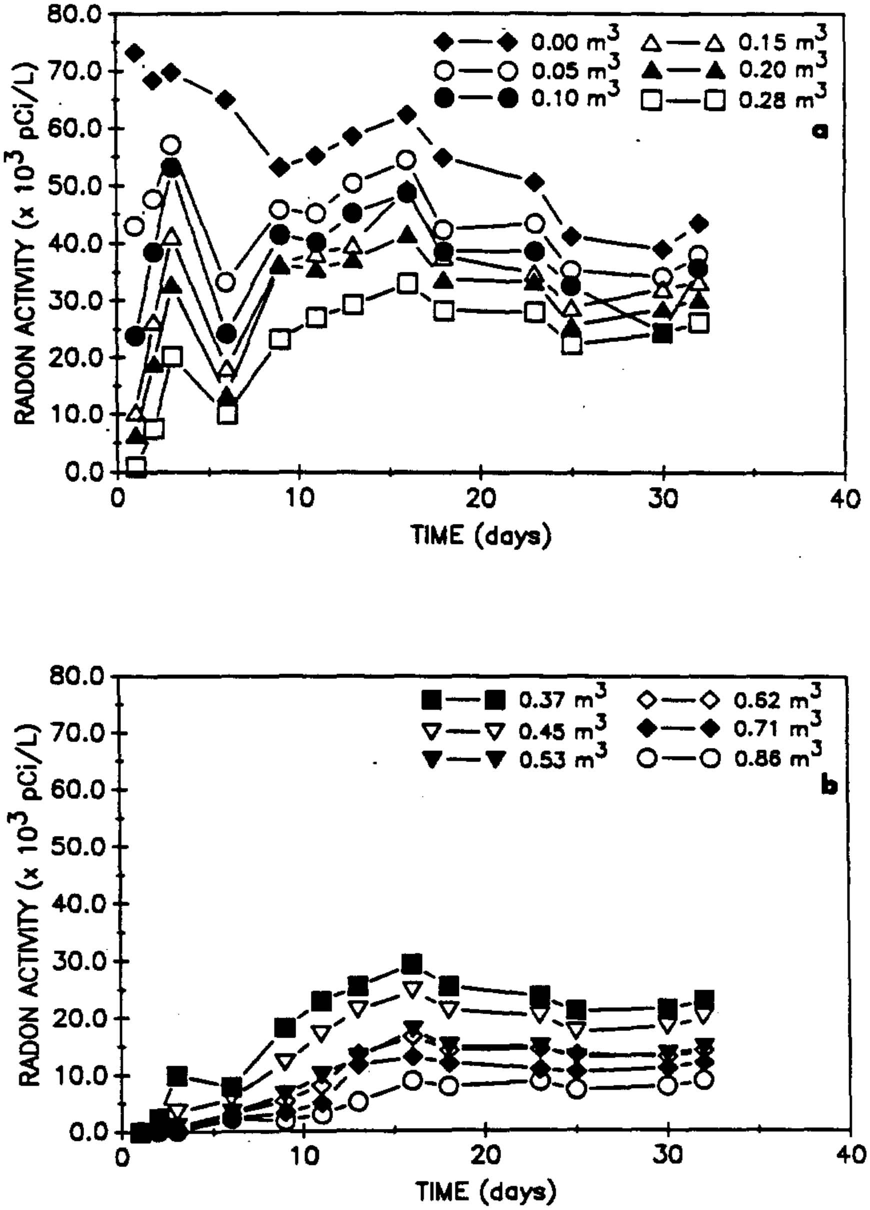


Figure 14. Phase I - Amherst, NH GAC system. Radon activity during first 32 days of operation for top 0.28 m^3 (a) and remaining 0.58 m^3 of GAC (b).

For the Amherst GAC system, the K_{ss} averaged 5.8 ± 1.44 hr^{-1} in the bottom 0.41 m^3 (when the radon loading and flowrate were fairly constant) once the unit was at secular equilibrium. These K_{ss} values are substantially higher than those predicted (3.02 hr^{-1} at 6-10°C) for this GAC (Lowry and Lowry, 1987) and may be a function of the column hydraulics and/or the raw water quality at the Amherst site.

Gamma/Beta Emissions--

Mont Vernon site--Results of the Geiger Müeller (GM) survey meter readings taken at the surface of the GAC units during Phase I are presented in Figure 15. The GM survey meter measures both gamma and beta emissions when the detector is unshielded. Two of the short-lived progeny of radon, lead-214 and bismuth-214, emit fairly high energy gamma radiation. This gamma activity penetrates the fiberglass tanks and is detected by the survey meter. For GAC #1, these emissions were fairly constant within 20 to 30 days of start-up corresponding to the radon activities observed. Gamma/beta emissions from GAC #2 increased after Day 5 corresponding to the breakthrough of radon from GAC #1. Figure 16 shows the correlation between the radon activity at the closest port to the gamma/beta emissions measurements during Phase I for GAC #1.

Amherst site--The GM survey meter reading results for Amherst are presented in Figure 17. The data from the top of the unit show some fluctuation, which was dampened through the system. Again, the radon activity was correlated with the gamma/beta emissions (Figure 18), however at this site a non-linear relationship appears to provide the best fit of the data.

Microbial Numbers--

These analyses were not performed during Phase I.

Iron and Manganese--

Mont Vernon site--On several occasions during the beginning of operation, the effluent concentrations of both iron and manganese exceeded the influent concentration (Figure 19). Thereafter, the influent iron and manganese were usually higher than the effluent from the GAC system. After Day 17, the majority of the influent and effluent data were at or near the detection limit (iron = 0.06 mg/L, manganese = 0.02 mg/L), so changes in iron and manganese could not be determined through the GAC.

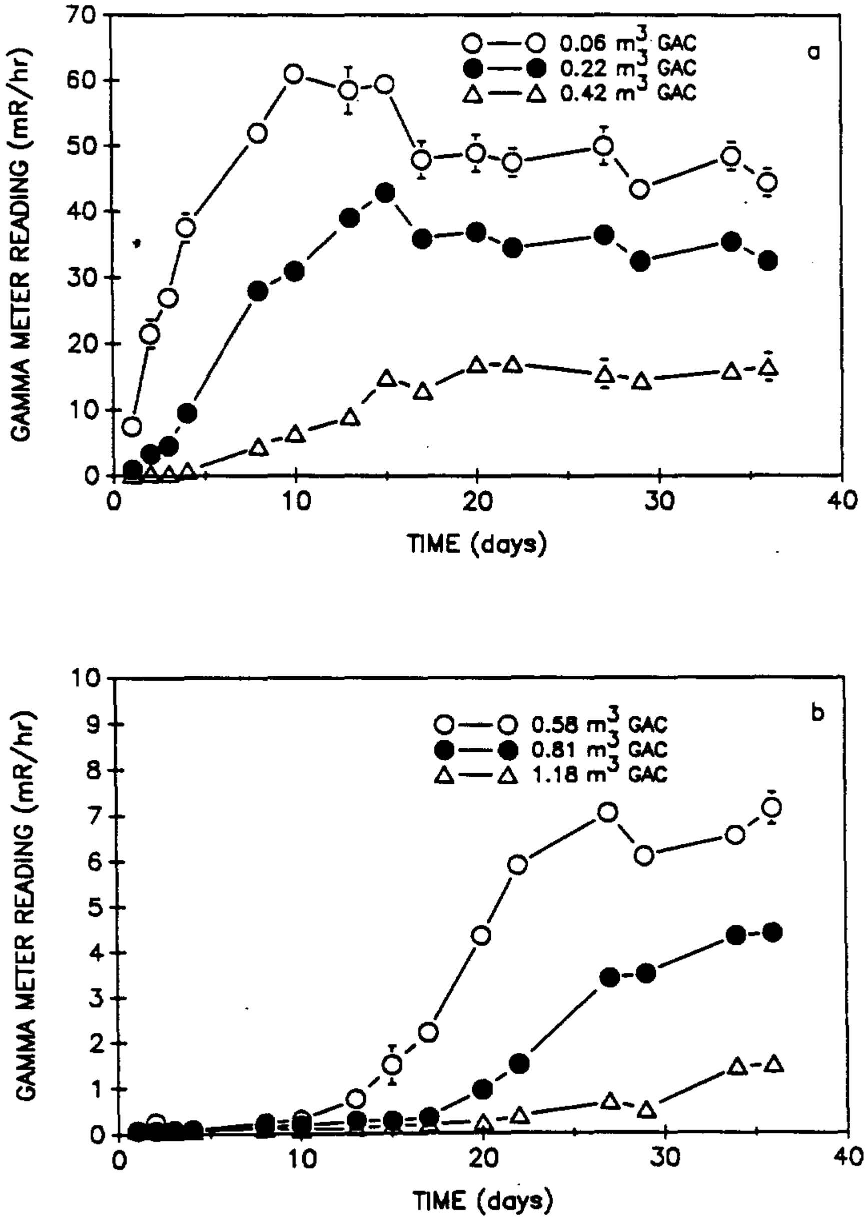


Figure 15. Phase I - Mont Vernon, NH GAC system. Gamma/beta emission measurements taken on contactor surfaces during first 36 days of operation for GAC #1 (a) and GAC #2 (b). Note difference in scale.

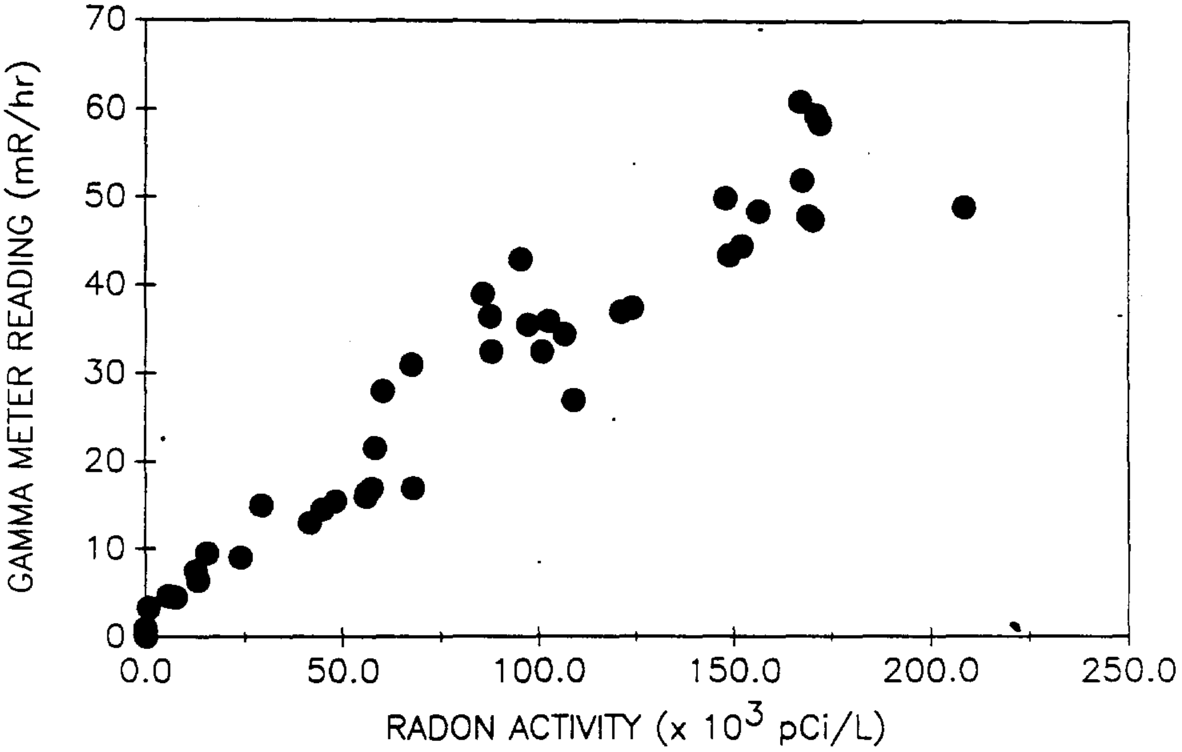


Figure 16. Phase I - Mont Vernon, NH GAC system. Gamma/beta emission measurements taken on the surface of GAC #1 as a function of influent radon activity to sections of GAC #1.

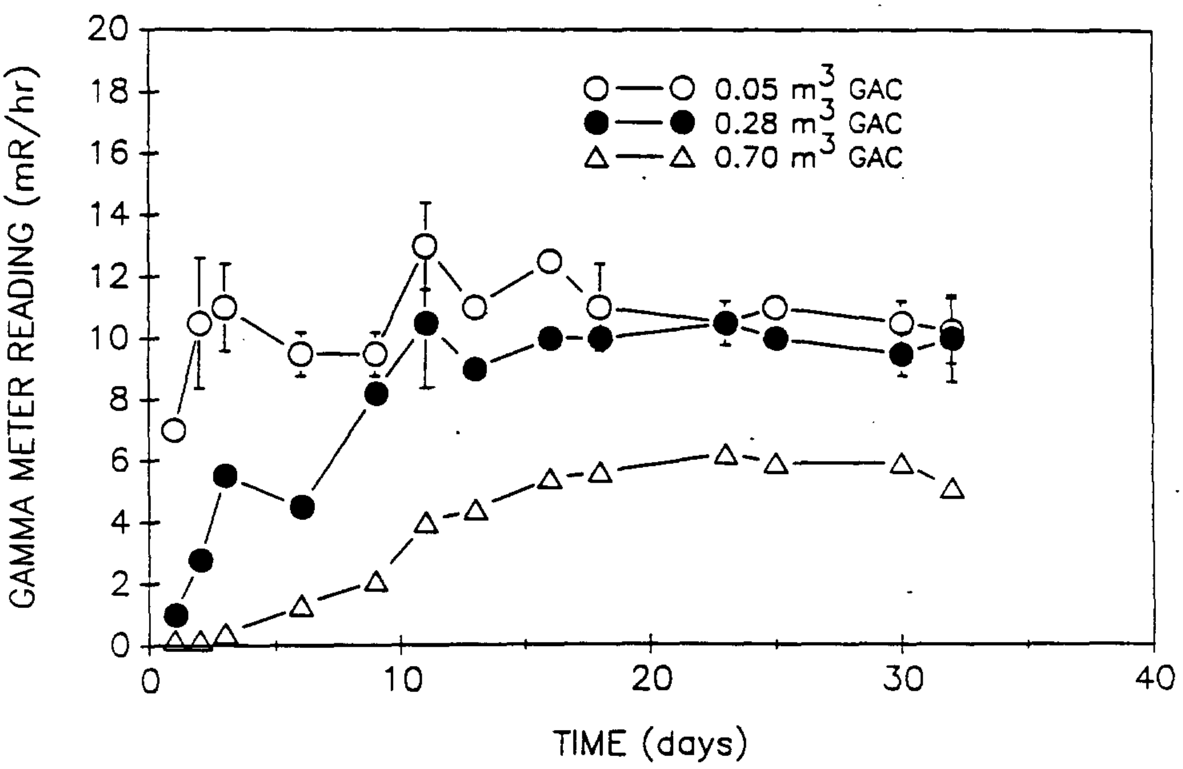


Figure 17. Phase I - Amherst, NH GAC system. Gamma/beta emission measurements taken on the contactor surface during first 32 days of operation.

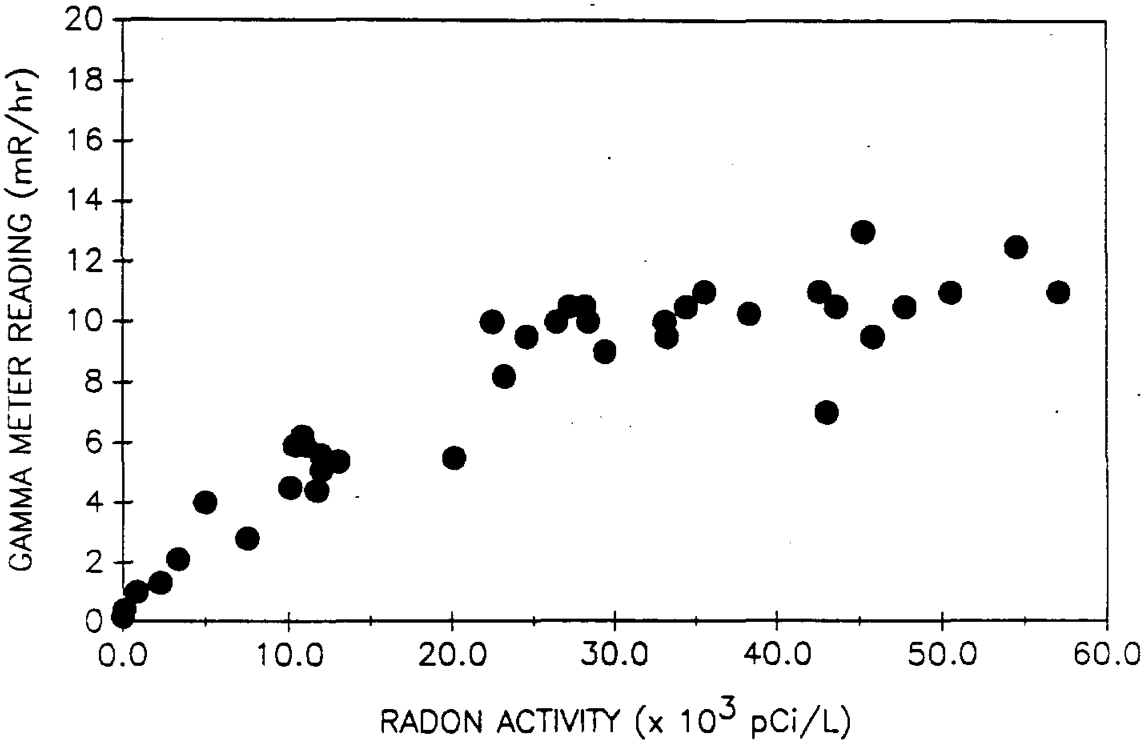


Figure 18. Phase I - Amherst, NH GAC system. Gamma/beta emission measurements taken on the contactor surface as a function of influent radon activity to sections of the GAC system.

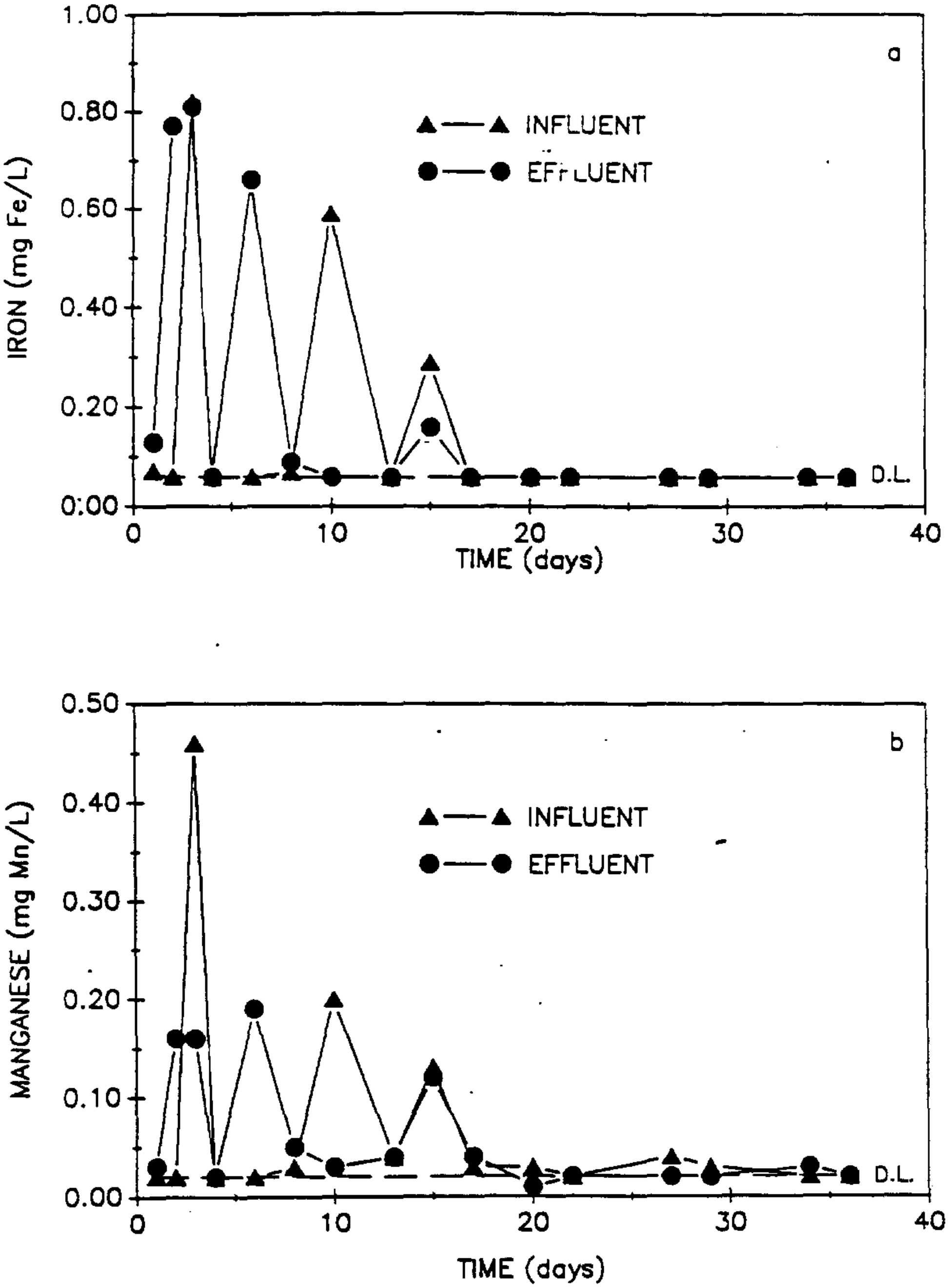


Figure 19. Phase I - Mont Vernon, NH GAC system. Iron (a) and manganese (b) data for the first 36 days of operation. (Detection limits: iron = 0.06 mg/L and manganese = 0.02 mg/L).

Amherst site--Influent iron concentrations were significantly higher (α = 0.05 and 0.01, StT) in the Amherst water than the Mont Vernon water (Figure 20). There was no significant difference (α = 0.05 and 0.10, StT) in raw water manganese concentrations between the sites (Figure 20). The average influent iron and manganese concentrations at Amherst were 0.50 ± 0.30 mg Fe/L and 0.09 ± 0.08 mg Mn/L, respectively. The average effluent concentrations were 0.19 ± 0.29 mg Fe/L and 0.06 ± 0.07 mg Mn/L. There was a significant difference at α = 0.05 and 0.02 (StT) in influent and effluent iron, but not at α = 0.01. Removal was observed for both metals through the GAC system which has been observed in other GAC units (Graese, 1987).

Phase II - Long-Term Monitoring

During Phase II, the GAC system at Mont Vernon was monitored weekly for 16 weeks, bi-weekly for 24 weeks and then monthly for two months. Between September 17, 1987 (Day 344) and October 15, 1987 (Day 372), a new well with a higher yield was drilled and replaced the existing wells as the water supply. The water in the new well contained less radon and DO, but considerably more uranium and radium and had a higher pH and alkalinity than the original water supply (Table 5). These data support observations by Boudette (NH State Geologist; Durham, NH; personal communication, 1988) that there may be considerable variation in radon activity in groundwater from neighboring wells, largely as a function of their depth within the aquifer.

The GAC system at Amherst was monitored weekly for 5 weeks and then bi-weekly for 12 weeks. Several problems were encountered with the water supply system after Day 122. The major problem was the very low well yield. In addition, the individual employed to monitor the water supply system repeatedly bypassed the GAC unit by mistake. As a result, the system was abandoned after Day 122.

pH and Alkalinity--

Mont Vernon site--Alkalinity and pH did not differ significantly (α = 0.05 and 0.01, StT) as the water flowed through the GAC system (Figure 21). The alkalinity of the influent water decreased from 49.5 mg/L as $CaCO_3$ on Day 36 to 26.3 mg/L as $CaCO_3$ on Day 43. A similar decrease was observed in the effluent alkalinity indicating that the GAC system was not affecting this parameter. Figure 22 shows typical profiles of pH and alkalinity through the GAC units on several days. Little difference in either parameter was observed from port to port. Day 372 reflects the change in pH and alkalinity resulting from the new water supply.

The continuous monitoring data for pH and alkalinity at Mont Vernon supports the hypothesis that the initial increase in those parameters during Phase I was related to the alkaline bearing residues in the GAC. As predicted by that hypothesis, and observed with the Amherst data, the alkaline material was eventually leached out (solubilized) and the influent and effluent pH and alkalinity were no longer affected by the GAC.

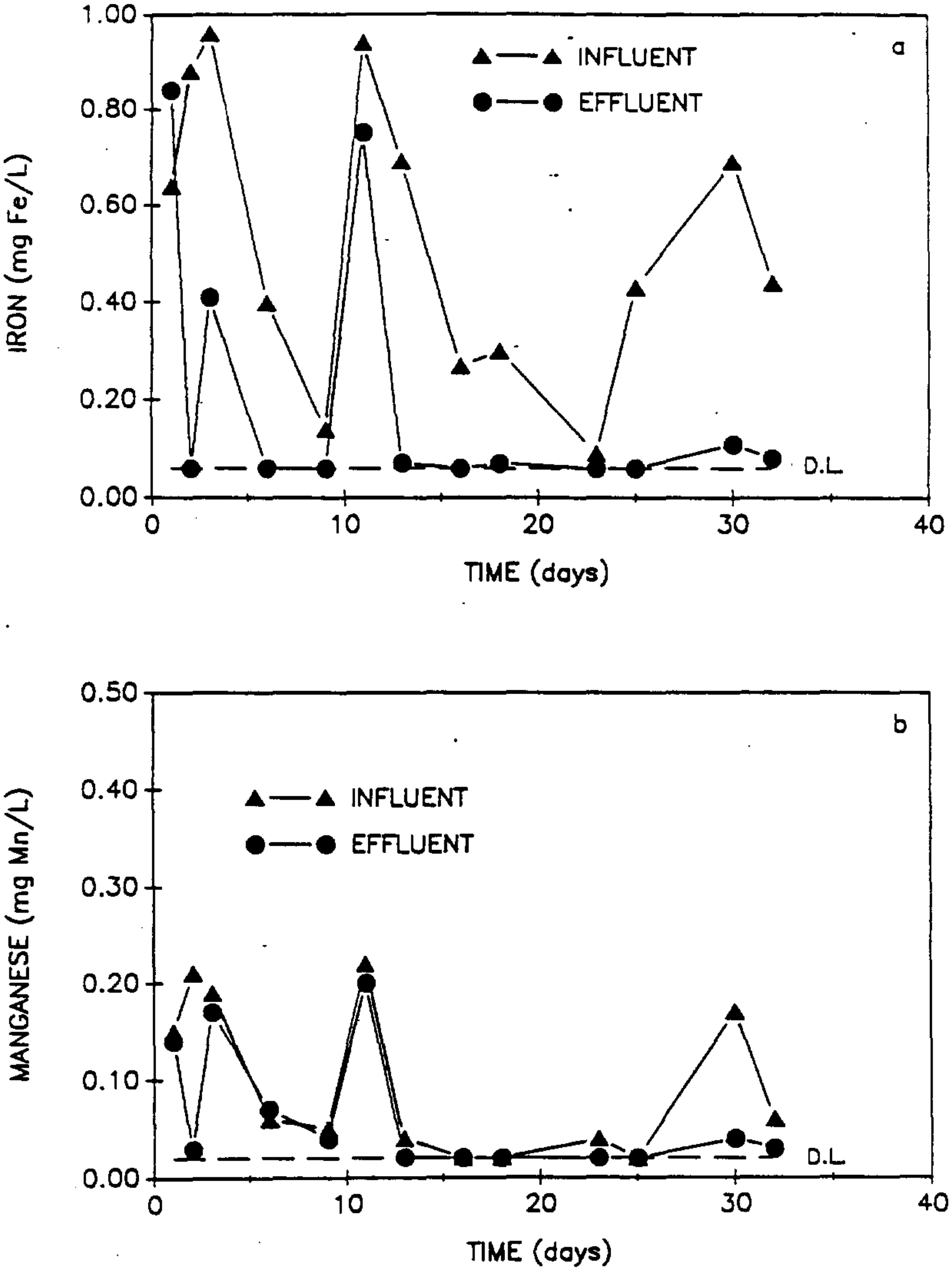


Figure 20. Phase I - Amherst, NH GAC system. Iron (a) and manganese (b) for the first 32 days of operation. (Detection limits iron = 0.06 mg/L and manganese = 0.02 mg/L).

TABLE 5. MONT VERNON WATER SUPPLY ORIGINAL WELLS VS. NEW WELL

	Original Wells*	New Well+
Radon (pCi/L)	210,500 ± 41,400	68,900 ± 1,400
pH	6.5 ± 0.2	7.7 ± 0.03
Temperature (°C)	10.2 ± 1.8	10.7 ± 1.5
Alkalinity (mg/L as $CaCO_3$)	22 ± 1.8	72 ± 3.7
Dissolved Oxygen (mg O_2/L)	10.3 ± 0.5	2.7 ± 0.3
Turbidity (NTU)	0.09 ± 0.05	0.14 ± 0.09
Microbial Populations (CFU/100 ml)	70,200 ± 99,000	52,300 ± 13,400
Iron (mg/L)	BDL**	BDL
Manganese (mg/L)	BDL	BDL
Uranium (pCi/L)	24.6 ± 3.6	528.7 ± 9.0
Radium (pCi/L)	1.4 ± 0.1	> 19++

* Based on 27 sampling events.

\+ Based on 3 sampling events.

** BDL = Below detection limit (0.06 mg Fe/L; 0.02 mg Mn/L).

++ Initial sample dilution insufficient.

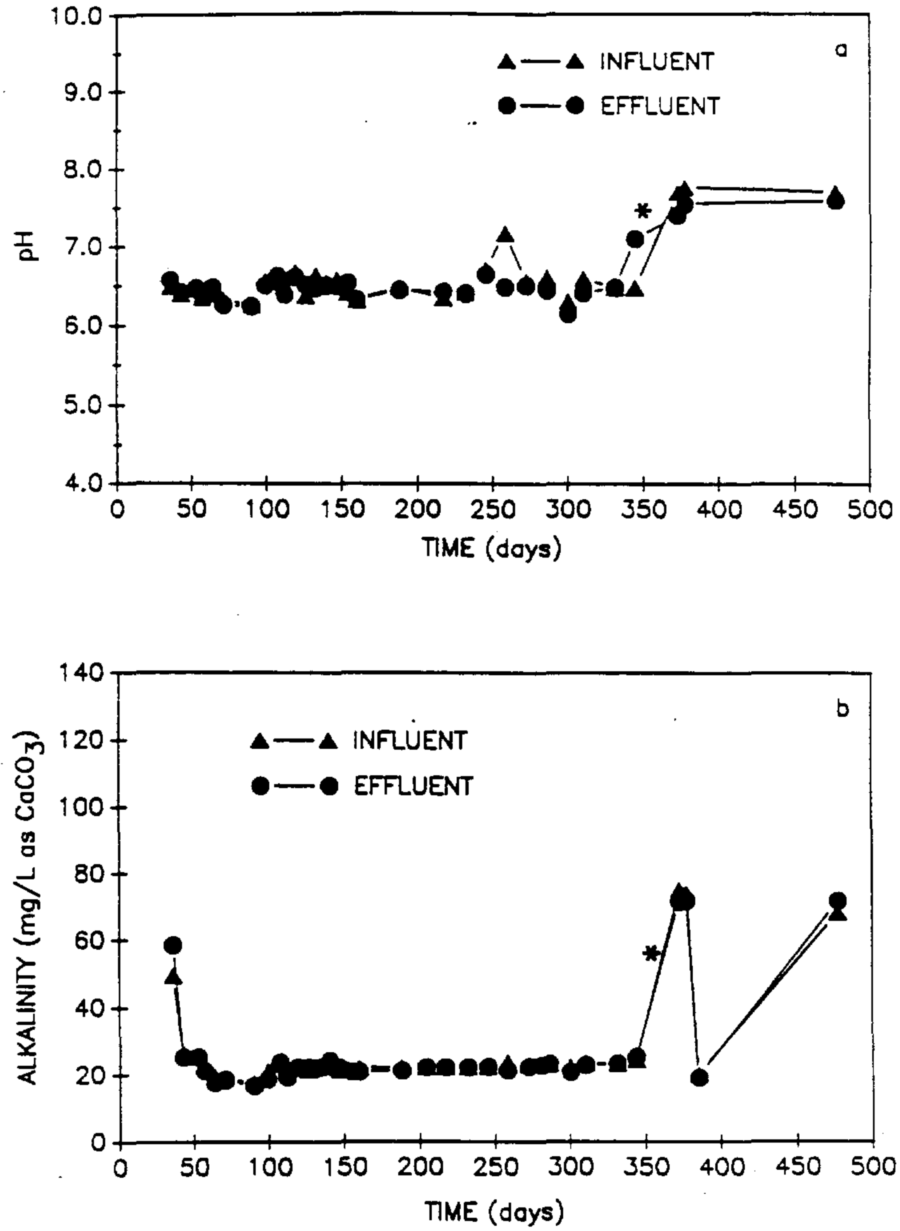


Figure 21. Phase II - Mont Vernon, NH GAC system. pH (a) and alkalinity (b) through 477 days of operation. (* new well)

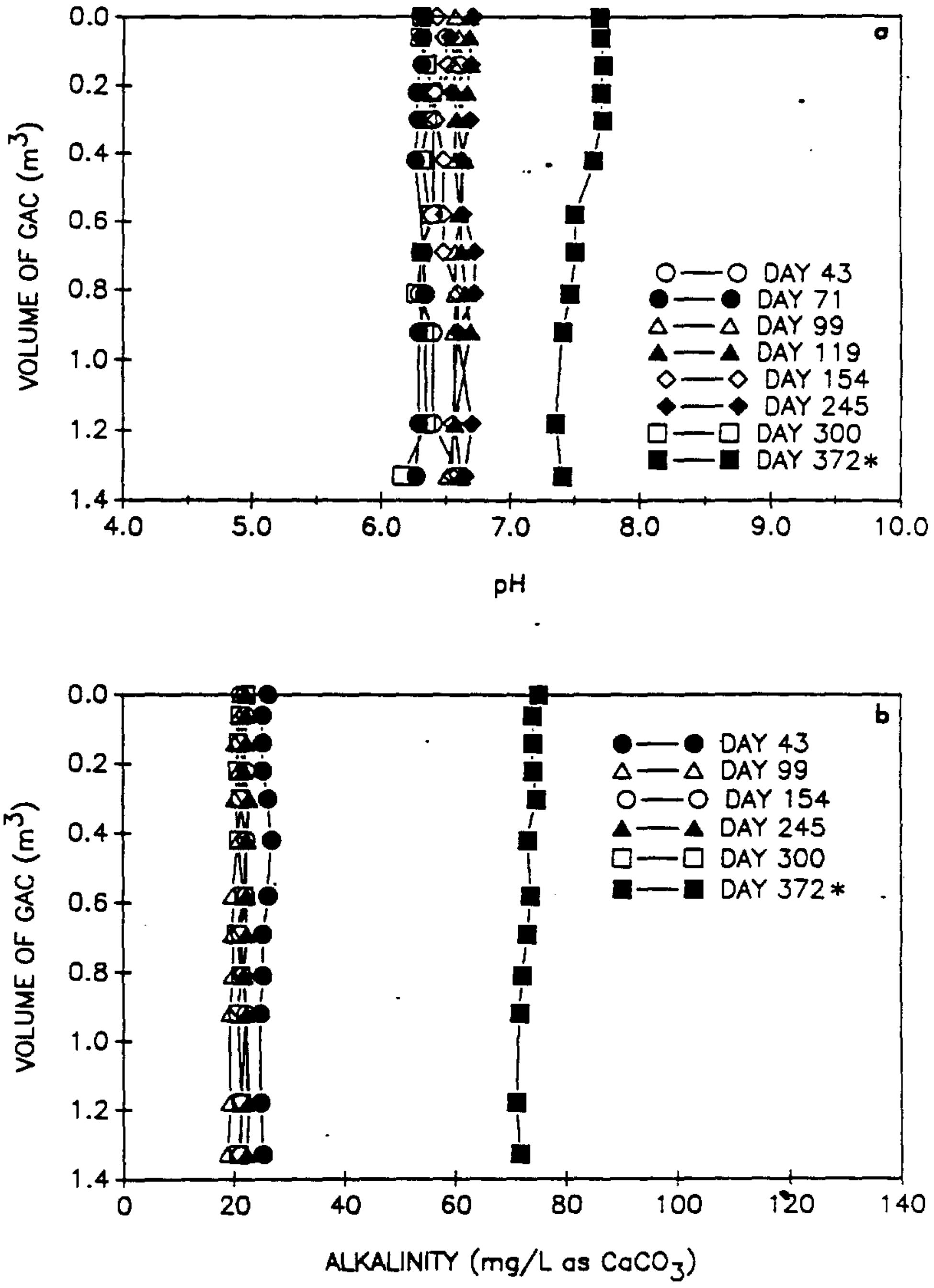


Figure 22. Phase II - Mont Vernon, NH GAC system. Representative profiles of pH (a) and alkalinity (b) through the GAC system. (*new well)

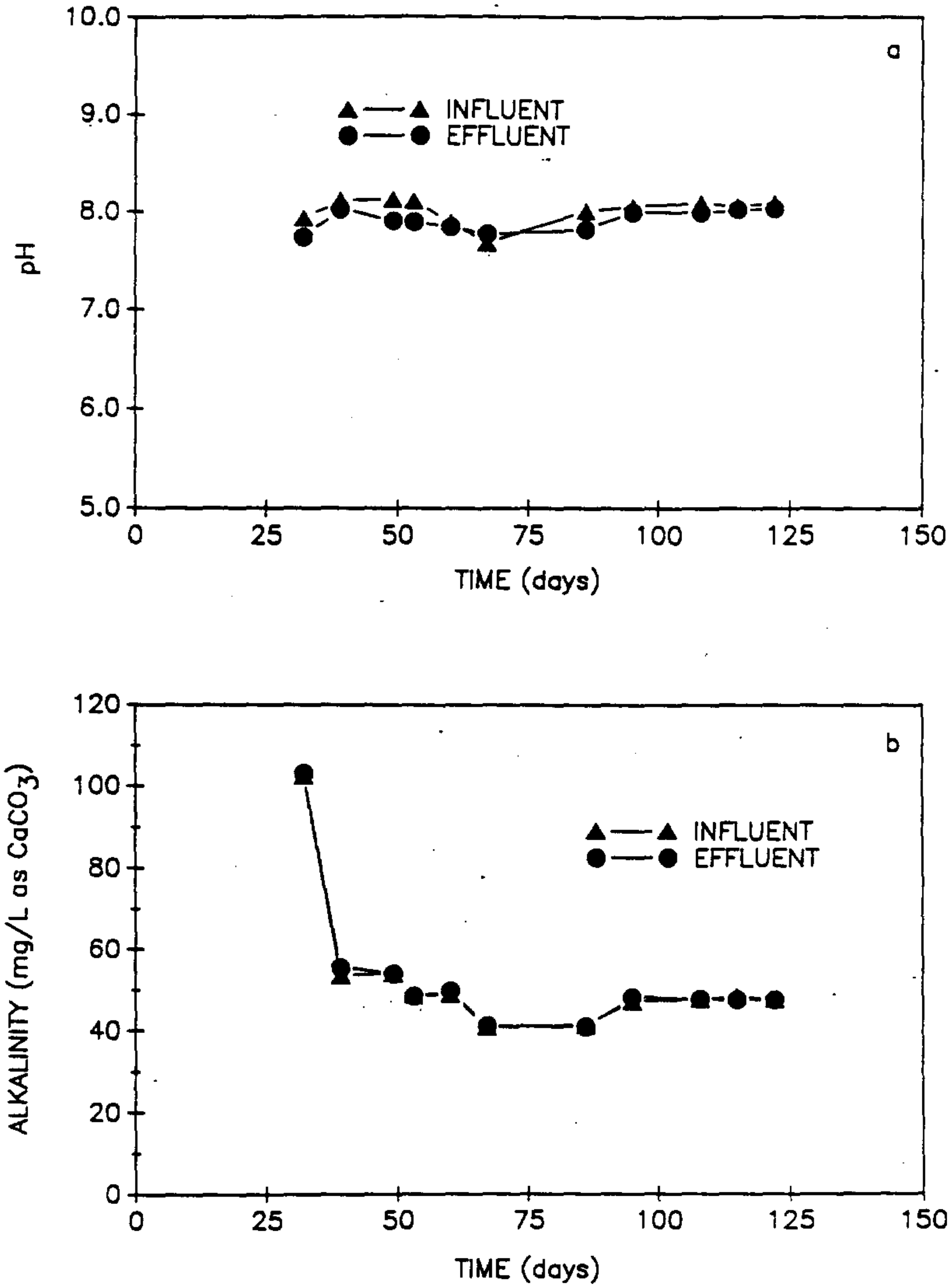


Figure 23. Phase II - Amherst, NH GAC system. pH (a) and alkalinity (b) through 122 days of operation.

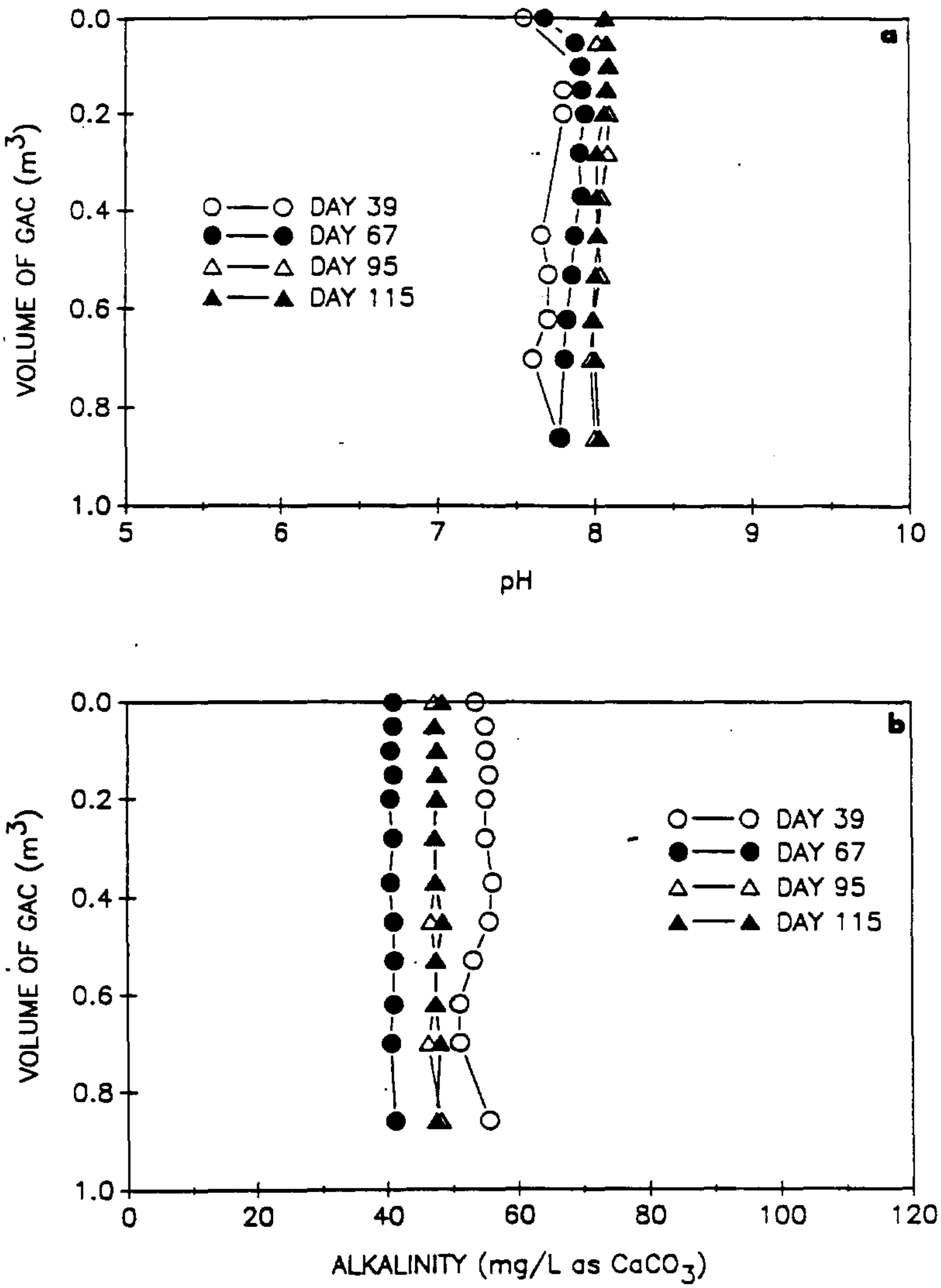


Figure 24. Phase II - Amherst, NH GAC system. Representative profiles of pH (a) and alkalinity (b) through the GAC system.

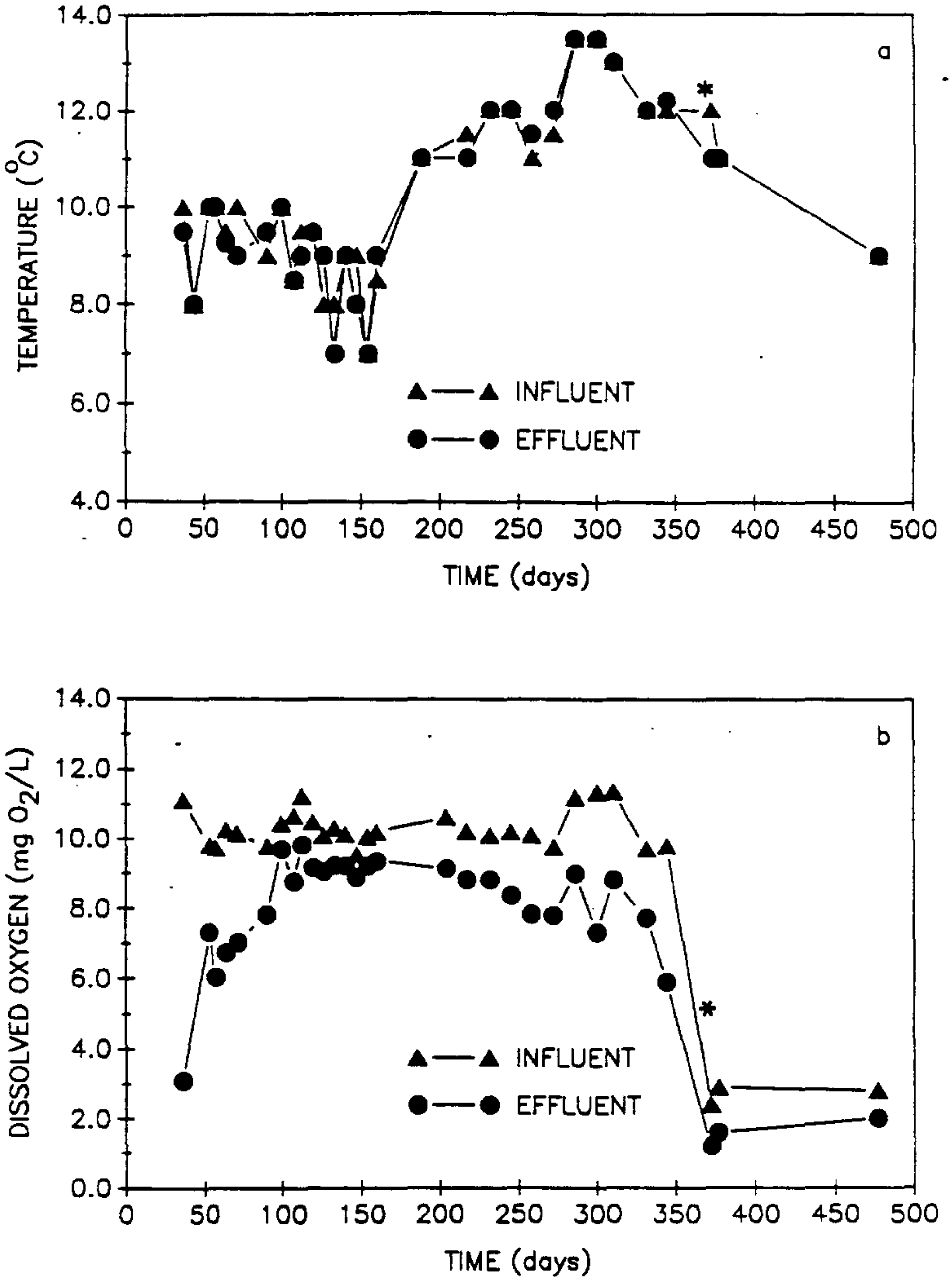


Figure 25. Phase II - Mont Vernon, NH GAC system. Temperature (a) and dissolved oxygen (b) through 477 days of operation. (* new well)

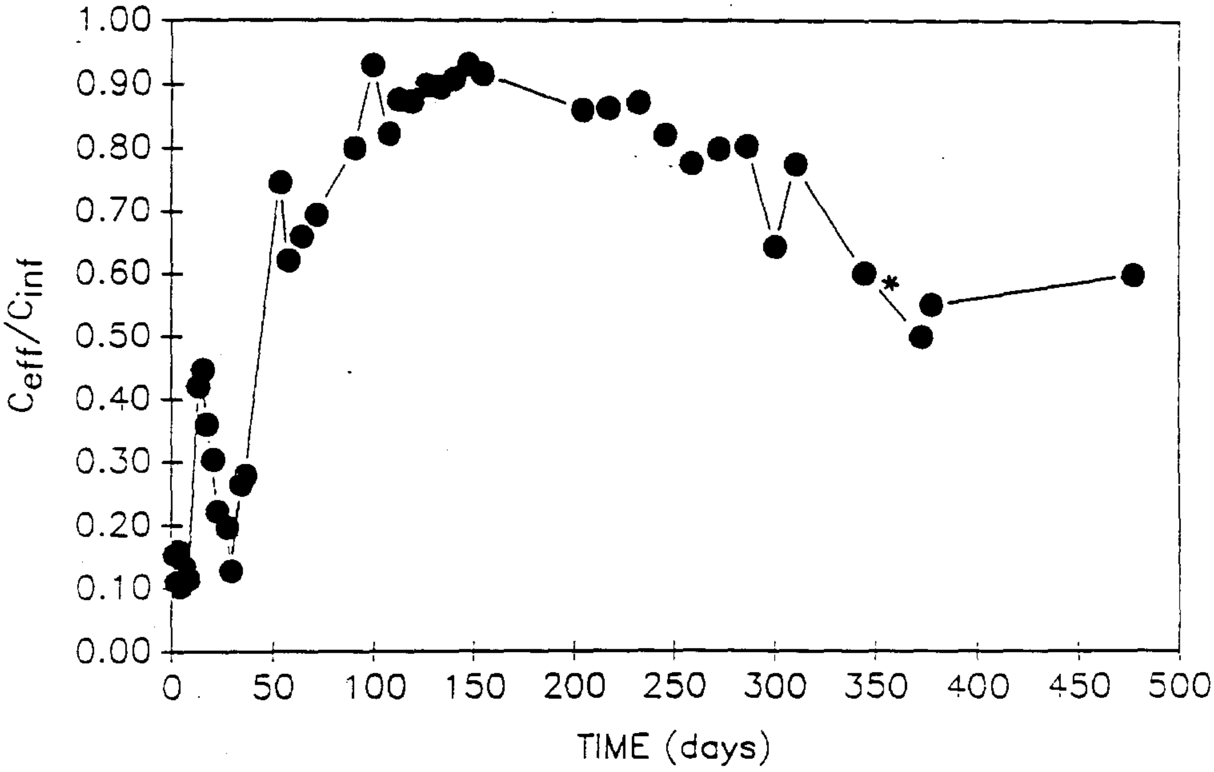


Figure 26. Phase II - Mont Vernon, NH GAC system. Dissolved oxygen breakthrough curve. (* new well)

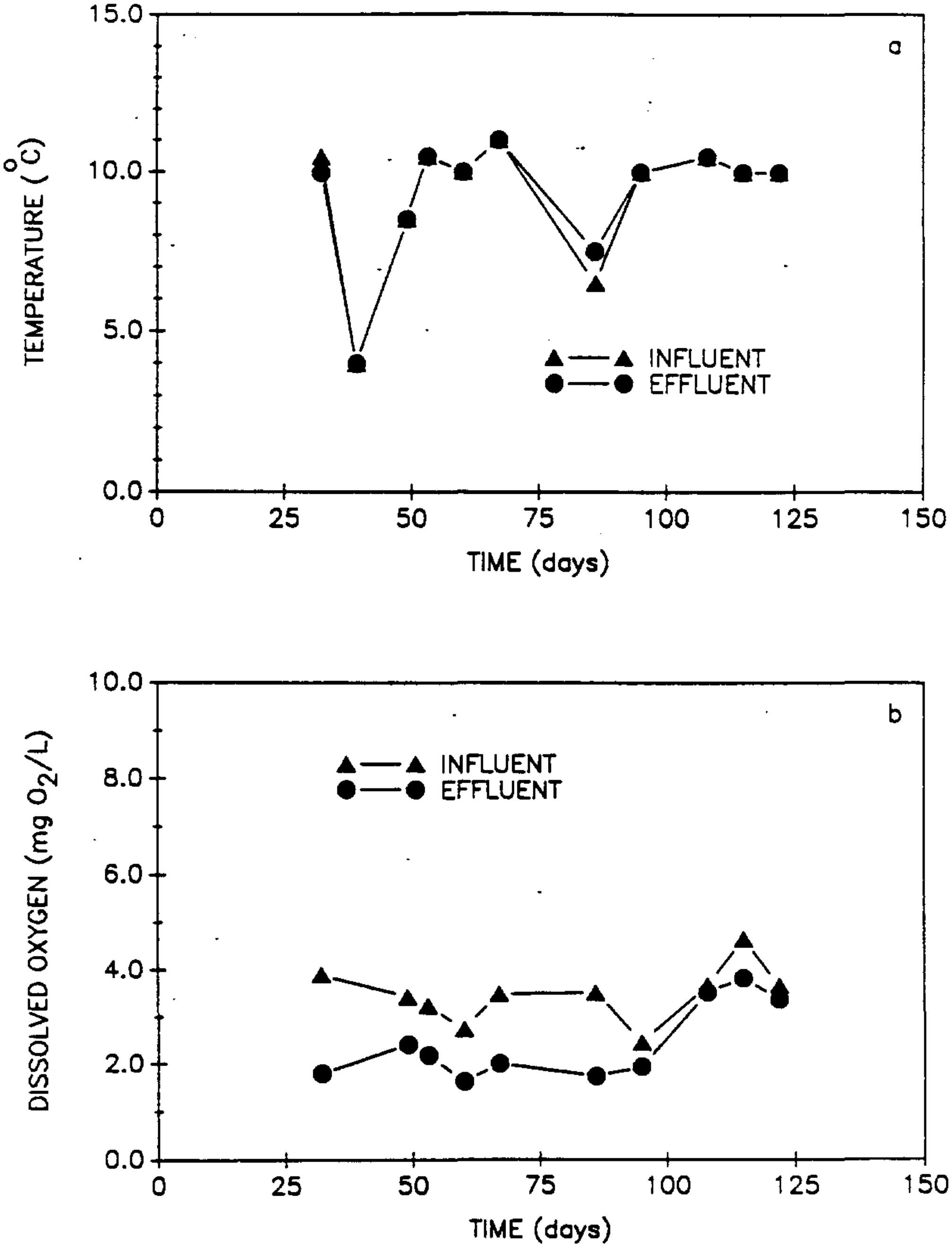


Figure 27. Phase II - Amherst, NH GAC system. Temperature (a) and dissolved oxygen (b) through 122 days of operation.

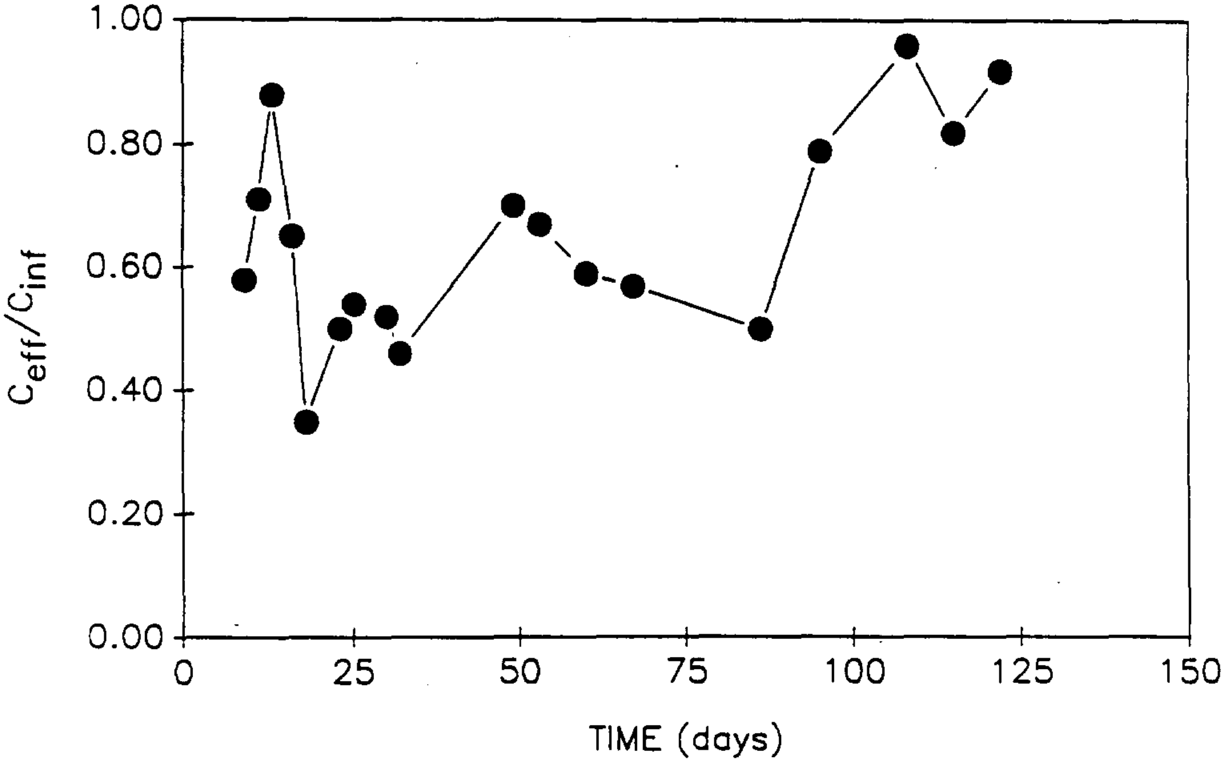


Figure 28. Phase II - Amherst, NH GAC system. Dissolved oxygen breakthrough curve.

- the Amherst unit contained 0.85 m^3 of GAC and was saturated with DO whereas the Mont Vernon unit contained 1.33 m^3 of GAC and continued to adsorb DO.

Turbidity--

Mont Vernon site--As observed during Phase I, turbidity was removed by the GAC units (Figure 29). However, due to the variability in the data and their closeness to the detection limit (0.05 NTU) differences were not significant ($\alpha = 0.05$ and 0.10, StT) between influent and effluent turbidity. Turbidity data taken from the sampling ports indicated there was no distinct trend of removal through the units. Accumulation of small amounts of turbidity may contribute to the potential for clogging the filter, increasing headloss, over long periods of operation. This is particularly important because GAC systems removing radon are not routinely backwashed.

Amherst site--The average influent turbidity (2.99 ± 2.28 NTU) was significantly higher ($\alpha = 0.05$ and 0.01, StT) than the average effluent (1.07 ± 0.50 NTU) (Figure 30). Profiles of turbidity through the system are shown in Figure 31. Turbidity removal was occurring primarily within the top of the GAC unit. The GAC units were not backwashed and thus the accumulation could have been substantial considering a daily flow of 61 m^3/day and an average removal of 1.92 NTU.

Flowrate--

Flowrate data for Phase II were calculated by dividing the total volume of water treated between sampling events by the number of days during that period. As a result, the data only give a gross estimate of the daily flowrate and must be used cautiously, especially during the latter portion of the study when sampling was less frequent.

Mont Vernon site--The average flowrate for the continuous monitoring period was 36 ± 12 m^3/day (Figure 32) which corresponds to an average EBCT of 53.3 min. A dramatic increase in water demand was observed during the period June through August (Day 245 to Day 310), reaching as high as 76 m^3/day (EBCT = 25.3 min). The GAC system was designed to handle an average flowrate of 25 m^3/day (EBCT = 77.9 min). The variability in flow suggests that selecting design flows for small communities using a limited database may be unrepresentative of the true flow variations which may occur. Large variations in flowrate may significantly impact effluent quality because contact time is so critical to GAC adsorption of radon.

Amherst site--The average water flowrate during Phase II was 61 ± 5 m^3/day yielding an average EBCT of 20.1 min and was relatively stable (Figure 33). This flowrate was similar to that observed during Phase I and was much greater than the 35 m^3/day (EBCT = 35.5 min) the system was designed to treat. This could have affected the effluent radon activity because of the shortened contact time within the GAC system. The flow at Amherst was not as variable as Mont Vernon's, but the system was not operated during the summer months when flowrates at Mont Vernon were higher.

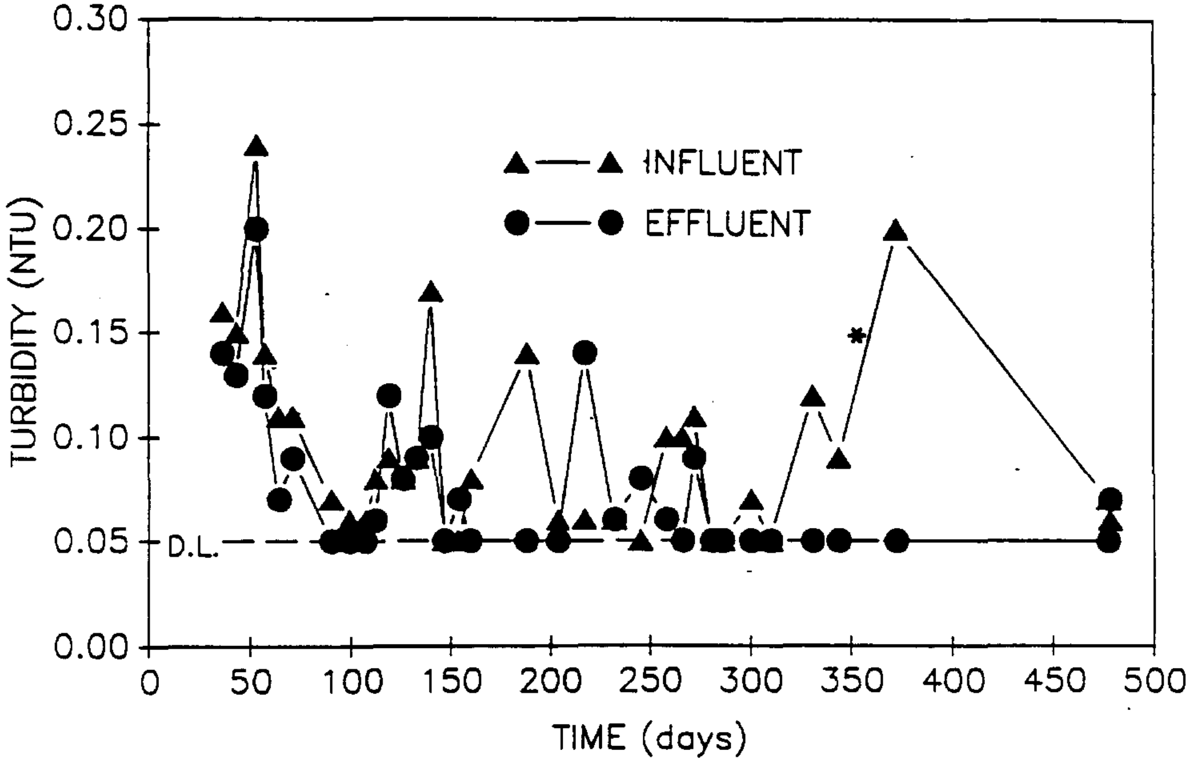


Figure 29. Phase II - Mont Vernon, NH GAC system. Turbidity through 477 days of operation (detection limit = 0.05 NTU). (* new well)

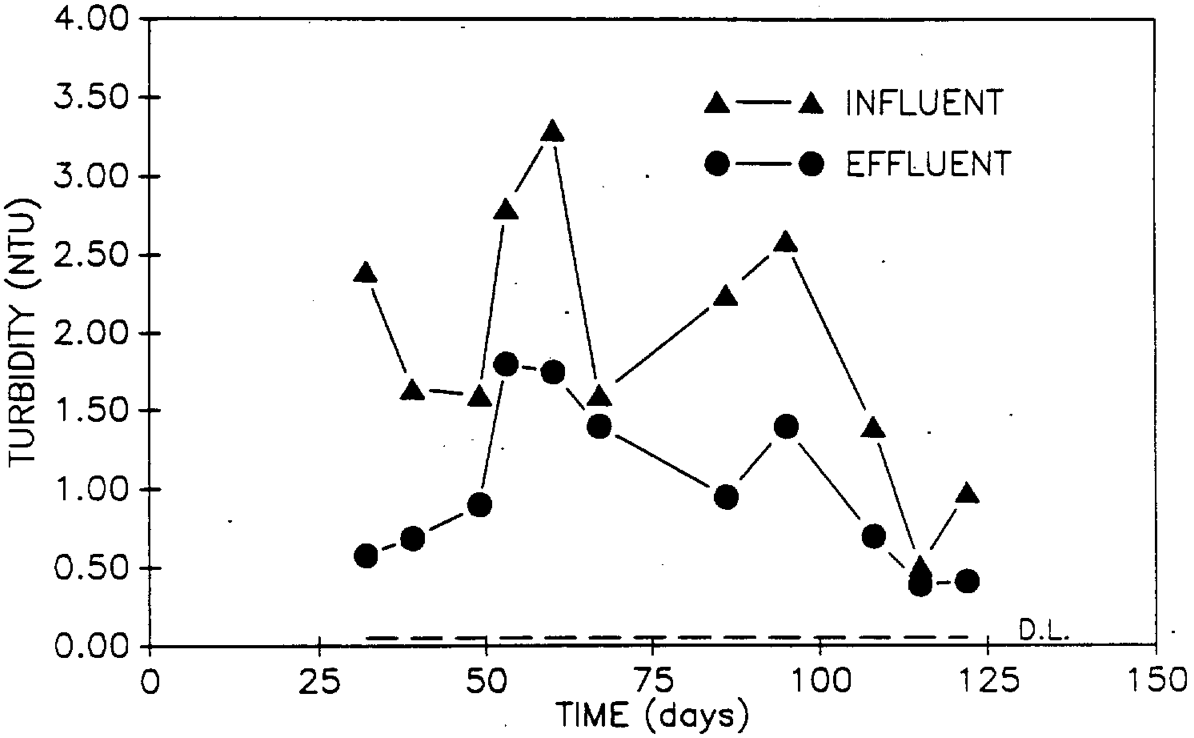


Figure 30. Phase II - Amherst, NH GAC system. Turbidity through 122 days of operation (detection limit = 0.05 NTU).

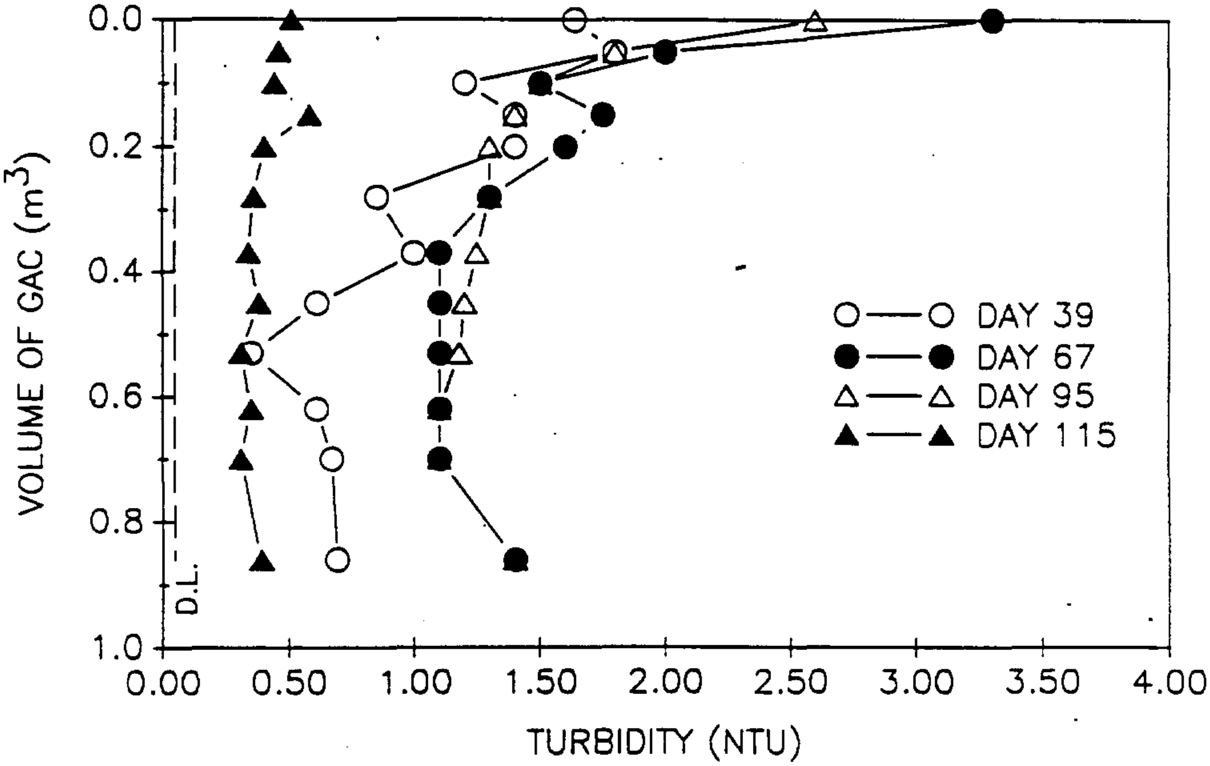


Figure 31. Phase II - Amherst, NH GAC system. Representative profiles of turbidity through the GAC system (detection limit = 0.05 NTU).

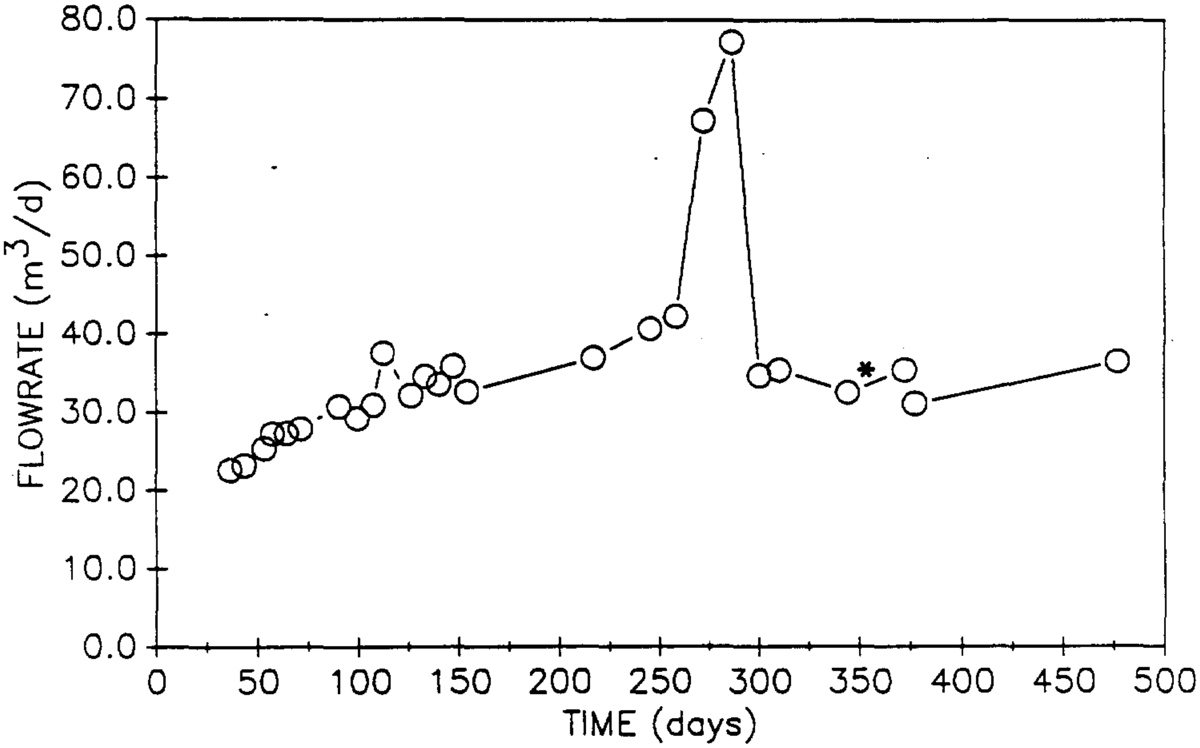


Figure 32. Phase II - Mont Vernon, NH GAC system. Water flowrate through 477 days of operation. (* new well)

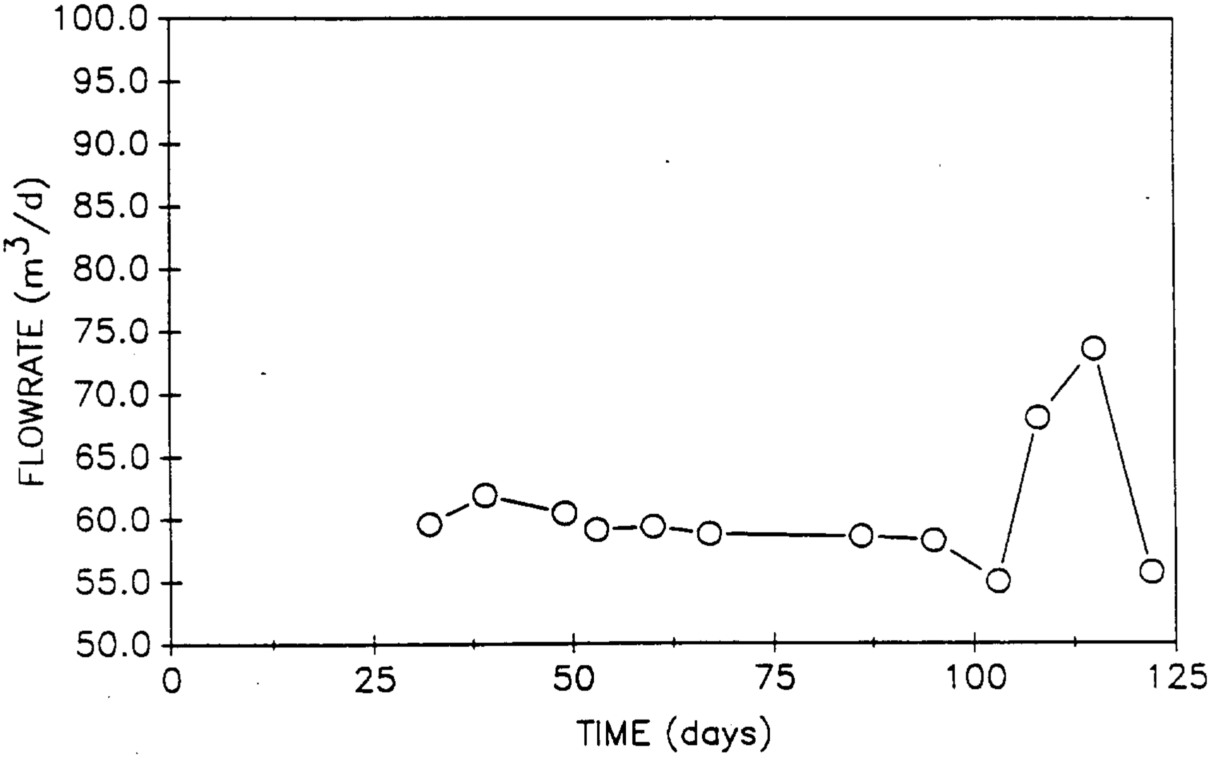


Figure 33. Phase II - Amherst, NH GAC system. Water flowrate through 122 days of operation.

Radon--

Mont Vernon site--With the exception of the 3.5 months at the end of the study when the new well was operating and the average influent activity was 68,900 ± 1,400 pCi/L, the influent radon activity remained higher than the design activity of 155,000 pCi/L. During the majority of Phase II, the average influent activity was 210,491 ± 41,384 pCi/L (Figures 34 and 35). The effluent quality varied between 4,750 to 68,400 pCi/L during Phase II.

Radon is unique among most of the contaminants treated using GAC because after a period of initial breakthrough, there is usually no further increase in the activity observed in the effluent (Lowry and Brandow, 1985). The GAC bed theoretically never becomes exhausted, hence, there is no need for regeneration or replacement. It is believed that this removal pattern occurs because radon decays within the bed.

Profiles of the radon activity through the GAC system are presented in Figure 36. Most of the plots exhibit exponential removal of radon as a function of bed volume which is consistent with the radon adsorption model developed by Lowry (1985). The exception to the pattern occurred late in the study when the new well was operating. Despite the significantly lower influent radon activity from the new well, the data show a very steep (almost straight) plot and an overall decrease in removal efficiency. The Day 477 data is an example of this trend. Although this decrease in removal efficiency corresponds to changes in other water quality parameters, the effects of raw water quality on radon adsorption by GAC are poorly understood.

The K_{ss} of the GAC at Mont Vernon was calculated for periods when flowrate and influent activity were relatively constant. These values averaged 2.37 ± 0.72 hr^{-1} as compared to the K_{ss} of 3.02 hr^{-1} at 6-10°C obtained in other tests with this type of GAC (Lowry and Lowry, 1987). Differences in the quality of the water treated by these systems and the column hydraulics may have accounted for these differences in K_{ss}.

Amherst site--The average influent radon activity during Phase II was 41,800 ± 15,200 pCi/L (Figure 37), which was not significantly different (α = 0.05 and 0.01, StT) from the design influent of 39,750 pCi/L. The influent radon activity was lower during this period than during Phase I and also showed a decreasing trend with time. The average effluent radon activity observed was 12,000 ± 5,900 pCi/L. Figure 38 presents profiles of radon activity through the Amherst GAC system. The activities observed in the section of the bed between 0.53 and 0.70 m^3 were sometimes lower than the effluent activity probably because air was entrained in the port tubing during sampling.

Again, the predicted (Lowry, 1985) exponential radon removal pattern was observed. K_{ss} values through the GAC unit average 5.7 ± 1.04 hr^{-1} during Phase II, which were considerably higher than the design K_{ss} of 3.02 hr^{-1} at 6-10°C previously obtained in other studies with this GAC (Lowry and Lowry,

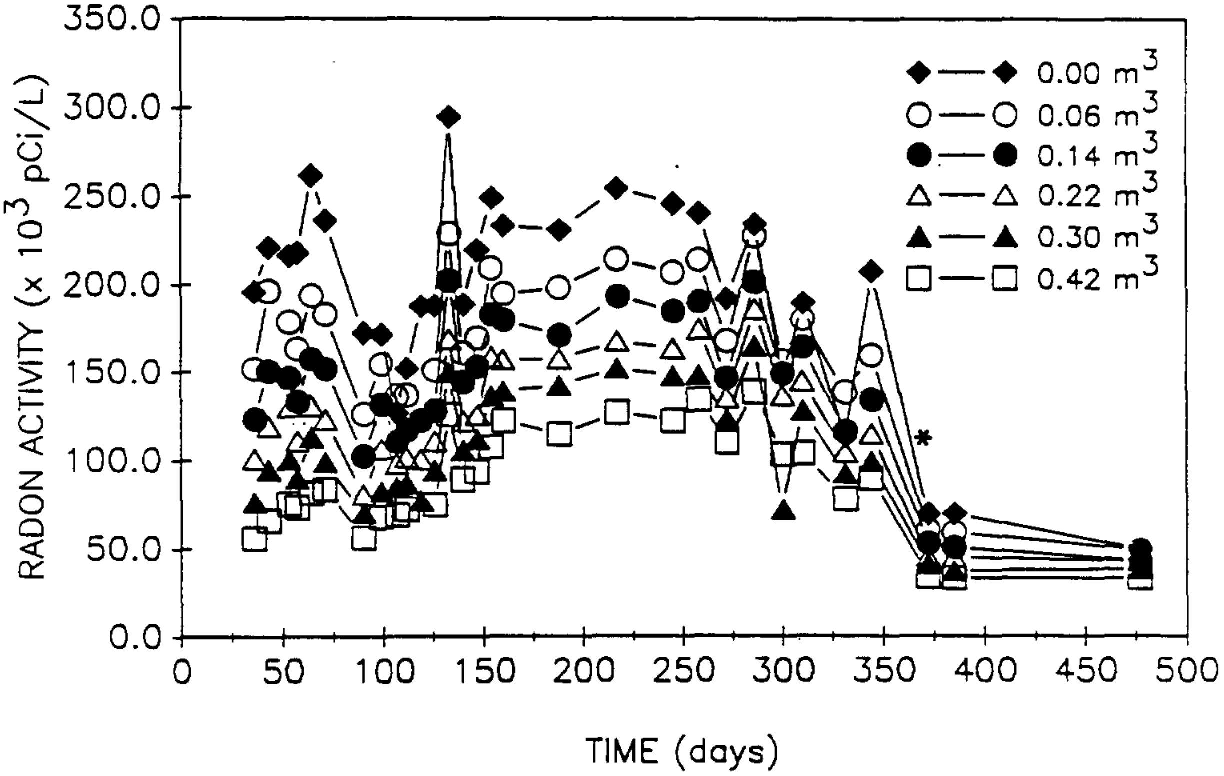


Figure 34. Phase II - Mont Vernon, NH GAC system. Radon activity through GAC #1 through 477 days of operation. (* new well)

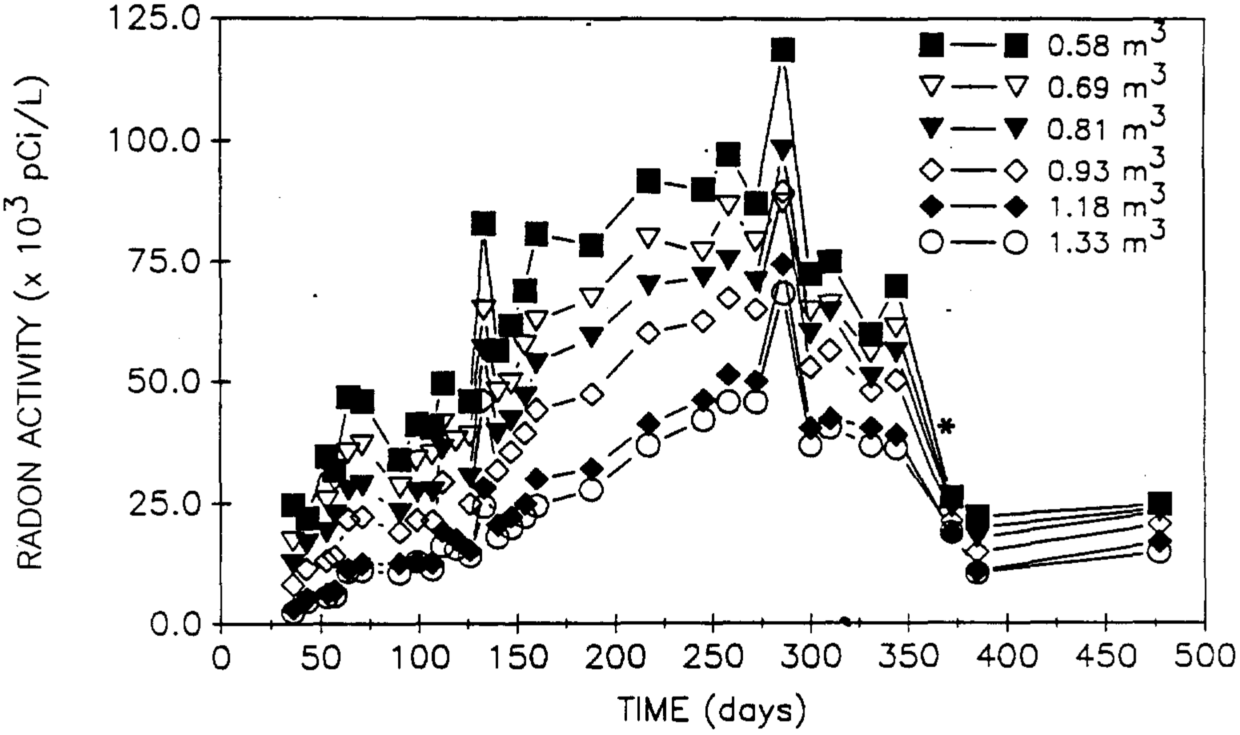


Figure 35. Phase II - Mont Vernon, NH GAC system. Radon activity through GAC #2 through 477 days operation. Note scale difference between Figure 34 and 35. (* new well)

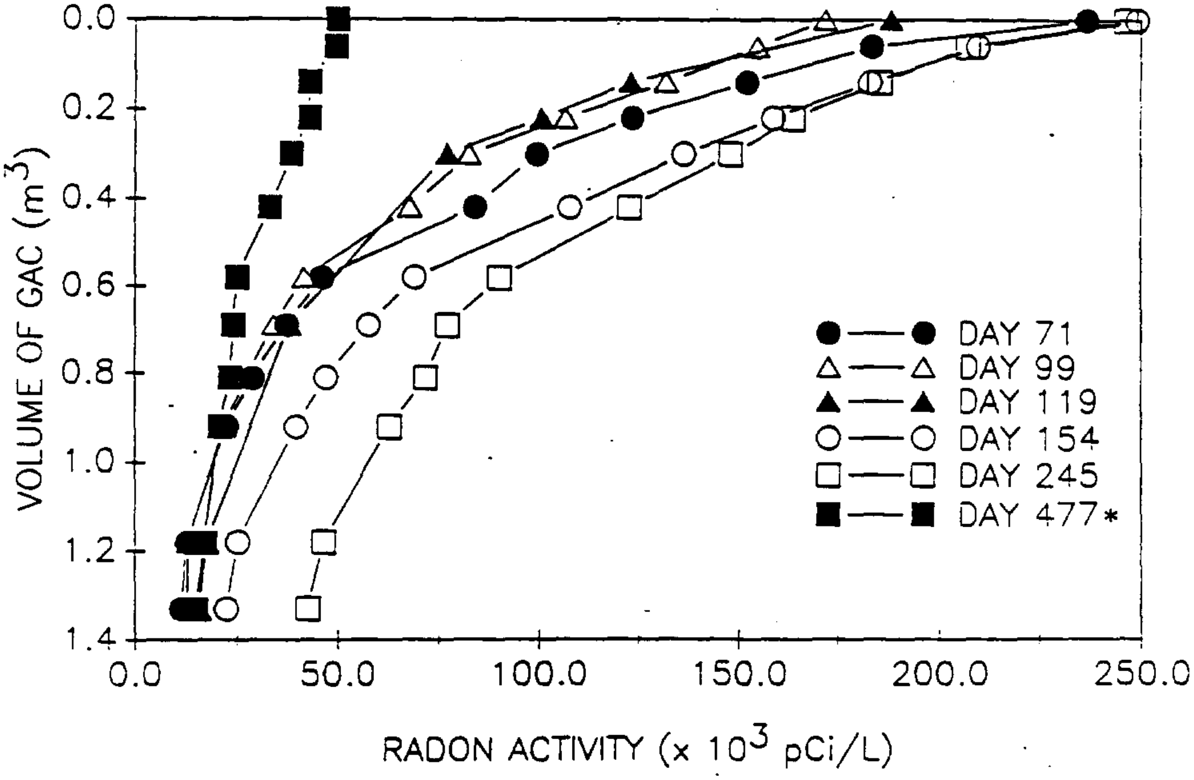


Figure 36. Phase II - Mont Vernon, NH GAC system. Representative profiles of radon activity through the GAC system. (* new well)

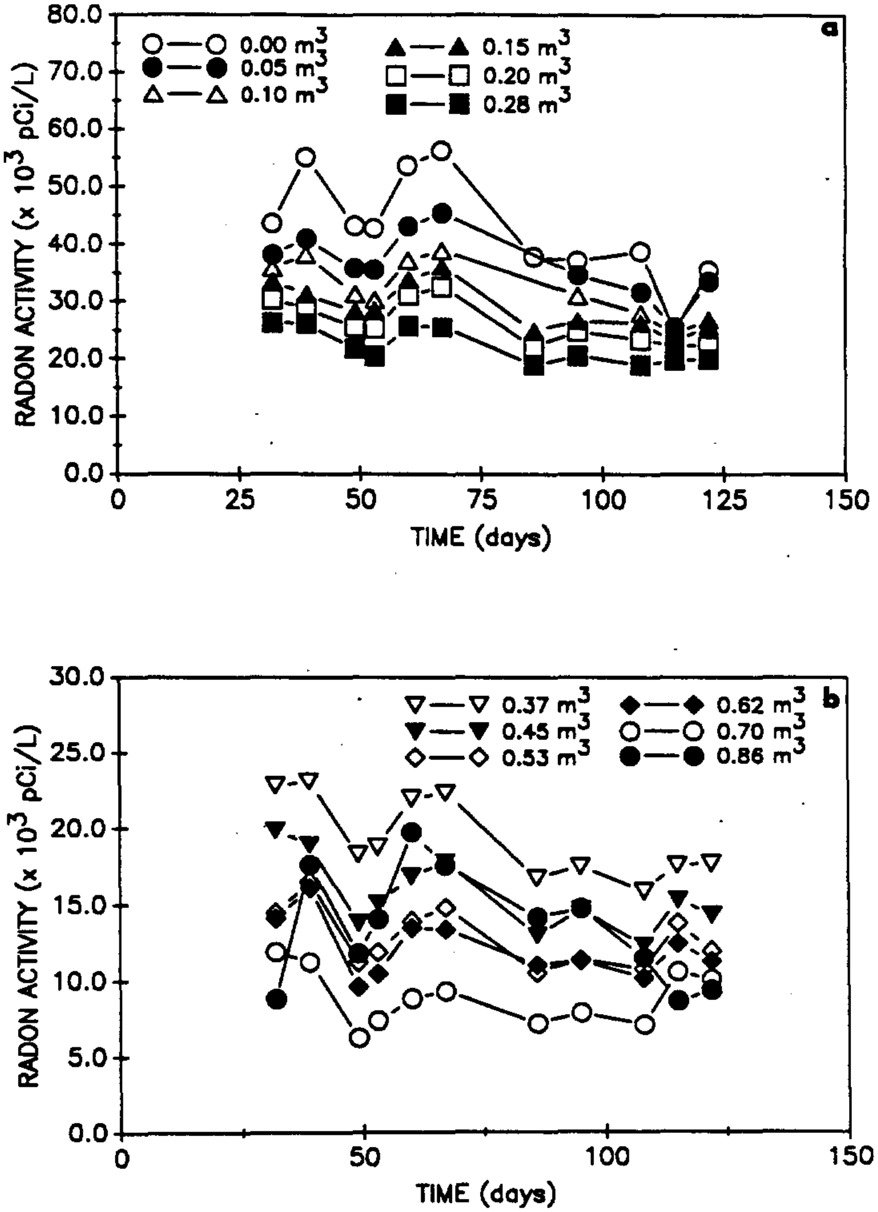


Figure 37. Phase II - Amherst, NH GAC system. Radon activity through the top 0.28 m³ of GAC (a) and the remaining 0.58 m³ of GAC (b) during 122 days of operation. Note difference in scales on ordinates in (a) and (b).

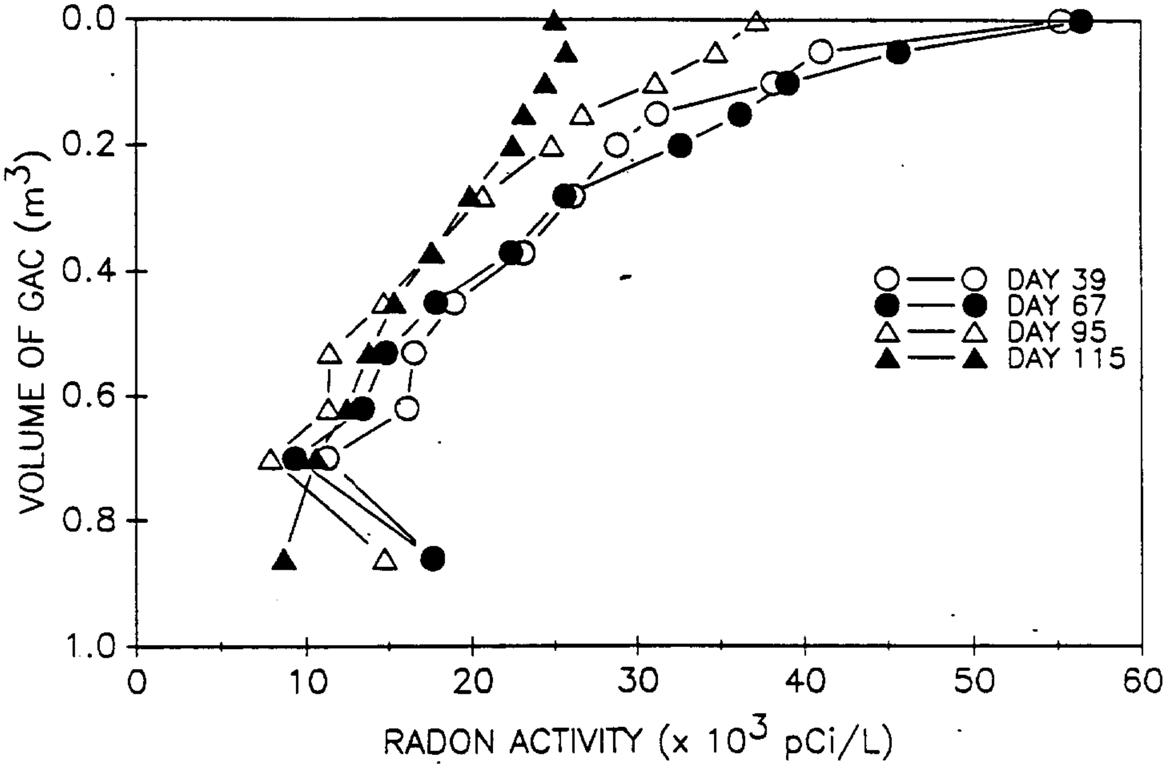


Figure 38. Phase II - Amherst, NH GAC system. Representative profiles of radon activity through the GAC system.

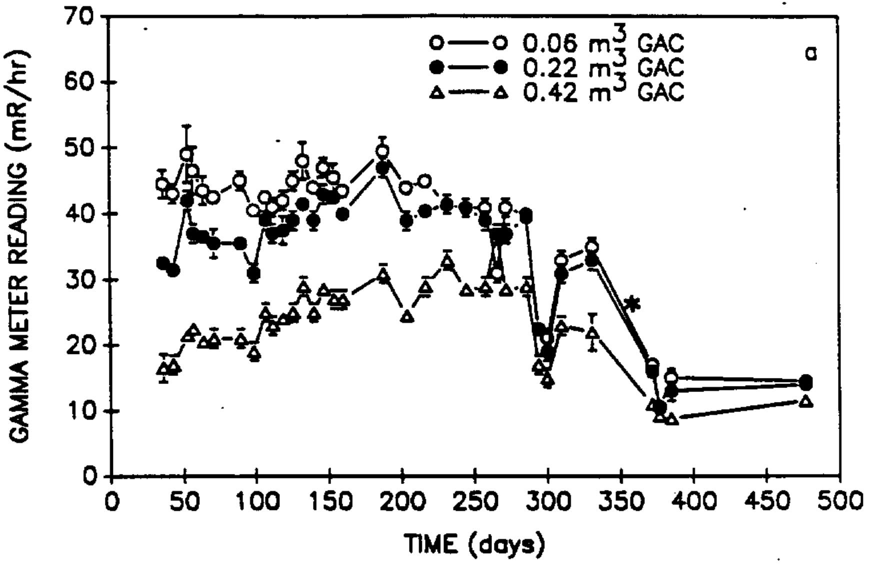
1987). The one exception was the K_{ss} data obtained during stable flow and radon activities in the GAC from 0.53 to 0.70 m^3. The K_{ss} in that region averaged 0.55 to 0.32 hr^{-1}, but may reflect problems with air entrainment in the sampling tubing.

One of the problems with designing GAC systems for radon removal is the need for data on influent radon activity and flowrate. These two parameters can vary widely in small community water supplies as demonstrated in this study. Therefore, design could be difficult unless a community had routinely monitored water quality and flowrate. Though the data shows the GAC systems can accommodate some variation in influent loading, changes in flow and/or radon activity typical in small community use of groundwater supplies could result in spikes of radon in the effluent exceeding the MCL. Further, since GAC system design is based on the K_{ss} (Lowry et al., 1987; Lowry and Lowry, 1987) and in our evaluations at Mont Vernon and Amherst this value was found to vary depending on site conditions (i.e., flowrate, radon concentration and raw water quality), it is recommended that pilot scale testing be performed to verify the GAC's capacity for radon at a given site prior to finalizing the system's design.

Gamma/Beta Emissions--

Mont Vernon site--Gamma/beta emissions in the top and middle of GAC #1 began to converge and were not significantly different (α = 0.05 and 0.01, StT) after 225 days of operation (Figure 39). Gamma readings for GAC #2 resembled the fluctuations observed for the radon activities in that unit. As observed during Phase I, there was a curvilinear relationships between influent radon activity and gamma/beta emissions readings (Figure 40). After 6 months of operation and before the new well was used, the relationships between gamma/beta emissions and the influent radon activity at the top of GAC #1 and #2 were 1 mR/hr = 5,570 pCi/L and 1 mR/hr = 6,549 pCi/L, respectively. These relationships are different than those observed by Lowry et al. (1988) (1 mR/hr ≡ 10,360 pCi/L) for point-of-entry units. The fluctuations in radon load (radon activity and flow) and removal could have had a tremendous effect on the gamma emissions through the Mont Vernon GAC units and the large GAC volume could account for some of the differences observed between the small community systems and the point-of-entry units. Perhaps, a mass balance approach would provide a better mechanism to compare emissions from different systems since it accounts for differences in the mass of GAC and radon mass loading. However, there is still a problem with geometries of the source and the differences in monitoring instrumentation which may make comparisons between GAC systems difficult. This may be less of a problem when comparing units with relatively similar flows, GAC masses, and contactor geometries.

Keene and Rydell (1989) cite a maximum 8 hr/day exposure applicable to residences of 0.058 mR/hr based on National Council on Radiation Protection guidelines. The data obtained at Mont Vernon indicates that gamma exposure in the 10^1 mR/hr range could occur at the units' surfaces. The background gamma/beta emissions measured in an area approximately 31 m away from the



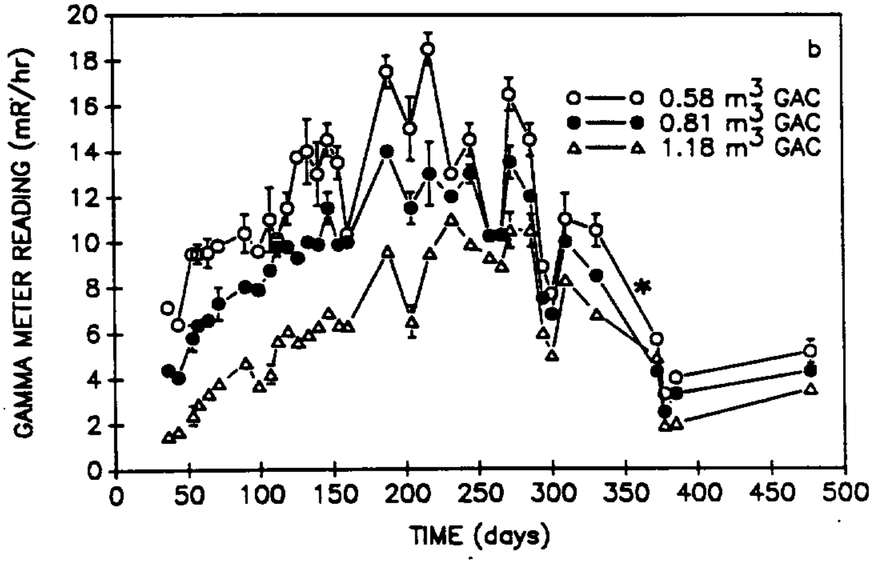


Figure 39. Phase II - Mont Vernon, NH GAC system. Gamma/beta emission measurements taken on the contactor surfaces through 477 days of operation for GAC #1 (a) and GAC #2 (b). Note difference in scales on ordinate in (a) and (b). (* new well)

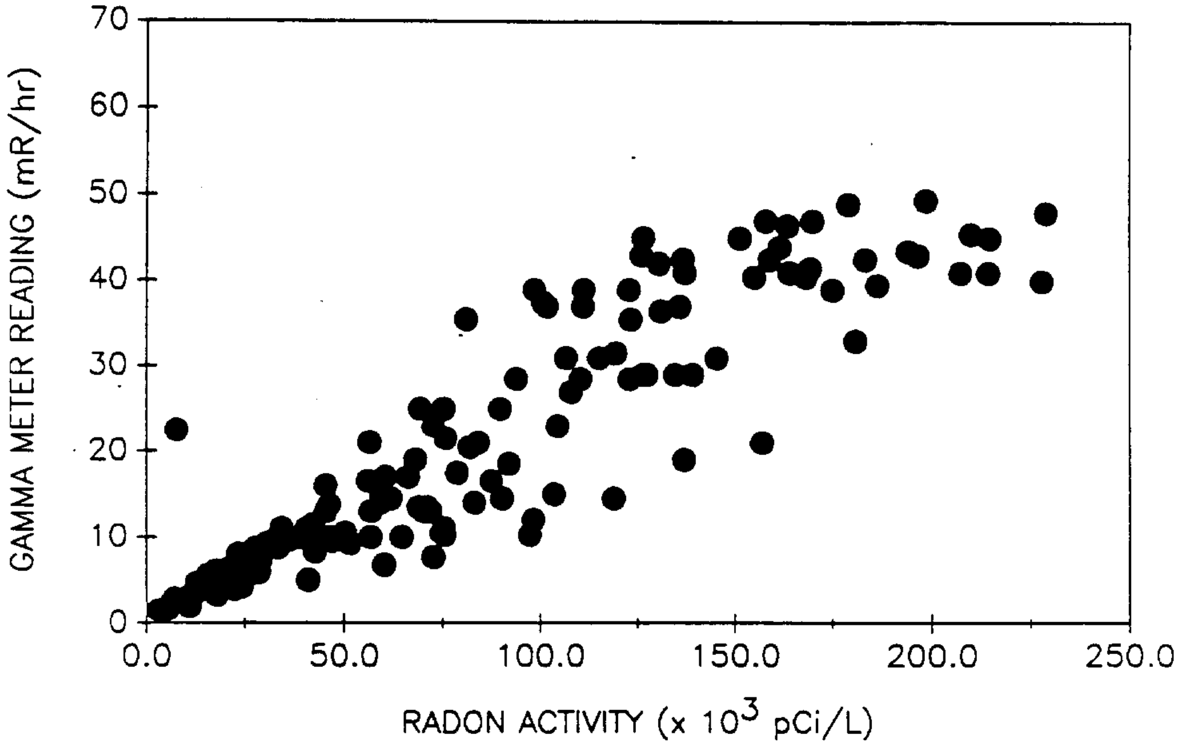


Figure 40. Phase II - Mont Vernon, NH GAC system. Gamma/beta emission measurements taken on the contactor surfaces as a function of effluent radon activity to sections of the GAC system.

pumphouse averaged 0.056 ± 0.028 mR/hr. The gamma/beta emissions measured in the vicinity of the GAC system decreased with distance from the units. 1.52 m away from GAC #1 the gamma/beta readings average 3.56 ± 0.97 mR/hr (same elevation as the middle of the tank), while 0.91 m from GAC #2 the readings averaged 1.2 ± 0.65 mR/hr (same elevation as middle of the tank).

To corroborate that gamma exposures were significant at the Mont Vernon site, dosimeter badges were placed in the pumphouse and on the surface of the GAC for 4 days. This was done when the influent flowrate and radon activity to the units were approximately 50 m^3/day and 53,304 pCi/L, respectively. The badge on the middle of GAC #1 measured 970 mREM, while a badge at the same elevation 0.61 m away from the unit registered 290 mREM. The badge on the middle of GAC #2 measured 400 mREM, while a badge 0.61 m away measured 160 mREM.

Exposures are highly dependent on shielding and the distance away from the source. Gamma exposures from small community GAC systems should be predicted based on radon removal (flowrate, influent radon activity, effluent quality, GAC volume and tank configuration) and should be minimized to meet accepted occupational safety and health standards.

Amherst site--The gamma/beta emissions during Phase II decreased gradually over the period as observed for the radon activities (Figure 41). Figure 42 shows the relationship between gamma/beta emissions and the radon activity at the nearest port. The non-linear relationship is similar to the one observed during Phase I for this system. Again, these differences may be a function of the mass of GAC, the radon loading/removal, the tank geometry and/or the instrumentation used. The background gamma/beta emissions at Amherst, approximately 9 m from the pumphouse, were 0.027 ± 0.031 mR/hr. Emissions in the doorway, approximately 1.2 m above the filter, averaged 0.33 ± 0.15 mR/hr.

Microbial Numbers--

Mont Vernon site--It has been well documented that GAC filters are capable of supporting microbial populations (Wilcox et al., 1983; Camper et al., 1985, 1986, 1987). Effluent standard plate count (SPC) microbial enumerations are often greater than influent SPC for GAC systems used in water treatment (Graese et al., 1987). The GAC provides a good surface for attachment and concentrates nutrients to support the microorganisms. The microbial numbers in the GAC effluent fluctuated greatly after 125 days of operation, but were significantly higher ($\alpha = 0.05$ and 0.01, StT) than the influent concentration (Figure 43).

Amherst site--The microbial numbers in the effluent from the GAC at Amherst were highly variable (Figure 44), as observed for the Mont Vernon site. There was some input of microorganisms to the unit from Day 53 to 67. It appears that the GAC at Amherst also supported a microbial population.

The microbial data collected for the GAC units at Mont Vernon and Amherst show significantly different conditions existed at each site. The Mont Vernon data indicates bacterial populations in the GAC effluent were significantly

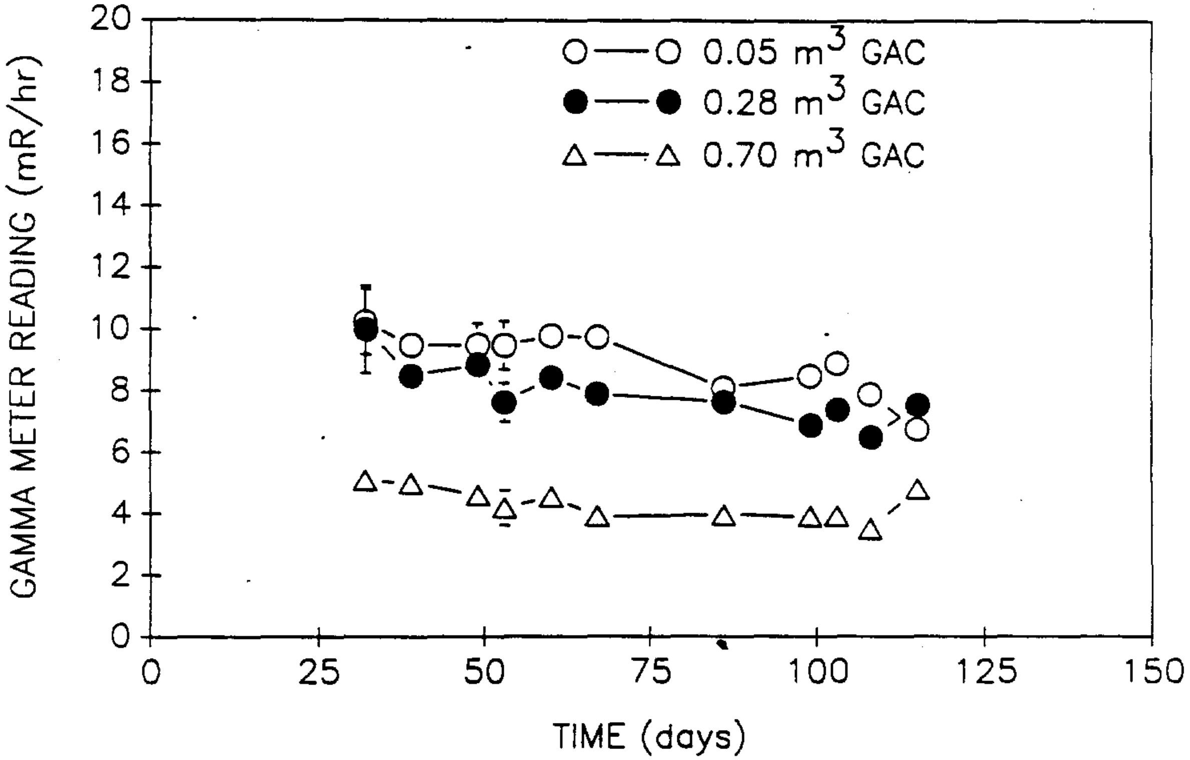


Figure 41. Phase II - Amherst, NH GAC system. Gamma/beta emission measurements taken on the contactor surface through 122 days of operation.

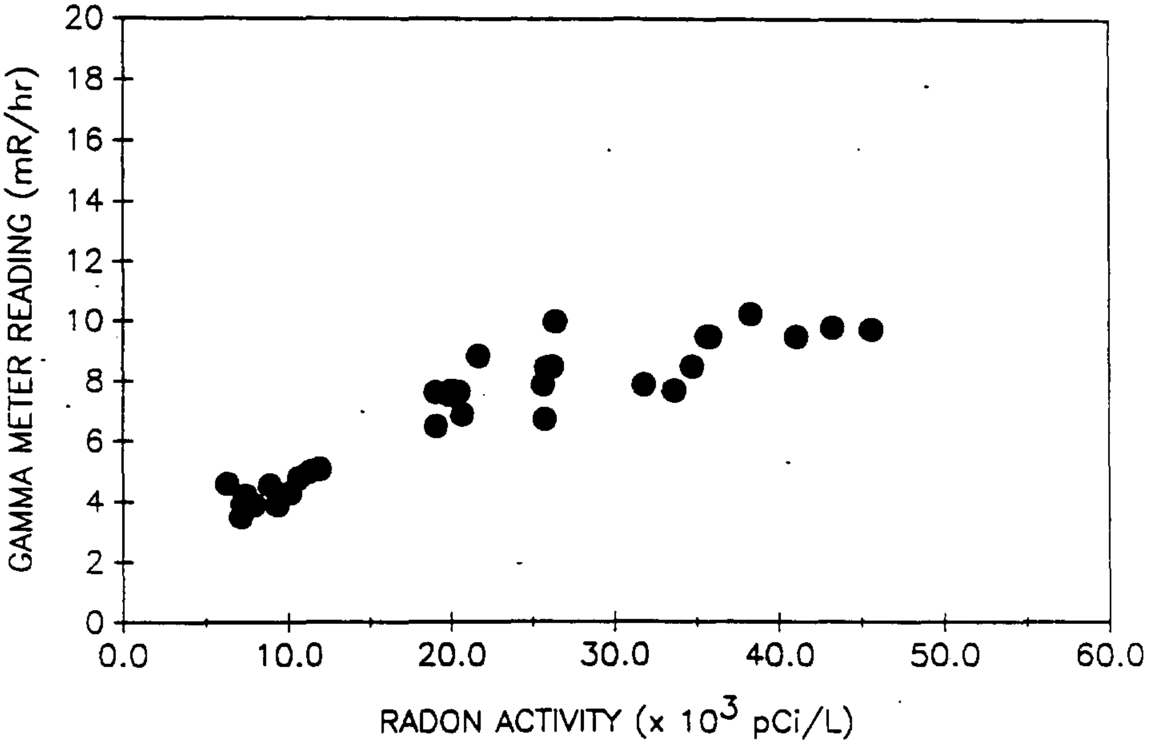


Figure 42. Phase II - Amherst, NH GAC system. Gamma/beta emission measurements as taken on contactor surface as a function of influent radon activity to sections of the GAC system.

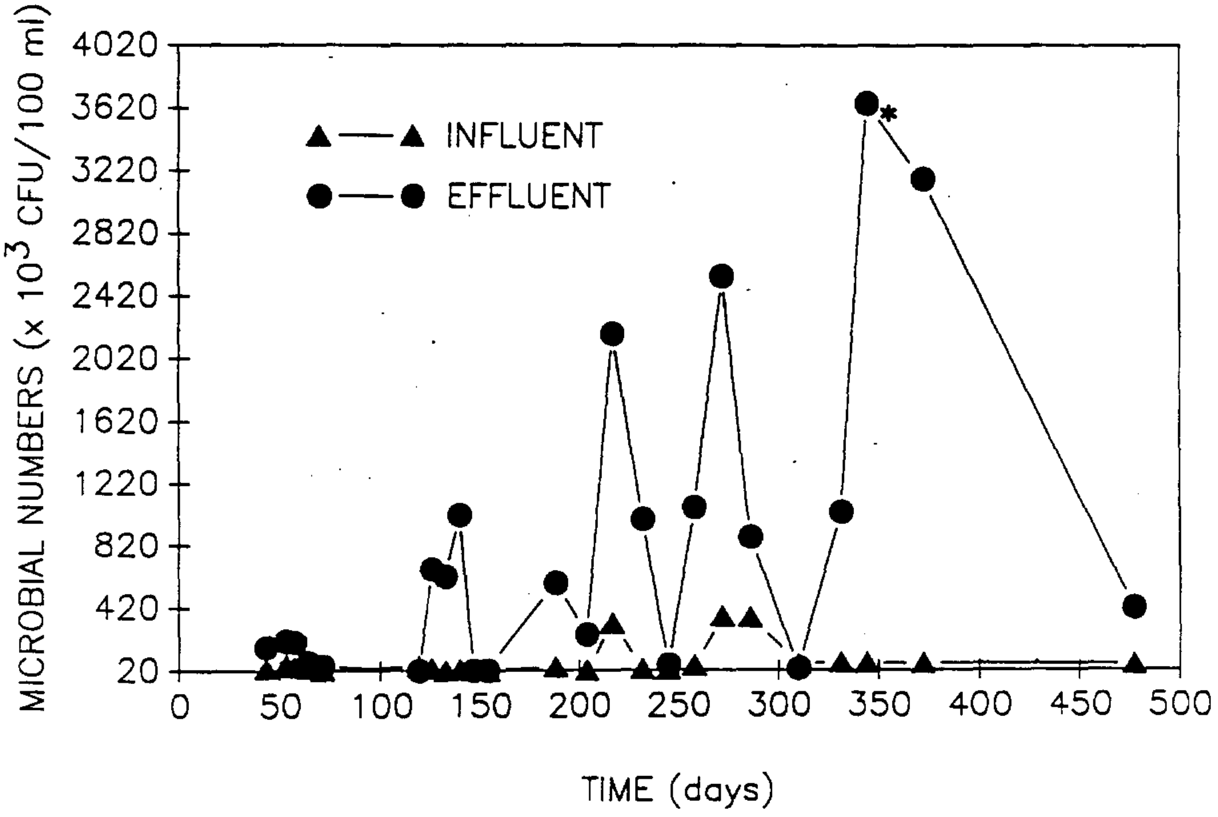


Figure 43. Phase II - Mont Vernon, NH GAC system. Microbial numbers through 477 days of operation (detection limit = 20,000 CFU/100 mL). (* new well)

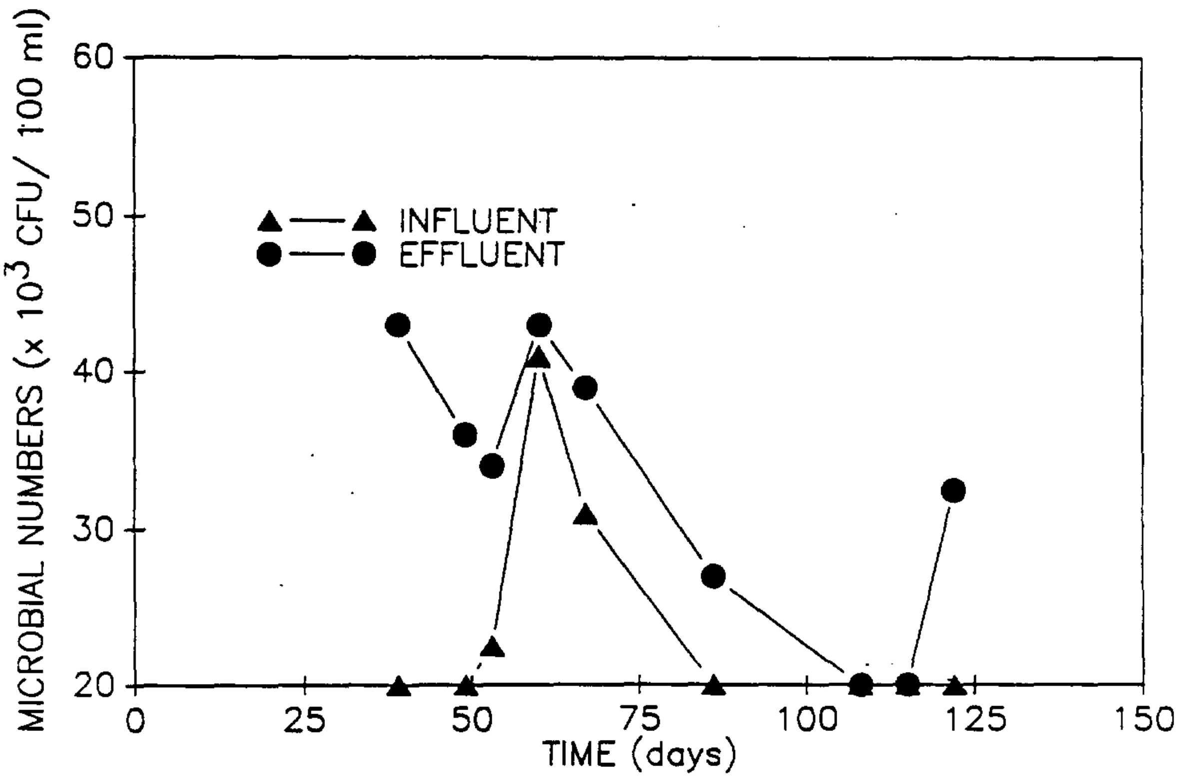


Figure 44. Phase II - Amherst, NH GAC system. Microbial numbers through 122 days of operation (detection limit = 20,000 CFU/100 mL).

higher than the typical ranges of 0 to 3,000 colonies/mL reported in the literature (Wilcox et al., 1983) for GAC columns used in water treatment. It is important to note that the bacterial concentrations found in the influent at Mont Vernon were elevated. Results for Amherst indicated that the microbial numbers for the GAC effluent ranged from about 200 to 400 colonies/mL. These values fall within the range of 0 to 600 colonies/mL reported by Bourbigot (1981) in effluent from a GAC column. The variations and fluctuation observed in the microbial number are analogous to those observed by other researchers (Wilcox et al., 1983) and may be related to seasonal effects.

Iron and Manganese--

Mont Vernon site--The iron and manganese concentrations during Phase II were mostly at or below the detection limit (Fe = 0.06 mg/L; Mn = 0.02 mg/L). On Day 43, the influent manganese concentration was 1.0 mg/L. Manganese was never detected in the GAC effluent. Iron concentrations were below detection limit in the effluent. Though the overall change in concentration, if it occurred, was probably small, the total accumulation of these metals in the GAC units may have been substantial considering the flow treated. For example, assuming an average flow of 36 m^3/day and a removal of 0.02 mg Fe/L, 262 g of Fe would have accumulated over a year. This may cause a problem in the units after several years of operation considering that backwashing is not normally recommended for GAC systems removing radon.

Amherst site--The water in Amherst contained significantly higher (α = 0.05 and 0.01, StT) concentrations of iron (0.78 ± 0.98 mg/L) than at Mont Vernon (Figure 45). Manganese concentrations averaged 0.10 ± 0.21 mg/L during Phase II and were significantly higher than Mont Vernon's at $\alpha = 0.10$, but not $\alpha = 0.05$ (StT). The average effluent iron and manganese concentrations were 0.16 ± 0.14 mg Fe/L and 0.08 ± 0.16 mg Mn/L, respectively. Figures 46 shows profile data for iron through the Amherst GAC unit. Iron was removed (influent vs. effluent iron significantly different at $\alpha = 0.05$, but not at 0.02; StT) on some days in the upper portion of the filter. There was no significant change (α = 0.05 and 0.10, StT) in manganese through the filter.

Uranium and Radium--

Mont Vernon site--The average uranium activity of the influent water from the original wells was 24.6 ± 3.6 pCi/L. Uranium removal followed the theoretical profile of a well adsorbed solute (Figure 47). Removal was not similar to radon because uranium species have very long half-lives (~10^5 to 10^9 yr). As a result, decay did not significantly impact the fate of sorbed uranium over the course of this study. On Day 43, all of the uranium (detection limit = 0.1 pCi/L) was removed after passing through 0.57 m^3 of GAC. Most of the uranium was removed by the top 0.57 m^3 of GAC on Day 300, with some removal occurring in the remaining 0.76 m^3. The removal profile over time indicated that the GAC bed would eventually be exhausted with respect to uranium.

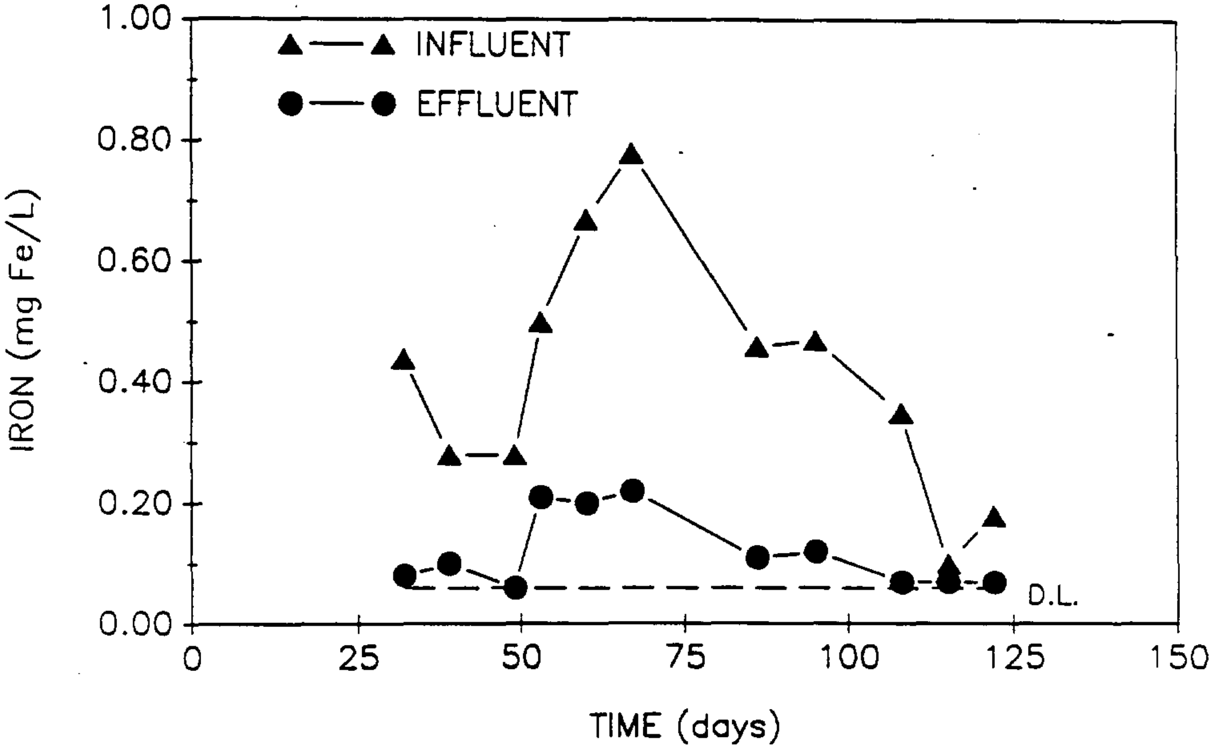


Figure 45. Phase II - Amherst, NH GAC system. Iron data through 122 days of operation (detection limit = 0.06 mg/L).

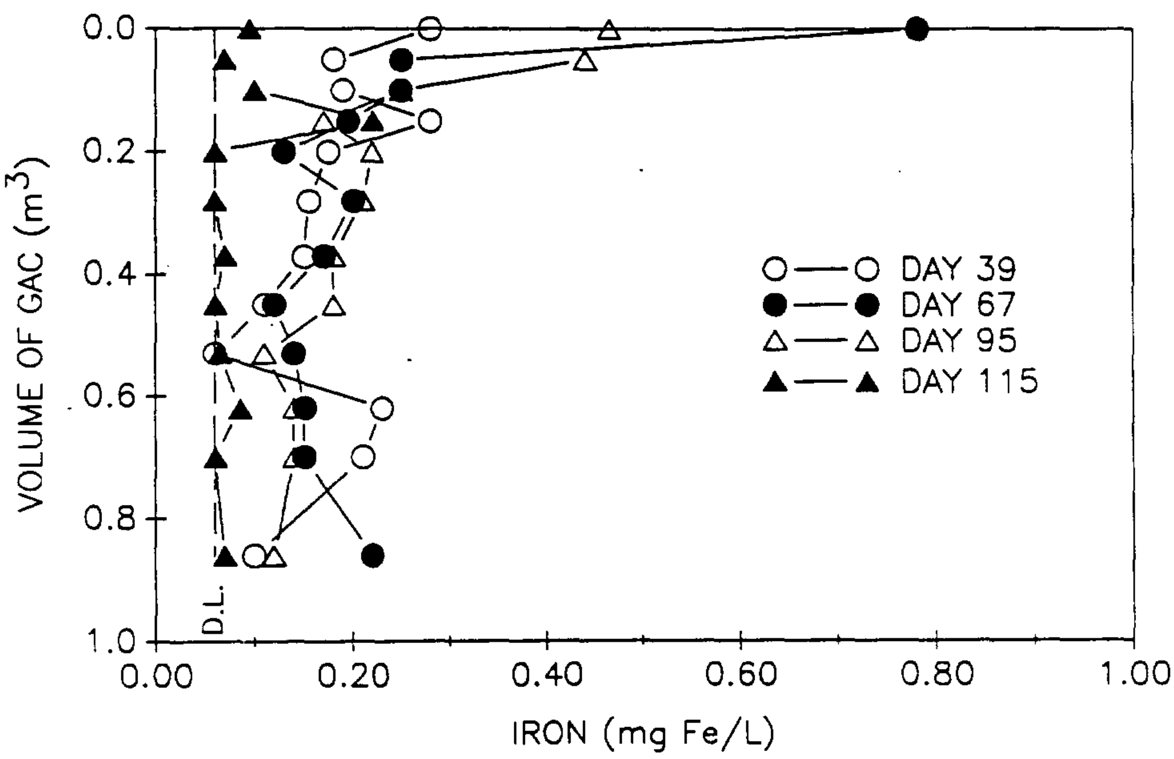


Figure 46. Phase II - Amherst, NH GAC system. Representative profiles of iron through the GAC system (detection limit = 0.06 mg/L).

The uranium activity increased dramatically (>500 pCi/L) when the new well was installed. Figure 47 shows the data collected during this period. By Day 372, the system had begun experiencing fluctuations in the uranium adsorption profile through the units. By Day 477, the influent uranium activity approached 500 pCi/L, however, there was little uranium removal.

The change in uranium removal exhibited by the GAC may have been due to a shift in uranium speciation resulting from the change in pH and alkalinity of the new water source. The effective removal of uranium by anion exchange (Jelinek and Sorg, 1988) suggests that uranium removal by GAC is based on coulombic (ionic or electrical) adsorption. Since researchers (Bean et al., 1964) have demonstrated that the surface of GAC is primarily nonpolar with a net negative charge, it follows that adsorption is at a minimum for negatively charged species and at a maximum for neutral species (Weber, 1972) assuming highly ionized positive species tend to be hydrophillic and therefore also reduce adsorption. Relating these facts to uranium speciation, Sorg (1988) reported that above pH 7 (new well = 7.7 ± 0.03) the negatively charged uranyl carbonate complex ($UO_2(CO_3)_2^{2-}$) is the predominant species whereas below pH ~ 6.8 (old well = 6.5 ± 0.2) the neutral UO_2CO_3 species predominates. Therefore, it is hypothesized that the shift in raw water pH and alkalinity when the new well was installed at Mont Vernon resulted in a shift in uranyl carbonate distribution from the favorably adsorbed neutral UO_2CO_3 to the poorly adsorbed $UO_2(CO_3)_2^{2-}$ species.

The radium in the water supply increased from 1.4 ± 0.1 pCi/L to >19 pCi/L when the new well was used (Figure 48). Data for influent and effluent radium were only available for a limited number of days. On Day 245, the effluent radium activity equaled the influent activity. When the new well began operating, the influent of 19.3 pCi/L on Day 372 was reduced to 2.3 pCi/L in the effluent. By Day 477, only 1.8 pCi/L of radium was removed by the GAC system (influent = 18.4 pCi/L, effluent = 16.6 pCi/L).

It is well documented that radium is poorly adsorbed by GAC (Sorg and Logsdon, 1978) since it has the greatest ionic and electropositive nature of all the Group II elements (Cotton and Wilkinson, 1980) rendering it extremely hydrophillic. Therefore, the small changes in radium concentration observed are not surprising. The observed removal of radium on Day 372 may be a result of physical filtration of radium containing particles by the GAC bed (Faust and Aly, 1987) or adsorption or ion exchange between the radium and precipitated solid phases (e.g., $Fe(OH)_3$) and/or organic matter deposited within the GAC bed.

Presently, there is no MCL set for uranium, but it is likely to be in the range of 20 to 40 pCi/L (EPA, 1989). The MCL for total radium is 5 pCi/L. Both uranium and radium present serious health concerns. Though GAC has some capacity for adsorbing these radionuclides, it seems to be sensitive to pH and alkalinity and it is not an alternative treatment technique for these radionuclides. In the Mont Vernon system, additional treatment for these radionuclides would be required to meet the MCLs. Adsorption of uranium and radium to the GAC may also be problematic because of their long half-lives.

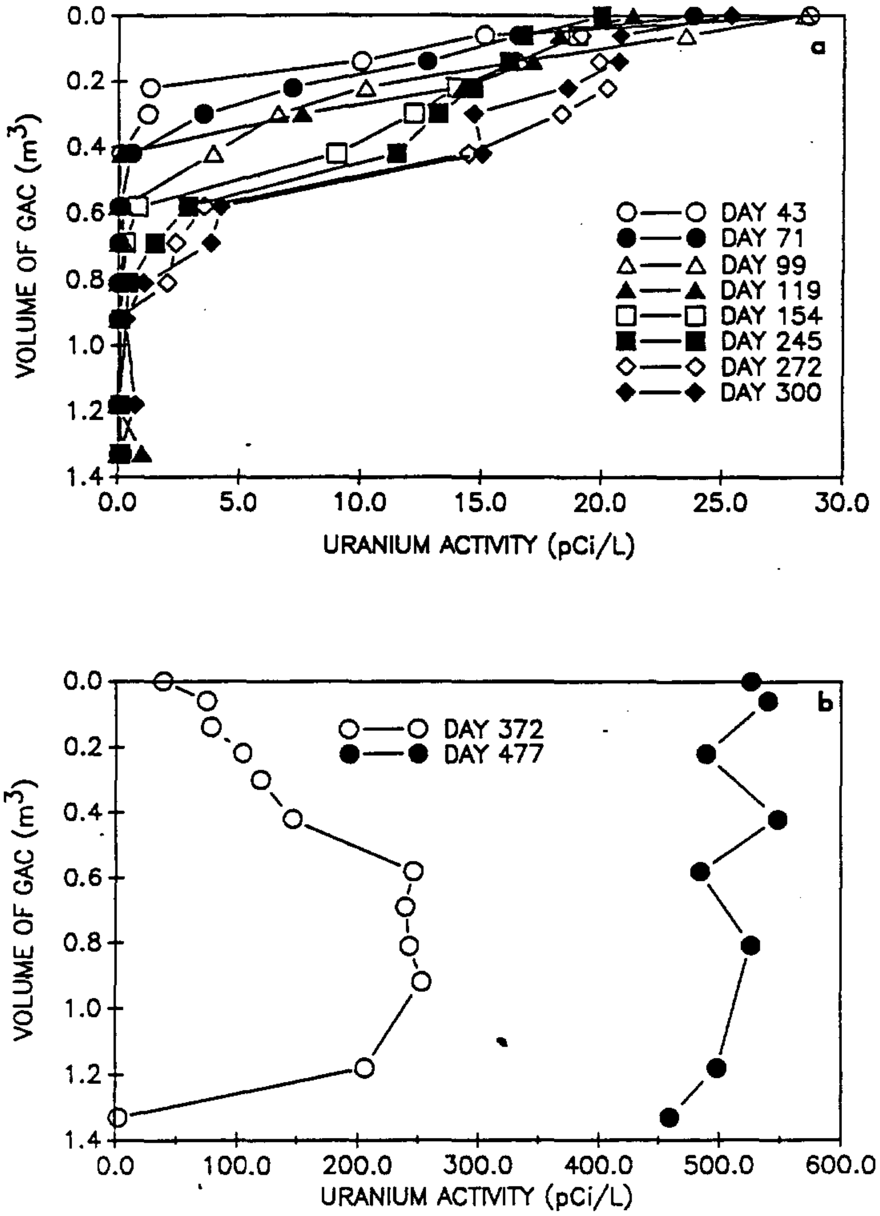


Figure 47. Phase II - Mont Vernon, NH GAC system. Representative profiles of uranium through the GAC system. Note differences in scales on abscissa. Original well (a) and new well (b).

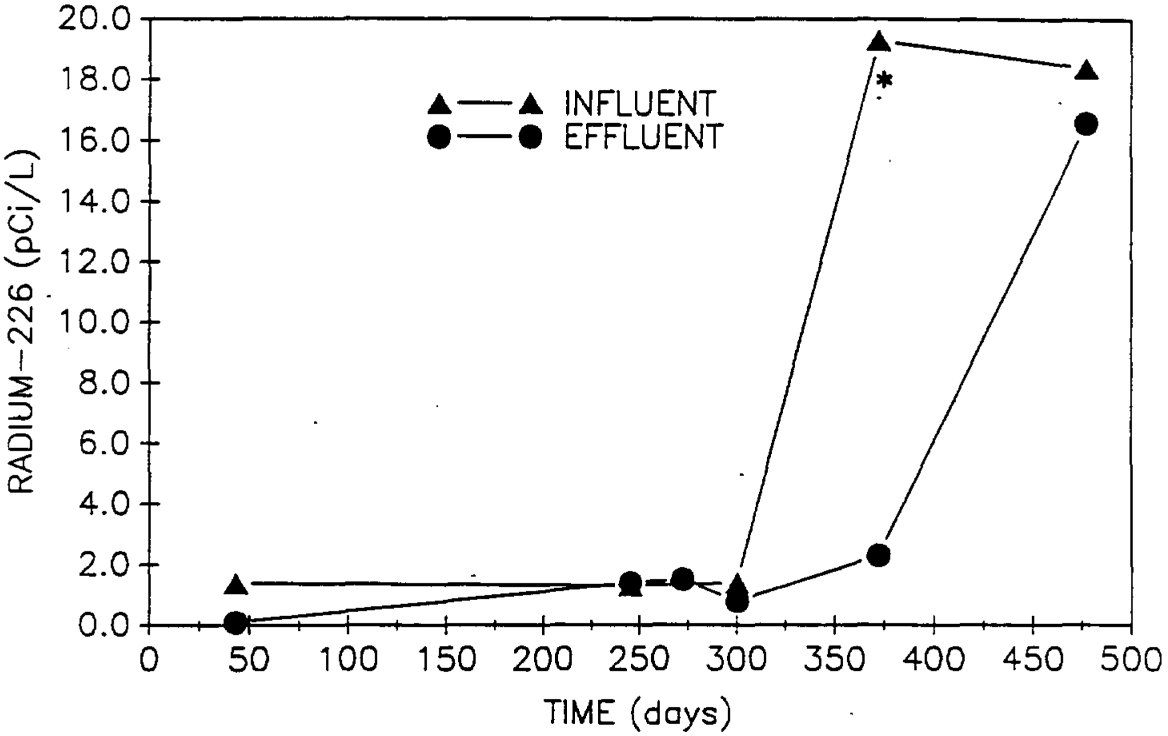


Figure 48. Phase II - Mont Vernon, NH GAC system. Radium-226 data through 477 days of operation. (* new well)

Accumulation of these radionuclides could lead to disposal problems with the GAC.

Amherst site--The uranium in the Amherst water supply averaged 34.5 ± 20.2 pCi/L. There was a steady decrease in the influent uranium activity during Phase II. The typical exponential removal profiles observed at Mont Vernon for uranium were not found at Amherst except during Day 39 (Figure 49). On many instances, higher uranium activities were observed within the unit than in the influent water. The pH and alkalinity of the Amherst water averaged 8.0 ± 0.14 and 48 ± 4.3 mg/L as $CaCO_3$, respectively during Phase II. As previously discussed, under these conditions, the negatively charged $UO_2(CO_3)_2^{2-}$ species will predominate and tend not to sorb to the GAC.

The only radium data available at Amherst was for Day 39 where the influent activity of 2.1 pCi/L was reduced to an effluent of 0.7 pCi/L. The limited database for radium prevented analysis of radium removal by the GAC system at Amherst.

Special Sampling Events

Diurnal Variations--

The effect of diurnal variations was investigated at the Mont Vernon site for a period of 12 hr on two consecutive days in July, 1987 (Day 266 and 267). Samples were taken from influent and effluent ports every hour for alkalinity, pH, temperature, dissolved oxygen, turbidity, iron, manganese and radon. Hourly measurements of total flow were also taken. Gamma readings, microbial numbers, uranium and radium analyses were not conducted during the diurnal studies.

As observed during Phase II, there was no change ($\alpha = 0.05$ and 0.10, StT) in pH and temperature as the water passed through the GAC units (Figures 50 and 51). The alkalinity (Figure 50) remained relatively constant during the diurnal study. The DO significantly ($\alpha = 0.05$ and 0.10, StT) decreased through the GAC system (Figure 51) by an amount comparable to that observed during Phase II. The turbidity was in the same low range observed during Phase II (Figure 52) and was significantly removed ($\alpha = 0.05$ and 0.10) through the units (0.09 ± 0.02 to 0.05 ± 0.01) on Day 266, but not Day 267. The change was relatively small and somewhat variable and the data were at or below the detection limit (0.05 NTU). The iron and manganese data were all below detection.

The water flowrate during this period was higher than the average (Figure 53). On Day 266, the flow averaged 49 ± 12 m^3/day (EBCT = 38.7 min) and was not significantly different ($\alpha = 0.05$ and 0.10, StT) than that observed on Day 267 (50 ± 8 m^3/day). There was some variation during the day, as would be expected in a small community. The influent radon activity was significantly different ($\alpha = 0.05$ and 0.01, StT) on the 2 days (Figure 53). On Day 266, it averaged 190,155 ± 3,739 pCi/L, while it was 165,013 ± 7,542 pCi/L on Day 267. In addition on Day 267, there was a steady decrease in influent activity over time. In spite of the differences in the influent activity, the effluent

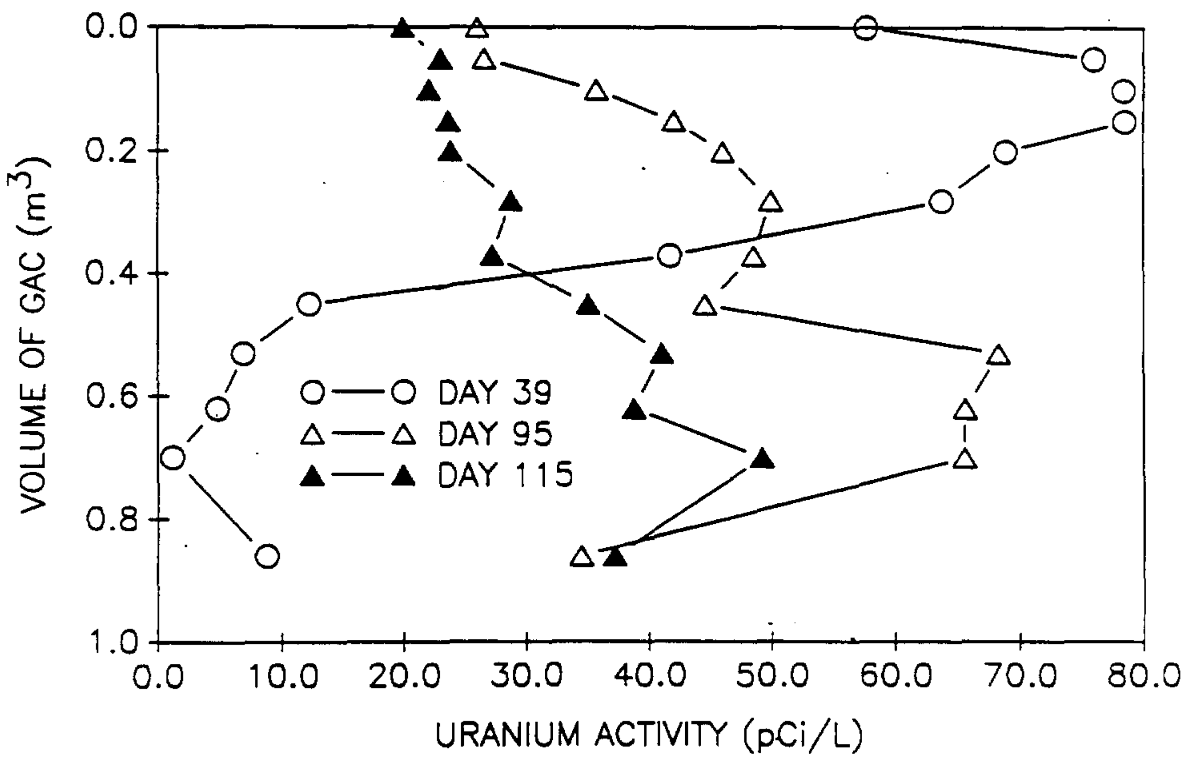


Figure 49. Phase II - Amherst, NH GAC system. Representative profiles of uranium through the GAC system.

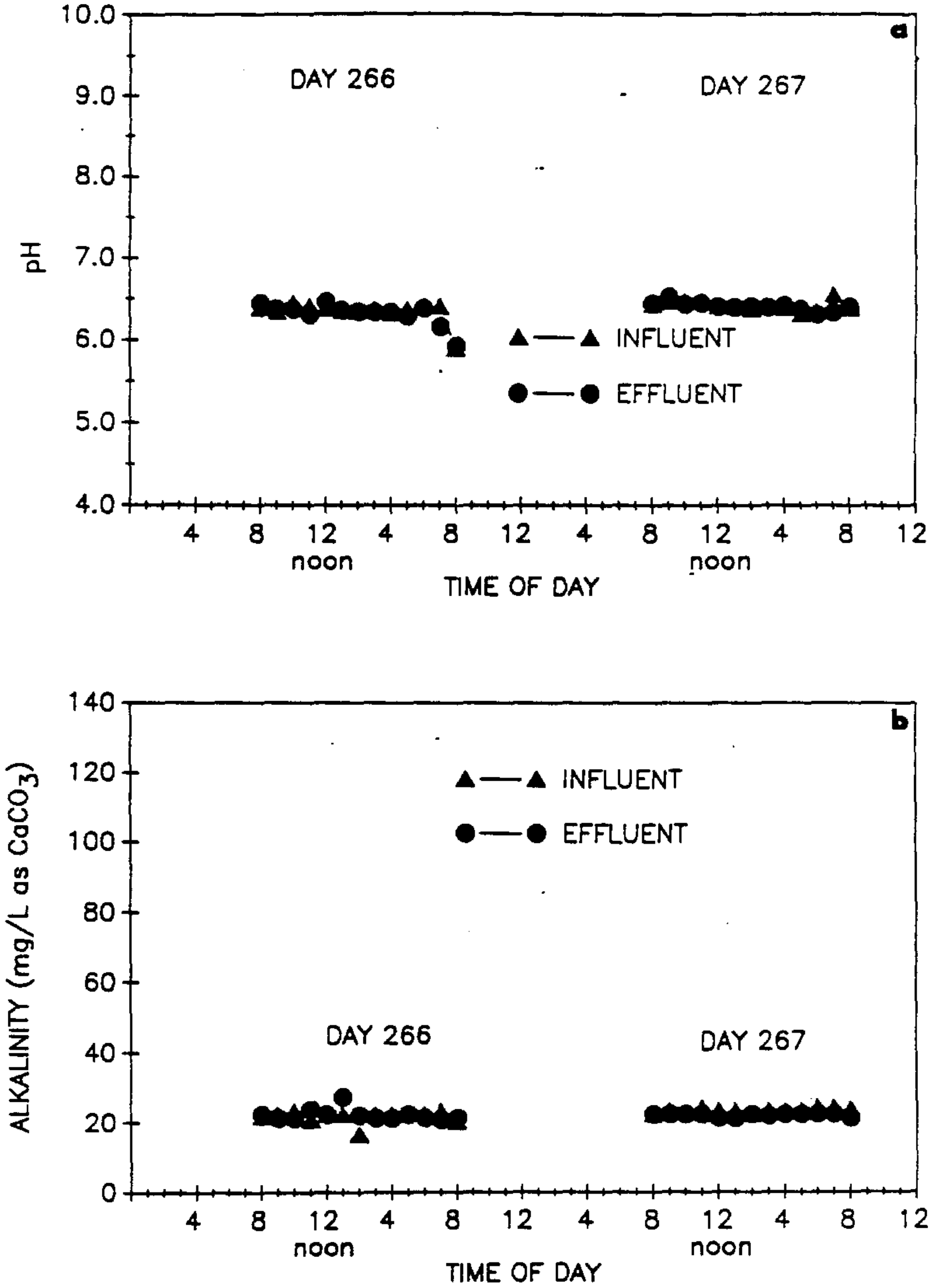


Figure 50. Diurnal variations - Mont Vernon, NH GAC system. pH (a) and alkalinity (b) data.

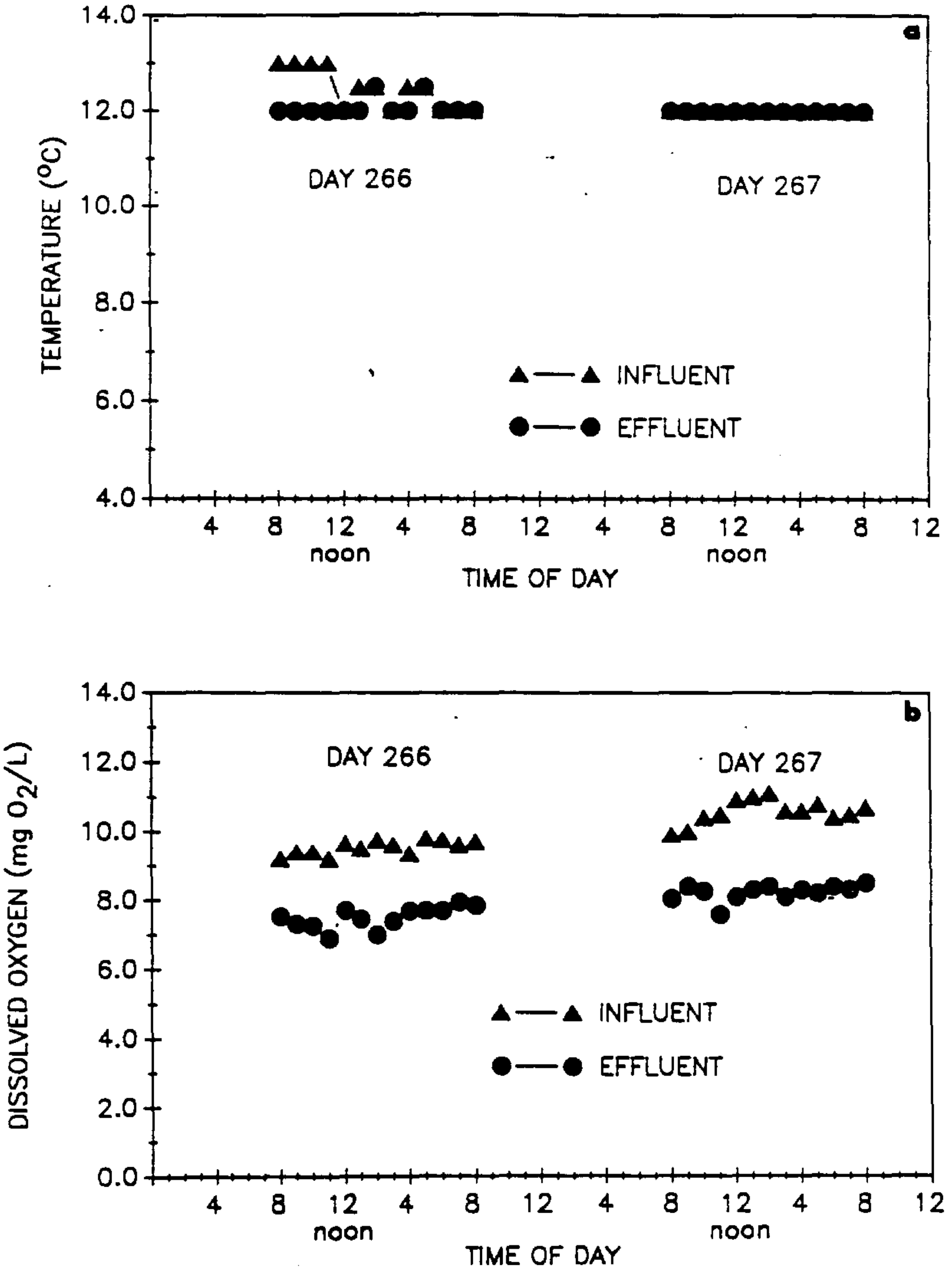


Figure 51. Diurnal variations - Mont Vernon, NH GAC system. Temperature (a) and dissolved oxygen (b) data.

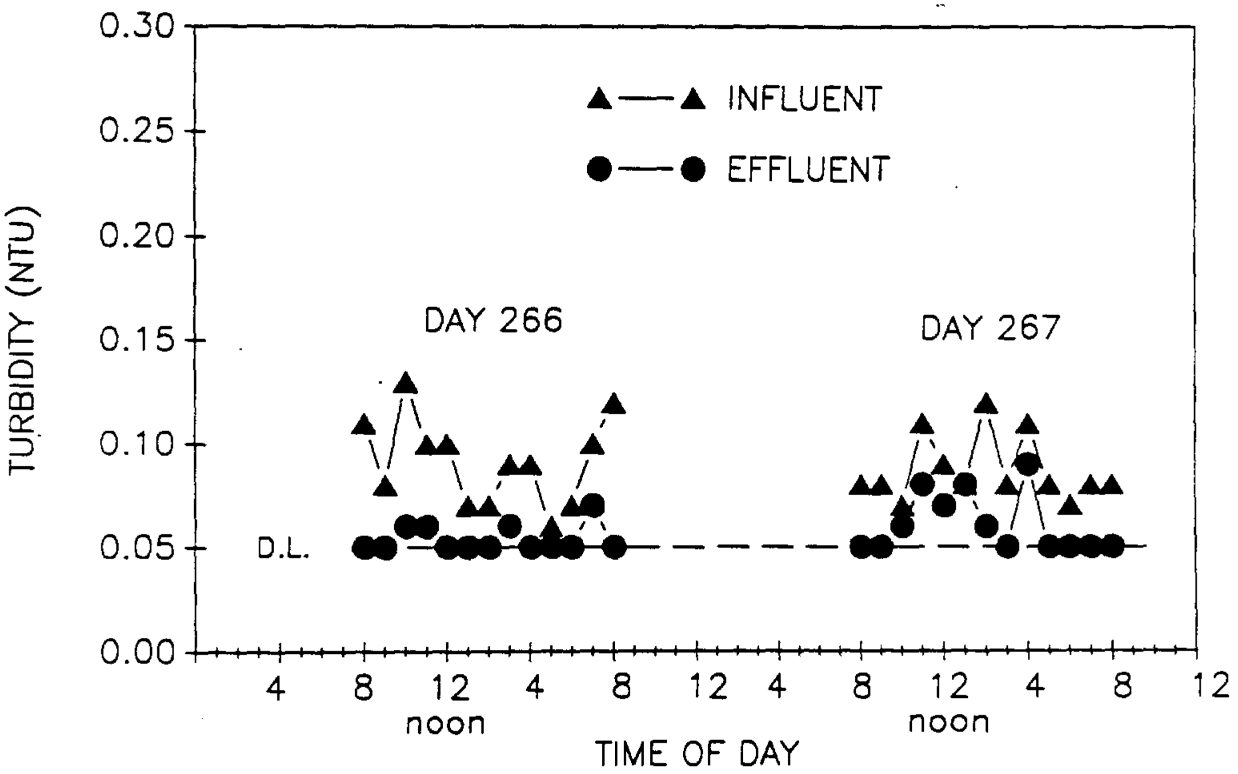


Figure 52. Diurnal variations - Mont Vernon, NH GAC system. Turbidity data (detection limit = 0.05 NTU).

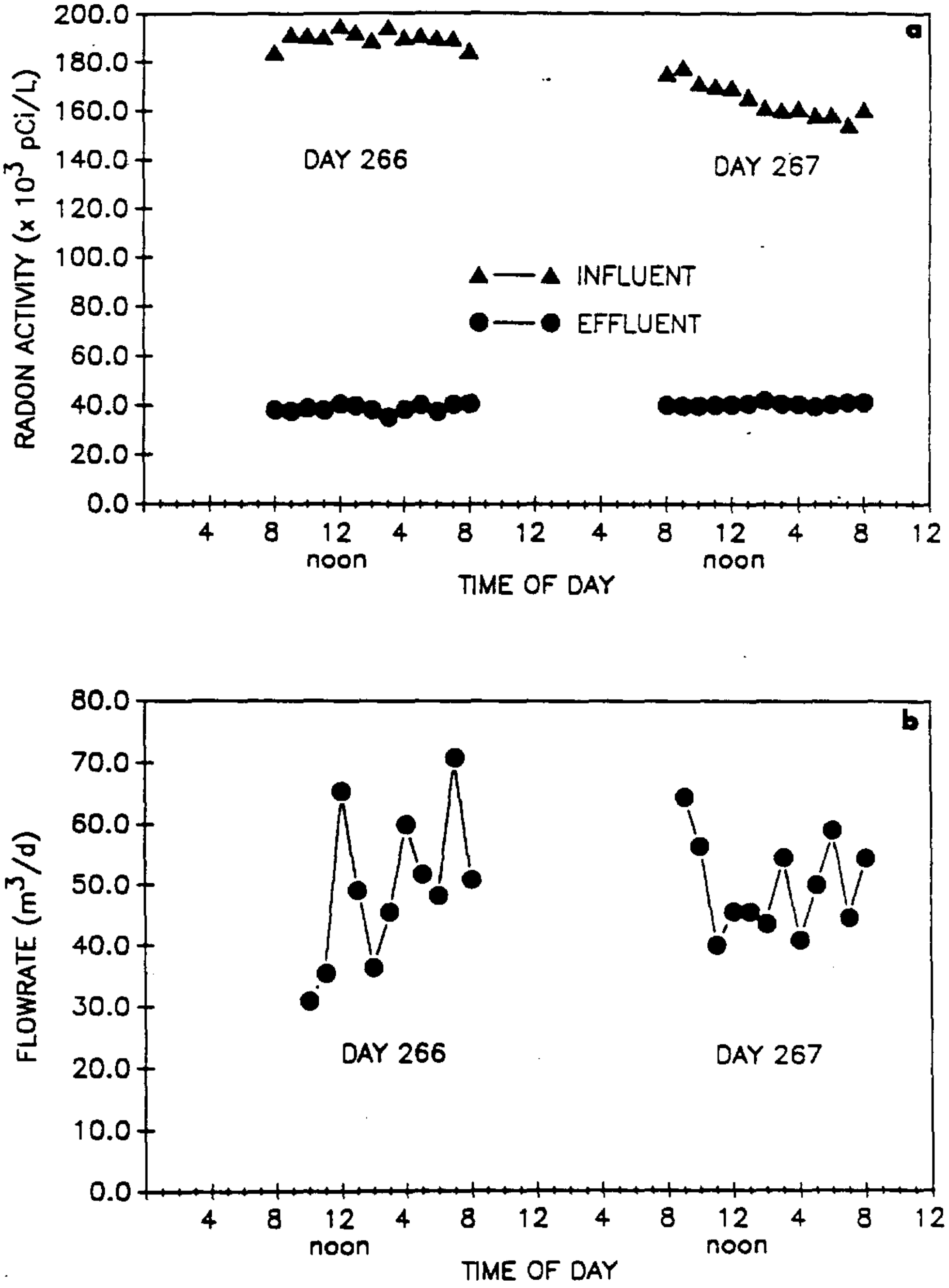


Figure 53. Diurnal variations - Mont Vernon GAC system. Radon activity (a) and water flowrate (b) data.

radon activity on the two days was not significantly different ($\alpha = 0.05$ and 0.10, StT). Although fluctuations in the influent conditions to the GAC systems were dampened during Phase I and II, the diurnal data indicate that the Mont Vernon units did not produce a better effluent when the influent radon activity decreased by ~25,000 pCi/L for a given water flowrate. The K_{ss} calculated for Day 266 (2.5 ± 0.6 hr^{-1}) was similar to the Phase II K_{ss} (2.4 ± 0.7 hr^{-1}). The Day 267 K_{ss} was, however, lower (2.0 ± 0.7 hr^{-1}). The fact that the effluent quality did not improve may have resulted because (i) the GAC system was not at steady state, (ii) radon was desorbing in response to the decrease in influent activity, and/or (iii) a water quality parameter(s) not monitored during the diurnal study was affecting radon adsorption. Further studies would need to be conducted to determine why the GAC system did not respond to the decreased influent loading.

High Flow--

The effect of high flow on the Mont Vernon system was investigated in July, 1987 (Day 281). Influent and effluent samples and measurements for flow, alkalinity, pH, temperature, turbidity, dissolved oxygen, microbial populations, iron, manganese and radon were taken every hour for 3 hr. The wells would not sustain a continuous flowrate at that level without depleting the community's water storage. As a result, the study was stopped after 3 hr. Radon samples were taken from all ports.

As observed in Phase II, there was no change ($\alpha = 0.05$ and 0.10, StT) in pH (Figure 54), alkalinity (Figure 54), and temperature (Figure 55) through the GAC units during the high flow study. DO was significantly reduced through the filter ($\alpha = 0.05$ and 0.10, StT) after 1 hr of operation, but not before that time (Figure 55). Turbidity remained low and was not significantly removed through the filter ($\alpha = 0.05$ and 0.01, StT). These changes are similar to those observed during Phase II. Iron and manganese were both below detection. The microbial numbers were variable (Figure 56) as observed in Phase II, but generally higher in the effluent than in the influent.

The flowrate during this part of the GAC evaluation averaged 73 ± 9.0 m^3/day (EBCT = 26.3 min) which was high compared to the flowrate during most of the study, but not during the period of June to August 1987. The flowrate for the high flow study was approximately 3 times the design of 25 m^3/day. Over the 3 hr period, the influent and effluent radon activity averaged 212,781 ± 11,164 pCi/L and 52,720 ± 574 pCi/L, respectively. These remained very consistent as did radon removal through the units (Figure 57). The K_{ss} for this period averaged 3.18 ± 1.39 hr^{-1}, compared with the 2.4 ± 0.72 hr^{-1} value during Phase II. This is closer to the 3.02 hr^{-1} at 6-10°C K_{ss} found previously with this GAC (Lowry and Lowry, 1987). Overall, the GAC system performed well during this study, producing a constant effluent radon activity in response to a high radon loading.

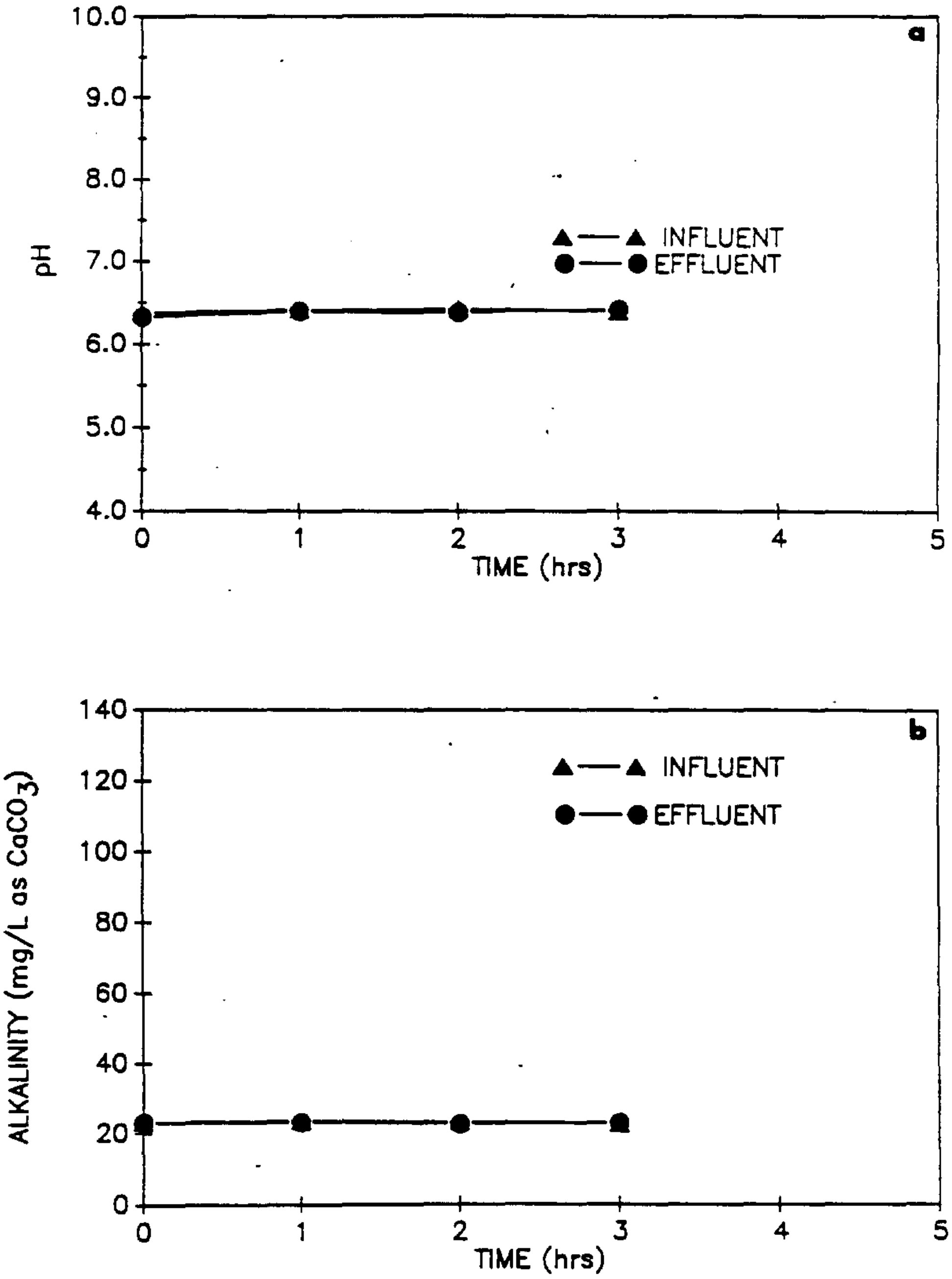


Figure 54. High flowrate - Mont Vernon, NH GAC system. pH (a) and alkalinity (b) data.

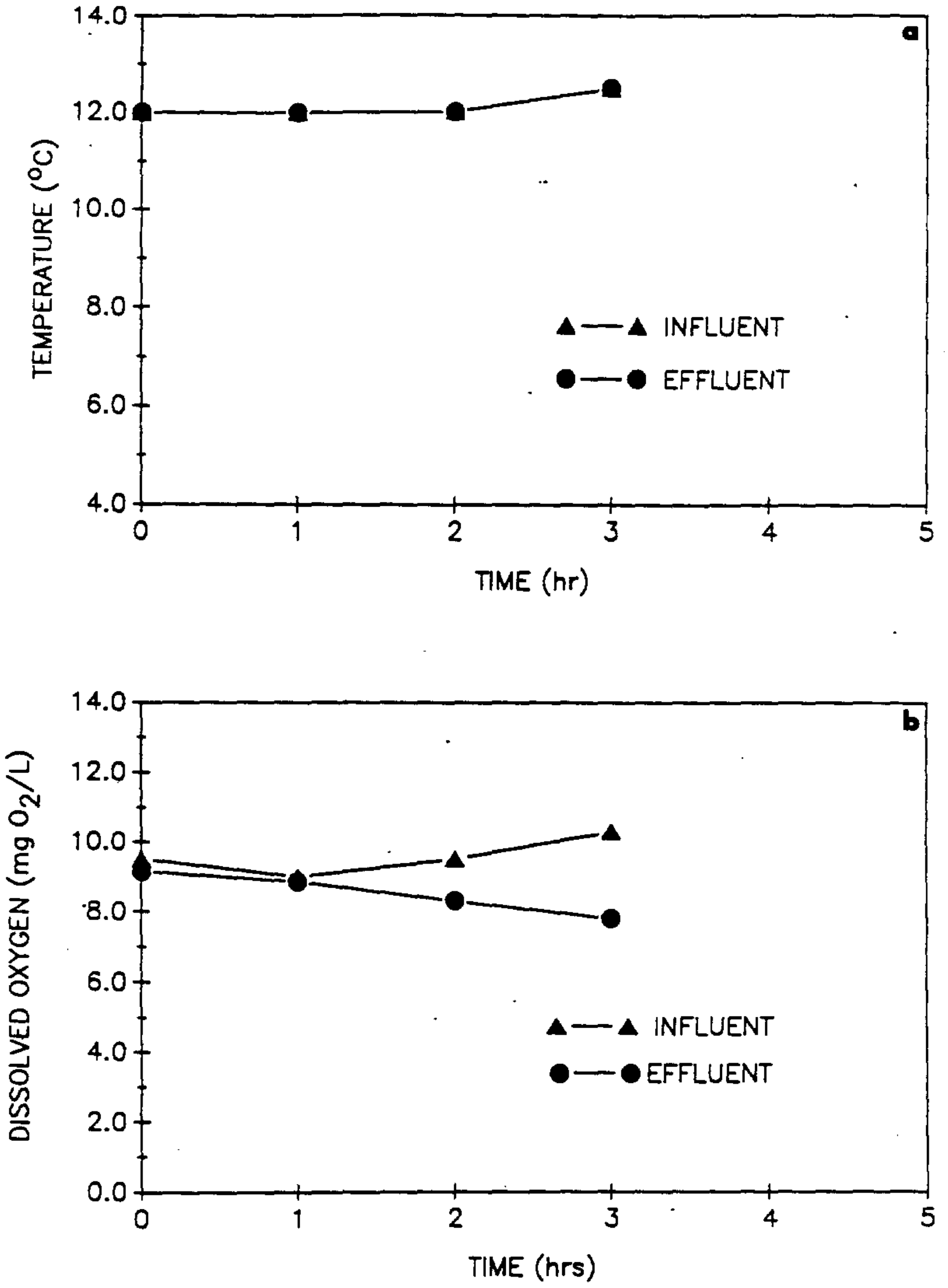


Figure 55. High flowrate - Mont Vernon, NH GAC system. Temperature (a) and dissolved oxygen (b) data.

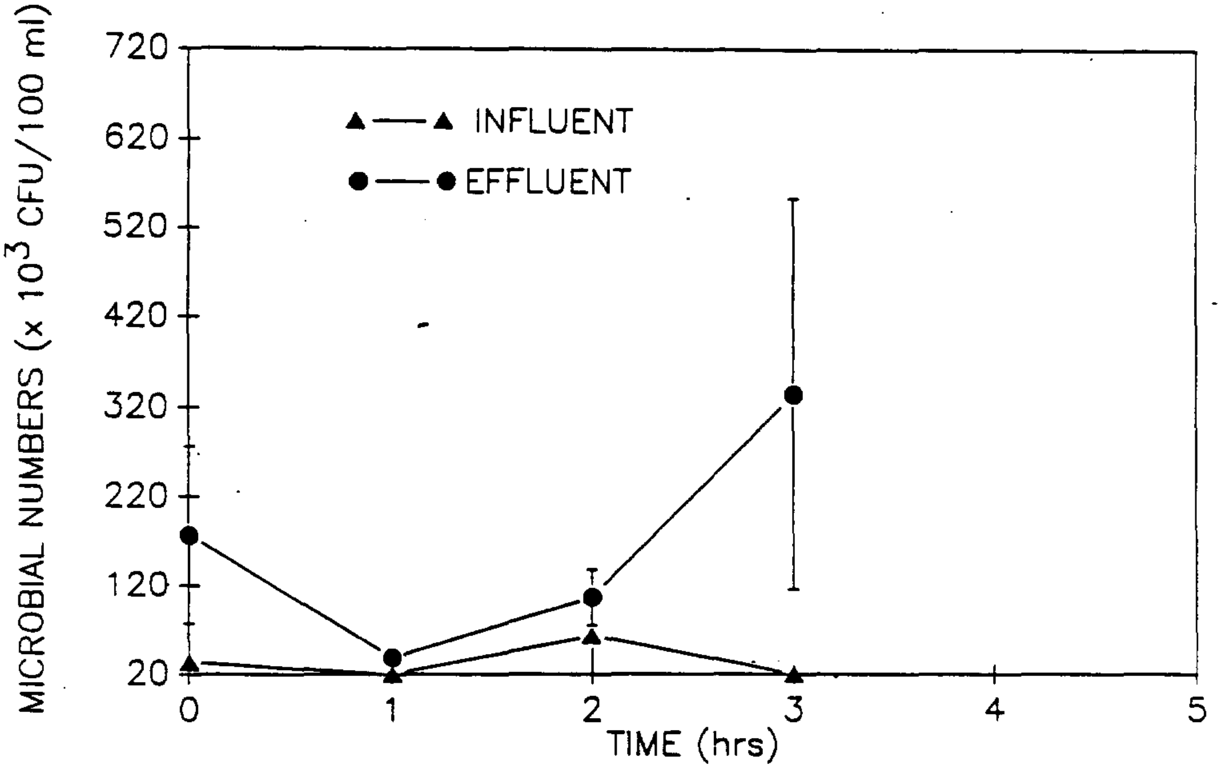


Figure 56. High flowrate - Mont Vernon, NH GAC system. Microbial numbers (detection limit = 20,000 CFU/100 mL).

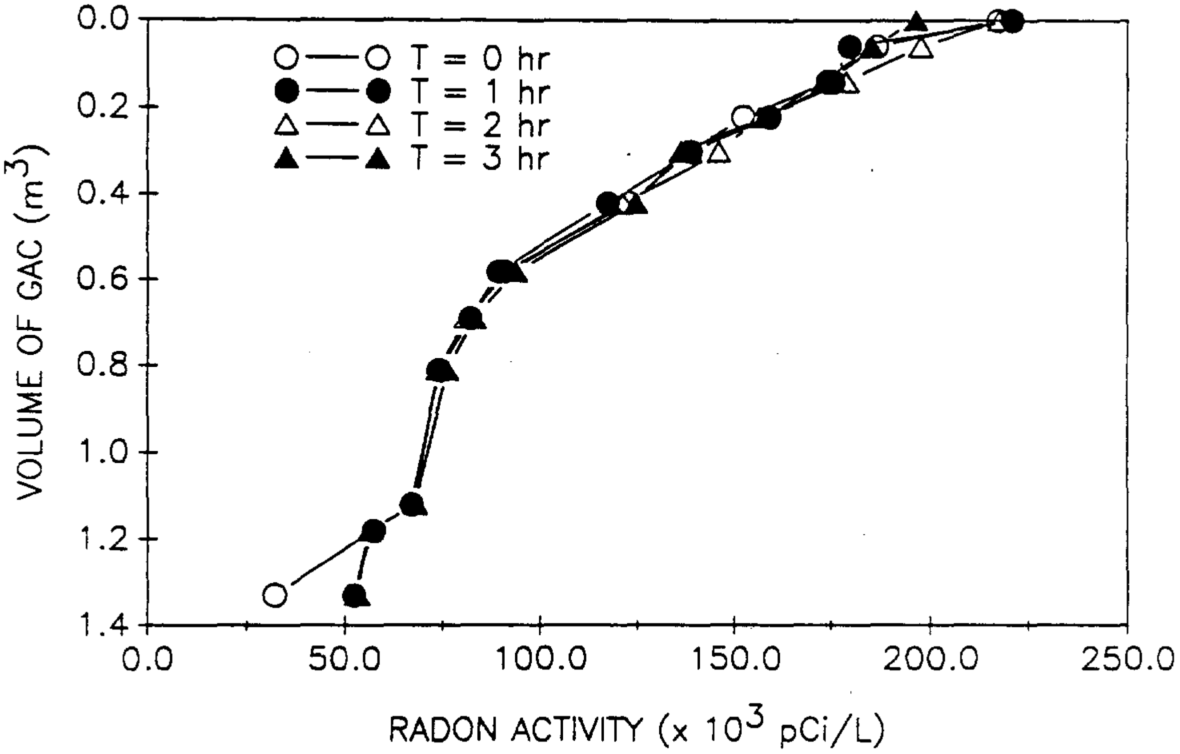


Figure 57. High flowrate - Mont Vernon, NH GAC system. Radon activities through the GAC system.

Coring--

Cores of the GAC material were obtained from the Mont Vernon and Amherst systems in late July 1987. The Mont Vernon system had been operating for 293 days before it was cored. The Amherst GAC unit had only operated for 122 days and it had been out of service for 171 days prior to coring. Each core was sectioned, and composite samples of each section were collected for percent moisture content, microbial populations, iron, manganese, uranium radionuclides, radium-226 and lead-210 analyses. Uranium and radium analyses were done by the State of New Hampshire and the University, while only the University made lead-210 measurements. All data were calculated in kg per dry weight of GAC (based on percent moisture content) and reflect a composite of the GAC volumes shown in Table 6. Samples of virgin GAC were also analyzed and the values obtained were subtracted out of all data presented.

Mont Vernon site--Three cores of GAC #1 and two cores of GAC #2 were taken from the Mont Vernon system after 293 days of operation. Manganese was below detection except in the bottom 0.27 m^3 of GAC in GAC #1 (2.7 mg/kg) and the bottom 0.41 m^3 of GAC #2 (2.9 mg/kg) (Figure 58). The highest iron accumulation was found in the top of GAC #2 (Figure 58). The iron accumulation pattern is likely the result of dissolved (ferrous) iron entering the filter, undergoing oxidation as it passed through the upper part of GAC #1 and being deposited as an insoluble (ferric) iron complex (e.g., $Fe(OH)_3$) in the bottom of GAC #1 and the top of GAC #2. It was impossible to compare the coring data to a theoretical removal as the iron concentrations in the water were usually below detection. However, the data indicate that accumulation can occur even with small changes in iron concentration because of the large volume of water passing through the units over time. As a result, backwashing may be required to prevent significant headloss. Microorganisms were growing throughout the two units (Figure 59), but were most concentrated in the bottom of GAC #2. The microbial numbers found in the GAC units at Mont Vernon are comparable to those reported by Wilcox et al. (1983), who found that the CFU per gram of GAC ranged from 3×10^5 to 6×10^7.

The estimated uranium adsorbed corresponded fairly well to the coring data (Table 7) considering these estimates were based on an average uranium activity and total flow through the unit and did not take into account the actual variations occurring at the site. It was not possible to do an accurate mass balance on any of the elements sampled because of the natural variation in influent water quality and flowrate occurring in the full-scale small community facilities.

The uranium 238 and 235 profiles (Figure 60) and the uranium removal profile obtained during Phase II sampling (Figure 47) both indicate uranium was removed exponentially through the GAC system. There was a progressive movement of the uranium adsorption front through the filter observed in Phase II (Figure 47) and the coring data indicated that the GAC was approaching exhaustion with respect to uranium so that eventually no additional removal would be occurring. The data indicate that uranium was being retained by the GAC and not desorbed, however, studies with careful regulation of influent

TABLE 6. GAC VOLUMES COMPOSITED IN CORING EVENT

Site	Volumes Composited (m^3)
Mont Vernon	0.00 - 0.14*
	0.14 - 0.30*
	0.30 - 0.57*
	0.57 - 0.69+
	0.69 - 0.93+
	0.93 - 1.33+
Amherst	0.00 - 0.20
	0.20 - 0.45
	0.45 - 0.85

* GAC #1.

\+ GAC #2.

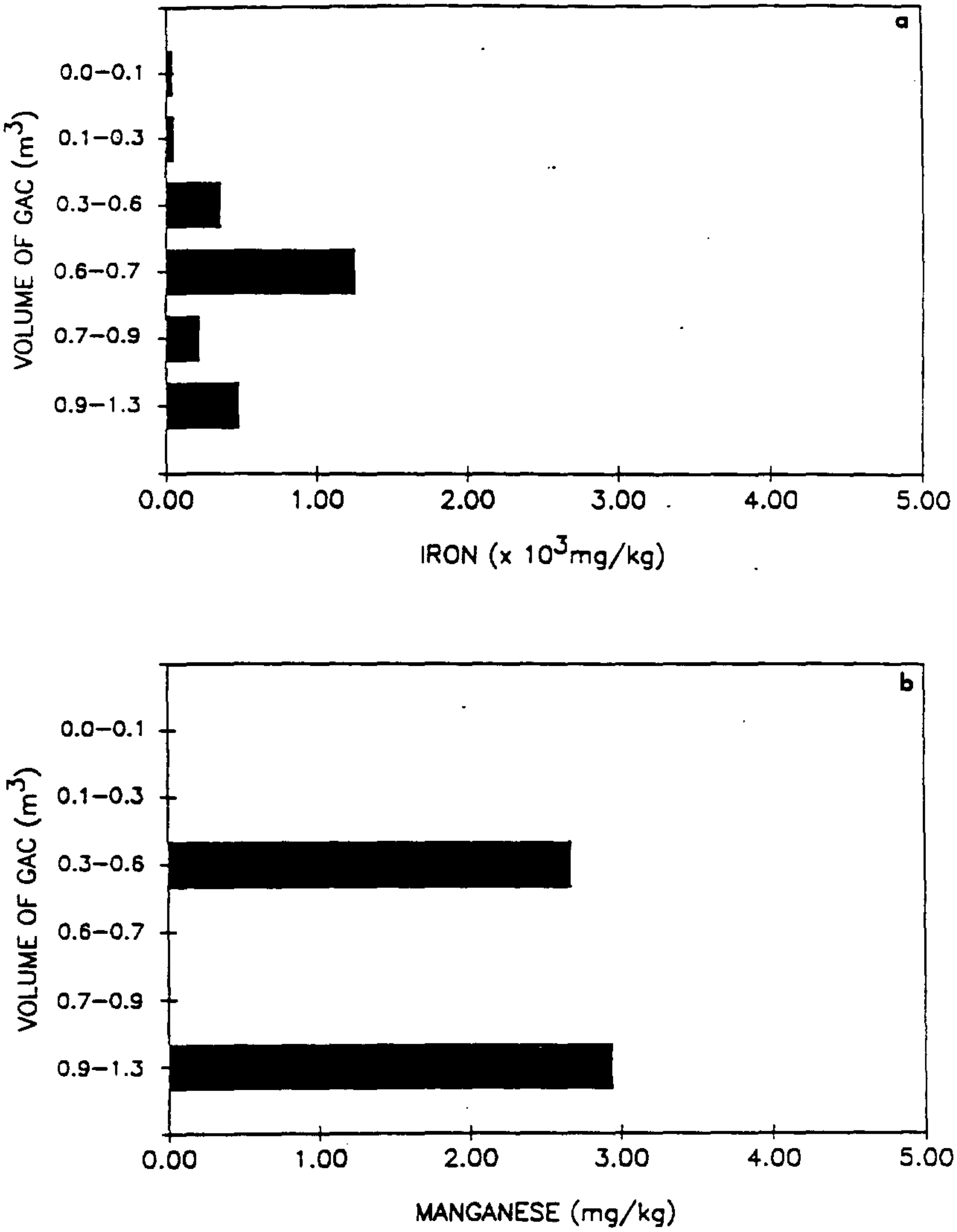


Figure 58. Coring event - Mont Vernon, NH GAC system. Iron (a) and manganese (b) found in GAC system.

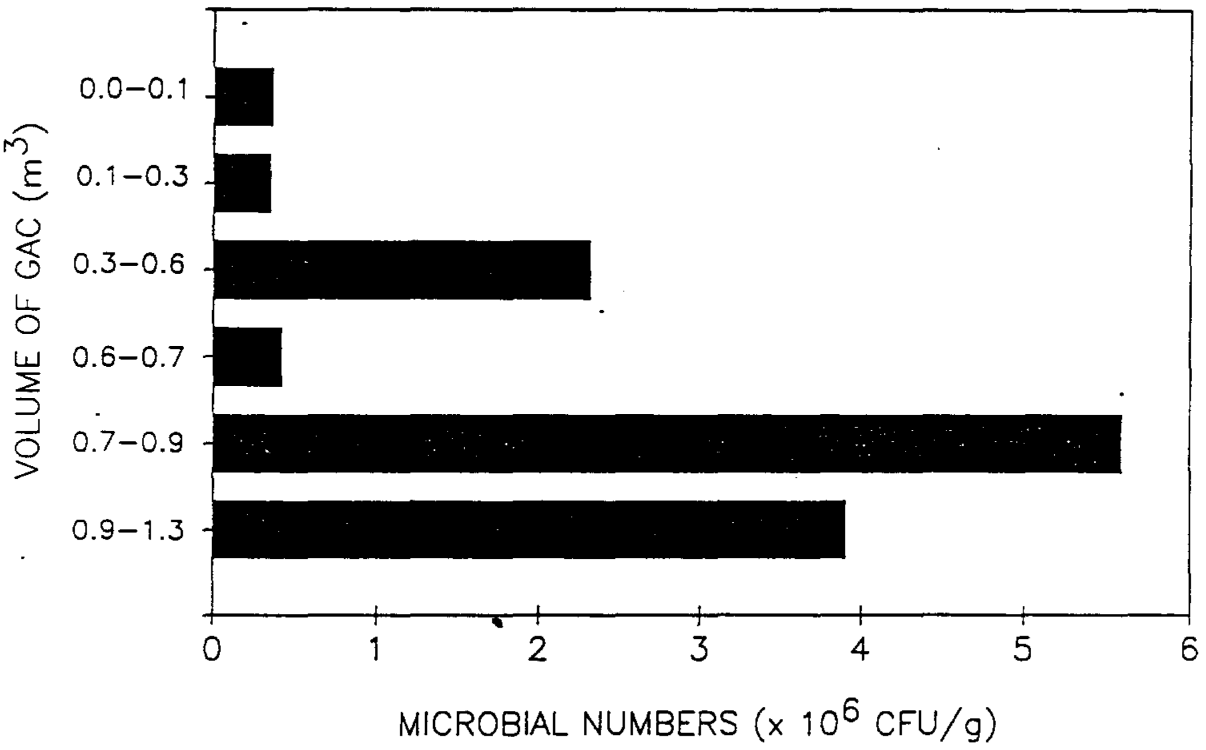


Figure 59. Coring event - Mont Vernon, NH GAC system. Microbial numbers found in the GAC system.

TABLE 7. COMPARISON OF THEORETICAL URANIUM* ADSORBED TO MONT VERNON CORING DATA

Volume of GAC (m^3)	Theoretical Adsorbed+ (pCi/kg)**	Coring Results (pCi/kg)	
		State++	University***
0.0 - 0.14	1,542,719	905,200	549,200
0.14 - 0.30	1,038,363	457,100	313,000
0.30 - 0.57	732,371	425,200	264,800
0.57 - 0.69	123,559	92,100	85,560
0.69 - 0.93	17,074	17,500	5,000
0.93 - 1.33	~0	598	290

[*] Coring data includes uranium-238 and uranium-235. Theoretical calculation based on total uranium as assessed by EPA Method 908.0 (EPA, 1980).

[+] Calculated using average total uranium activity and water flowrate measured during operating period.

[**] pCi/kg dry weight of GAC, based on percent moisture data.

[++] State of New Hampshire Division of Public Health Services Laboratory.

[***] University of New Hampshire Institute of Earth, Oceans and Space Laboratory.

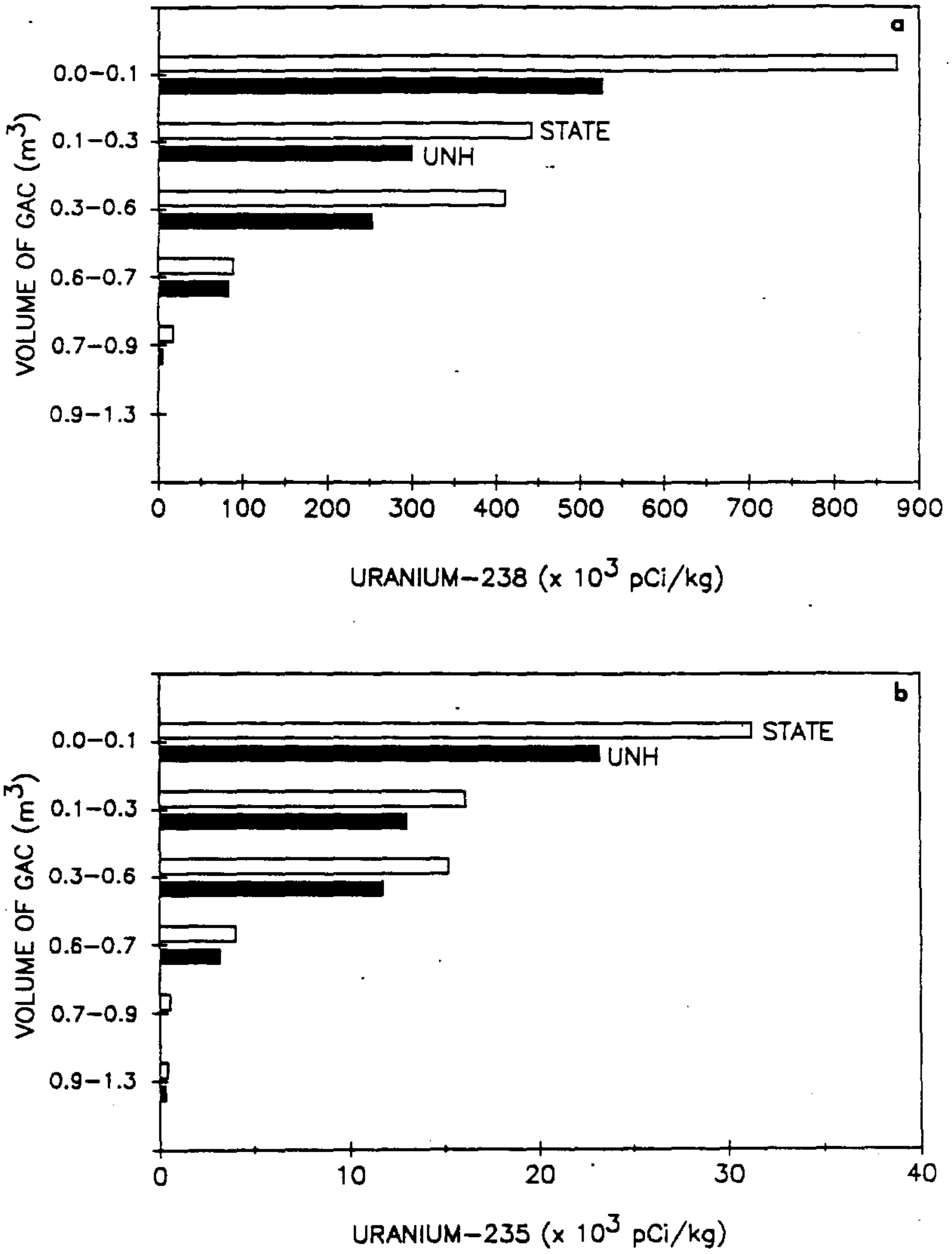


Figure 60. Coring event - Mont Vernon, NH GAC system. Uranium-238 (a) and uranium-235 (b) found in the GAC system.

activity, flowrate and water quality would be needed to verify this observation.

There was very little radium-226 data collected during Phase II, hence comparisons of theoretical adsorbed radium and the coring data could not be made. The cores (Figure 61) indicated that significant quantities of radium were found throughout the entire GAC system. Since the removal of radium by GAC adsorption alone is considered unfavorable, (Sorg and Logsdon, 1978) it is hypothesized that the radium found in the GAC cores may have been removed by adsorption or ion exchange reactions occurring with other solid phases (e.g., $Fe(OH)_3$, $Mg\ CO_3$) or organic matter deposited on the GAC.

The lead-210 coring data was indicative of the retention of radon progeny on the GAC. As observed for the radon removal profiles during Phase II (Figures 34 and 35), the sorbed lead-210 content decreased exponentially through the unit (Figure 61). A comparison of the theoretical lead-210 adsorbed and lead-210 coring data (Table 8) indicated that most, if not all of the radon progeny were retained by the GAC. As with the other comparisons, it was impossible to make a completely accurate mass balance because of variations in the raw water quality and flowrate. Further study under more controlled conditions would be required to demonstrate whether retention is complete or affected by variations in the raw water loading.

Rubin and Mercer (1981) reported that GAC adsorption capacities for (non-radioactive) lead ranged from 6.2×10^4 to 1.9×10^6 µg/kg at pH 6.5 and from 2.1×10^6 to 1.8×10^8 µg/kg at pH 8.0 based on Langmuir adsorption isotherms. Comparison of these data to the lead-210 coring results (Table 8) indicates that the GAC's capacity for lead-210 at Mont Vernon was at least 10^6 times higher than the actual lead-210 loading. This supports the hypothesis that lead-210 was completely retained by the GAC units. However, it should be noted that lead removal by GAC is highly dependent upon pH (Netzer and Hughes, 1984). Therefore, the ability of the GAC to retain lead-210 may be highly dependent on pH and raw water quality and should not be assumed without verification.

The GAC in both units, except for the bottom 0.41 m^3 of GAC #2, exceeds the limit for uranium-238 (58,410 pCi/kg; 2.5×10^{-5} Ci/m^3) as outlined by the New Hampshire Radiological Health Program de minimus regulations (1983). All materials which exceed de minimus levels are considered to be low level radioactive wastes. Ultimately, it appears all of the GAC at Mont Vernon would have exceeded the de minimus standards if monitoring had continued. All of the GAC is below the de minimus level (58,410 pCi/kg; 2.5×10^{-5} Ci/m^3) for uranium-235. All of the material presently exceeds the de minimus limit (44.39 pCi/kg; 1.9×10^{-8} Ci/m^3) for radium-226. The State of New Hampshire does not have a de minimus standard for lead-210. When the GAC at Mont Vernon needs to be replaced, it will have to be handled as a low level radioactive waste. This impacts the cost of replacement of the GAC system when it occurs and also has implications with respect to handling and public policy. It

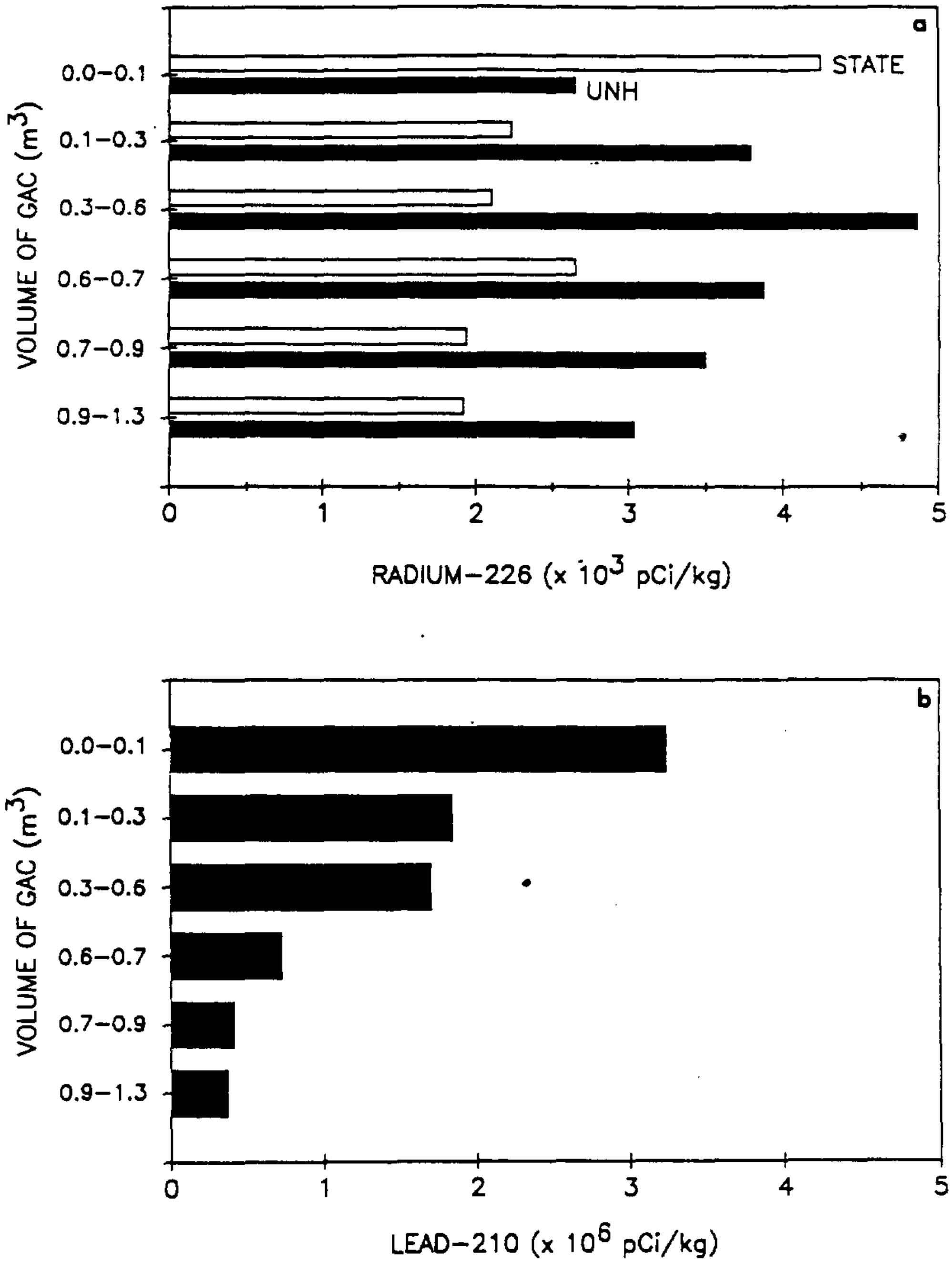


Figure 61. Coring event - Mont Vernon, NH GAC system. Radium-226 (a) and lead-210 (b) found in the GAC system.

TABLE 8. COMPARISON OF THEORETICAL LEAD-210* ADSORBED TO MONT VERNON LEAD-210 CORING DATA

Volume of GAC (m^3)	Total Radon-222 Removed (pCi/kg)+	Lead-210 Theoretical Adsorbed (pCi/kg)+	Lead-210 Coring Results (pCi/kg)+	Lead-210 Coring Results (μg/kg)
0.0 - 0.14	1.12×10^{10}	5.57×10^{6}	3.43×10^{6}	0.045
0.14 - 0.30	6.54×10^{9}	3.26×10^{6}	1.94×10^{6}	0.026
0.30 - 0.57	4.63×10^{9}	2.31×10^{6}	1.80×10^{6}	0.024
0.57 - 0.69	2.59×10^{9}	1.29×10^{6}	7.57×10^{5}	0.010
0.69 - 0.93	2.89×10^{9}	1.44×10^{6}	4.28×10^{5}	0.006
0.93 - 1.33	1.35×10^{9}	6.75×10^{5}	3.83×10^{5}	0.005

* See Appendix C for methods of calculation. The total theoretical lead-210 activity in GAC #1 = 8.24×10^{8} pCi and GAC #2 = 3.32×10^{8} pCi. The total measured lead-210 in GAC #1 = 5.46×10^{8} pCi and GAC #2 = 1.50×10^{8} pCi (based on coring results).

\+ pCi/kg dry weight of GAC, based on percent moisture data.

should be noted that the State of New Hampshire has stringent de minimus standards and that in many other states this material would not be considered to be a low level radioactive waste.

Amherst site--Two cores were taken of the Amherst GAC material. The microbial populations were not monitored because the unit had been out of service for approximately 6 months when the coring event occurred. Substantial amounts of manganese and iron (Figure 62) had accumulated. The data showed higher accumulations of these metals at the top of the filter with a decreasing trend through the unit, probably because these metals were entering the filter as precipitates due to oxidation in the atmospheric storage tank prior to the GAC. The accumulation of iron and manganese was greater than the accumulation at Mont Vernon considering the Amherst unit had operated 6 months less than the Mont Vernon GAC. This was expected as the raw water concentrations of iron and manganese and the flowrate were higher at Amherst.

The coring data for uranium 238 and 235 (Figure 63) and the Phase II uranium data (Figure 49) at Amherst both indicated that the GAC did not effectively adsorb uranium. In comparison to the coring data obtained at Mont Vernon, the Amherst GAC adsorbed very little uranium. The discrepancy may be explained by the different pH's at the two sites (Amherst = 8.03 ± 0.14; Mont Vernon = 6.5 ± 0.2). As previously discussed, uranium species are predominately anionic soluble carbonate complexes in natural waters from pH 7 to 10 (Sorg, 1988), while below pH ~ 6.8, the neutral UO_2CO_3 species predominates. Therefore, it is assumed that the poorly adsorbed anionic species predominated at Amherst whereas the favorably adsorbed neutral species predominated at Mont Vernon, which would account for the observed differences.

The radium-226 coring data (Figure 64) indicate that this radionuclide was retained by the GAC in a manner similar to that at Mont Vernon. A comparison with the theoretical radium-226 adsorbed could not be made because only one analysis was made during Phase II at Amherst.

The lead-210 coring data (Figure 64) again showed an exponential pattern similar to the radon removal observed during Phase II (Figure 37). The theoretical comparison (Table 9) yielded similar values considering variation in radon activity and flowrate were not known. The amount of lead-210 adsorbed was much less than that observed at Mont Vernon because the Amherst GAC unit was only operated for 4 months.

The Amherst GAC exceeded the State of New Hampshire de minimus levels for uranium-238 (58,410 pCi/kg; 2.5×10^{-5} Ci/m³) and radium-226 (44.39 pCi/kg; 1.9×10^{-8} Ci/m³) throughout the filter. As a result, all of the GAC at Amherst was classified as a low level radioactive waste. The uranium-235 concentrations were below de minimus standards (58,410 pCi/kg; 2.5×10^{-5} Ci/m³) and lead-210 is not regulated in New Hampshire.

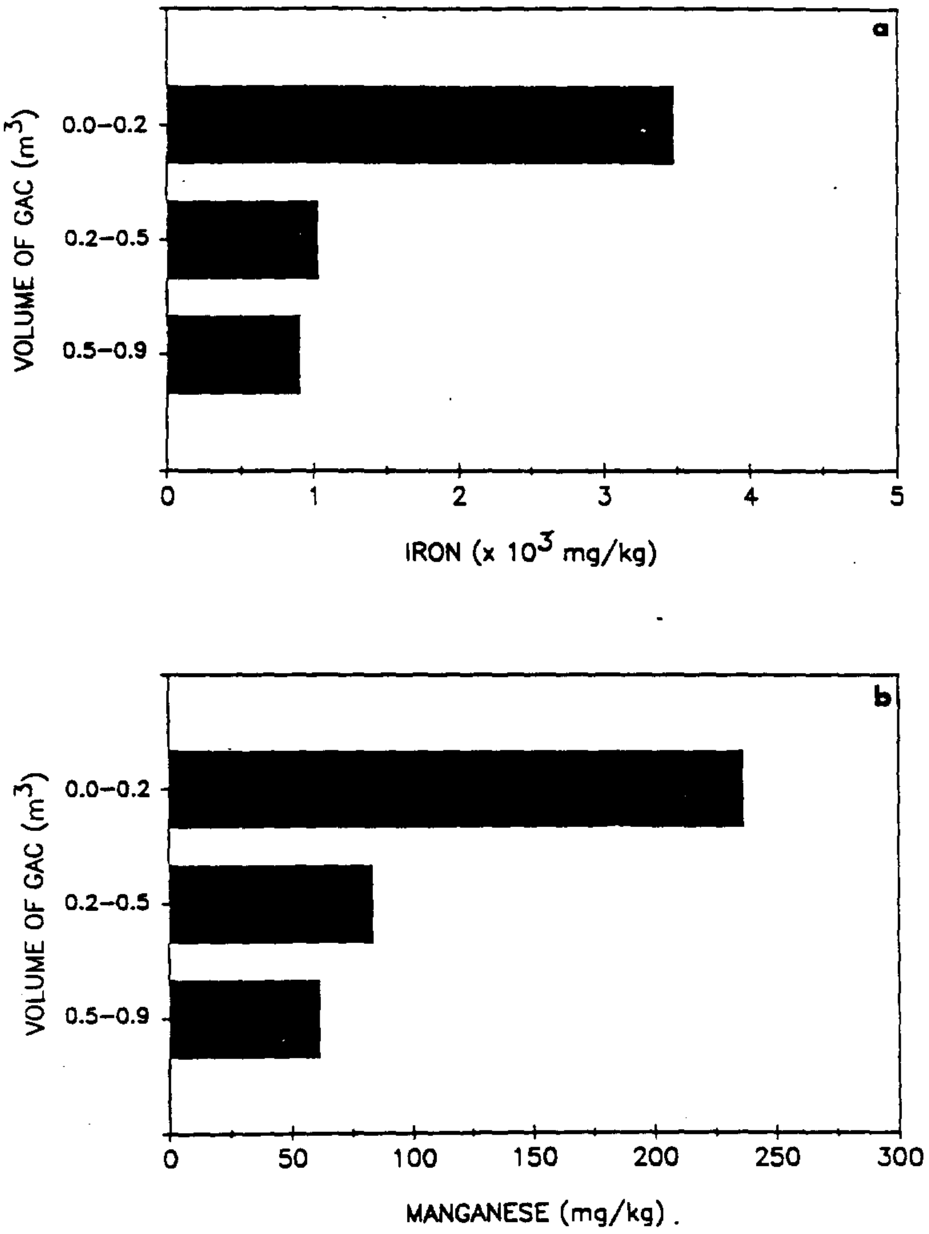


Figure 62. Coring event - Amherst, NH GAC system. Iron (a) and manganese (b) found in the GAC system.

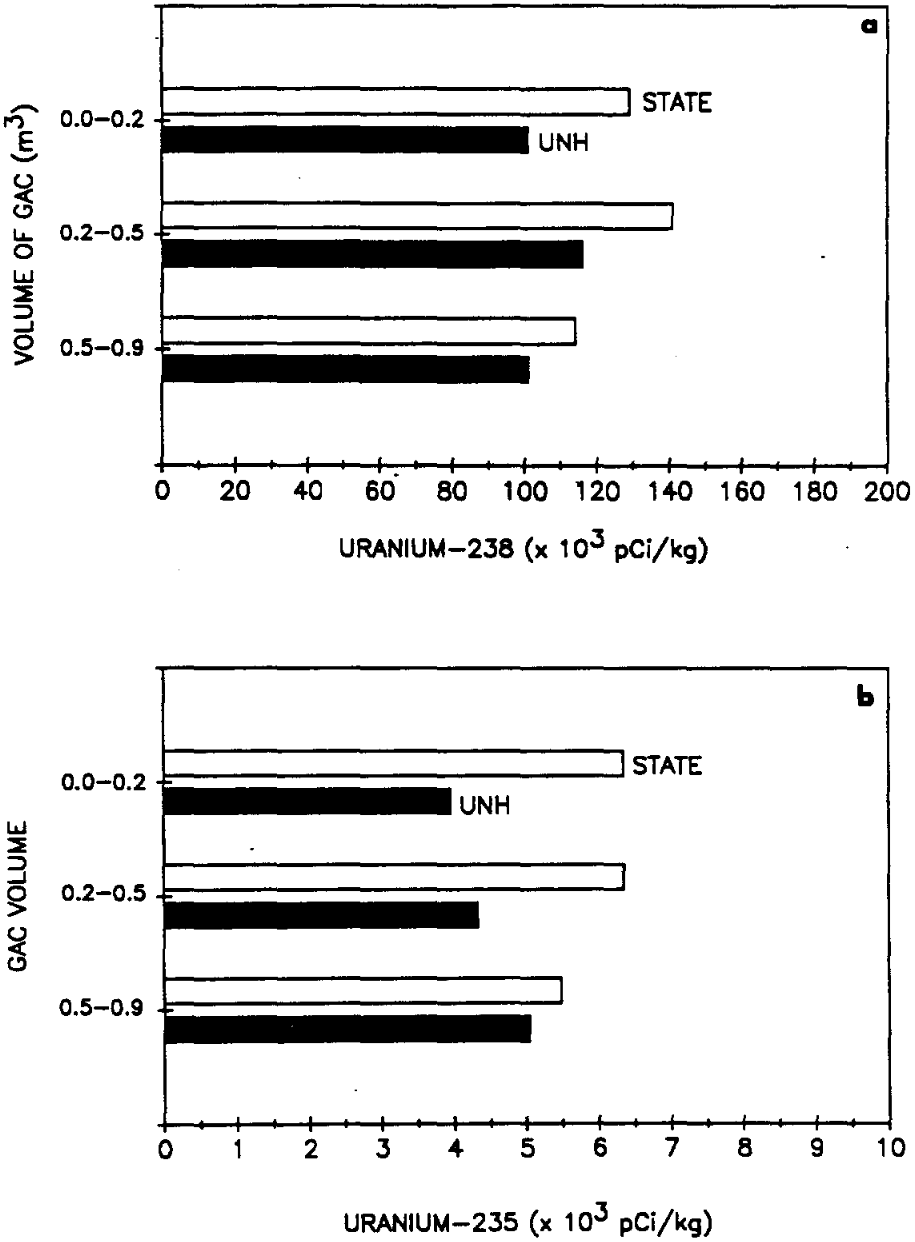


Figure 63. Coring event - Amherst, NH GAC system. Uranium-238 (a) and uranium-235 (b) found in the GAC system.

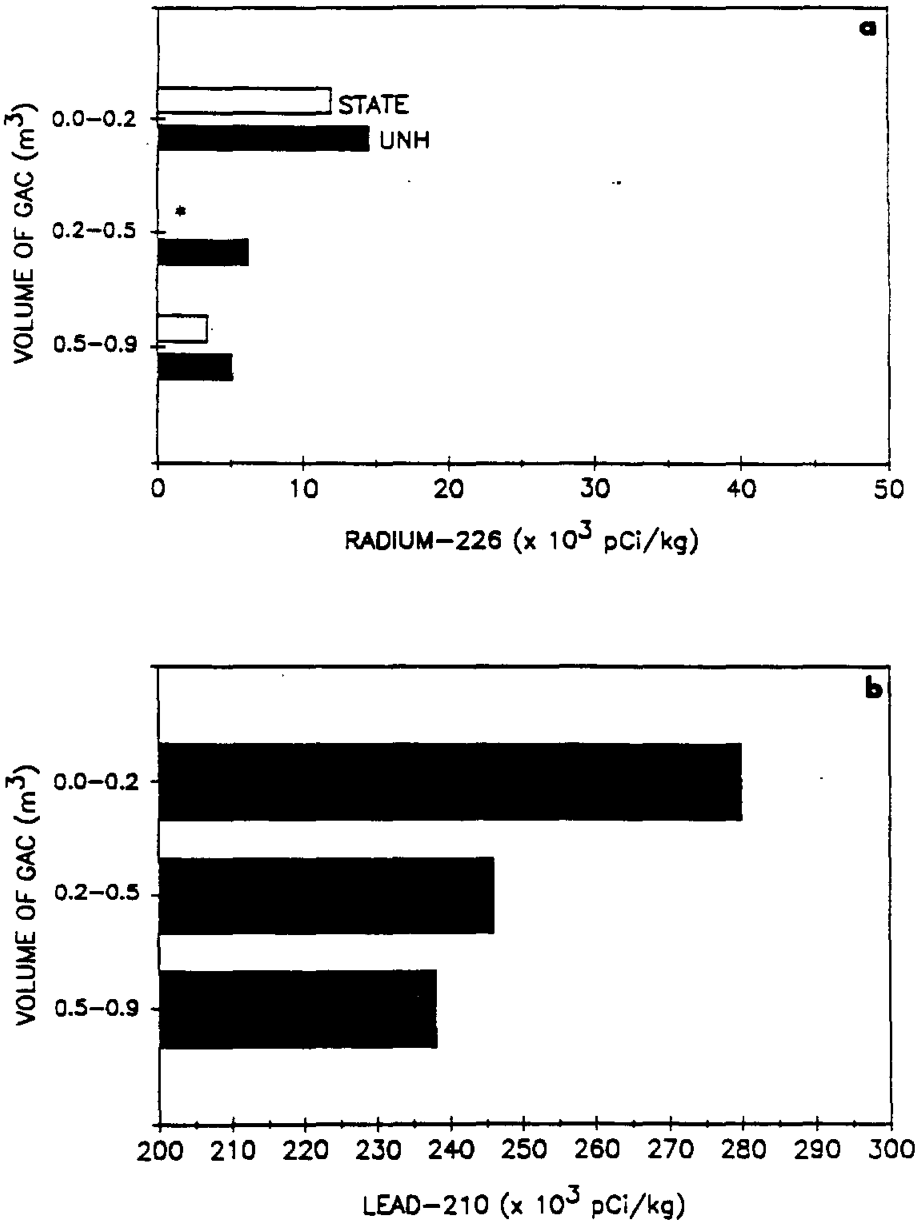


Figure 64. Coring event - Amherst, NH GAC system. Radium-226 (a) and lead-210 (b) found in the GAC system. (* Data from State laboratory unavailable.)

TABLE 9. COMPARISON OF THEORETICAL LEAD-210* ADSORBED TO AMHERST LEAD-210 CORING DATA

Volume of GAC (m^3)	Total Radon-222 Removed (pCi/kg)+	Lead-210 Theoretical Adsorbed (pCi/kg)+	Lead-210 Coring Results (pCi/kg)+	(μg/kg)
0 - 0.20	1.48×10^9	7.39×10^5	2.97×10^5	0.004
0.20 - 0.45	8.66×10^8	4.32×10^5	2.59×10^5	0.003
0.45 - 0.85	4.93×10^8	2.46×10^5	2.51×10^5	0.003

* See Appendix C for methods of calculation. The total theoretical lead-210 activity in GAC bed = 1.52×10^8 pCi. The total measured lead-210 in the GAC bed = 9.61×10^7 pCi (based on coring results).

+ pCi/kg dry weight of GAC, based on percent moisture data.

Backwashing--

The GAC filters at Mont Vernon were backwashed individually in January 1988 (Day 477) using the raw water. The original backwash rate proposed (12.2 m^3/m^2-hr) caused the GAC to start flowing out of the filter. The backwash rate was immediately decreased to approximately 55 m^3/day (5 m^3/m^2-hr for GAC #1 and 3.5 m^3/m^2-hr for GAC #2) which eliminated this problem. Each unit was backwashed for 10 min as originally proposed. Normally, backwashing flowrates for GAC range from 29-49 m^3/m^2-hr for 10 to 15 min (Montgomery, 1985), but these flows would have been too high to prevent loss of GAC from the Mont Vernon filter since it was filled near capacity (36-53 cm freeboard). Samples for pH, alkalinity, DO, turbidity, iron, manganese and microbial numbers were taken immediately before and 1.3 and 24 hr after backwashing (~1 and 18 empty bed volumes, respectively). Samples for radon activity were taken before and at 1.3, 2.6, 3.9, and 24 hr after backwashing (~1, 2, 3 and 18 empty bed volumes, respectively). Samples were also taken of the backwash water and those were analyzed for iron, manganese, uranium and radon.

The water exiting the filters during backwashing was initially very turbid, especially from GAC #1. After 10 min of backwashing, the water from GAC #2 was clear, but the GAC #1 water remained somewhat turbid. The backwash water from the first minute of operation contained 0.68 mg Fe/L and 0.14 mg Mn/L from GAC #2 and 129.5 mg Fe/L and 1.33 mg Mn/L from GAC #1. After 10 min of backwashing, the iron concentration in the backwash water from GAC #2 remained the same and the manganese was below detention. At the end of backwashing, the iron and manganese concentrations in the GAC #1 backwash water were 4.37 mg/L and 0.11 mg/L, respectively. The backwashing had a major impact on removing iron and manganese precipitates from GAC #1, where they appeared to be most concentrated. These data seem to contradict those obtained from the coring (Figure 58) which indicated most of the iron was in the top of GAC #2. It is possible that the new well at Mont Vernon, installed after coring and in operation 3.5 months before backwashing contained higher amounts of iron which precipitated in the top of GAC #1. Another plausible explanation for the results might be the difference between iron associated with the pore water (i.e., removed by backwashing) and that which is adsorbed to the GAC (i.e., measured by coring). The uranium (~500 pCi/L) and radon (~28,000 - 43,000 pCi/L) activities measured in the backwash water were similar to those in the raw water from the new well at Mont Vernon indicating that these radionuclides were not desorbing from the GAC.

There was no significant change in the pH (Figure 65), alkalinity (Figure 65) and dissolved oxygen (Figure 66) obtained before and after backwashing (α = 0.05 and 0.10, StT). The turbidity profile improved primarily in the top 0.22 m^3 of GAC #1 after 24 hr with a decrease in turbidity from approximately 5.0 NTU to 0.10 NTU, showing a positive effect of backwashing. The impact of backwashing on the iron and manganese profiles (Figure 67) was not as positive. The backwashing appeared to move iron and manganese precipitates to the top of GAC #1, but did not improve the concentrations of these metals over 24 hr. There was some change in the microbial numbers (Figure 68) in the top of the GAC #1 after backwashing, however, caution must be used when interpreting these data because of the variability in microbial counts normally obtained from GAC filters.

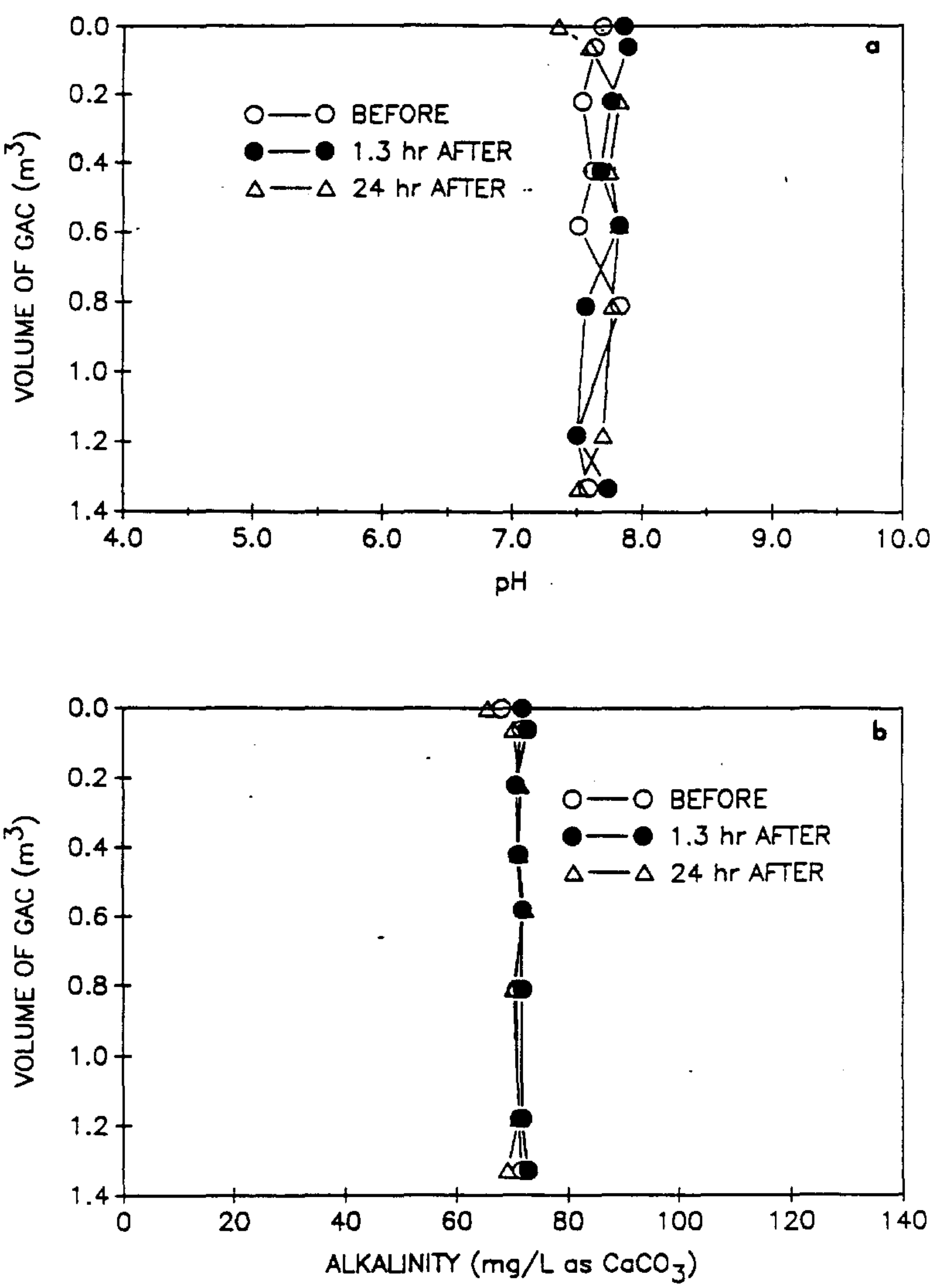


Figure 65. Backwash event - Mont Vernon, NH GAC system. pH (a) and alkalinity (b) profiles before and after backwashing.

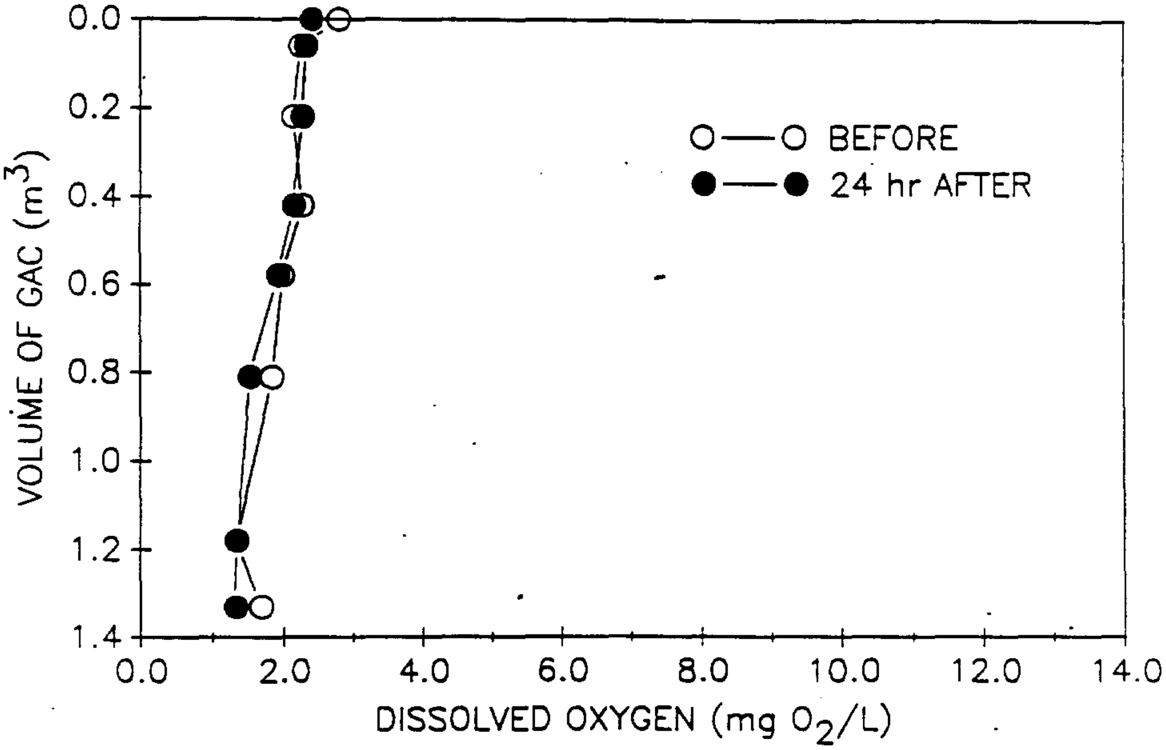


Figure 66. Backwash event - Mont Vernon, NH GAC system. Dissolved oxygen profile before and after backwashing.

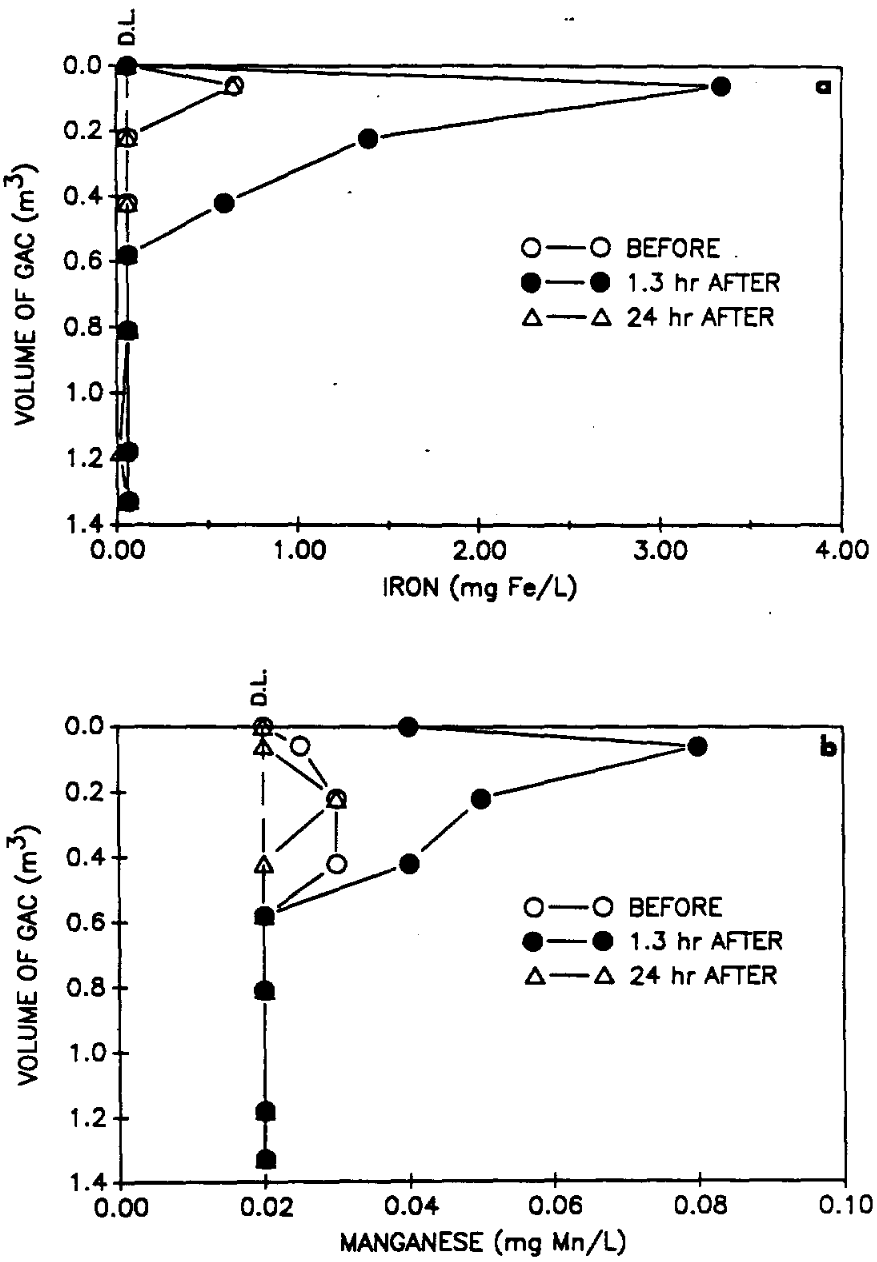


Figure 67. Backwash event - Mont Vernon, NH GAC system. Iron (a) and manganese (b) profiles before and after backwashing. (Detection limits: iron = 0.06 mg/L, manganese = 0.02 mg/L).

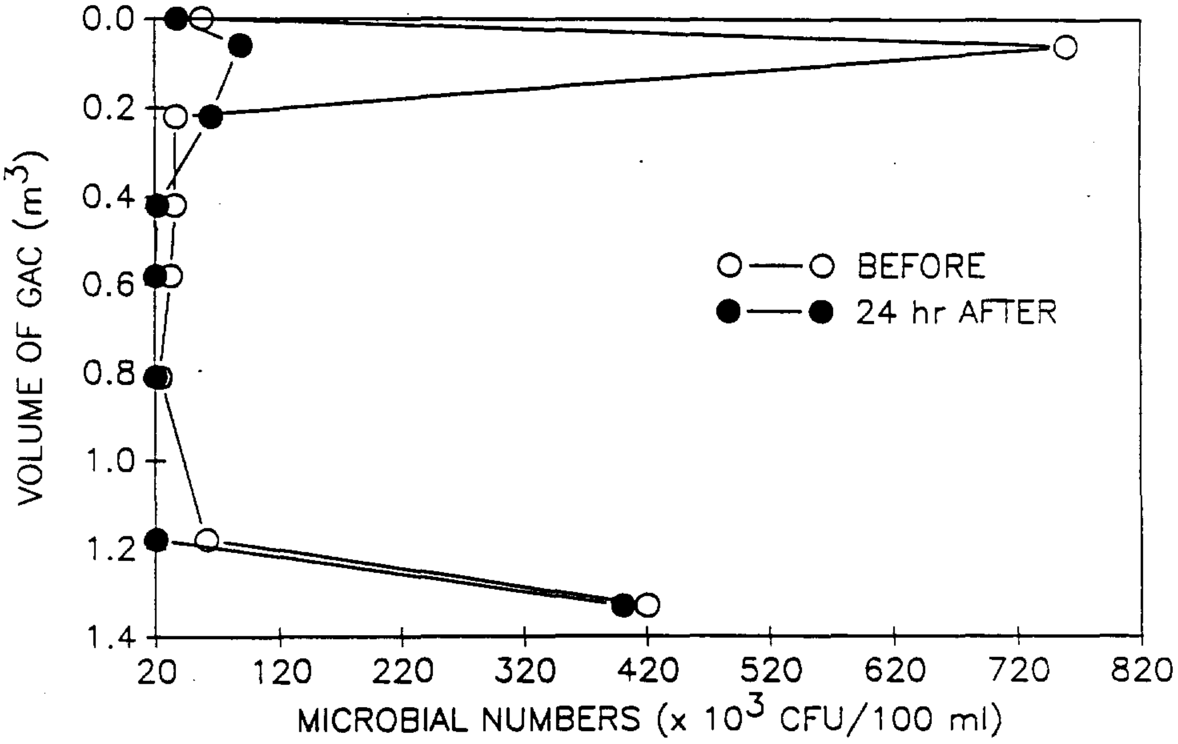


Figure 68. Backwash event - Mont Vernon, NH GAC system. Profile of microbial numbers before and after backwashing. (Detection limit = 20,000 CFU/100 mL).

The radon activity through the filter did not change as a result of the backwashing (Figure 69). There was no statistically significantly difference ($\alpha = 0.05$ and 0.10, StT) in the influent and effluent radon activities obtained before or after backwashing. In a study of point-of-entry units, Lowry and Brandow (1985) determined that frequent backwashing is detrimental to radon removal because of desorption of radon from the bottom of the bed for a brief period after the backwash event. EPA (1987) noted that backwashing is detrimental to GAC point-of-entry units because it mixes the carbon in the tank. This effect was not observed in this study, however, the units were larger, there was no testing after 24 hr, and this was a single backwash event using a low backwash flowrate.

A more thorough study of backwashing needs to be done, coordinated with GAC coring before and after the backwashing events. In addition, sampling should be conducted over a longer period of time before and after backwashing to better determine the impact of the process. A complete understanding of how backwashing affects radon removal is imperative because it appears that backwashing to remove contaminants such as iron and manganese is essential to insure long term filter operation with minimal headloss. Even with pretreatment, GAC filters are usually backwashed to remove microorganisms and usage of GAC for radon removal in natural waters would probably not be an exception to this practice. In conventional practice, GAC units are backwashed with treated water, however, in small community systems this would require additional water storage adding significantly to the cost.

ECONOMICS

A detailed economic evaluation of the technologies examined in this project, for removal of radon from small community water systems, has been performed using conventional engineering economic practices. The economic evaluation assumes all community water system components (e.g., wells, storage, pumping, distribution piping) are already in-place and only the costs related to the installation and operation of the radon removal technologies are presented. Costs have been divided into two major categories: capital costs and operation and maintenance (O&M) costs. Capital costs have been further divided to include direct costs and indirect costs. Direct costs include process equipment costs plus support equipment costs (e.g., installation). All direct costs were obtained from the records of actual expenditures made during this project, hence, they are site specific. All equipment and material costs are free on board (FOB), thus, shipping costs have not been included. The indirect costs were estimated as fixed percentages of the total direct costs (TDC). These fixed percentage rates were developed by the USEPA Office of Drinking Water (Cummins, 1987) and are presented solely to provide a basis for evaluating the economic feasibility of each technology and for comparing technologies. The indirect cost estimates include: sitework (15% of TDC), engineering costs (15% of TDC), contractor overhead and profit (12% of TDC), legal and financial fees (2.5% of TDC), interest during construction (6% of TDC), and contingencies (15% of TDC).

Operation and maintenance costs include power, labor, maintenance and administrative costs. Power costs were calculated based on the equipment horsepower, operating efficiency, operation period and electrical power cost.

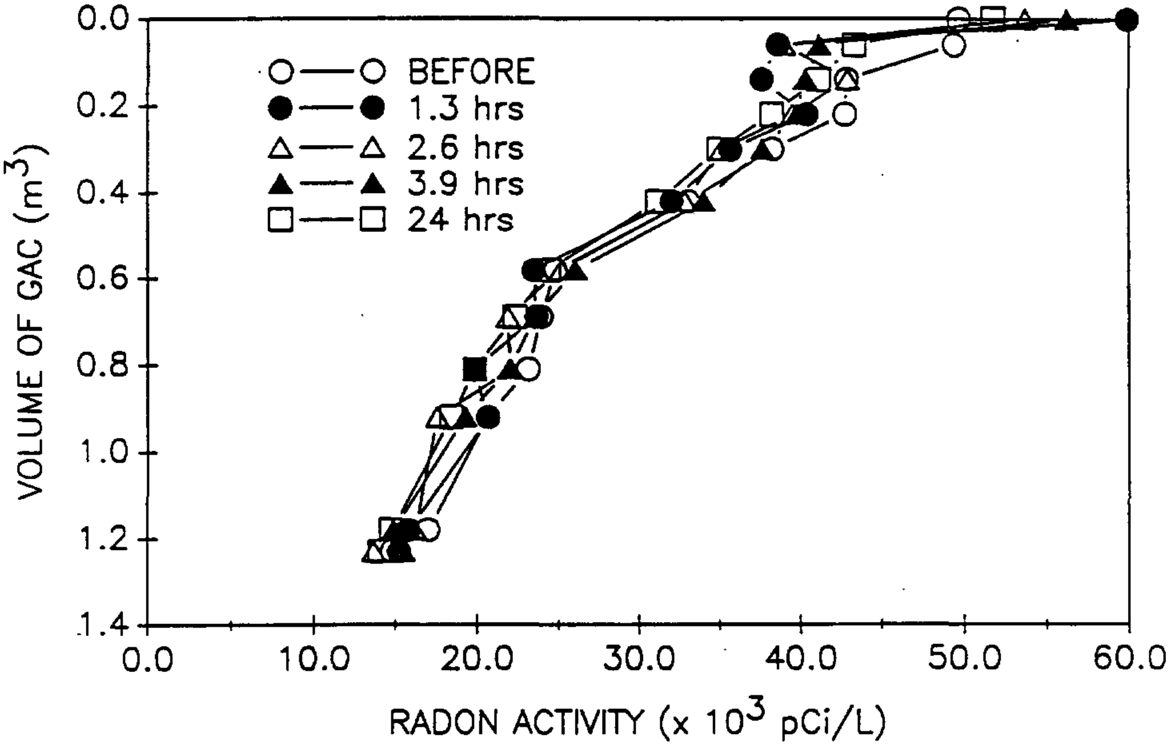


Figure 69. Backwashing event - Mont Vernon, NH GAC system. Radon activity profiles before and after backwashing.

An electrical power cost of $0.10157/kW-hr was used based on Public Service of New Hampshire (PSNH) Class G rate scale for a small community water supply with a demand of less than 5 kW (PSNH, 1989). Labor, equipment maintenance and administrative costs were calculated based on USEPA estimates (Cummins, 1987) as follows: a labor cost of $0.03/1,000 gallons was assumed; the annual equipment maintenance costs were estimated as the sum of 10% and 4% of the mechanical and non-mechanical process equipment capital costs, respectively; and the administrative costs were calculated as the sum of 20% and 25% of the labor and maintenance costs, respectively.

For comparative purposes, annual costs and production costs were developed for each treatment system. These costs were expressed in conventional units to be consistent with those reported in other studies. The annual cost was computed as the sum of the amortized capital cost and the O&M cost. The amortized capital cost is the total capital cost amortized over a 20 year time period at a 9% interest rate. A 20 year time period was chosen since it reflects a typical engineering design life for water treatment systems (Montgomery, 1985). The interest rate of 9% reflects the current (June 1989) U.S. Treasury Bill (T-Bill) interest rate. Production costs ($/1,000 gallons) were calculated by dividing the annual cost by the volume of water treated per year assuming the system was operated at the design flow.

Based on the findings of this study, that iron and manganese have the potential to foul the radon removal technologies, an additional cost for pretreatment to remove iron and manganese was estimated and used to compute the resulting production cost of radon removal with pretreatment. For comparative purposes, it was assumed that iron and manganese removal would be achieved via cation exchange. The production cost of this cation exchange system is estimated for design flows up to 38 m^3/day to be $1.50/1,000 gallons based on quotations from leading ion exchange system suppliers. This cost estimate is comparable to those reported by Reid, Lassovszky and Hathaway (1985) for use of cation exchange to remove radioactivity from drinking water. It should be noted that the cost of ion exchange for removal of iron and manganese does not include disposal of the resin regenerant or the spent ion exchange cartridges (used for non-regenerated systems). These costs may be substantial since the regenerant stream or spent cartridges are likely to contain accumulated radionuclides (e.g., uranium and radium) and require special handling and disposal. However, there is insufficient data to provide a sound basis for estimating the costs of handling and disposal of regenerant or spent cartridges. Further, it should be noted that the selection of the most cost effective pretreatment technique for iron and manganese removal is site specific and may not always involve the use of ion exchange.

All cost figures were updated to second quarter 1989 dollars using the ENR Construction Cost Index (CCI). The 1967 base year CCI index for second quarter 1989 is 426.

The cost estimates presented in this report are intended to give a general indication of the economics of the radon removal technologies studied. Specific treatment requirements will vary from site to site and may not be accurately reflected by the systems or assumptions used in this economic analysis. Further, any plans for selection, design and operation of

TABLE 10. COST ESTIMATE FOR GAC TREATMENT AT MONT VERNON, NH

Item	Total Cost (2nd. Quarter 1989 dollars)*
CAPITAL COSTS	
DIRECT COSTS	
Process Equipment	
1 - 91 cm diameter fiberglass pressure vessel	$ 1,301
1 - 76 cm diameter fiberglass pressure vessel	905
1.33 m^3 of Barnebey Cheney Type 1002 GAC ($2091/$m^3$)	2,782
2 - PVC Distributor Systems ($98/system)	196
1 - Rockwell flow rate recorder/ totalizer	1,036
1 - Brooks Rotometer	443
1 - Differential Pressure Meter+	600
Total Process Equipment Cost	$ 7,263
Support Equipment	
Installation	195
Plumbing	1,545
Electrical	98
Total Support Equipment Cost	$ 1,838
TOTAL DIRECT COST	$ 9,101
INDIRECT COSTS	
Sitework	$ 1,365
Engineering	1,365
Contractor Overhead and Profit	1,092
Legal and Financial Fees	228
Interest	546
Contingencies	$ 1,365
TOTAL INDIRECT COST	$ 5,961
TOTAL CAPITAL COST	$15,062
AMORTIZED CAPITAL COST (Annual)	$ 1,650

(continued)

TABLE 10 (continued)

Item	Total Cost (2nd Quarter 1989 dollars)*
ANNUAL O&M COSTS	
Power	NA**
Maintenance	
Mechanical	208
Non-mechanical	207
Labor	71
Administrative	118
TOTAL ANNUAL O&M COST	$ 604
TOTAL ANNUAL COST	$ 2,254/year
TOTAL PRODUCTION COST (Design flow 25 m^3/day)	$0.95/1,000 gallons
PRETREATMENT COST	$1.50/1,000 gallons
TOTAL PRODUCTION COST WITH PRETREATMENT	$2.45/1,000 gallons
GAC DISPOSAL COST++	$0.19/1,000 gallons
TOTAL PRODUCTION COST WITH GAC DISPOSAL	$1.14/1,000 gallons
TOTAL PRODUCTION COST WITH PRETREATMENT AND GAC DISPOSAL	$2.64/1,000 gallons

[*] Based on ENR Construction Cost Index, Base Year 1967 = 100.

[+] Pressure meters were not used during the research, but should be required for process control in full scale facilities.

[**] Additional power costs incurred by operation of the GAC system are considered negligible.

[++] Based on estimated disposal cost of $335/$m^3$-yr (assuming it is classified and regulated as a low level radioactive waste) and the design flow.

full-scale systems should be based on large scale, on-site pilot data reviewed by design professionals familiar with the specific site requirements.

The cost estimates for the GAC system installed in Amherst, NH are shown in Table 10. Based on the design flow of 34 m^3/day the production cost for GAC treatment at Amherst was $0.56/1,000 gallons. The production costs for this GAC system with pretreatment for iron and manganese and/or GAC disposal were also computed. The total production cost with pretreatment and GAC disposal for Amherst, NH is estimated to be $2.15/1,000 gallons.

The total annual O&M cost estimate for Amherst was $551. It should be noted that the O&M cost estimate assumes that power costs are negligible. This assumption is valid providing that the cost of the system backwashing is neglected. It is estimated that to backwash the GAC system at Amherst using a hydraulic loading rate of 4.9 m^3 H_2O/m^2-hr for a 15 min period would have a power cost of $0.09/backwashing event. Therefore, if the backwashing frequency is less often than once per day, which the data in this study indicates it should be, the cost of backwashing at Amherst would be less than $0.01/1,000 gallons.

The costs of GAC disposal, assuming it must be handled as a regulated low level radioactive waste, were computed based on an estimate (in 1989 dollars) of $4238/$m^3$ of GAC obtained from General Dynamics Services Company, Groton, CT. Based on the work of Lowry and Brandow (1985), it is assumed that with proper pretreatment and backwashing, the GAC would not require replacement or regeneration. Therefore, this disposal cost would be incurred at the end of the 20 year design life. Thus, the disposal cost was adjusted to a future cost using the ratio of ENR CCI over the previous 20 year period (1969 vs. 1989). This resulted in an estimated future cost of $17,163/$m^3$ of GAC or $14,580 for disposal of the 0.85 m^3 of GAC used in Amherst. The future cost was then translated to an annual cost assuming 9% interest over 20 years. The resulting annual cost was $335/$m^3$/yr or $285/yr. This annual cost represents an additional production cost of $0.09/1,000 gallons at Amherst.

Cost estimates for the GAC facility at Mont Vernon, NH are presented in Table 11. The costs of GAC treatment at Mont Vernon were estimated in the same manner as described for the Amherst facility. The total production cost including pretreatment and GAC disposal at Mont Vernon was estimated to be $2.64/1,000 gallons. The major differences in cost between Amherst and Mont Vernon relate to the additional equipment (i.e., pressure vessel and GAC) required to treat the higher radon loading at Mont Vernon. In particular, the additional capital and disposal costs of the GAC at Mont Vernon account for 95% of the increased costs.

The cost estimates presented for GAC treatment of radon containing well water at Amherst and Mont Vernon, NH should be interpreted with the following stipulations. (i) The costs of using GAC will be highly dependent on the influent flow and radon activity as well as the treated water radon standard which must be attained. The costs presented for Amherst are based on system design specifications of 84% radon removal at a flow of 34 m^3 and a 39,750 pCi/L average influent radon activity (0.85 m^3 GAC). Mont Vernon was designed

TABLE 11. COST ESTIMATE FOR GAC TREATMENT AT AMHERST, NH

Item	Total Cost (2nd Quarter 1989 dollars)*
CAPITAL COSTS	
DIRECT COSTS	
Process Equipment	
1 - 91 cm diameter fiberglass pressure vessel	$ 1,301
0.85 m^3 of Barnebey Cheney Type 1002 GAC ($2091/$m^3$)	1,776
1 - PVC Distributor System	98
1 - Rockwell flow rate recorder/ totalizer	1,036
1 - Brooks Rotometer	543
1 - Differential Pressure Meter+	600
Total Process Equipment Cost	$ 5,354
Support Equipment	
Installation	215
Plumbing	1,611
Electrical	111
Total Support Equipment Cost	$ 1,937
TOTAL DIRECT COST	$ 7,291
INDIRECT COSTS	
Sitework	$ 1,094
Engineering	1,094
Contractor Overhead and Profit	875
Legal and Financial Fees	182
Interest	437
Contingencies	$ 1,094
TOTAL INDIRECT COST	$ 4,776
TOTAL CAPITAL COST	$12,067
AMORTIZED CAPITAL COST (Annual)	$ 1,322

(continued)

TABLE 11 (continued)

Item	Total Cost (2nd Quarter 1989 dollars)[*]
ANNUAL O&M COSTS	
Power	NA[**]
Maintenance	
Mechanical	218
Non-mechanical	127
Labor	100
Administrative	106
TOTAL ANNUAL O&M COST	$ 551
TOTAL ANNUAL COST	$ 1,873/year
TOTAL PRODUCTION COST (Design flow 34 m^3/day)	$0.56/1,000 gallons
PRETREATMENT COST	$1.50/1,000 gallons
TOTAL PRODUCTION COST WITH PRETREATMENT	$2.06/1,000 gallons
GAC DISPOSAL COST[++]	$0.09/1,000 gallons
TOTAL PRODUCTION COST WITH GAC DISPOSAL	$0.65/1,000 gallons
TOTAL PRODUCTION COST WITH PRETREATMENT AND GAC DISPOSAL	$2.15/1,000 gallons

[*] Based on ENR Construction Cost Index, Base Year 1967 = 100.

[+] Pressure meters were not used during the research, but should be required for process control in full scale facilities.

[**] Additional power costs incurred by operation of the GAC system are considered negligible.

[++] Based on estimated disposal cost of $335/$m^3$-yr (assuming it is classified and regulated as a low level radioactive waste) and the design flow.

SECTION 5

DIFFUSED BUBBLE AERATION

INTRODUCTION

Diffused bubble aeration has been used effectively to remove many volatile contaminants from drinking water. It is easy to operate and the equipment can be retrofit into existing tanks (as long as sidewall depth effects are considered). Unlike the GAC system, it does not concentrate radon and its progeny and as a result is not a source of gamma emissions. Unlike the packed tower, it does not require a large tower structure to be built. However, in contrast to packed tower aeration, diffused bubble aeration requires contact times of several minutes or more and higher volumetric A:W ratios to achieve equivalent removals. The disadvantages most often associated with diffused bubble aeration include: high costs of operation (aeration system), release of radon to the atmosphere, potential for oxidation of iron, the need for additional pumping or repressurization of the water and excessive noise of the air blower.

Diffused bubble treatment for radon removal has been studied previously for household (Lowry and Brandow, 1981) and small community water supplies (Perkins and Hewett, 1988). The studies using small community supplies (Perkins and Hewett, 1988) were limited to an average influent radon activity of about 16,560 pCi/L and an average water flow of 0.11 m^3 H_2O/min with a volumetric A:W ratio = 10:1. For the UNH study, the average influent radon activity was 71,937 ± 8,766 pCi/L and two water flow ranges were monitored: 0.045-0.047 m^3 H_2O/min and 0.102-0.106 m^3/min at volumetric A:W ratios = 2:1, 3:1, 5:1, 7:1, 10.5:1, 15:1, and 20:1. These A:W ratios and water flowrates were chosen to determine the effect of air and water flowrates on radon removal efficiency. Costs, the effects of water quality, such as iron and turbidity, and stack gas emissions were also considered during the diffused bubble study.

METHODS AND MATERIALS

Description of Units

The diffused bubble aeration system, designed by Dr. Jerry Lowry (Unity, ME), consisted of 3 polyethylene tanks aligned in series, each with a holding capacity of 1022 L (Figure 70). The system was installed in a small community water supply in Derry, NH. An insulated, plywood building protected the system from adverse weather and vandalism. Aeration was provided by an air blower with a 2.66 m^3/min capacity which forced ambient outdoor air into the tanks via Schedule 40, 3.8 cm diameter PVC piping connected to the diffusers

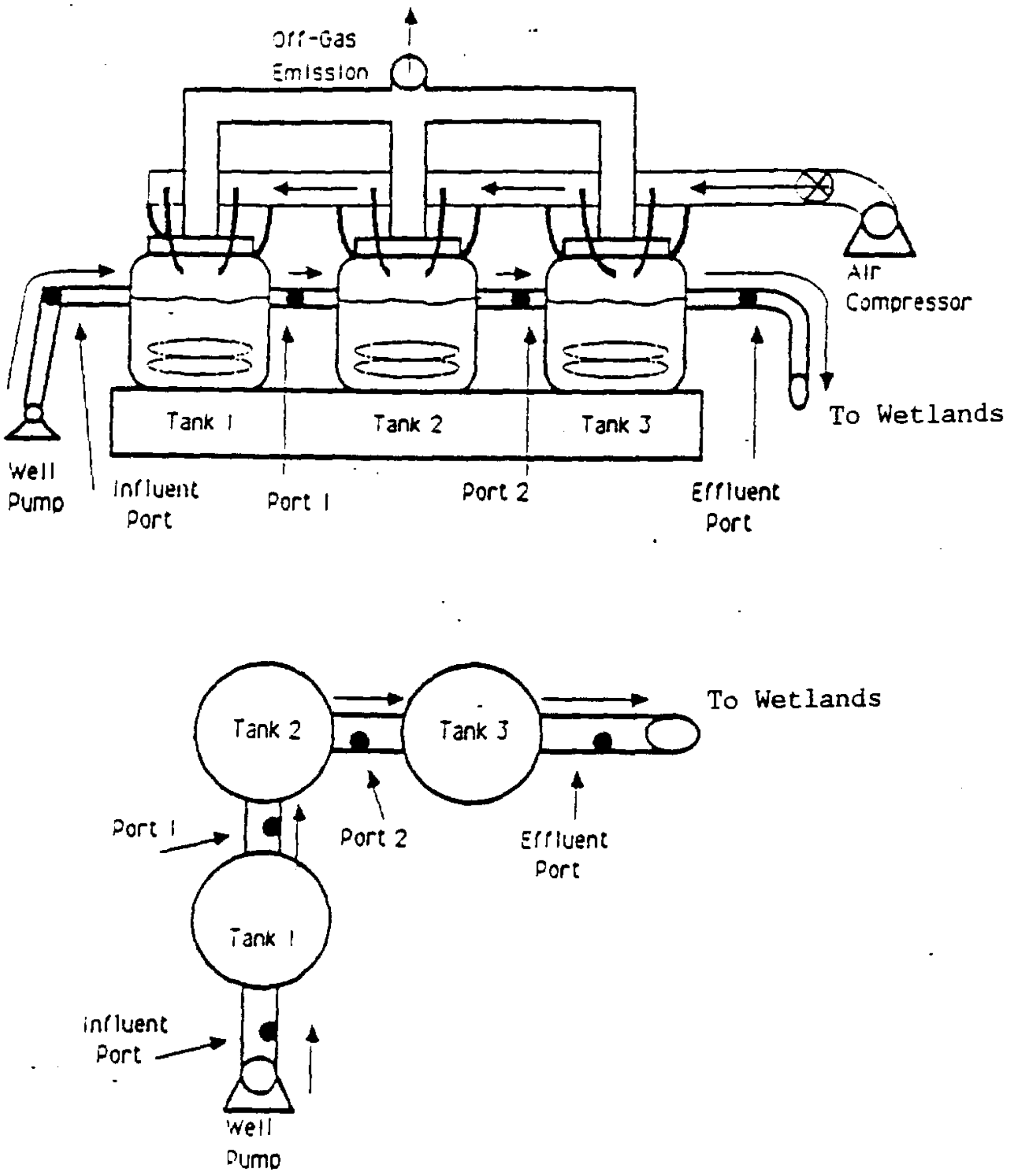


Figure 70. Schematic of the diffused bubble aeration system. Derry, NH.

which consisted of 1.90 cm diameter coiled plastic tubes with 0.038 cm holes drilled in their underside. The distance between the holes ranged from 0.5 to 1.6 cm. Air flow was regulated with a globe valve located on the influent air line. The diffusers were located about 79 cm below the water's surface and 36 cm from the bottom of each tank. Once the radon was stripped from the water, the contaminated air flowed from the top of each tank through Schedule 40 3.18 cm PVC piping and was vented outside of the building.

Water was pumped directly from the bedrock wells into the system through 3.18 cm plastic hose. The water from each upstream tank overflowed into the adjacent downstream tank by simple gravity displacement. Since the water supply was not being used by the community, treated water bypassed the atmospheric storage tank and flowed from the third tank by gravity through a Schedule 40, 5.1 cm PVC pipe to a wetland area outside of the pumphouse. The first two tanks maintained higher water volumes, of about 38 liters due to the expansion of the water column by the air.

An influent water sampling port was located in the influent line to the first tank (Figure 70). The 5.1 cm PVC connections between Tanks 1 and 2 and Tanks 2 and 3 were tapped using ball valve faucets with 1.1 cm diameter hose connections to provide sampling ports. The hoses attached to the ports each had the neck of a polyethylene funnel (10 - 15 cm maximum opening) fitted into their free end. During sampling, the mouth of the funnel was upright so that the water from the port filled it and flowed over its lip. In this way a radon sample could easily be collected from the funnel with little disturbance at a low water flowrate. An effluent sampling port was similarly constructed and tapped into the 5.1 cm PVC pipe carrying effluent water from Tank 3 to the wetland. The PVC connections between tanks were located 114 cm above the bottom of each tank.

Experimental Design

The system allowed comparison of the removal efficiencies over a wide range of influent radon activities (Tank 1: 60,843 to 86,355 pCi/L radon, Tank 2: 10,096 to 80,271 pCi/L and Tank 3: 1,767 to 74,112 pCi/L). Two flow ranges were obtained by manually operating one (low flow = 0.047 ± 0.00053 m^3 H_2O/min) or two wells (high flow = 0.10 ± 0.0019 m^3/min). Radon activities averaged 65,487 ± 5,657 pCi/L during high water flow and 78,385 ± 6,120 pCi/L during low water flow. The two water flowrates and the two radon activities resulted in applied radon loading rates averaging 6,819 ± 548 nanoCuries (10^3 pCi) per minute (nCi/min) for high water flow and 3,639 ± 295 nCi/min for low water flow. A:W ratios of 2:1, 3:1, 5:1, 7:1, 10.5:1, 15:1, and 20:1 were tested for both water flowrates. The tanks were drained at the end of each run and refilled with raw water immediately before the start of a new run.

Sampling Plan

The parameters monitored for the diffused bubble aeration system are shown in Table 12. Samples were collected at 0, 15, 30, 60 and 90 min at both water flowrates and also at 120 and 150 min for the low flowrate. Water flowrate was monitored using an in-line water flowmeter/totalizer located in

TABLE 12. DIFFUSED BUBBLE AERATION SAMPLING PARAMETERS

	Radon	Alkalinity	Turbidity	Temperature	Iron/Mang.*	Flowrate
Water Samples						
– Influent	X	X	X	X	X	X
– Port 1	X					
– Port 2	X					
– Effluent	X	X	X	X	X	X
Air Samples						
– Influent						X
– Building				X		
– Effluent	X					

*Iron and manganese.

the influent piping directly before the diffused bubble system. Air flowrate was monitored using a pitot tube installed in the influent air pipe and connected to a Dwyer^R Magnehelic pressure difference gauge. Air samples for radon analysis were taken before each run (background) and approximately 1 hr into the run. A 0.97 cm diameter plastic tube, which was placed 20 cm inside the off-gas (air effluent) pipe, was used to pump air samples to a Pylon AB-5 monitor.

RESULTS AND DISCUSSION

Radon

Overall System Performance--

Figures 71 and 72 show percent removals of applied radon over time for each run (A:W ratio and high or low water flowrate combination). Percent removals were calculated by dividing the mass of radon removed per time (i.e., applied radon loading (Q_w x C_{inf}) - radon remaining (Q_w x C_{eff})) by the applied radon mass loading (Q_w x C_{inf}) and multiplying by 100. (This is equivalent to $(C_{inf} - C_{eff})/C_{inf}$ because Q_w factors out of the equation.) A mean influent radon activity was computed for each run and used in the percent removal calculation. This approach is valid since there was <15% variation in influent activity during a run.

ANOVA analyses were run on the data to determine when the system was at steady state for each run. This determination was necessary because the three tanks in the diffused bubble system were filled with raw water at the beginning of a run. Steady state was defined as the minimum time after which effluent radon activities at a given condition (A:W ratio and Q_w) were not significantly different at $\alpha = 0.05$. However, true steady-state conditions could not be achieved because there were changes in the raw water quality and water flowrate from the wells over time. Table 13 shows the time required to reach steady state removal at all conditions tested. The times required for the high water flowrate conditions were shorter than those for the lower flowrate. For all subsequent statistical analysis of the diffused bubble system, different conditions were compared using data collected at 60 and 90 min for high flow and 90, 120 and 150 min for low flow. These times were chosen because for all A:W ratios tested the radon removals were at steady state.

For a diffused bubble system of a given construction, the percent removal of the radon applied is controlled by (i) the contact time between the contaminated water and diffused air (the hydraulic detention time), (ii) the aeration system's K_La (which is temperature dependent), (iii) the concentration of the contaminant in the air, and (iv) the influent concentration of the contaminant in the water.

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*Q_w = water flowrate (volume/time)

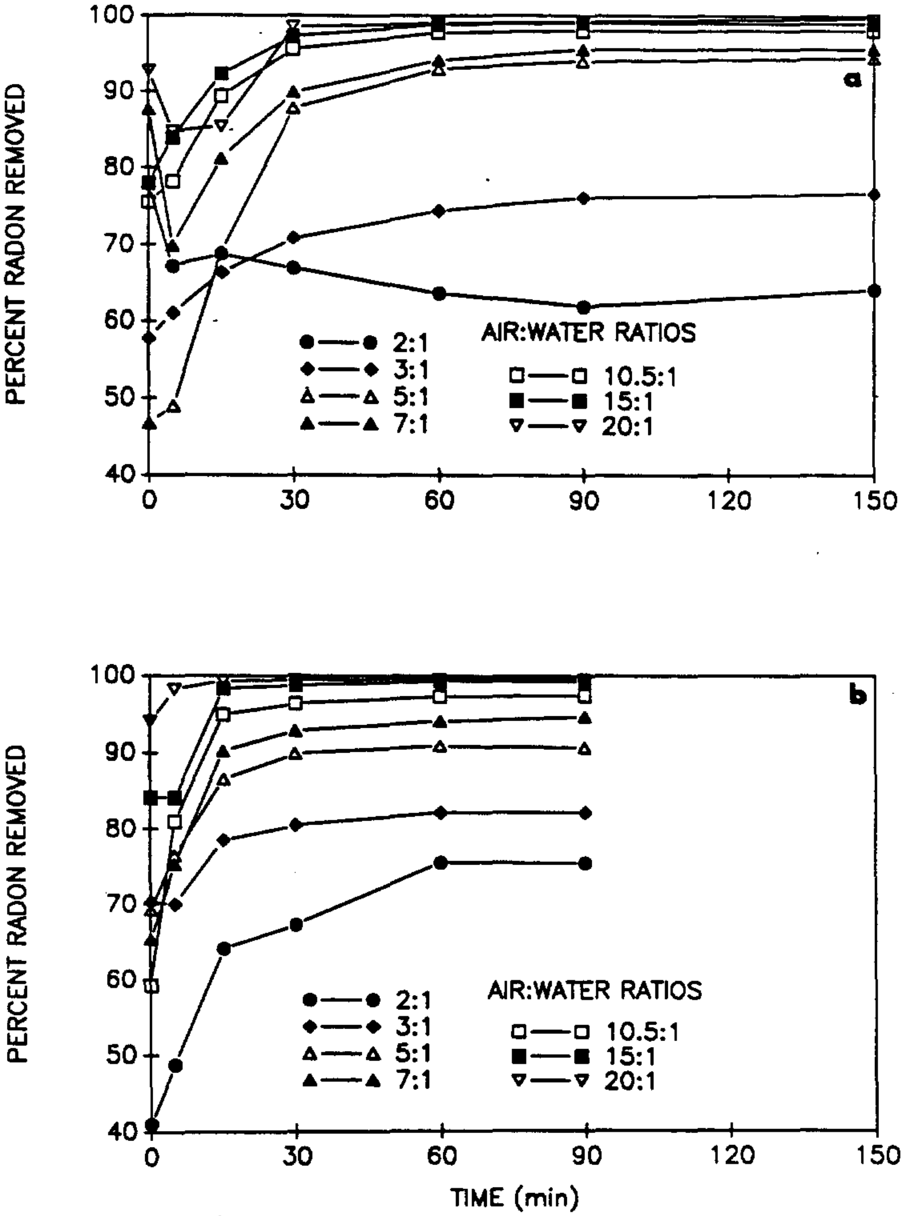

Figure 71. Diffused bubble aeration - Derry, NH. Percent radon removal as a function of time for varying A:W ratios.

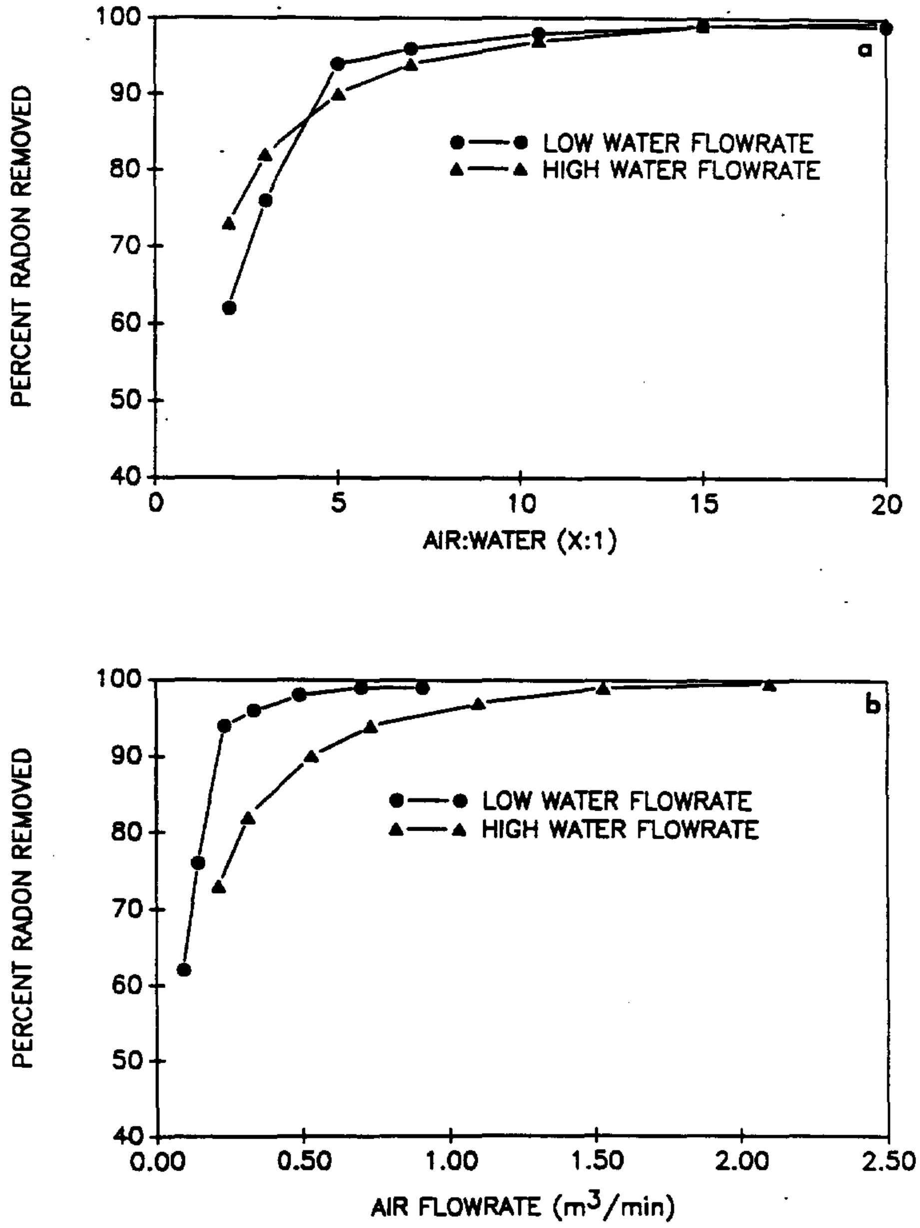

Figure 72. Diffused bubble aeration - Derry, NH. Percent radon removal as a function of A:W ratios (a) and air flowrate (b) for the low and high water flowrate.

TABLE 13. TIME REQUIRED TO ACHIEVE STEADY STATE OPERATION* DURING THE DIFFUSED BUBBLE EVALUATIONS

	Time (min)	
A:W Ratio	Low Water Flowrate	High Water Flowrate
2:1	60	60
3:1	90	60
5:1	90	60
7:1	90	60+
10.5:1	60	60
15:1	30	15
20:1	30	15

* Minimum time after which there was no significant difference in effluent quality achieved in diffused bubble system, $\alpha = 0.05$ (ANOVA).

+Not significantly different at $\alpha = 0.04$, significant different at $\alpha = 0.05$.

Figure 71 shows the overall percent removals of radon applied vs. A:W ratio for the high and low water flowrates using data after steady state conditions were achieved. As the A:W ratio increased from 2:1 to 5:1 for each flowrate, there was a sharp increase in radon removal. However, above 5:1, there was much less improvement in efficiency with large increases in A:W ratio.

When operating at A:W ratios of 5:1 and greater (at both high and low water flowrates), the overall radon removal efficiency ranged from 90.0% to >99.6%; the greatest efficiency was obtained at A:W ratios of 15:1 and 20:1. At A:W ratios of 10.5:1 and greater for the low flowrate and 15:1 and greater for the high flowrate, there was no significant difference in removal efficiency ($\alpha = 0.05$ and 0.01, ANOVA). Hence, for the diffused bubble system tested at the given conditions of radon loading, the lowest A:W ratios to yield (statistically significant) maximum radon removal for low and high water flowrates were in the range 7:1 to 10.5:1 and 10.5:1 to 15:1, respectively. From the practical viewpoint of designing the diffused bubble system, the A:W ratio should be based on the most cost effective blower size and mode of operation. Given the design water flowrate of 37.85 m^3/day and assuming a minimum A:W of 20:1 (0.53 m^3/min) with an estimated static pressure of 38 cm-H_2O, the most cost effective (capital and power rating) input horsepower size for the blower would be 0.5 hp (R.G. Wilson; Aladdin HVAC, Corp; personal communication; 1989). Using this blower represents a conservative engineering design (i.e., a 1.13 m^3/min air flow is delivered) since the resulting A:W ratio would be 43:1. The data suggests such a high A:W ratio would achieve removals of $\geq$99.6% with an influent averaging 72,000 pCi/L.

At the lowest A:W ratio (2:1) at both water flowrates, the low air flow (<0.23 m^3/min) severely limited bubble distribution in the tanks, resulting in lower radon removal (Figure 73). At air flowrates of <1.0 m^3/min, the percent removal achieved with the low water flowrate was significantly greater than the higher water flowrate in spite of the significantly higher radon activities entering the tank during the low flow runs (High $\equiv$ 65,487 ± 5,657 pCi/L vs. Low $\equiv$ 78,385 ± 6,120 pCi/L) ($\alpha = 0.05$ and 0.01, StT). This difference may have been a function of the longer detention time in the units at the low water flowrate condition. The overall mass removal of radon (pCi/min) was, however, considerably greater at the higher water flowrates for comparable air flowrates (Figure 73). These results reflect the greater mass loading ($Q_w \times C_{inf}$) of radon to the units which negated the effect of the decreased detention time by increasing the rate of mass transfer (i.e., higher concentration gradient $C_{liquid} - C_{gas}$) particularly in Tanks 2 and 3.

Previous studies of diffused bubble aeration performance on removal of radon are comparable to those obtained with the system at Derry. Perkins and Hewett (1988) found that a diffused bubble system employing 61 cm diameter stainless steel bowl diffusers containing fritted plastic diffuser media could achieve percent removals of radon activities of 95% and greater. Their system was run at an A:W = 10:1, and a 0.23 m^3 H_2O/min water flow, with a hydraulic detention time of 60 min and a raw water radon activity of 16,560 pCi/L. Their setup consisted of 2 manholes, 2.4 m in diameter and 2.4 m deep so that

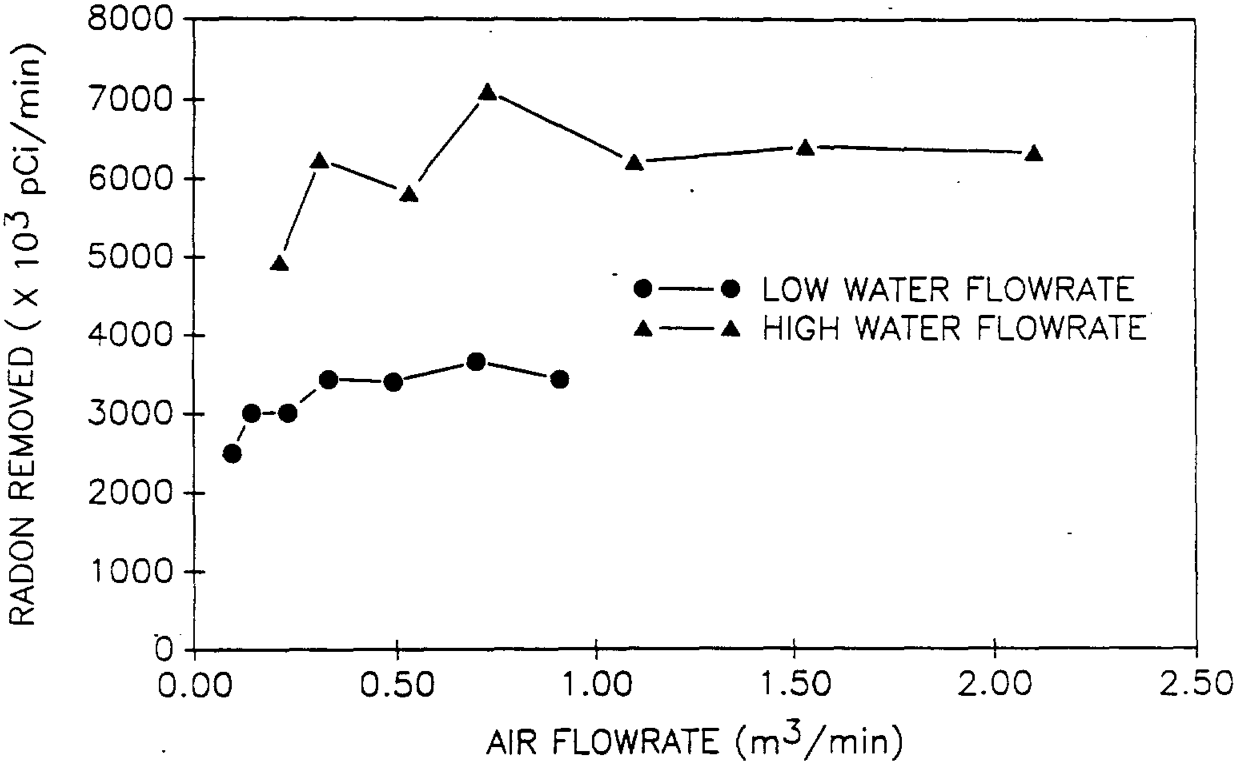

Figure 73. Diffused bubble aeration - Derry, NH. Radon activity removed (x 10^3 pCi/min) as a function of an air flowrate for the low and high water flowrates.

sidewall depth was more of a factor than in the Derry system. A third manhole, 1.8 m in diameter and 2.4 m deep, acted as a clear well.

Lowry and Brandow (1981) also found that 95% removal of radon activity could be achieved with diffused aeration. The system they tested was a static (Q_w = 0 m^3/min) tank (132 - 151 L), usable in point-of-entry applications. The influent radon activity was 30,000 pCi/L, and yielded an effluent activity of 1,100 pCi/L with a detention time of 30 min and an air flow of 0.014 m^3/min. They estimated that 95-100% removal of applied radon activity could be achieved for a continuous flow system after 60 min of operation.

In more recent research, Lowry et al. (1987) tested a 454L vessel with fine bubble diffusers using an aeration rate of 1,400 L/hr and attained 99% removal with a 60 min aeration period (A:W ratio = 3.4:1). A multi-stage bubble aeration system (Lowry et al., 1988) has been installed in 22 point-of-entry applications and has achieved radon removals of 92 to 99.9% with influents ranging from 20,000 pCi/L to over 500,000 pCi/L.

Though results from all of these diffused bubble systems are similar to the Derry results, extrapolating from individual system performance at a given loading and with different construction (including diffusers) and water quality to other water supplies, especially those with considerably lower radon mass loadings, should be avoided because of variations in mass transfer characteristics.

Individual Tank Performance--

The performance of the individual tanks was analyzed to better understand what was occurring within the diffused bubble system. It was designed to act as three completely stirred tank reactors in series. In initial testing of the diffused bubble system at the Derry site, the results indicated that there were problems with the construction (Figure 74). Tank 2 removed $\leq$18% of the total radon regardless of A:W ratio or water flowrate. In addition, at the lower A:W ratios (2:1, 3:1, 5:1), Tank 1 was also not performing well. Visual observation indicated that air flow was somewhat restricted to Tank 2, but this was hardly noticeable. Dye studies indicated that hydraulically all tanks were completely mixed and not short circuiting even at the lowest A:W ratios and high water flowrate. The theoretical detention time of 9.6 min for each 1022 L tank at the high water flowrate of 0.11 m^3 H_2O/min showed excellent agreement with the actual detention times shown in Figure 75. The air supply lines to all tanks were moved to a single header, which improved the removal in Tank 1, but had no effect on Tank 2. Upon careful examination of the system, the brackets holding the diffuser in Tank 2 were found to be 7.6 cm lower on one side of the circular tank than on the other. Tank 1 was also raised 1.0 cm to be level with Tanks 2 and 3. These improvements yielded markedly better performance in Tank 2, however, equal air distribution to Tank 1 continued to be a problem at air flows <0.23 m^3/min. These data indicate that equal tank and diffuser elevations (i.e., equal air distribution) in small community multiple tank systems are crucial in achieving efficient radon removal. Further, diffusers must be operated at air flowrates within their design specifications.

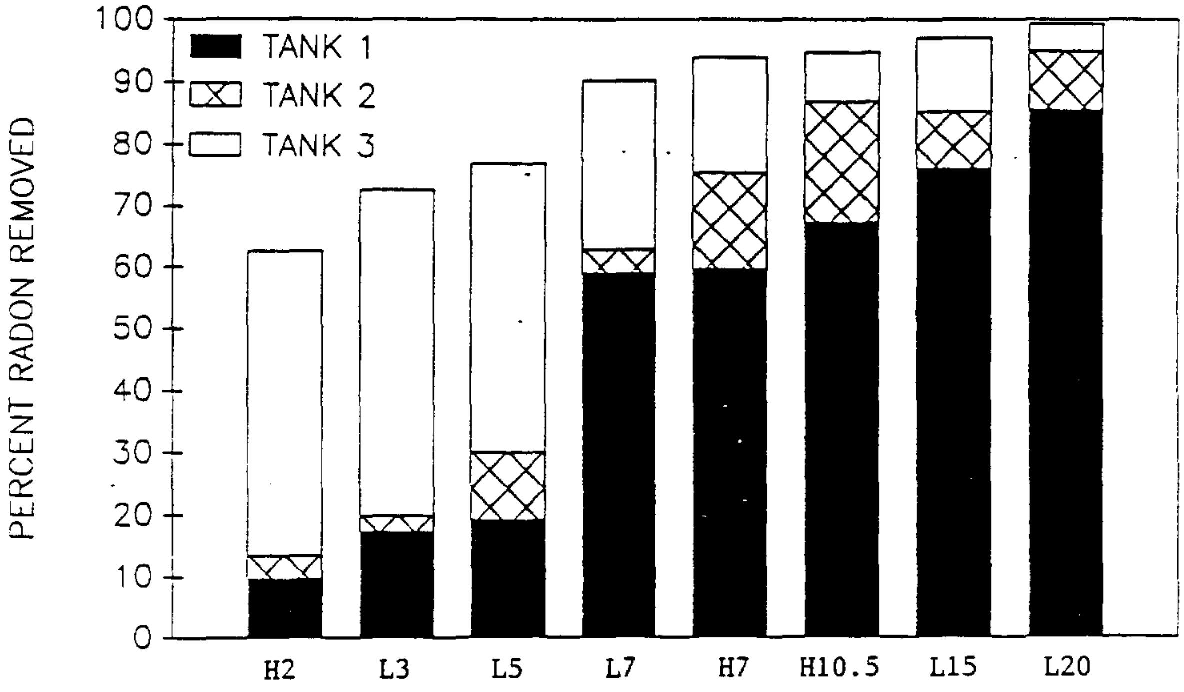

Figure 74. Diffused bubble aeration - Derry, NH. (Before repairs to aeration systems). Percent radon removal for high (H) and low (L) water flowrates and varying A:W ratios in each tank.

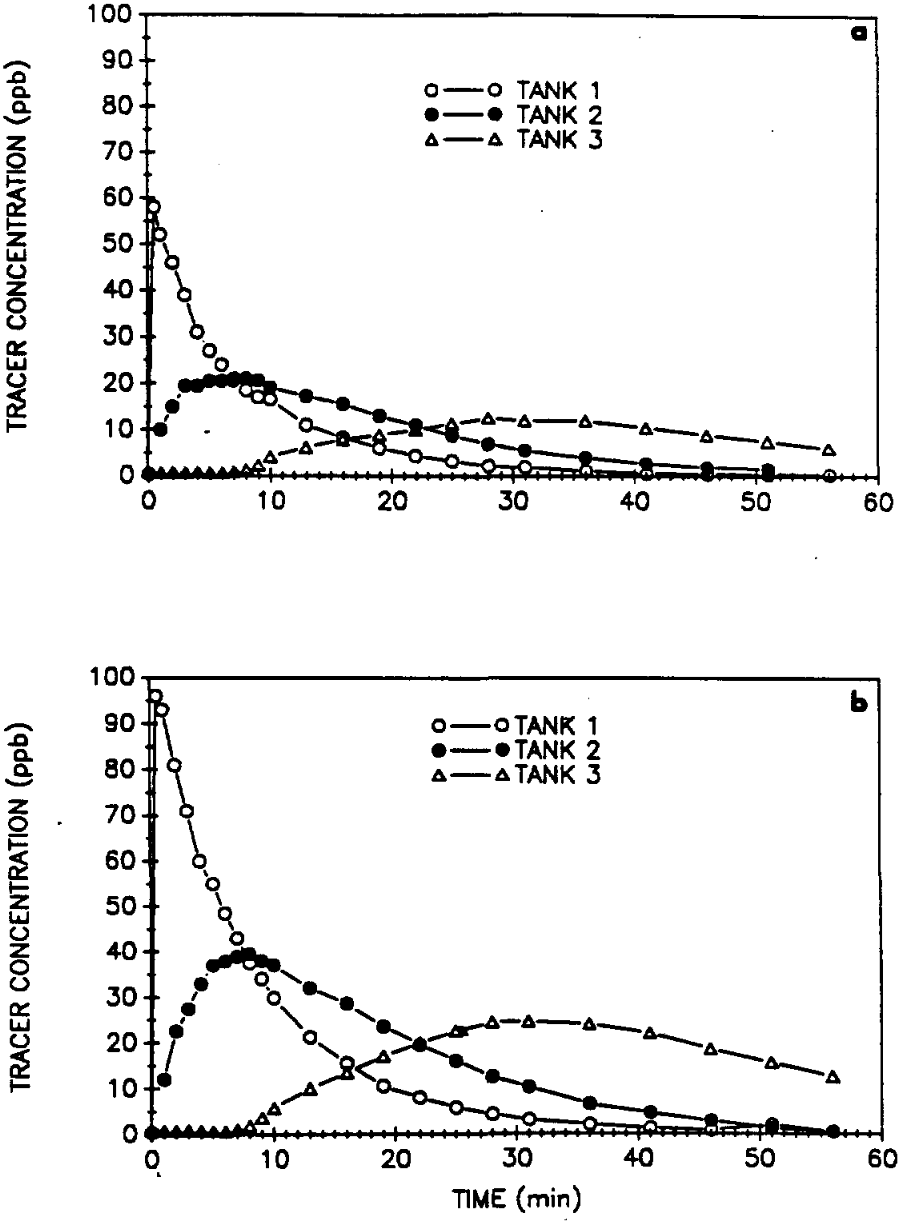

Figure 75. Diffused bubble aeration - Derry, NH. Dye concentration as a function of time in each tank. (a) water flowrate = 0.106 m^3/min, A:W ratio = 3:1. (b) water flowrate = 0.106 m^3/min, A:W ratio = 10.5:1.

After these problems were corrected, the diffused bubble system yielded the results shown in Figure 76. At the lowest A:W ratio, the percent radon removal in the third tank was greater than in the first or second tanks indicating the continued difficulty in getting equal air distribution in a small system unless the construction details are more sophisticated. As the A:W ratio increased, the efficiency in the first tank increased from 39% at A:W = 5:1 (high water flowrate) to 85% at A:W = 20:1 (high water flowrate). As air flowrates increased, the radon removal observed in Tank 1 increased (Figures 76 and 77). However, as the removal increased in the first tank, Tanks 2 and 3 became increasingly less efficient with respect to the overall process and served as polishing units. The data suggest that for the diffused bubble system tested at water flowrates of 0.047 and 0.10 m^3 H_2O/min radon removal was highly dependent on the radon activity and therefore the concentration entering the tank.

Radon removal in the diffused bubble system appeared to be a function of mass transfer. As the radon activity became progressively lower through the series of tanks, the driving force ($C_{liquid} - C_{gas}$) decreased, limiting removal. It is also possible that in some applications where radon activity is very low in the water, the activity in the air could be significant, making gas phase mass transfer limited. Similarly, if the radon concentration in the air above the tanks is allowed to accumulate (i.e., poor exhaust gas venting), transfer limitations may also occur. Though mass transfer may make it more difficult to achieve low effluent activities, the diffused bubble system tested produced water with effluent activities of about 1,849 to 280 pCi/L for A:W ratios of 10.5:1 and greater (Table 14). These data suggest that it may be possible to meet an MCL of 200 pCi/L if the air flowrate is high, and/or there is a long contact time.

Radon removal in diffused bubble systems is dependent on many factors not evaluated in this study including: sidewall depth, bubble size and diffuser type, air and water temperature, and tank and diffuser configuration. Though it appears that most diffused bubble systems tested (Perkins and Hewett, 1988; Lowry and Brandow, 1981 and this study) can achieve high percent radon removal, pilot scale testing of a given configuration may be required especially when (i) retrofitting diffusers into existing tanks in a small community water system, (ii) treating water with low influent radon activity and/or (iii) attempting to meet low MCLs.

Mass Transfer Coefficients--

Theoretically, the global liquid mass transfer coefficient, K_La, for a given diffused bubble system is primarily a function of the air flowrate (Q_g) which can affect the bubble diameter (d_b), the contact time as the bubble rises which is a function of sidewall depth (H_L) and bubble velocity, the water characteristics such as viscosity (μ_w) and density (ρ_w), the contaminant diffusivity compared with oxygen (β), oxygen diffusivity in water (D_o), and

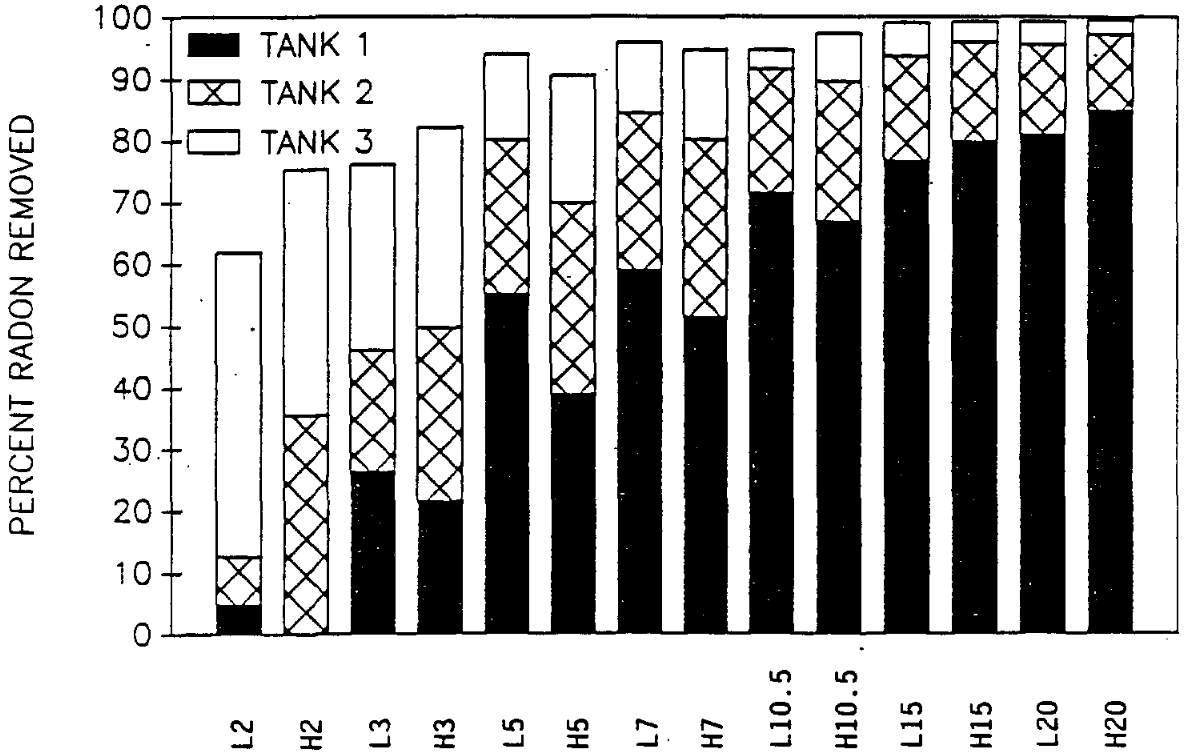

Figure 76. Diffused aeration - Derry, NH. (After improvements to aeration system). Percent radon removal as a function of high (H) and low (L) water flowrates and varying A:W ratios in each tank.

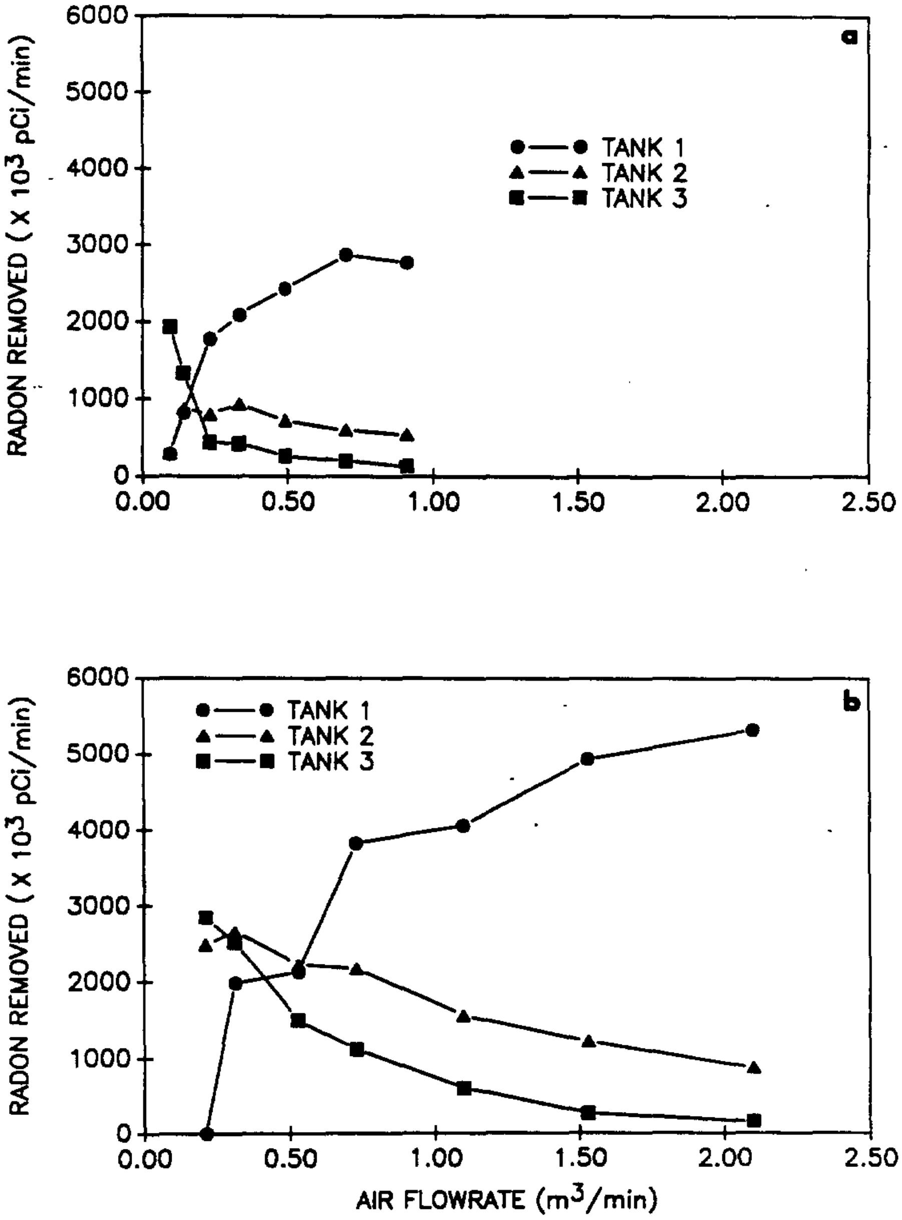

Figure 77. Diffused bubble aeration - Derry, NH. Radon activity removed (x 10^3 pCi/min) as a function of air flowrate for low (a) and high (b) water flowrates in each tank.

TABLE 14. EFFLUENT RADON ACTIVITIES FROM THE DIFFUSED BUBBLE SYSTEM OPERATING AT A:W RATIOS OF 10.5:1 AND GREATER

Water Flowrate	A:W Ratio	Radon Activity (pCi/L) Influent*	Radon Activity (pCi/L) Effluent
Low	10.5:1	74,986 ± 5,894	1,541 ± 87
	15:1	79,799 ± 2,878	777 ± 16
	20:1	76,609 ± 4,995	556 ± 49
High	10.5:1	61,352 ± 7,406	1,849 ± 48
	15:1	69,392 ± 6,003	564 ± 73
	20:1	60,843 ± 4,729	280 ± 14

*Raw water radon activity from wells to diffused bubble system.

the water temperature (since the water density, viscosity, and diffusivity are all temperature dependent). The K_La can be expressed as:

$$K_La = \frac{6\beta' \, H_L^{2/3} Q_g \rho_w^{1/2} D_o^{1/2}}{d_b \, V_L \, (\mu_w)^{1/2}} \qquad \text{(eq. 5)}$$

where $\beta' = \beta/H_L^{1/3}$ and V_L = the liquid volume of the tank (Schroeder, 1977).

For a given diffused bubble system, assuming well mixed conditions, the fraction of the radon removed ($1-C_{eff}/C_{inf}$) can be estimated using the following steady state formula (Eckenfelder,1989):

$$1 - \frac{C_{eff}}{C_{inf}} = 1 - \left[1 + \frac{Q_g}{Q_L} H_c \, (1 - e^{-\phi})\right]^{-1} \qquad \text{(eq. 6)}$$

where C_{eff} = effluent radon activity in water at steady state, C_{inf} = raw water radon activity, Q_g = air flowrate, Q_L = liquid flowrate, H_c = dimensionless Henry's constant, and ϕ = bubble saturation parameter defined as:

$$\phi = \frac{K_La \; V_L}{H_c \; Q_g} \qquad \text{(eq. 7)}$$

The ϕ term accounts for the fractional saturation of the air bubble with respect to radon.

Figure 78 shows the relationship between air flowrate and the fraction of radon removed for each tank at high and low water flowrates. Theoretically, the fraction removed for each tank (assuming identical design and operation) should be the same (eq. 6 and 7). However, Tank 3 consistently had significantly higher removals ($\alpha = 0.01$, StT) than Tanks 1 and 2 under all conditions except the A:W ratio of 20:1 for the high water flowrate (Table 15). Further, Tank 2 had significantly ($\alpha = 0.01$, StT) higher removals than Tank 1 in most cases. These observed differences may have been a function of the assumption that all tanks received an equal air flowrate. The construction and operation of the air distribution system appeared to favor Tank 3, with Tank 1 receiving the least flow. The decreased air flowrate to Tank 2 and especially to Tank 1 would account (eq. 6 and 7) for the decreased removals observed in those tanks. As predicted by eq. 6 and 7, the removals increased significantly ($\alpha = 0.05$, StT) as air flowrate increased in all cases except high water flowrate (A:W = 15:1 and 20:1) and low water flowrate (A:W = 2:1 and 15:1 and 20:1) which were only different at $\alpha \geq 0.10$ (StT).

Differences in tank volume, diffuser type and configuration, air flowrate, sidewall depth, water quality and temperature make comparisons of K_La values (eq. 5) and removals (eq. 6 and 7) among diffused bubble systems

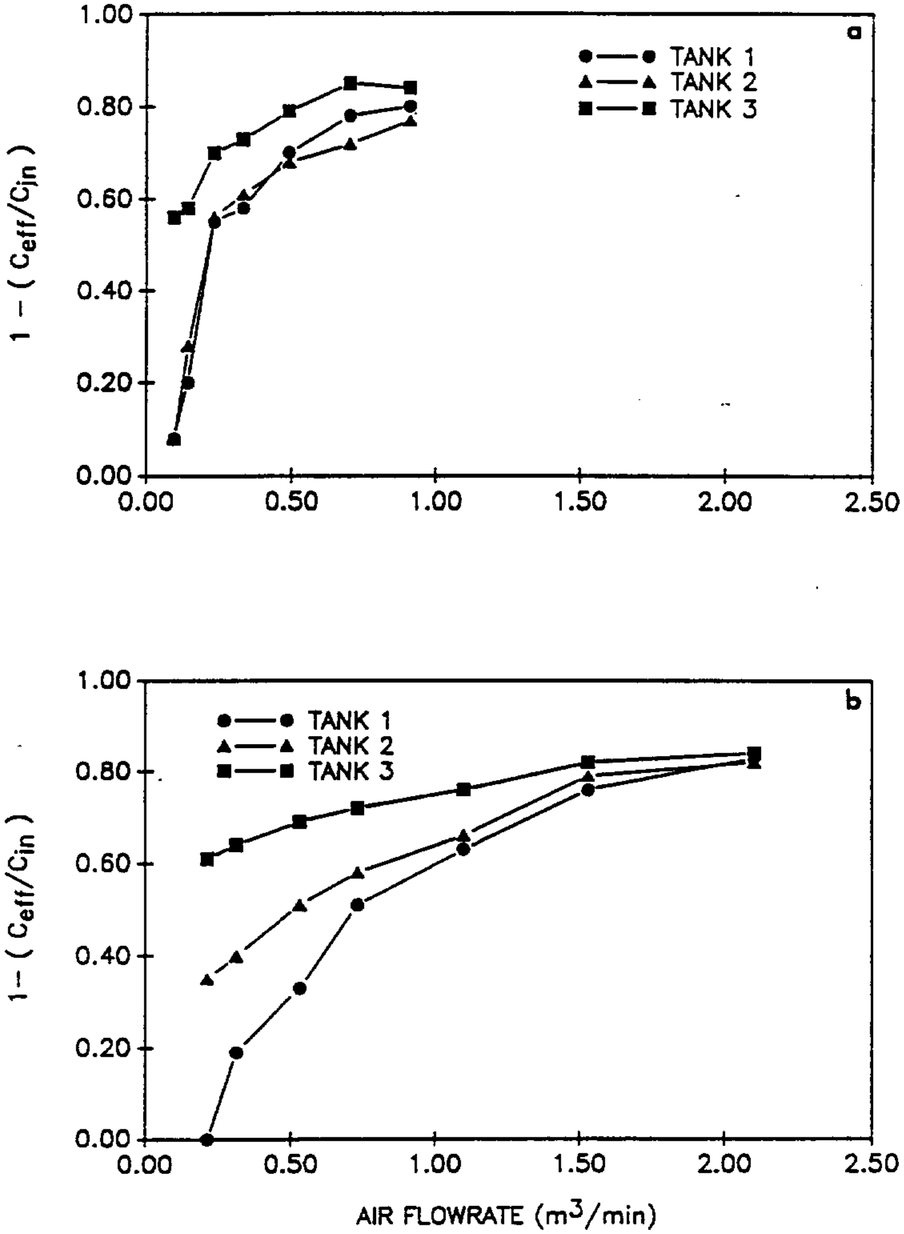

Figure 78. Diffused bubble aeration - Derry, NH. Fraction of radon removed ($1-C_{eff}/C_{inf}$) as a function of air flowrate for low (a) and high (b) water flowrates in each tank.

TABLE 15. STATISTICAL COMPARISONS* OF FRACTION OF RADON REMOVED $(1 - \frac{C_{eff}}{C_{inf}})$ IN TANKS IN THE DIFFUSED BUBBLE SYSTEM

Water Flowrate	A:W Ratio	Statistical Relationship	Significance Level α
Low	2:1, 5:1, 10.5:1	Tank 1 = Tank 2	0.05 & 0.10
	3:1, 7:1, 15:1, 20:1	Tank 2 > Tank 1	0.05 & 0.01
	2:1, 3:1, 5:1, 7:1, 10.5:1, 15:1, 20:1	Tank 3 > Tank 1 Tank 3 > Tank 2	0.05 & 0.01
High+	3:1, 5:1, 7:1, 10.5:1	Tank 2 > Tank 1 Tank 3 > Tank 1 Tank 3 > Tank 2	0.05 & 0.01
	15:1	Tank 2 > Tank 1 Tank 3 > Tank 1 Tank 3 > Tank 2 Tank 3 = Tank 2	0.05 & 0.01 0.05 & 0.01 0.05 0.01
	20:1	Tank 1 = Tank 2 Tank 3 = Tank 1 Tank 3 > Tank 2 Tank 3 = Tank 2	0.05 & 0.10 0.05 & 0.10 0.05 0.01

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* As assessed by ANOVA.

\+ High water flowrate and A:W ratio of 2:1 not included because of air flow distribution problems in Tank 1.

inappropriate (Eckenfelder, 1989; Perkins and Hewett, 1988; Lowry and Brandow, 1981).

Off-Gas Radon Activities--

Stack emissions were monitored to determine if the off-gas radon activities from the diffused bubble system would affect the air quality of the surrounding environment. The measured off-gas radon emission rates were plotted against the theoretical off-gas radon emissions rates (Figure 79). Theoretical off-gas rates were found by multiplying the actual difference between the measured influent and effluent water radon activities in the diffused bubble system by the measured air flowrates for a given operating condition. The measured activities were 10% to 50% lower than those predicted by theoretical calculation. The differences may have been a function of (i) variation in radon loading over the runs, (ii) lack of replicate sampling, (iii) slight variations in the air flowrate, (iv) dilution of the sample collected with ambient air at the end of the stack and/or (v) air sampling problems from streamlining of the off-gas flow around the sampling tube.

As the A:W ratio and air flowrate increased, there was a decrease in the radon activity in the off-gas (Figure 80). The increased air flow diluted the mass of radon removed from the water in increasingly larger volumes of air. Higher air flowrates would be expected to decrease the off-gas activity even further. For the system tested, the off-gas activity (3,361 to 18,356 pCi/L) would need to be diluted 10^4 to 10^5 times to be similar to radon activities found in the ambient air at the site (0.1 to 0.15 pCi/L) (NH Bureau of Radiological Health, personal communication, 1986).

Iron and Manganese

As observed with radon, there was a difference in the influent total iron and manganese concentrations in the raw water between the high and low water flowrate tests (Low Water Flowrate: 0.14 ± 0.11 mg Fe/L and 0.10 ± 0.05 mg Mn/L; High Water Flowrate: 0.31 ± 0.28 mg Fe/L and 0.32 ± 0.04 mg Mn/L). This probably occurred because of differences in the wells over time at the Derry site. In all cases, except low water flowrate A:W = 3:1 (L3:1), there was no significant change in total iron through the diffused bubble system ($\alpha = 0.05$ and 0.10, StT). At L3:1, the influent iron 0.31 ± 0.01 mg/L decreased to below detection (0.06 mg/L) in the effluent. In all cases, except for high water flowrate A:W = 3:1 (H3:1), there was also no significant change ($\alpha = 0.05$ and 0.10, StT) in total manganese through the system. At H3:1 the influent manganese decreased from 0.30 ± 0.14 mg/L in the influent to 0.13 ± 0.04 mg/L in the effluent. Overall, there was little change in the total concentration of either metal through the system. The effects of aeration on metal oxidation would not be noticeable in a total metals analysis unless settling were occurring in the completely mixed diffused bubble system, which is doubtful. In the future, soluble and total metal concentrations should be analyzed to determine if oxidation and precipitate formation are factors. This would be problematic if it occurred because of potential accumulation of sludges in the atmospheric storage tank and fouling of the diffusers.

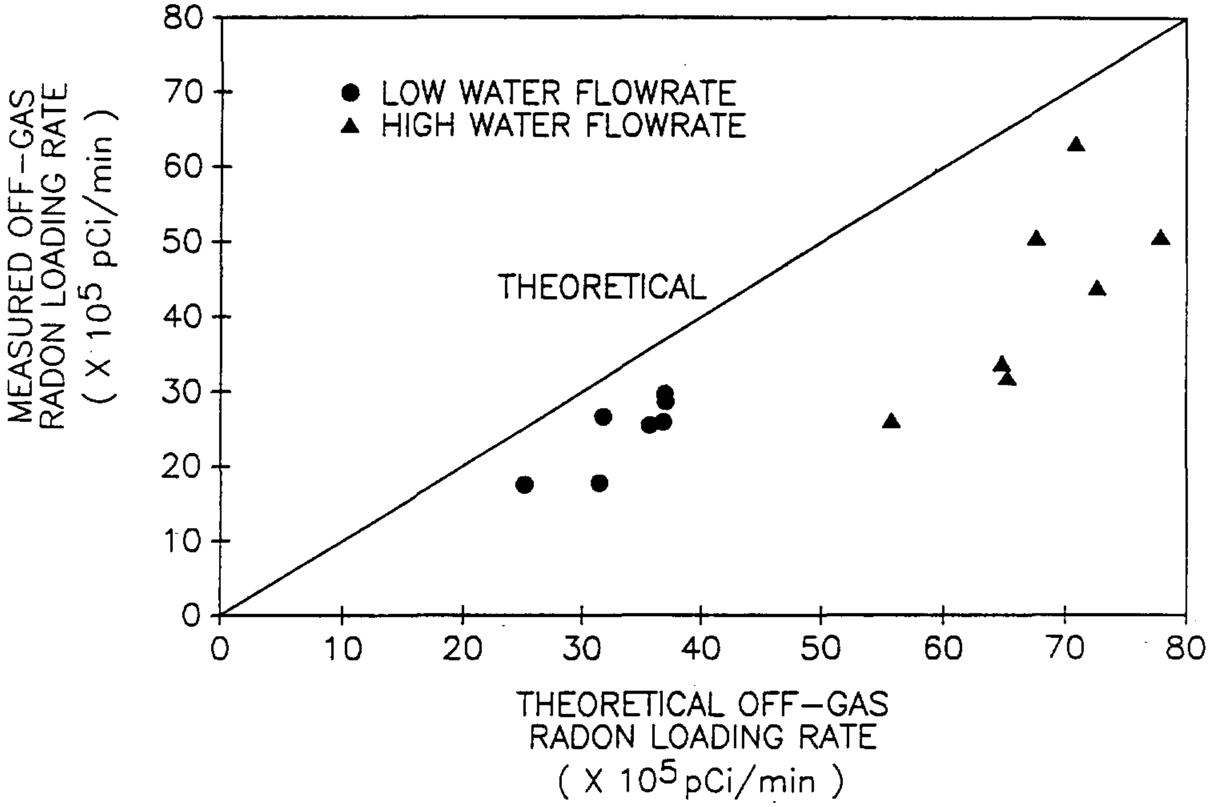

Figure 79. Diffused bubble aeration - Derry, NH. Off-gas radon activity (x 10^5 pCi/min) as a function of theoretical off-gas radon activity (x 10^5 pCi/min).

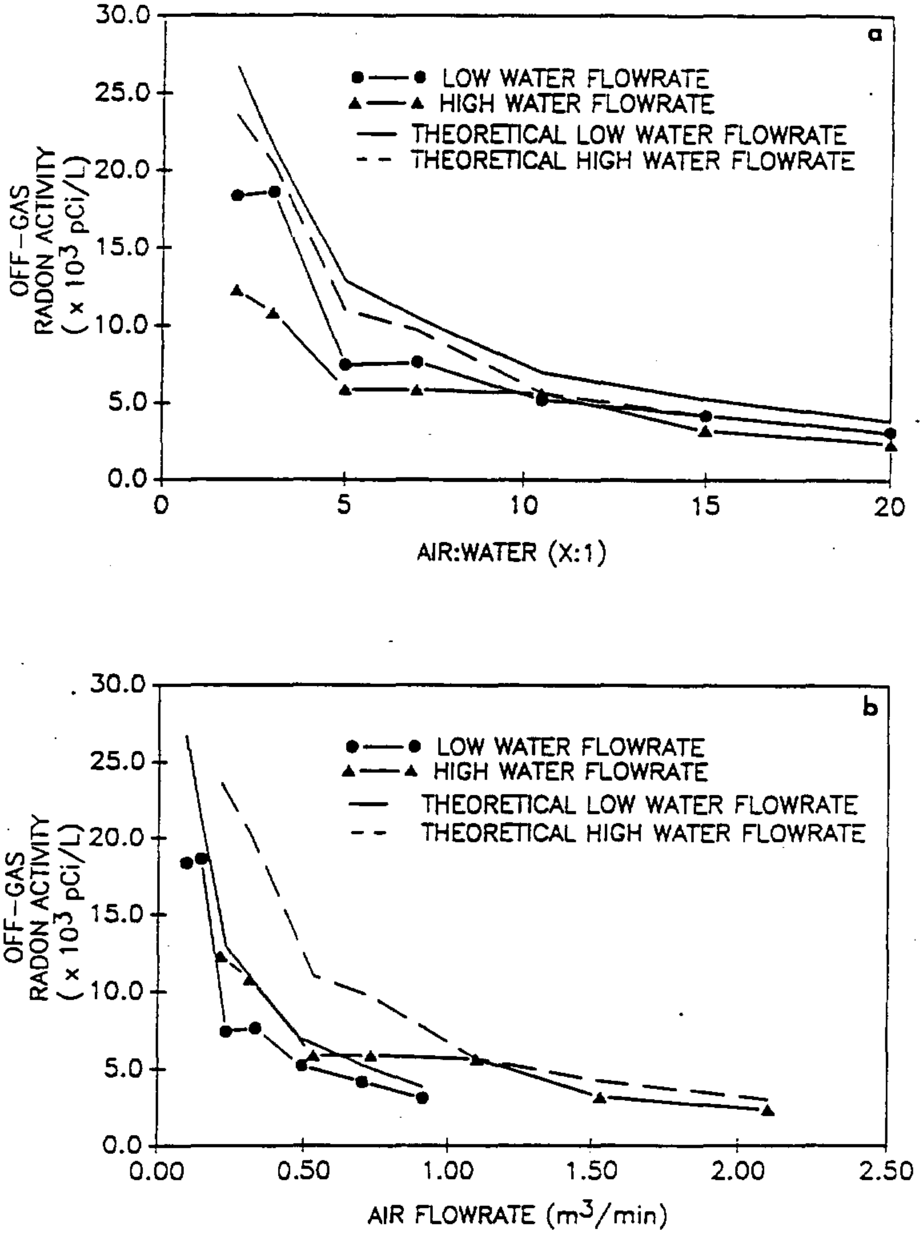

Figure 80. Diffused bubble aeration - Derry, NH. Off gas radon activity as a function of A:W ratio (a) and air flowrate (b) for the low and high water flowrates.

Turbidity, Alkalinity and pH

The influent turbidity averaged 0.20 ± 0.09 NTU and 0.08 + 0.07 NTU for the high and low water flowrates, respectively. With the exception of the low water flowrate at an A:W ratio of 2:1, there was no significant difference in turbidity through the units at all A:W ratios and low water flowrate ($\alpha = 0.05$ and 0.10, StT). At the high water flowrate for A:W ratio 2:1 to 5:1, there was no significant change in turbidity through the units ($\alpha = 0.05$ and 0.01, StT). However at A:W ratios of 7:1 and greater significant increases in turbidity occurred (Table 16). It appears that at these air flowrates, there may have been some increase in precipitate formation within the units, but this cannot be corroborated with the iron data because the latter does not reflect changes in soluble iron. It is unlikely that manganese precipitates would have formed at this pH (6.25 ± 0.09) and detention time (30 to 60 min).

There was no clear pattern in the alkalinity data (Table 17) and in most cases there was no significant change in this parameter through the units. Again there was a difference in raw water alkalinity between the high water flowrate and the low water flowrate runs (41 ± 3.9 mg/L as a $CaCO_3$ vs. 79 ± 4.0 mg/L as $CaCO_3$, respectively). The only decrease in alkalinity occurred at the low water flowrate and an A:W ratio of 20:1 (81 ± 0.64 to 71 ± 1.0 mg/L as $CaCO_3$), but is likely the result of variations in the analytical technique.

The pHs of the high and low water flowrates were 6.3 ± 0.09 and 6.5 ± 0.20, respectively. In all cases, except the high water flowrate and A:W ratio of 2:1, there was a significant increase in pH through the units at $\alpha = 0.05$ (except low water flowrate and A:W ratio of 3:1 which was at $\alpha = 0.10$) and at $\alpha = 0.01$ (except high water flowrate and A:W ratios of 3:1 and 10.5:1). As shown in Table 18, the effluent pH increased as A:W ratio increased. The increases probably resulted from an increased removal of CO_2 from the water with larger air flowrates.

ECONOMICS

The cost estimates for the diffused bubble aeration system used at Derry, NH are shown in Table 19. The economic analysis was performed using the techniques described in Section 4. The total production cost for the diffused bubble system used at Derry, treating 37.85 m^3/min, was estimated to be $0.64/1,000 gallons. If pretreatment for iron and manganese via ion exchange is required (see Section 4) then the total production cost with pretreatment would be $2.14/1,000 gallons. The capital and O&M costs for the diffused bubble system are in close agreement with those reported by Malcolm Pirnie, Inc. (1988). The latter study estimated the capital and O&M costs for a plant treating an average daily flow of 49 m^3/day (227 m^3/day design flow) to be $12,200 and $1,000/year, respectively.

The major costs of the diffused bubble system result from the capital costs of process equipment and the resulting annual O&M costs of this equipment. The diffused bubble performance data obtained in this study indicates that the same process equipment can achieve comparable radon removals at flows up to 150 m^3/day (0.10 m^3 H_2O/min) which is far in excess of the 38 m^3/day design flow. Therefore, the production costs for diffused

TABLE 16. INFLUENT AND EFFLUENT TURBIDITY FOR THE HIGH WATER FLOWRATE DIFFUSED BUBBLE SYSTEM RUNS (A:W RATIO 7:1 AND GREATER)

A:W Ratio	Turbidity (NTU) Influent	 Effluent	Statistical* Relationship	
7:1	0.13 ± 0.04	1.21 ± 0.25	Inf < Eff	α^+ = 0.05 & 0.01
10.5:1	0.18 ± 0.05	1.29 ± 0.12	Inf < Eff	α = 0.05 & 0.01
15:1	0.27 ± 0.01	1.33 ± 0.06	Inf < Eff	α = 0.05 & 0.01
20:1	0.36 ± 0.01	2.23 ± 0.81	Inf < Eff	α = 0.05 & 0.01

*As assessed by Student's t test.

+α: significance level.

TABLE 17. STATISTICAL COMPARISON* OF INFLUENT AND EFFLUENT ALKALINITY IN THE DIFFUSED BUBBLE SYSTEM

Water Flowrate	A:W Ratio	Statistical Relationship	Significance Level α
Low	2:1, 3:1, 5:1, 7:1, 10.5:1, 15:1	Inf = Eff	0.05 & 0.10
	20:1	Inf > Eff	0.05 & 0.01
High	2:1, 3:1, 5:1, 7:1	Inf = Eff	0.05 & 0.10
	10.5:1	Inf = Eff Inf < Eff	0.05 0.10
	15:1	Inf > Eff Inf = Eff	0.05 0.01
	20:1	Inf = Eff	0.05 & 0.10

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*As assessed by Student's t test.

TABLE 18. EFFLUENT pHs FOR DIFFUSED BUBBLE RUNS

Water Flowrate	A:W Ratio	Effluent pH
Low	2:1	6.8 ± 0.02
	3:1	6.8 ± 0.11
	5:1	7.5 ± 1.08
	7:1	7.4 ± 0.05
	10.5:1	7.7 ± 0.04
	15:1	7.7 ± 0.05
	20:1	7.8 ± 0.09
High	2:1	6.5 ± 0.04
	3:1	6.8 ± 0.18
	5:1	6.9 ± 0.07
	7:1	6.9 ± 0.16
	10.5:1	6.8 ± 0.15
	15:1	7.4 ± 0.13
	20:1	7.2 ± 0.20

TABLE 19. COST ESTIMATE FOR THE DIFFUSED BUBBLE SYSTEM AT DERRY, NH

Item	Total Cost (2nd Quarter 1989 dollars)*
CAPITAL COSTS	
DIRECT COSTS	
Process Equipment	
3 - Polyethylene tanks ($370.94/tank)	$ 1,113
1 - Fuji ring compressor	606
1 - Float valve	155
1 - Control panel	207
1 - Air pump filter assembly	31
1 - Air vent assembly	108
1 - Air manifold	205
3 - Air diffuser assemblies ($358.67/diffuser)	1,076
1 - Booster/transfer pump+	800
1 - Rockwell flow rate recorder/totalizer	1,036
Total Process Equipment Cost	$ 5,337
Support Equipment	
Installation	150
Plumbing	1,123
Electrical	700
Total Support Equipment Cost	$ 1,937
TOTAL DIRECT COST	$ 7,310
INDIRECT COSTS	
Sitework	$ 1,097
Engineering	1,097
Contractor Overhead and Profit	877
Legal and Financial Fees	183
Interest	439
Contingencies	$ 1,097
TOTAL INDIRECT COST	$ 4,790
TOTAL CAPITAL COST	$12,100
AMORTIZED CAPITAL COST (Annual)	$ 1,326

(continued)

TABLE 19 (continued)

Item	Total Cost (2nd Quarter 1989 dollars)[*]
ANNUAL O&M COSTS	
Power	310[**]
Maintenance	
Mechanical	391
Non-mechanical	57
Labor	110
Administrative	134
TOTAL ANNUAL O&M COST	$ 1,002
TOTAL ANNUAL COST	$ 2,328/year
TOTAL PRODUCTION COST (Design flow 37.85 m^3/d)	$0.64/1,000 gallons
PRETREATMENT COST	$1.50/1,000 gallons
TOTAL PRODUCTION COST WITH PRETREATMENT	$2.14/1,000 gallons

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[*] Based on ENR Construction Cost Index, Base Year 1967 = 100.

[+] Although a transfer pump was not required at this site, it is likely to be required at most locations.

[**] Cost based on input power requirements of 1 hp each for the air blower and the transfer pump, operating period of 5.6 hr/day for each, and an electric rate of $0.10157/kW-hr.

bubble systems may decrease markedly as design flows increase. (N.B., the most cost effective blower is a 0.38 kW constant speed blower for the range of flows evaluated in this study, so capital and power costs for the blower would be constant.)

A major assumption implicitly made in the economic analysis is that the radon present in the off-gas will disperse in the wind currents and will not require additional treatment. Although the economic analysis does account for the cost of venting the radon above the average home roof line (i.e., 11 m), no additional air treatment costs are included. If subsequent treatment is required to meet federal or state air quality standards, then the cost estimates could be significantly higher.

SECTION 6

PACKED TOWER

INTRODUCTION

Packed tower aeration has several advantages when used as a treatment technique for removing radon from water. It is rapid, requiring contact times of seconds to minutes. It also uses fairly low volumetric A:W ratios, often less than those used for diffused bubble aeration. In addition, a tower can be easily retrofit to most buildings without requiring much floor space. Unlike GAC systems, it does not concentrate radon and its progeny and as a result is not a source of gamma emissions. The disadvantages most often associated with packed tower aeration include: high costs of operation (aeration system), release of radon to the atmosphere, potential oxidation of iron, accumulation on and sloughing of bacteria from the packing media, and the need for additional pumping or repressurization of water. Other potential problems may be the excessive noise of the air blower and in northern climates, freezing conditions causing ice formation with the tower and pipes.

Cummins (1987) evaluated the radon removal efficiency of packed tower aeration for small community water supplies using several packing heights (0.15 to 5.33 m), and A:W ratios (0.05:1 to 15:1). These field studies used a 0.61 m diameter tower packed with plastic saddles receiving an influent radon activity ranging from 4,100 to 6,200 pCi/L. At A:W ratios of 2:1, 5:1 and 15:1, the tower achieved 95, 99.1 and 99.4 percent removal, demonstrating it could efficiently remove radon yielding effluent activities <50 pCi/L. Dixon and Lee (1987, 1989) reported radon removals of 95-97% using a packed tower 4.88 m in height and 1.37 m in diameter with an A:W ratio of 50:1 (packing medium unknown). The influent radon activity to the tower, which was designed to remove volatile organic compounds, was 646-783 pCi/L.

METHODS AND MATERIALS

The research was conducted in two phases. Phase I evaluated the effect of various parameters on the tower's performance including liquid loading rate, volumetric A:W ratio, packing media and influent radon activity. Phase II involved continuous operation over 6 to 8 hr periods at one condition.

Description of Packed Tower System

The packed tower aeration system, designed by North East Environmental Products (Lebanon, NH), treated the water supply at the same mobile home park in Mont Vernon, NH used in the GAC study. The system consisted of an 5.49 m tall, 0.30 m diameter, stainless steel tower containing randomly packed plastic media (Figure 81). It was supported within a wooden housing and set on a concrete base. Water was provided to the top of the tower directly from 1 or 2 bedrock wells via 3.18 cm PVC piping. A nozzle located 15.24 cm above the top of the media distributed the water. Air entered the tower 1.07 m from the bottom (0.15 m below the media) and was supplied from a manually-triggered American Fan (Fairfield, OH) single phase industrial electric air blower via a 10.16 cm plastic tube connection. The countercurrent air flow aided in the release of the radon from the water. A 10.16 cm plastic hose connected to the top of the tower carried the effluent gas (off-gas) outside. A 0.63 cm plastic tube was inserted 15.24 cm into the end of the hose and was used to sample radon activity in the off-gas.

The treated water was collected in a reservoir at the base of the tower below the media. Electrodes were placed in this reservoir so that when the water level reached a height of 0.91 m, a transfer pump would empty it to a level of 22.86 cm. The treated water was pumped into the community's 7,571 L atmospheric holding tank. During Phase II, the effluent water was wasted to a field behind the pumphouse. Wasting was necessary to operate the system continuously without flooding the pumphouse.

An in-line water flowmeter was located in the 3.18 cm PVC pipe that carried the effluent water to the atmospheric tank or outside the pumphouse. A rotometer capable of measuring water flowrates ranging from 0.97 - 9.65 m^3/hr was located on the influent line to the tower. Pressure and temperature gauges were placed in the influent and effluent water lines and in the influent air flow line. A pitot tube was placed in the influent air flow line and attached to a pressure gauge to measure air flowrates, which were regulated by an in-line globe valve.

The tower contained 3 sampling ports located within the media, as well as influent and effluent sampling ports. Each sampling port consisted of a stainless steel, half cylinder trough (0.95 cm diameter) which protruded about 15.24 cm into the tower. Media sampling ports were located 0.46 m, 1.98 m, and 3.81 m below the influent nozzle. Plastic tubing (1.27 cm diameter) carried the sample water from each port to a sampling board located at the base of the tower. The influent port was tapped into the influent 3.81 cm PVC line just before the water entered the tower and the effluent port was tapped into the reservoir 5.08 cm from the bottom. 0.97 cm plastic tubing transported the influent and effluent water from the ports to the sampling board. A 10.16 cm diameter plastic funnel was attached to the free end of each sampling hose. During sampling, the mouth of the funnel was upright so that the water from the hose filled it and overflowed its lip. This facilitated collection of radon samples even at low water flowrates.

Experimental Design

The Mont Vernon raw water had an average radon activity of 190,134 ± 48,237 pCi/L throughout the packed tower study. Phase I of the study consisted of several separate 3 hr runs designed to determine the tower's radon removal efficiency for a variety of operating conditions. A second phase tested one of these conditions over two 8 hr and one 6 hr continuous

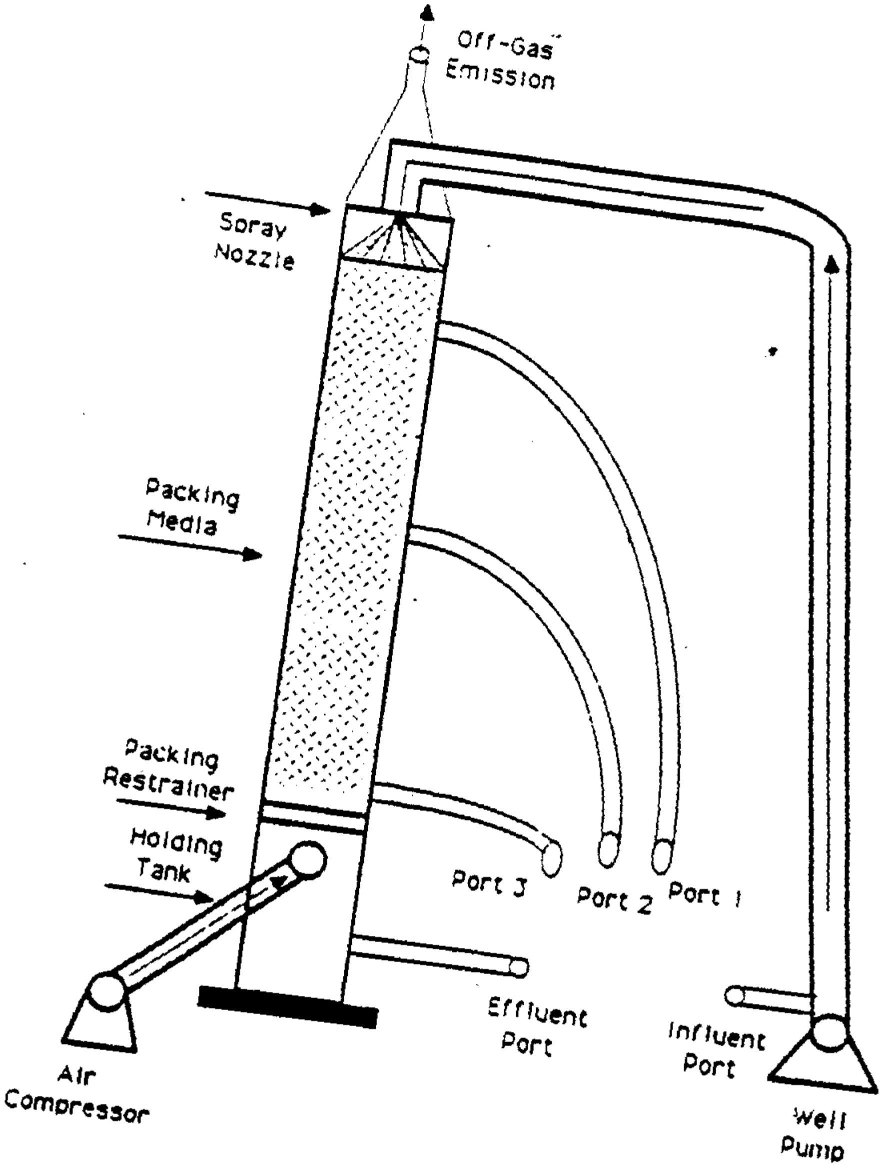

Figure 81. Schematic of the packed tower aeration system. Mont Vernon, NH.

flow periods. The duration of these continuous flow periods was based on access to the water supply as regulated by the community.

Comparisons were made among combinations of four parameters: packing type, packing height, liquid loading rate, and volumetric A:W flow ratios. During Phase I, three types of packing media were tested: 5.08 cm diameter Glitsch^R mini rings (packing factor (C_f) = 30), 2.54 cm diameter Glitsch^R saddles (C_f = 33), and 2.54 cm diameter Koch^R pall rings (C_f = 45) using the ports installed at the 3 depths within the tower to monitor removal. The packing heights were 0.30, 1.83, 3.66 and 3.75 m when the tower was filled with mini rings and saddles, and 0.15, 1.68, 3.51, and 3.60 m when filled with pall rings. Two liquid loading rate ranges were obtained by adjusting the liquid flow into the tower. Manually activating one well provided the low flowrate range of 0.18 to 1.07 m^3 H_2O/hr (liquid loading rate of 2.5 to 14.7 m^3H_2O/m^2-hr), while 2 wells provided a high flowrate range of 0.74 - 2.55 m^3 H_2O/hr (liquid loading rate of 10.2 to 35.0 m^3 H_2O/m^2-hr). Variations in flowrate were the result of the low yield of the wells at Mont Vernon during this testing period. Air flows were adjusted to obtain approximate volumetric A:W ratios of 20:1, 10:1, and 5:1 for each liquid flowrate range, media type, and packing height. A:W ratios of 2:1 and 1:1 were subsequently tested for the high flowrate, using the pall rings.

Phase II was conducted using both wells with an average water flowrate of 1.46 ± 0.20 m^3 H_2O/hr and an A:W ratio of 2:1. Pall rings were used as packing media since they were the last medium tested in Phase I and had yielded good radon removals. The influent radon activity during this phase of the research averaged 203,675 ± 22,412 pCi/L.

Sampling Plan

The parameters monitored for the packed tower aeration system are shown in Table 20. During Phase I, water samples/measurements for radon, alkalinity, turbidity, iron, manganese, temperature, pressure, and flowrate, and air samples/measurements for pressure, temperature, and flowrate were taken at 15, 30, 60, 120, and 180 min. Air temperature, barometric pressure, and relative humidity were measured inside the pumphouse at the beginning of each run. Background air samples for radon analysis were taken at the beginning of each run, and off-gas emissions were monitored once each sampling run, usually after two hours of operation.

During Phase II, water samples/measurements for radon, alkalinity, turbidity, iron, manganese, temperature, pressure, and flowrate and air samples/measurements for pressure, temperature, and flowrate were all taken at initial startup and each hour thereafter (over the 6 to 8 hr run time). These parameters were monitored during Phase II to determine the effects of aeration over longer operational periods. Relative humidity, temperature, and barometric pressure of the air in the building were measured at the beginning of each run. Background air samples for radon were taken at the beginning of each run and off-gas emissions were measured halfway through and at the end of each continuous operation period. All temperatures were measured using

TABLE 20. PACKED TOWER AERATION SAMPLING PARAMETERS

	Radon	Alkalinity	Turbidity	Iron/Mang.[*]	Flowrate	Temp.[+]	Pressure	Barometric Pressure	Relative Humidity
Water Samples									
- Influent	X	X	X	X		X	X		
- Port 1	X								
- Port 2	X								
- Port 3	X								
- Effluent	X	X	X	X	X	X	X		
Air Samples									
- Influent					X	X	X		
- Building						X		X	X
- Effluent	X								

[*] Iron and manganese.
[+] Temperature.

in-line thermometers. Water and air pressures were also measured using in-line gauges. Water flowrates were measured using an in-line flowmeter. Air flowrates were measured using a pitot tube placed in the influent air line, which was attached to a Dwyer^R magnehelic differential pressure gauge (Michigan City, IN). Pitot tubes were placed in PVC pipes of sizes 1.27, 2.54, 5.08 and 10.16 cm, depending on the required air flow. The magnehelic had a gauge reading 0 to 1.27 cm (water).

RESULTS AND DISCUSSION

Radon

Phase I - Evaluation of A:W Ratio, Packing Media and Water Flowrate--

Overall System Performance--The experimental design for the packed tower experiments was supposed to evaluate the effect of water flowrate (liquid loading), air flowrate (A:W ratio) and packing media on radon removal. Unfortunately, during Phase I, the wells at Mont Vernon were pumping erratically (summer and early fall 1988). At that time, New Hampshire had received little rainfall and the groundwater table was depressed. In spite of the fact that the wells in Mont Vernon were deep (61-91 m) and believed to be in a bedrock aquifer, they appeared to be affected by the lower water table. Their yields were low and the radon activities associated with the water fluctuated greatly on a daily basis.

As a result, the liquid loading rates (i.e., water flowrates) and radon activities for packed tower runs made during Phase I were not comparable (Figures 82 and 83). For example, the high water flowrate runs had significantly different flowrates for each packing medium within a given A:W ratio. Similarly, the runs with a given packing medium at the different A:W ratios often had very different radon activities in the raw water further confounding comparisons.

The only factor which was well regulated among the trials was the air flowrate which could be controlled by the sampling team. In addition, influent radon activities were often similar for more than one of the sampling times within a run at a given test condition (Figure 84). For purposes of data analysis when there was <8.0% difference between the raw water radon activities in a given run they were considered to be similar and were pooled for use in statistical analysis.

Determinations of treatment efficiency could not be made based on the percent mass of radon removed per time [i.e., $(Q_w \times C_{inf} - Q_w \times C_{eff})/Q_w\ C_{inf} \times 100$] because of the constant variations in water flowrate. Therefore, all percent removals were expressed in terms of the change in radon activity [$(C_{inf} - C_{eff})/C_{inf} \times 100$]. Due to the very short residence time of water in the tower, comparison of influent and effluent activities at a given sampling time was justified because short term variations in influent activity were minimal.

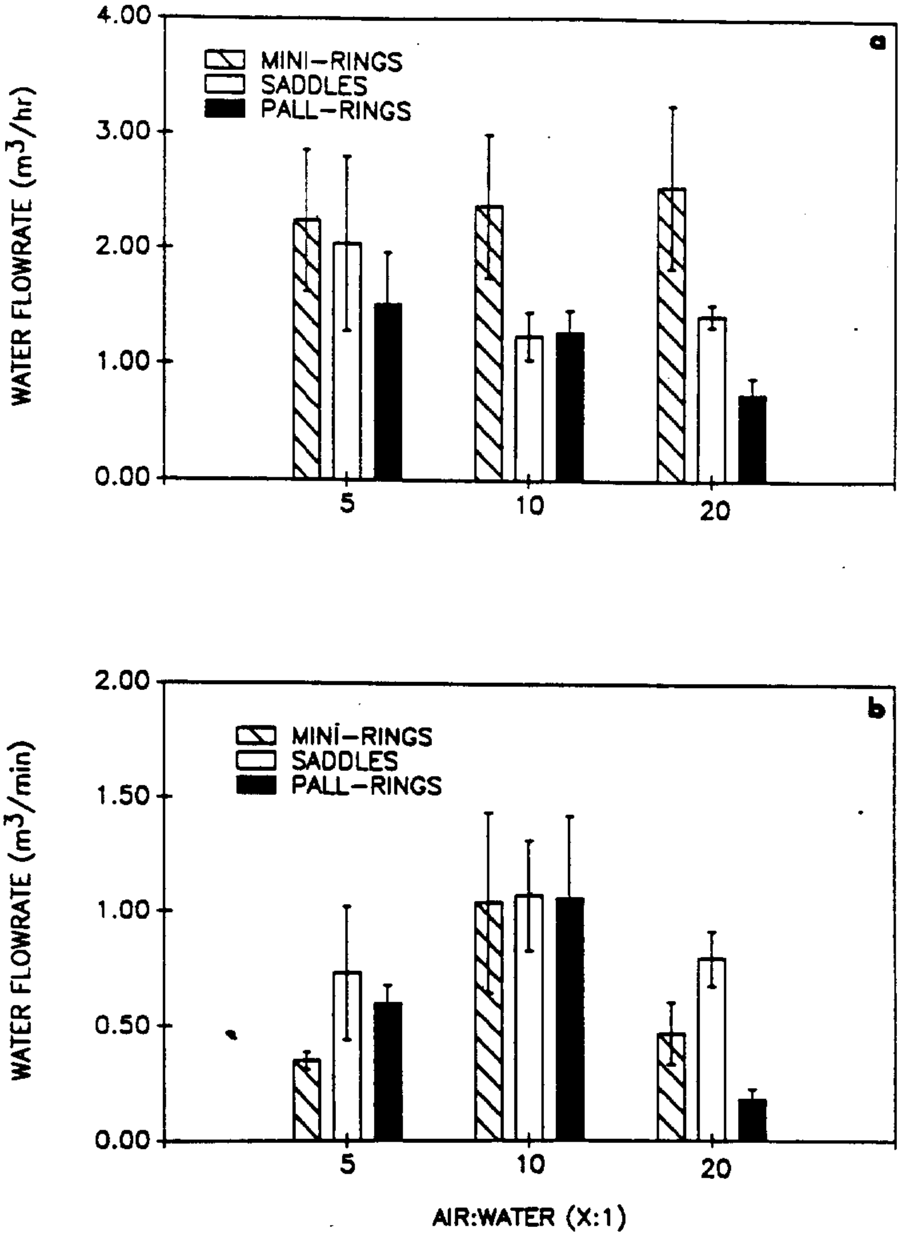

Figure 82. Phase I - Packed tower aeration. Mont Vernon, NH. Water flowrate as a function of A:W ratio for each packing medium. (a) high water flowrate, (b) low water flowrate. Note differences in scale on ordinate.

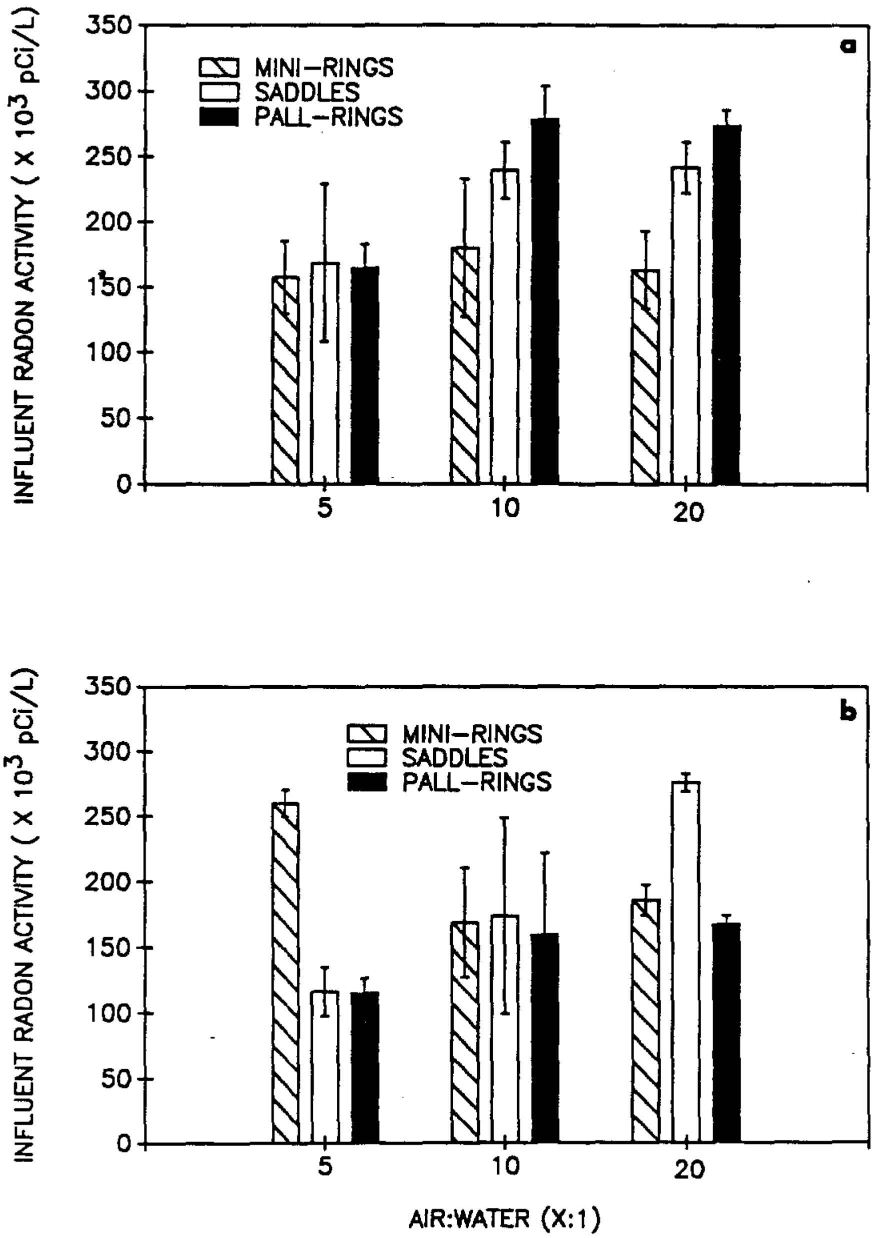

Figure 83. Phase I - Packed tower aeration. Mont Vernon, NH. Influent radon activity as a function of A:W ratio for each packing medium. (a) high water flowrate, (b) low water flowrate.

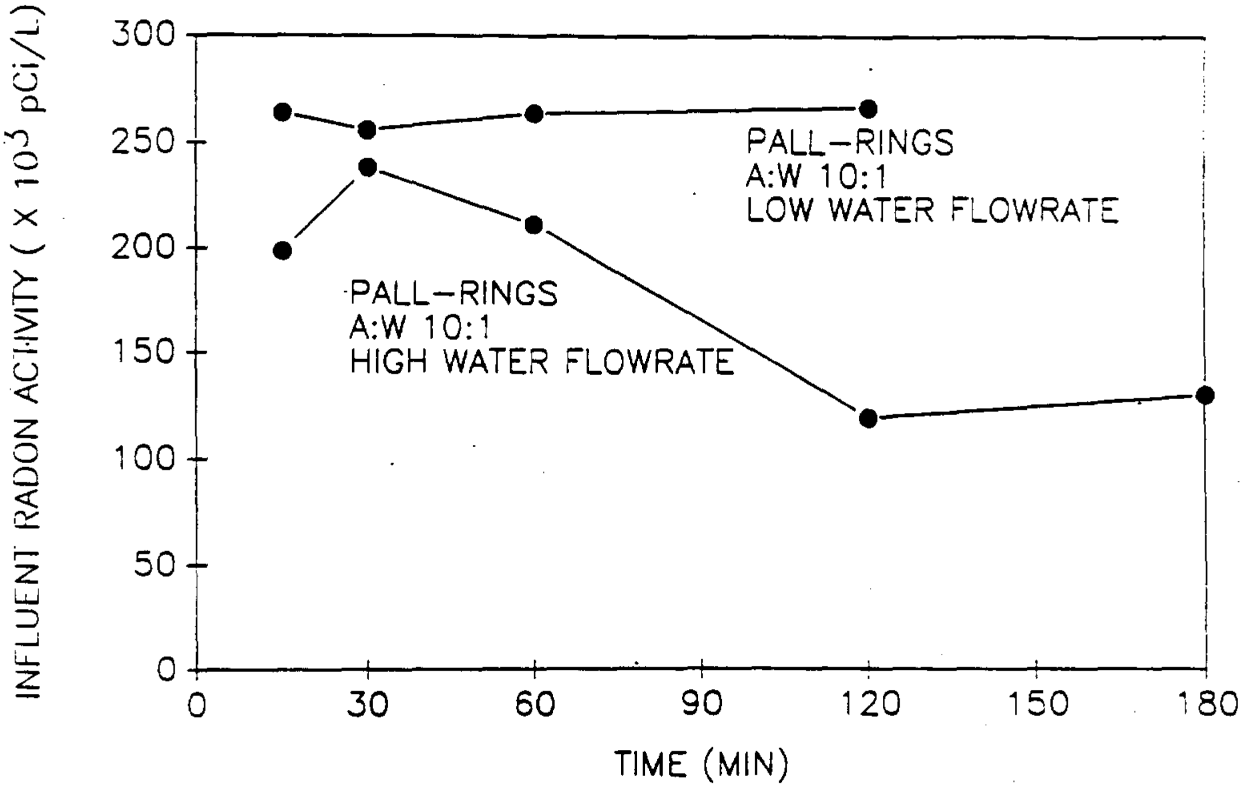

Figure 84. Phase I - Packed tower aeration. Mont Vernon, NH. Influent radon activity as a function of time for two runs.

Though the fluctuations in water flowrate and radon activity prevented comparisons of the effects of A:W ratio and packing type, there was relatively little difference in the overall percent radon removal observed among the conditions tested (Figure 85). For a packed tower aeration system, mass transfer is usually less sensitive to changes in air flowrate than to changes in liquid loading rate and interfacial surface area provided by the media (Kavanaugh and Trussell, 1980). The percent radon removals only varied from 92.7 to 99.8, which is surprising considering the variation in water flowrate (0.18 to 2.6 m^3/hr), radon activity (115,255 to 278,488 pCi/L) and packing type. The resilience of the packed tower system is encouraging considering that many small communities required to install radon treatment systems may experience variations in water flowrate and radon activity similar to those observed at the Mont Vernon site. At the air flowrates tested, the changes in liquid loading and mass loading of radon probably did not impact the effluent activity because (i) radon is a highly volatile gas and thus is readily transferred from water to air in the tower and (ii) the tower (~3.66 m of packing media) was deep enough to compensate for the large variations in loading (i.e., the tower contained far more theoretical transfer units than required since radon is so volatile).

It is important to note that the height of the packed tower (5.49 m) and the range of liquid loading rates (2.5 to 35 m^3/m^2-hr) used in this study were in the lower range of those used in water treatment for volatile organic contaminant removals where typical heights and liquid loadings are 4.3 to 12.2 m and 13.7 to 105 m^3/m^2-hr, respectively. Therefore, the effects of using greater tower heights and/or liquid loadings on effluent radon activity should be considered, particularly for systems required to meet stringent effluent radon design goals.

Performance within Tower Sections--Figure 86, profiles of percent radon removal obtained within the tower during the various runs, shows that a significant amount of the treatment occurred in the top 0.30 m of the mini rings (59.1 - 72.5%) and saddles (45.4 - 53%). In this section, the driving force ($C_L - C_G$) for mass transfer of radon from the liquid concentration (C_L) to the gas phase concentration (C_G) was the greatest. In addition, the end effect of the nozzle spraying the water over the packing probably aided in release of the radon to the air. Likewise, the water flowing through the very top section of the packing was experiencing the most turbulent flow conditions in the tower. It is well documented in the literature (Perry and Chilton, 1983) that end effects in packed tower air strippers resulting from liquid distributors, free fall and turbulence can increase removal of the contaminant. This pattern is less clear with the pall ring (38.9 - 47.3%) data in which the water only passed through 0.15 m of packing before reaching the first sampling port.

Through the next approximately 1.52 m of mini-rings and saddles, the radon removal efficiency decreased, ranging from 23.2 to 42.0%. With the pall rings, the removal efficiency of the middle section was as good or better than the top section (43.0 - 50.6%). It is possible that most of the removal still occurred in the top 0.30 m of the pall rings, but because of differences in packing heights and sampling depths, this could not be determined. The last

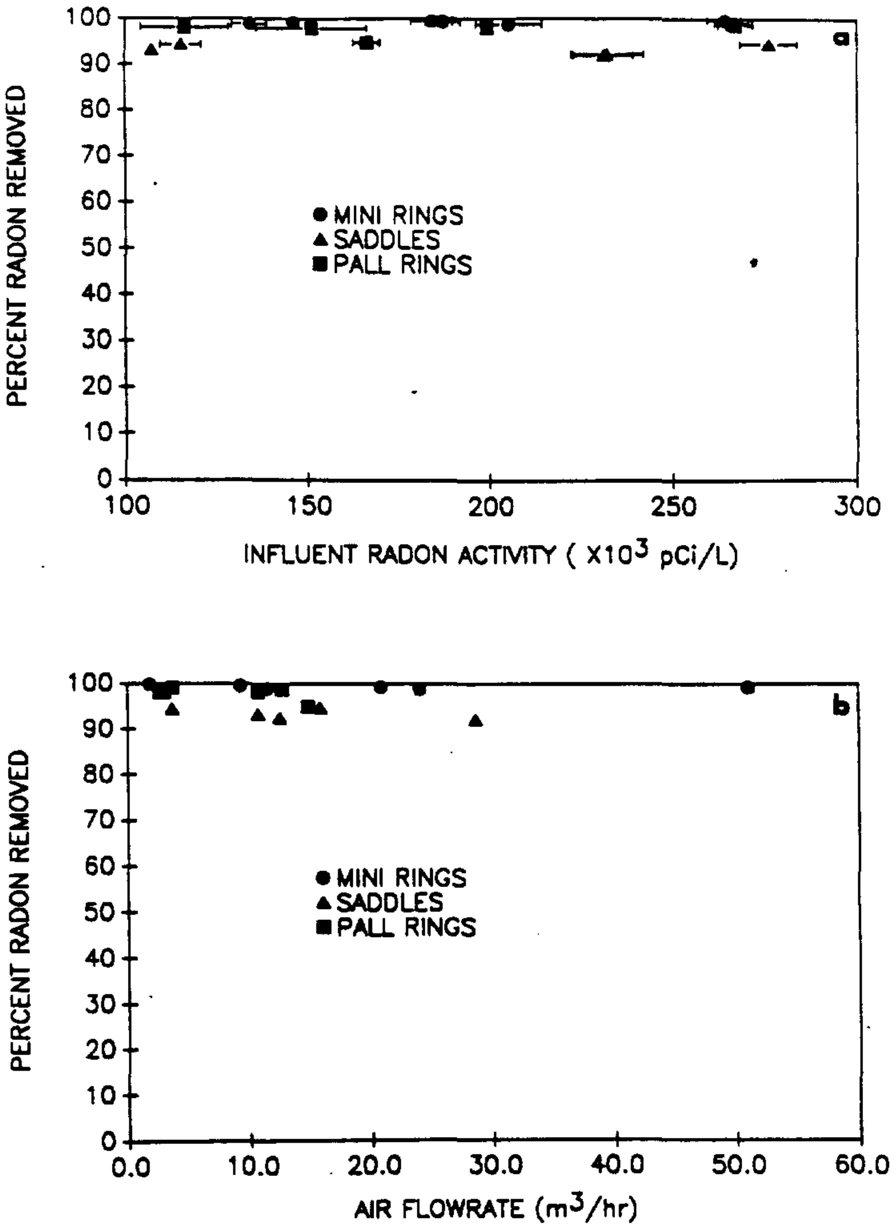

Figure 85. Phase I - Packed tower aeration. Mont Vernon, NH. Percent radon removal as a function of influent radon activity (a) and air flowrate (b) for each condition.

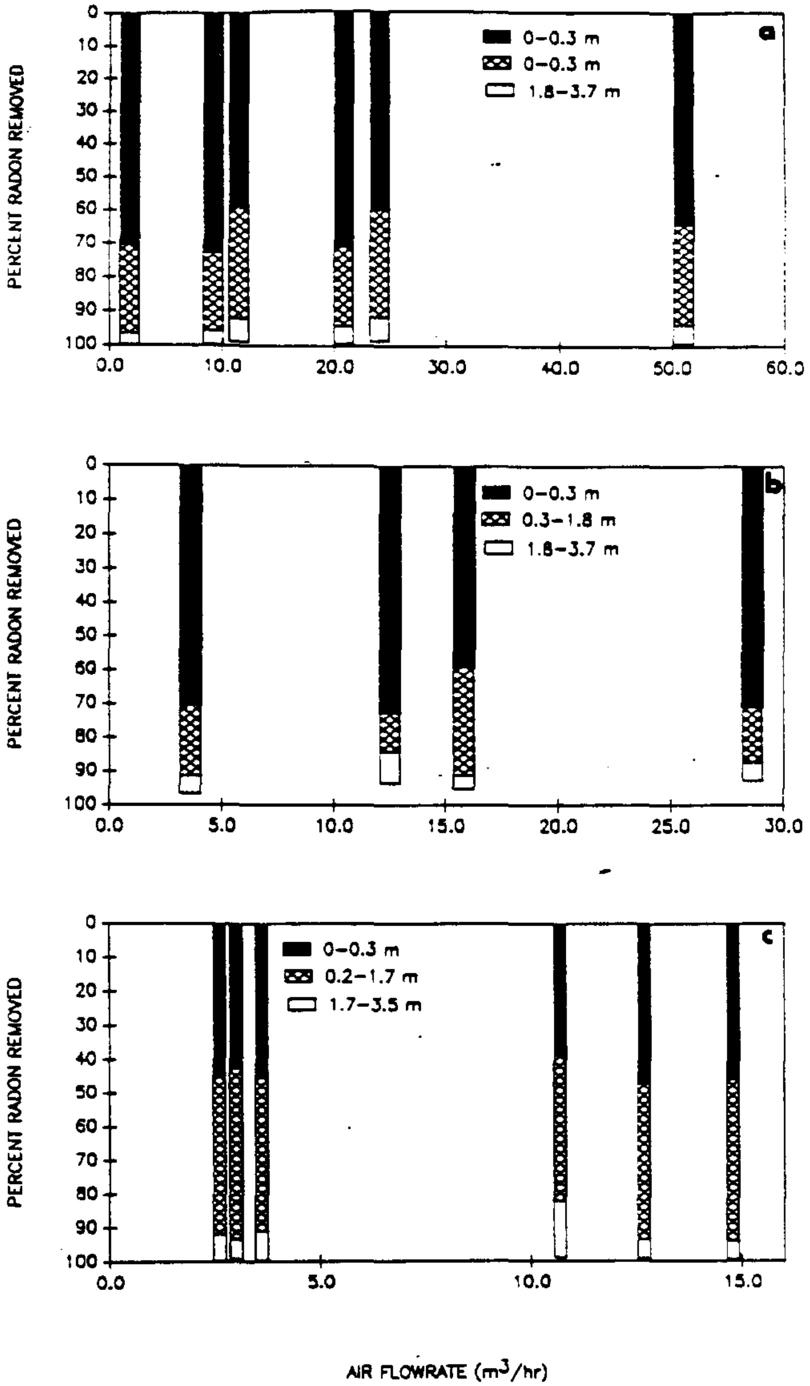

Figure 86. Phase I - Packed tower aeration. Mont Vernon, NH. Percent radon removal as a function of air flowrate within the tower for mini rings (a), saddles (b) and pall rings (c).

1.83 m of packing, in all cases, removed less than 10% of the radon from the water, except with the pall rings at an air flowrate of 10.7 m³/hr (16.9%). The lower radon removal efficiency with depth in the tower probably resulted from the decreased mass transfer due to the lower driving force ($C_L - C_G$). In addition, the liquid flow conditions in these sections were probably far less turbulent than in the top section, especially because there was no redistribution of the liquid flow within the tower at mid-depth. The decreased turbulence can increase the liquid film thickness, which in turn decreases the mass transfer of radon from the water to the air.

Mass Transfer Coefficients--The global liquid mass transfer coefficient, K_La, within a packed tower can be expressed as a function of (i) the molecular diffusion coefficient of the contaminant in water (D_w), (ii) the viscosity (μ_w) and density (ρ_w) of the water, (iii) the liquid loading rate (L), and (iv) empirical constants (m and n) dependent on the type and size of the packing media as shown by the correlation developed by Sherwood and Holloway (1940):

$$K_La = \left[m\left(\frac{L}{\mu_w}\right)^{(1-n)}\left(\frac{\mu_L}{\rho_w D_w}\right)^{1/2}\right] D_w \qquad \text{(eq. 8)}$$

K_La is also dependent on the water temperature which affects μ_w, ρ_w and D_w. Therefore, the K_La within a packed tower operating at a given liquid loading rate with a specific randomly distributed packing is not a function of packing height. However, the K_La for a given system can be calculated based on data collected from continuous flow operation by solving the mass balance-derived packed tower equations:

$$R = HG/P_tL \qquad \text{(eq. 9)}$$

$$NTU = R/(R - 1)\ln[(C_{in}/C_{out}(R - 1) + 1)/R] \qquad \text{(eq. 10)}$$

$$HTU = Z/(NTU) \qquad \text{(eq. 11)}$$

$$K_La = L/(HTU)C_m \qquad \text{(eq. 12)}$$

where Z = packing height, NTU = number of transfer units, R = stripping factor, H = Henry's Constant, P_t = ambient pressure, HTU = height of a transfer unit, L = liquid loading rate, G = air loading rate, K_La = overall mass transfer coefficient, and C_m = density of the liquid. (N.B., the units for Henry's constant must be in atmospheres in this equation).

The K_La values calculated for the top sections of the tower with all of the media (Figure 87) were substantially higher than those for the lower two

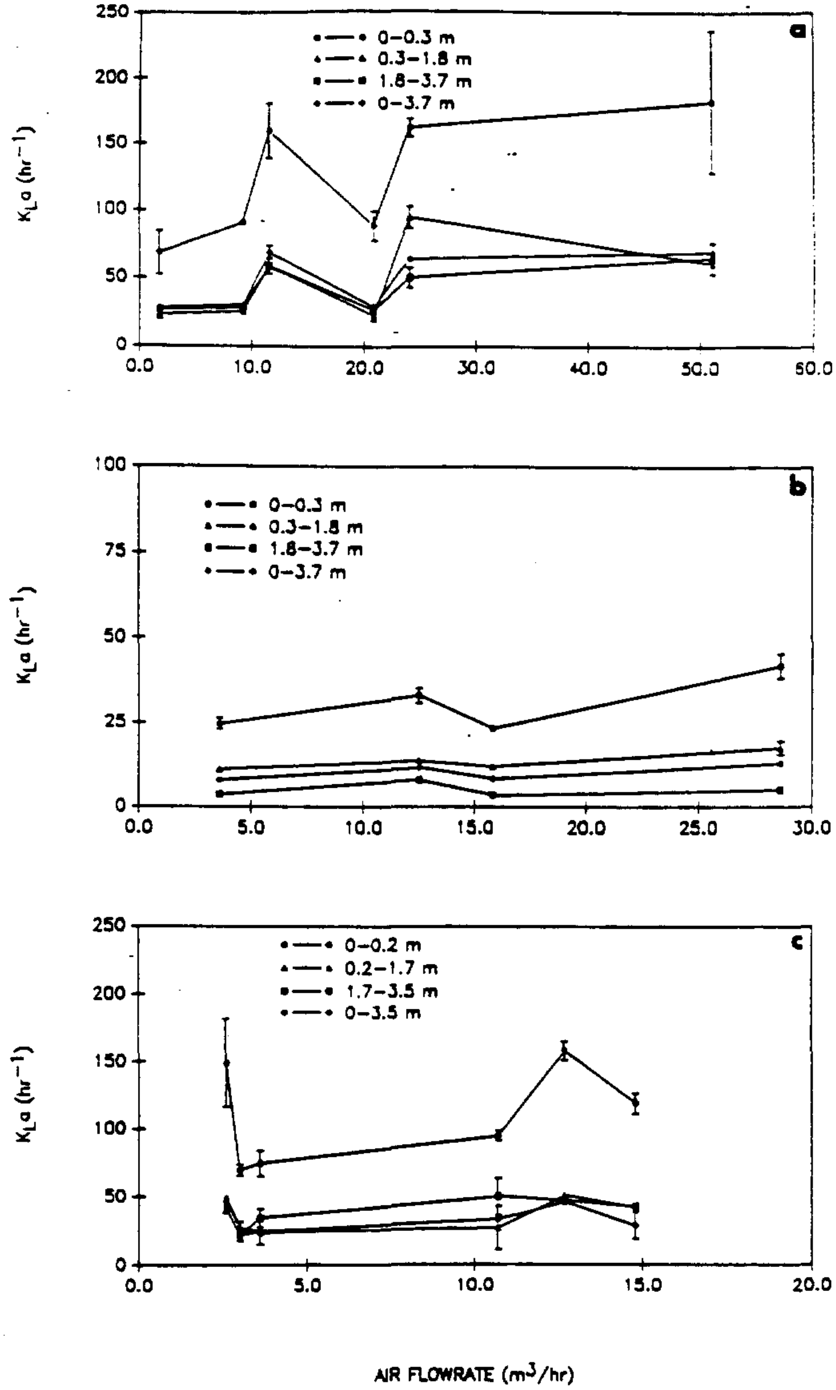

Figure 87. Phase I - Packed tower aeration. Mont Vernon, NH. K_La as a function of air flowrate within the tower for mini rings (a), saddles (b) and pall rings (c). Note differences in scale on ordinate and abscissa.

sections and the overall K_La. (The calculated K_La values only serve as an indication of the true K_La because of the variations in the liquid loading rate during the runs. The values shown were calculated using the average L for each run.) These higher K_La values result from the end effects of the spray nozzle and the turbulent flow in the top section. As seen in Figure 87, the K_La values for the bottom two sections and the overall K_La were very similar and are a better representation of the mass transfer coefficients for the packed tower operating at a given liquid loading rate and with a specific packing medium. (Statistical comparisons were not justified because of the variations in L).

Factors Affecting Packed Tower Operation--The data indicate that mass transfer limitations may be a major factor in designing towers to achieve very low radon activities. These limitations primarily arise from the change in radon concentration (activity) with depth in the tower reducing the driving force. Therefore, in order to achieve low effluent activities, the packing height (and hence, the contact time) would need to be substantial. This was demonstrated by Cummins (1987), who obtained very low effluent activities (<50 pCi/L) with a raw water activity of 4,100 - 6,200 pCi/L at A:W ratios of 2:1, 5:1 and 20:1, but only with a large packing height (5.33 m). For small communities such as Mont Vernon, which have moderate to high radon activities, packed towers may be impractical because mass transfer limitations will dictate the use of extremely large packing heights to meet a 200 pCi/L MCL. As a result of mass transfer limitations, pilot scale testing over the expected range of water flowrates and if possible, radon activities should be conducted in most cases.

The air intake for the blower used at the Mont Vernon site was located inside the building. This was done to decrease problems associated with pumping very cold outside air into the tower during the winter. However, this practice can have repercussions with respect to radon removal efficiency. In the Mont Vernon pumphouse, the air contained as much as 100 pCi/L of radon. Such a high air activity will decrease the driving force (C_L - C_G) and thus the mass transfer of radon, especially when radon activity in the water is low. Therefore, when designing packed tower systems in northern climates, one must evaluate the cost effectiveness of using cold or preheated outside air versus using pumphouse air which may have a significant radon activity.

Though the variations in water flowrate and influent activity prevented detailed statistical analysis of the performance of the 3 packing media, there was a lower overall percent removal observed during Phase I for the saddles (94.4 ± 1.71%) compared with the mini rings (99.2 ± 0.49%) and pall rings (99.3 ± 0.33%). These differences, also observed in the K_La values for the media, occurred even though the range of water flowrates and raw water radon activities tested were similar (Table 21). When the saddles were removed from the tower, it appeared that water was ponding in this packing. Perhaps the ponding occurred because the saddles are primarily solid pieces of plastic,

TABLE 21. RANGE OF INFLUENT RADON ACTIVITIES AND WATER FLOWRATES DURING PHASE I

Packing Medium	Influent Radon Activity (pCi/L)	Water Flowrate (m^3/hr)
Mini Rings	146,630 - 262,753	0.35 - 2.55
Saddles	114,996 - 276,191	0.72 - 2.04
Pall Rings	116,546 - 267,306	0.18 - 1.52

while the mini and pall rings have many openings to facilitate water and air flow. Further testing is probably warranted to determine if there is a significant difference in radon removal efficiency among the packing media.

Phase II - Radon Removal--

Phase II runs were conducted to determine if the packed tower could operate continuously, achieving similar results to those obtained during Phase I. The packed tower was operated on 3 separate days for 6 to 8 hr during the winter of 1988. The water flowrate was much more constant averaging 1.46 ± 0.20 m^3 H_2O/hr. The pall rings were used and the A:W ratio was set at 2:1. The influent activity on the 3 days averaged 181,614 ± 32,781; 202,989 ± 36,429; and 226,422 ± 52,870 pCi/L, respectively. Figure 88 shows overall percent removals of applied radon over time for each of the three days. As observed in Phase I, removal efficiency remained fairly constant regardless of the variation in influent activity. Effluent activities were 4,882 ± 1,659, 3,558 ± 1,311 and 5,173 ± 1,667 pCi/L, respectively. As observed during Phase I with the pall rings, the majority of the radon removal occurred in the top (43.3 ± 2.4%) and middle (46.8 ± 1.3%) sections of the tower (Figure 89), while the bottom 1.83 m removed only a small percentage (7.5 ± 0.41%) of the radon applied. The K_La values were also similar to those obtained at an A:W ratio of 2:1 (pall rings) during Phase I (Figure 90).

The Phase II tests were conducted during the winter, in part, to determine whether colder air and water temperatures would affect performance. During Phase II, air and water temperatures as low as 7°C and 4°C, respectively did not appear to decrease the removal over that obtained during Phase I when the temperatures were 21.2 ± 1.64°C and 13.1 ± 1.4°C, respectively. However, the tower and its piping must be protected from freezing conditions. The tower was enclosed in a wooden structure attached to the Mont Vernon pumphouse. This structure channeled cold ambient air into the building. There were a few extremely cold nights (<-18°C) during the operating period and some of the Schedule 40 PVC piping used to plumb the tower and the glass water flowrate rotometer cracked, even though neither of them contained water. (All water was drained from the system when the tower was not operating.) This experience indicates that in full-scale operation at small communities in northern climates, provisions should insure that pipes and/or towers are insulated to prevent freezing, especially during periods of low or intermittent water flow.

Off Gas Monitoring

Off-gas emissions were monitored to determine the radon activity exiting the tower. In most cases for Phase I and II, the radon activity measured in the air using a Pylon AB-5 monitor was less than the theoretical radon activity (Figures 91 and 92) calculated using a mass balance approach (Q_WC_W/Q_AC_A). There are several explanations for this discrepancy. The wide fluctuations in water flowrate and influent radon activity made calculation of the theoretical radon activity in the off-gas difficult. However, the fluctuations were probably not great enough to account for recoveries consistently less than 50%. There may have been problems associated with the

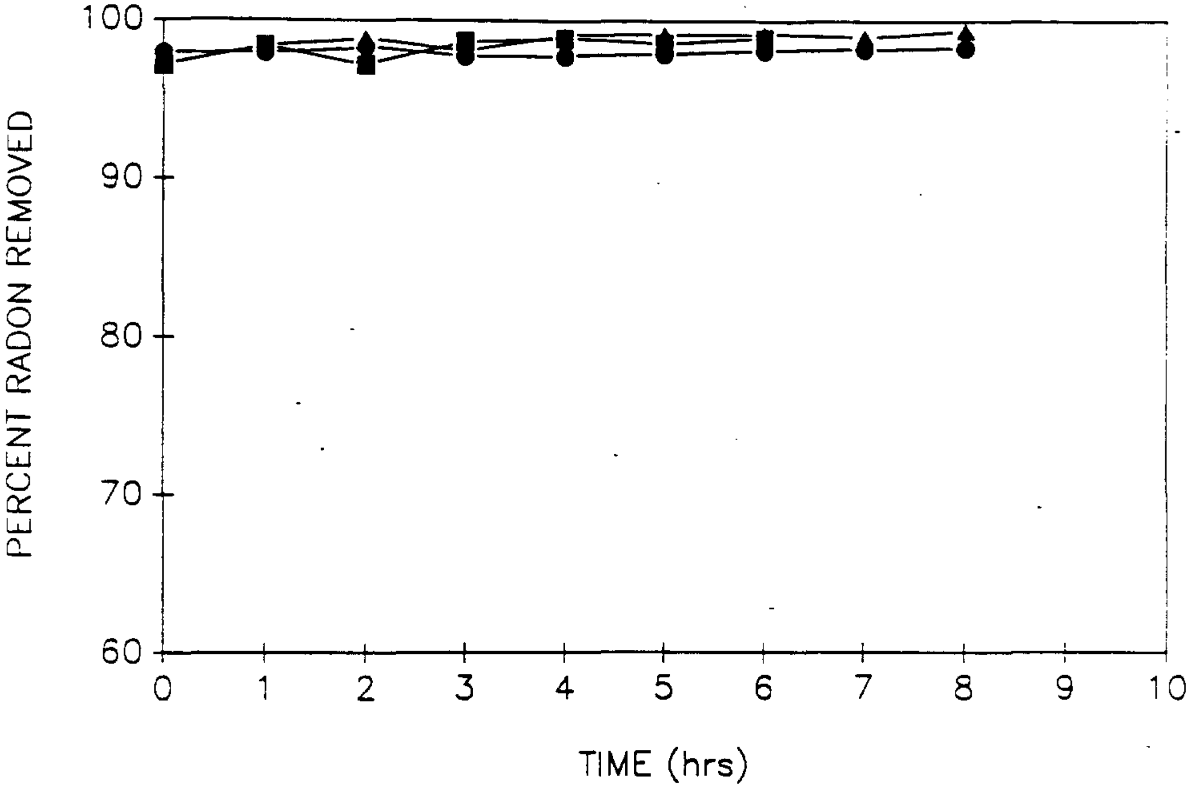

Figure 88. Phase II - Packed tower aeration. Mont Vernon, NH. Percent radon removal as a function of time for each day of testing (pall rings, water flowrate = 1.46 ± 0.2 m^3 H_2O/hr, A:W = 2:1).

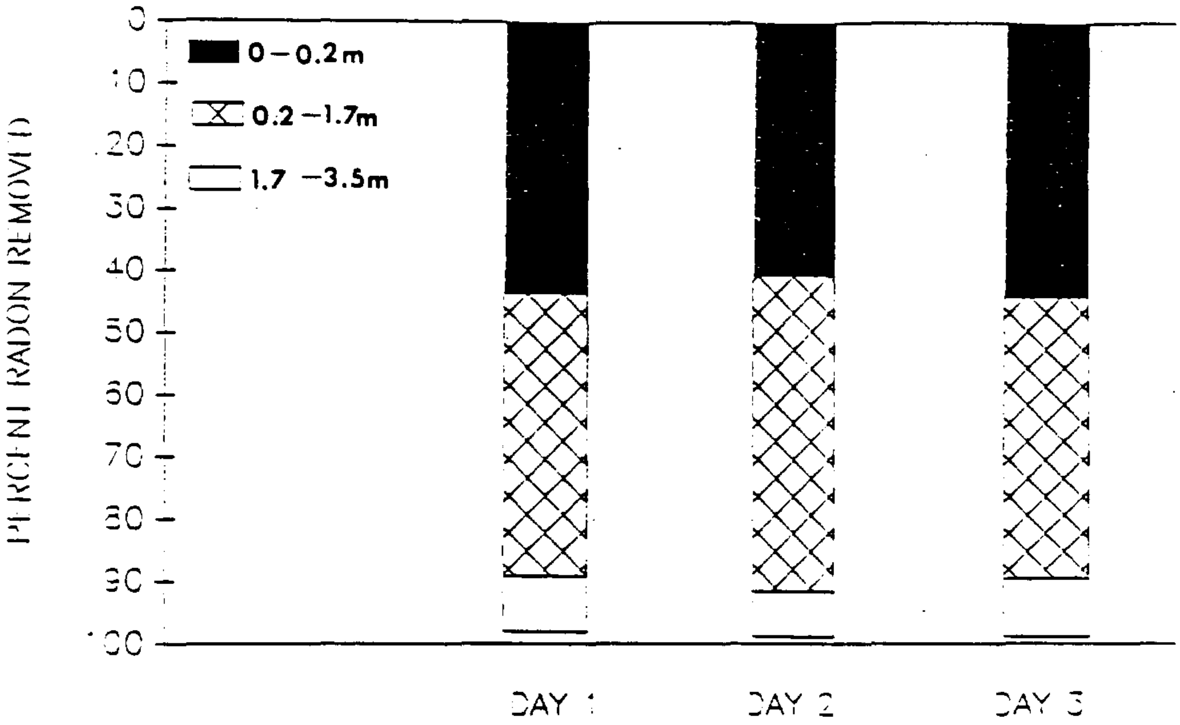

Figure 89. Phase II - Packed tower aeration. Mont Vernon, NH. Percent radon removed within the tower for each day of testing. (Pall rings, water flowrate = 1.46 ± 0.2 m^3 H_2O/hr, A:W ratio = 2:1).

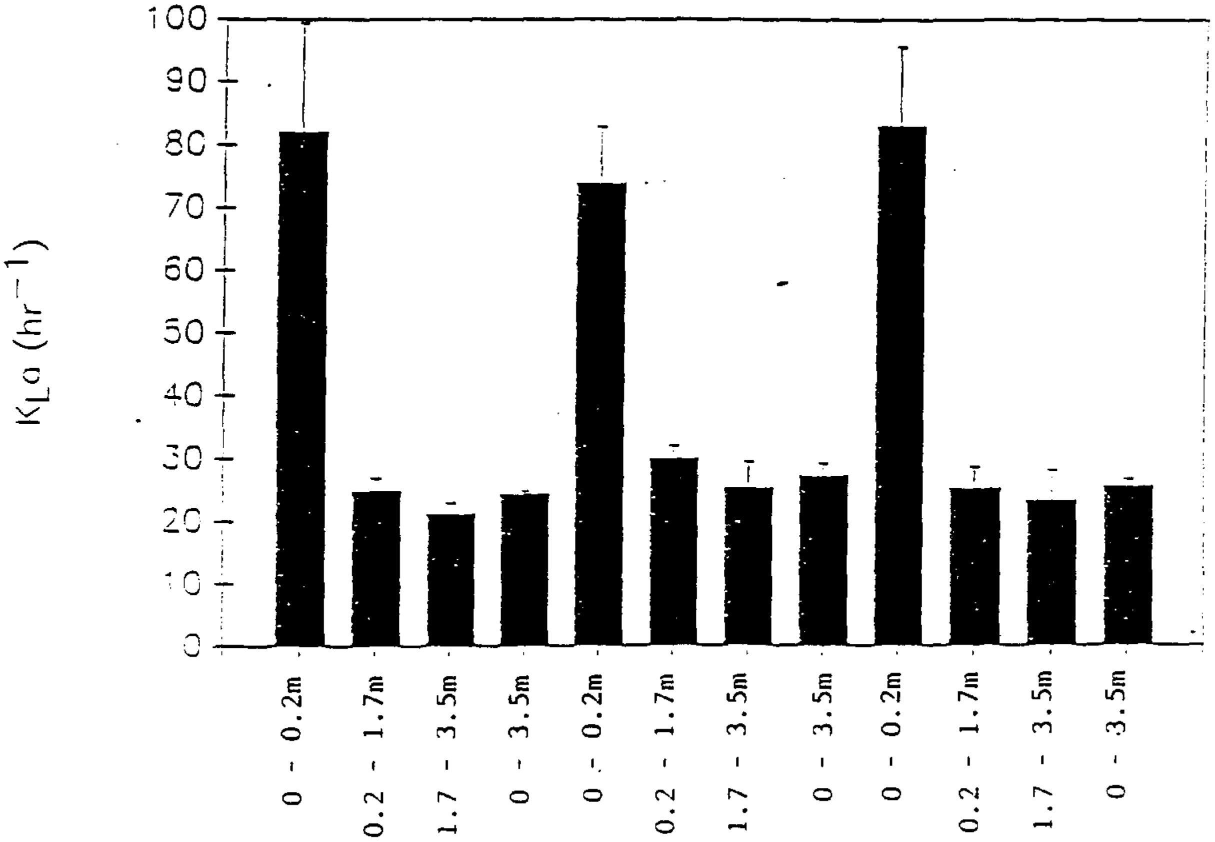

Figure 90. Phase II - Packed tower aeration. Mont Vernon, NH. K_La values within the tower.

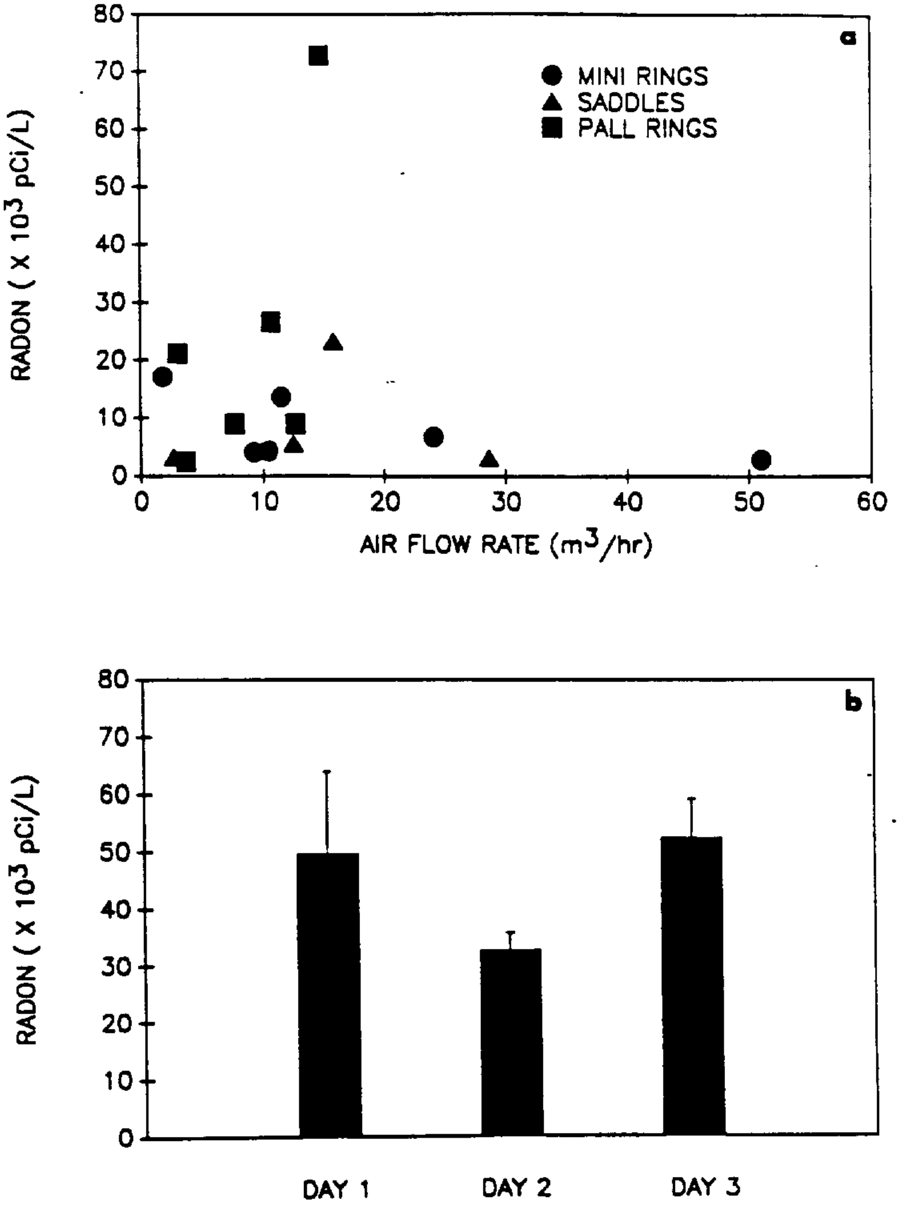

Figure 91. Packed tower aeration - Mont Vernon, NH. Percent off-gas recovery in Phase I as a function of air flowrate for each packing medium (a) and percent off-gas recovery in Phase II (b).

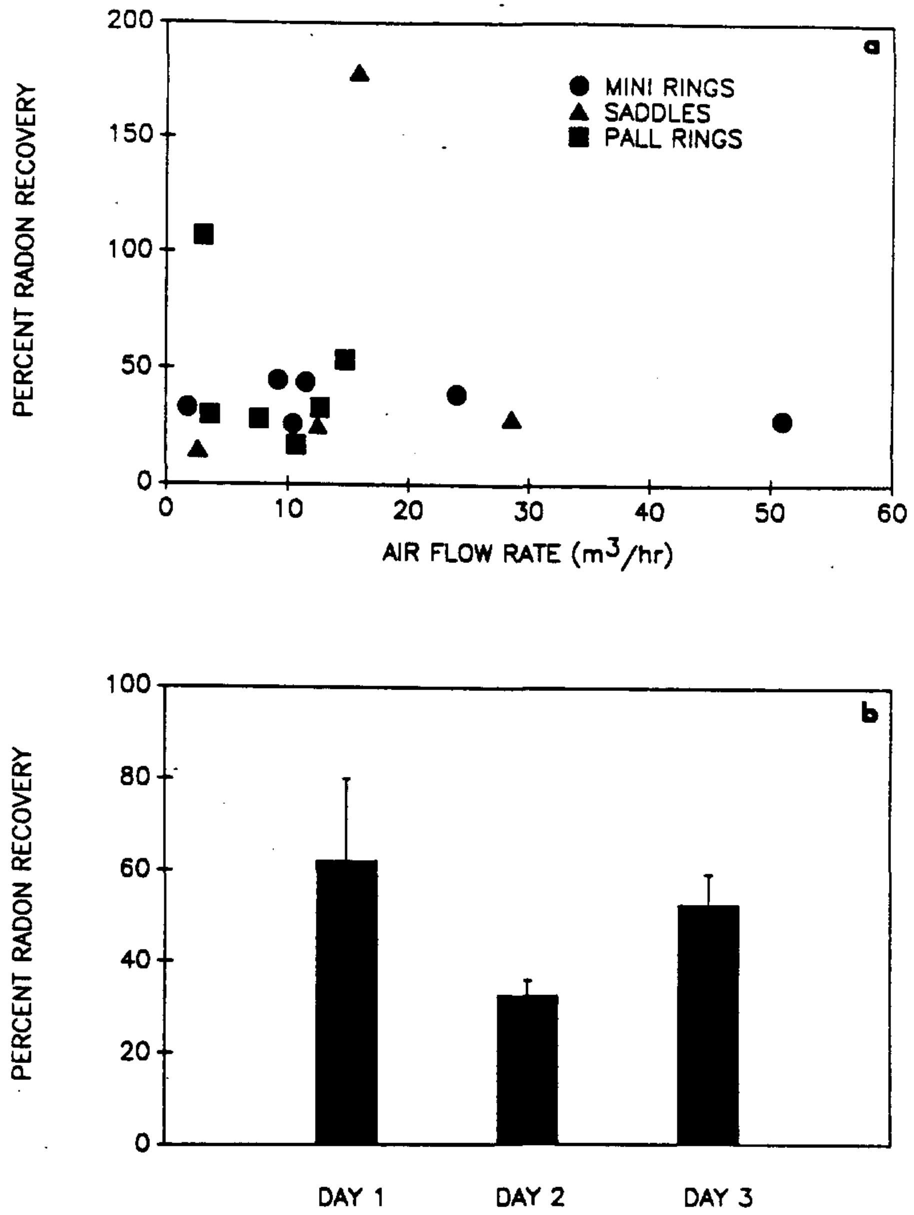

Figure 92. Packed tower aeration - Mont Vernon, NH. Off-gas radon activity in Phase I as a function of air flowrate for each packing medium (a) and off-gas radon activity in Phase II (b).

collection of the air samples. The off-gas exited the top of the tower via a 10.16 cm diameter black plastic hose. The 0.69 cm diameter air sampling tube, attached to the side of the outlet hose, protruded approximately 0.30 m inside it. The inlet of the sampling tube faced into the off-gas flow. It is possible that the boundary layer effects caused the off-gas air to divert around the intake for the Pylon resulting in the low activities recorded.

In all cases, the off-gas radon activities were extremely high (Figures 91 and 92) and would require a large amount of dilution to approach the activity of ambient air (average of 0.1 to 0.15 pCi/L) (NH Bureau of Radiological Health, personal communication, 1986). In future research, the radon activity surrounding the building should be monitored in addition to the emissions at the top of the tower to determine the dilution rate and the effects of the off-gas on air quality.

Iron and Manganese

The influent total iron concentrations ranged from below detection (0.06 mg/L) to 1.70 mg/L. In almost all of the samples taken at Mont Vernon during operation of the packed tower, the influent iron concentrations were below detection. As a result, the effect of the tower on iron oxidation could not be determined. The average total manganese concentrations in the raw water (0.06 ± 0.01 mg/L) did not change significantly through the tower ($\alpha = 0.05$ and 0.01, StT). This is not surprising since the tower's aeration system was not expected to change the total iron and manganese concentrations unless the metals accumulated in the reservoir or within the packing. In future research on the packed tower system, changes in soluble iron and manganese should be determined to see if precipitate formation is occurring. Pretreatment of waters containing significant amounts of iron and manganese may be necessary to prevent accumulation of metal precipitates within the tower system and problems associated with oxidized iron and manganese in the distribution system.

Turbidities, Alkalinities, and pH

Turbidity, alkalinity, and pH were monitored to determine if the packed tower treatment system had any effect on these basic water quality parameters. Influent turbidities ranged from below detection (0.05 NTU) to 0.95 NTU during Phase I and from 0.05 to 2.80 NTU during Phase II. Effluent values ranged from below detection to 0.80 NTU for Phase I and from 0.05 to 2.00 NTU during Phase II, which was not significantly different from the influent ($\alpha = 0.05$ and 0.10, StT). This was expected since the media were not expected to contribute to the turbidity of the water nor were they expected to remove a significant amount of turbidity from the water.

Influent alkalinities to the packed tower ranged from 18 to 32 mg/L as $CaCO_3$ during Phase I and from 20 to 28 mg/L as $CaCO_3$ during Phase II, while effluent alkalinities ranged from 19 to 30 mg/L as $CaCO_3$ during Phase I and from 13 to 24 mg/L as $CaCO_3$ during Phase II. No significant difference was detected between the influent and the effluent alkalinity ($\alpha = 0.05$ and 0.10, StT), except for two runs where the differences were statistically significant at $\alpha = 0.10$, but the change in alkalinity was less than 0.7 mg/L as $CaCO_3$. Changes in alkalinity due to air stripping were not expected at Mont Vernon based on raw water characteristics.

At all conditions, the tower raised the pH of the water significantly ($\alpha = 0.05$ and 0.01, StT). Influent pH values ranged from 6.3 to 6.9 during Phase I and from 6.4 to 6.7 during Phase II, while effluent pH values ranged from 6.6 to 7.7 during Phase I and from 6.9 to 7.5 during Phase II. The aeration process within the packed tower probably caused the increase in pH by removing CO_2 from the water. Even though the pH was raised significantly, the resulting effluent pH was not outside the 6.5 - 8.5 acceptable range for drinking water.

ECONOMICS

Cost estimates for the packed tower treatment system operated in Mont Vernon, NH, were developed using the approach described in Section 6 and are presented in Table 22. These data indicate that the total production cost for the packed tower system was $0.60/1,000 gallons. If pretreatment for iron and manganese is required then production cost estimates would increase to $2.10/1,000 gallons (Section 4). The production cost for the packed tower system is comparable to that reported by Malcolm Pirnie, Inc. (1988). The latter report concluded that the production cost to treat an average flow of 49 m^3 H_2O/day (227 m^3 H_2O/day design flow) would be $0.58/1,000 gallons for 99% radon removal.

It should be noted that a major component of the packed tower system cost at Mont Vernon is related to support equipment costs. Since support equipment includes a variety of site specific factors (e.g., installation), one would expect this cost to be highly variable from site to site. In addition, performance data for the packed tower system indicates that this facility can be operated at flows up to 61 m^3 H_2O/day (i.e., a continuous flowrate of 2.54 m^3 H_2O/hr and still achieve radon removals comparable to those achieved at the 38 m^3 H_2O/day design flow. Therefore, the production cost of the packed tower system may decrease as design flows increase (for the range of flows observed in this study).

In addition, as was discussed for the diffused bubble system (Section 5), the cost estimates for the packed tower aeration system only include venting the radon contaminated off-gas to the air at a height equivalent to the average home roof line (i.e., 10.7 m). Therefore, if treatment of the off-gas for radon is required to meet air standards, system costs could increase significantly.

TABLE 22. COST ESTIMATE FOR THE PACKED TOWER SYSTEM AT MONT VERNON, NH

Item	Total Cost (2nd Quarter 1989 dollars)[*]
CAPITAL COSTS	
DIRECT COSTS	
Process Equipment	
1 - Stainless steel column	$ 511
10 cu. ft. plastic packing media (\$668/m^3)	189
1 - Air flow meter	47
1 - Bete fog nozzle	28
1 - Demister pad	102
1 - Air blower	348
2 - Air pressure gauges (\$34.86/gauge)	70
1 - Air vent assembly	100
4 - Temperature gauges (\$22.62/gauge)	91
1 - Booster/transfer pump	800
1 - Rockwell flow rate recorder/totalizer	1,036
Total Process Equipment Cost	$ 3,322
Support Equipment	
Installation[+]	2,350
Plumbing	1,133
Electrical	1,381
Total Support Equipment Cost	$ 4,864
TOTAL DIRECT COST	$ 8,186
INDIRECT COSTS	
Sitework	$ 1,228
Engineering	1,228
Contractor Overhead and Profit	982
Legal and Financial Fees	205
Interest	491
Contingencies	$ 1,228
TOTAL INDIRECT COST	$ 5,362
TOTAL CAPITAL COST	$13,548
AMORTIZED CAPITAL COST (Annual)	$ 1,484

(continued)

TABLE 22 (continued)

Item	Total Cost (2nd Quarter 1989 dollars)*
ANNUAL O&M COSTS	
Power	232**
Maintenance	
Mechanical	252
Non-mechanical	32
Labor	110
Administrative	93
TOTAL ANNUAL O&M COST	$ 719
TOTAL ANNUAL COST	$ 2,203/year
TOTAL PRODUCTION COST (Design flow 37.85 m^3)	$0.60/1,000 gallons
PRETREATMENT COST	$1.50/1,000 gallons
TOTAL PRODUCTION COST WITH PRETREATMENT	$2.10/1,000 gallons

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[*] Based on ENR Construction Cost Index, Base Year 1967 = 100.

[+] Installation includes cost of poured concrete base for tower and cost to remove and reroute existing utilities to facilitate new construction.

[**] Power costs are based on input power of 0.5 Hp for the blower and 1 Hp for the pump, an operating period of 5.6 hr/day for each, and an electric rate of $0.10157/kW-hr.

SECTION 7

LOW TECHNOLOGY ALTERNATIVES

INTRODUCTION

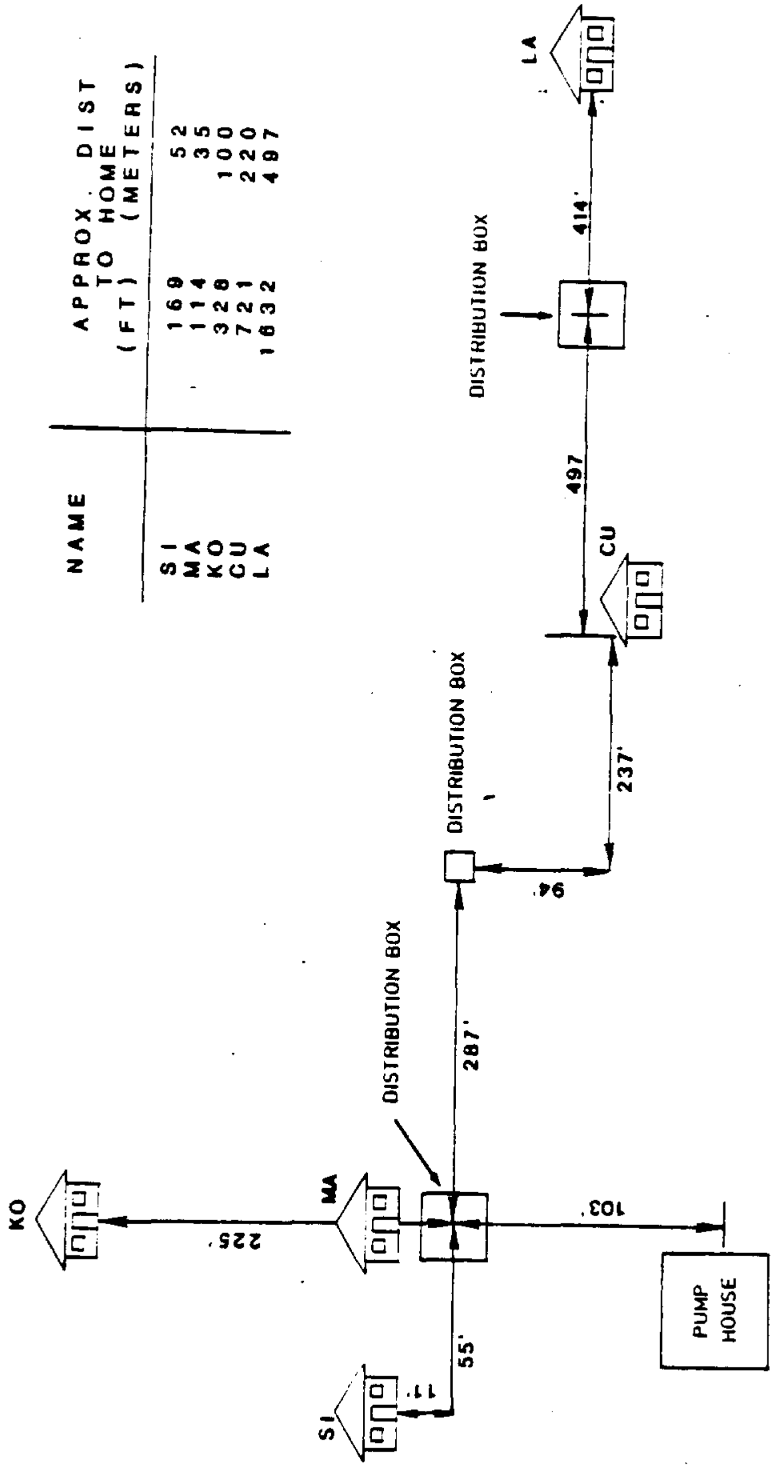
Many very small communities, required to treat their groundwater supplies for radon, will face severe financial problems paying the capital and O&M costs associated with installing GAC, diffused bubble or packed tower aeration systems. The objective of this part of the study was to evaluate the potential of some low technology treatment alternatives to reduce radon levels in drinking water. Low technology systems should (i) be easily retrofit into existing facilities, (ii) require little initial capital investment, (iii) require little routine maintenance, and (iv) have lower operating costs than the conventional radon technologies. In a typical small community water supply in rural New England, water is pumped to an atmospheric storage tank (7571-37,854 L) and then repressurized in a second storage tank (7571-37,854 L). The existing atmospheric storage tank could be readily adapted to low technology radon removal techniques such as coarse bubble aeration. The study included pilot-scale experiments monitoring radon reduction in (i) a distribution system and (ii) in a flow-through reservoir system with a variety of influent control devices and a coarse bubble (>500 μm diameter) aeration system. During this phase of the project, radon was the primary contaminant of concern and thus the techniques evaluated were monitored exclusively for radon removal.

METHODS AND MATERIALS

Loss in a Distribution System

This study was conducted at the Mont Vernon, NH site. The influent radon concentration during this phase of the project ranged between 150,000 to 250,000 pCi/L. The GAC unit was treating the water at Mont Vernon during this period and the GAC effluent ranged between 22,000 to 27,000 pCi/L. The exact design of the distribution system at the site was not available, however, it was determined that the pipes ranged from 3.8 to 5.1 cm diameter (I.D.) and were constructed of black polyethylene. The network (Figure 93) serves approximately 33 mobile homes. Distances between the homes and connections were approximated using a surveyor's tape.

The loss of radon in the distribution system was assessed with both untreated and GAC-treated water at 5 homes (Figure 93). The homes, where samples were collected, were chosen to give an overall assessment of the potential effectiveness of passive removal of radon in the distribution system tested.



NAME	APPROX. DIST TO HOME (FT)	APPROX. DIST TO HOME (METERS)
SI	169	52
MA	114	35
KO	328	100
CU	721	220
LA	1632	497

Figure 93. Schematic of the Mont Vernon, NH distribution system.

Radon samples were taken in duplicate from the kitchen faucet in each home after all aeration devices were removed. The first two sets of data were collected when the homes were receiving GAC-treated water. When the untreated water test was conducted, the GAC system was by-passed beginning 1.5 to 3 hr before sampling commenced. This lag time insured that the distribution system was completely flushed of treated water. Samples were always taken between 1700 and 1930 hr because of the higher water use during that time and for the convenience of the homeowners. The high use period was chosen because it would simulate the worst case scenario (i.e., shortest detention time in the system).

Loss During Storage

A 1.8 m diameter fiberglass tank was placed outside of the pumphouse at the Derry, NH site. During the study period, a centrifugal pump was used to continuously supply water from the atmospheric storage tank at Derry to the pilot scale fiberglass storage tank located approximately 13.7 m away. In-line brass flow controllers were installed in the pump's effluent line. Three flow controllers were used during the course of the study (1.3 m^3/day, 3.3 m^3/day, and 4.9 m^3/day).

The storage reservoir (Figure 94) was filled to yield a water depth of 0.6 m (total volume = 1628 L as determined in a dye study). The reservoir was covered with a plywood lid which had a hole (0.3 m x 0.30 m) in the center to allow for overhead entry of raw water. A plywood baffle (0.91 m x 0.96 m) was suspended in the water near the effluent port to prevent short circuiting. The modes of influent entry during the study were: (i) entry at the bottom of the reservoir, (ii) discharge 0.61 m above the reservoir water level, (iii) discharge 0.61 m above the reservoir water level using a spray attachment and (iv) discharge 0.61 m above the reservoir level through a venturi apparatus to add air to the stream. In addition to the four tests with varied types of influent entry, a coarse bubble aeration system was added to entry types (i) and (ii). Influent flowrates of 1.3 m^3/day, 3.3 m^3/day and 4.9 m^3/day (theoretical detention times of approximately 8, 12, and 30 hr, respectively) were tested for all operating conditions. Duplicate influent and effluent water samples were taken at 0.5 hr intervals over a 3 hr period and were taken only after the system had been operated for at least 3 detention times at each condition.

A dye test was conducted to determine the hydraulic detention time of the low technology tank. Rhodamine B was added to the tank, after it had filled with raw water, and the contents were mixed with a large paddle to achieve homogeneity. Then the flow of water through the system was started and a time = 0 sample of the effluent and tank contents was taken. The effluent dye concentrations were measured, using a Turner Model C10-005 fluorometer (Sunnyvale, CA), after 0.25, 0.5, 0.75, 0.88, 1 and 1.5 theoretical detention times. Studies were conducted using the 1.3 m^3/day and 4.9 m^3/day flow controllers.

The spray attachment (entry mode iii) was a standard garden hose screw-type adjustable nozzle. The diameter of the spray was set at

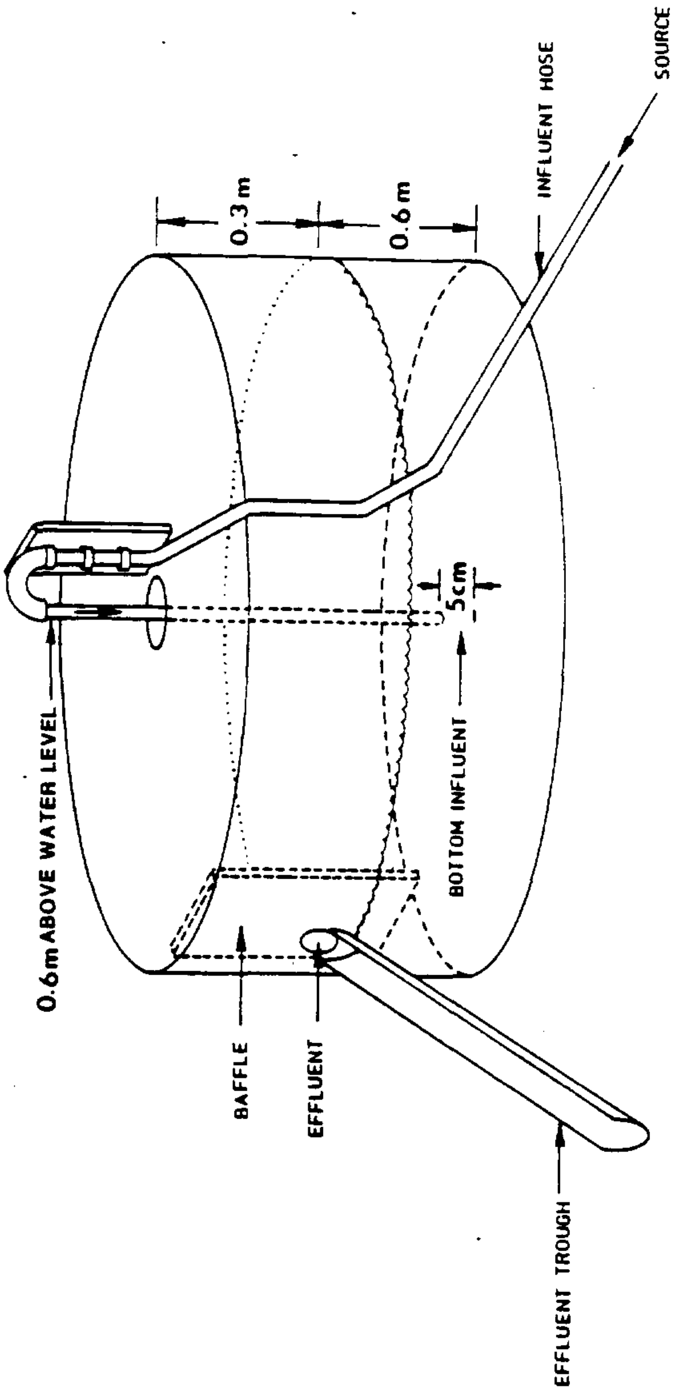


Figure 94. Schematic of the pilot-scale atmospheric storage tank. Derry, NH.

approximately 15 cm when it reached the water's surface. At the low flow of 1.3 m^3/day, a fine spray could not be achieved and thus data are not available for that condition. In tests where the coarse bubble aeration system was used, a laboratory-made apparatus was added to the reservoir. Air was supplied by a small laboratory air pump (capacity = 0.031 m^3/min, max. pressure = 414 kPa) and pumped through 0.64 cm (I.D.) clear plastic tubing. The main air line to the storage reservoir was subdivided into four 81 cm symmetrical radial arms using plastic connectors. The radial arms were plugged at the terminal ends with small corks. Small holes were pressed in the wall of the radial arm tubing, every 10 cm, using a steel-plated 11 mm long thumbtack which made 635 µm (I.D.) holes. The bubbles emanating from the tubing appeared to be approximately 4.8 mm diameter and were produced at an average rate of 2 to 3 per second from each hole. The venturi device (entry mode iv) used was designed for a laboratory faucet. The lowest flow accommodated by commercially available venturi devices was 27 m^3/day and therefore, could not be retrofit to the reservoir system.

RESULTS AND DISCUSSION

Loss in a Distribution Systems

The results obtained from the distribution system evaluations are summarized in Table 23. The overall loss during distribution was low (0 to 18.8%), over a distance of 497 m (hydraulic detention time 0.7 to 17.9 min). The greatest reduction (15.6%) in radon activity occurred at the home located furthest from the pumphouse and therefore where the water had the longest hydraulic detention time in the system. The differences observed, however, were not always statistically significant (Table 24).

During the evaluation with the untreated water, faucets were allowed to flush for several minutes, but it is possible that at site KO an insufficient time was allowed to completely clear the piping and the very small home storage tank. During some sampling, especially at sites KO and CU, it was very difficult to sample from the faucet without entraining air in the sampling syringe, even when using the universal hose connector suggested by EPA (1978).

The loss of radon due to decay alone would be minimal in the distribution system at Mont Vernon because of the short detention time (0.7 to 17.9 min) at the 82 m^3/day water flowrate. The community's daily flowrate is usually a minimum of 27 m^3/day, with a daily maximum of 82 m^3/day. At these flow conditions the loss of radon from decay would range from <0.01% to 0.67% in the homes sampled. Although the sampling techniques may have accounted for some of the observed radon removal, the mobile home park is known to have many leaks in its distribution system which may account for the radon loss.

The data suggest that flow in a small community distribution system (3.8-5.1 cm diameter pipes) at distances up to 497 m and detention times of 18 min or less will not significantly reduce the radon activity of the raw water. It is doubtful that this mechanism would be useful in lowering the radon levels to meet proposed standards unless a very small decrease was needed (10 to 20%). However, data were collected at Mont Vernon during periods of high

TABLE 23. LOSS IN THE DISTRIBUTION SYSTEM

Site	Distance (m)	Hydraulic Detention Time* (min)	Percent Radon Removal RUN 1+	RUN 2	RUN 3
MA	35	0.7-1.2	0	0	2.1
SI	52	1.0-1.8	**	**	N/A
KO	100	2.0-3.6	10.8	7.3	++
CU	220	4.4-7.8	4.1	9.2	11.8
LA	497	10.0-17.9	15.6	**	7.5

* Hydraulic detention time range for 82 m^3/day water flowrate assuming 3.8 cm and 5.1 cm diameter piping.

\+ Runs 1 and 2 with GAC-treated water. Run 3 with untreated water.

** Radon level slightly higher than influent.

N/A Homeowners not available during sampling.

++ Invalid sampling procedure (inadequate flushing or sample aeration).

TABLE 24. STATISTICAL COMPARISON[*] OF RADON REMOVALS WITH TIME IN A DISTRIBUTION SYSTEM

Site	Hydraulic Detention Time[**] (min)	Statistical Comparisons vs. System Influent[+]		
		RUN 1[++]	RUN 2[++]	RUN 3[++]
MA	0.7-1.2	INF = MA (α=0.05&0.10)	INF = MA (α=0.05&0.10)	INF = MA (α=0.05&0.10)
SI	1.0-1.8	INF = SI (α=0.05&0.10)	INF = SI (α=0.05) INF>SI (α=0.10)	N/A
KO	2.0-3.6	INF > KO (α=0.05&0.01)	INF > KO (α=0.05) INF = KO (α=0.01)	N/A
CU	4.4-7.8	INF > CU (α=0.05) INF = CU (α=0.01)	INF > CU (α=0.05) INF = CU (α=0.01)	INF > CU (α=0.05&0.01)
LA	10.0-17.9	INF > LA (α=0.05&0.01)	INF = LA (α=0.05) INF < LA (α=0.10)	INF > LA (α=0.05) INF = LA (α=0.01)

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[*] As assessed by Student's t test. α: significance level.

[+] INF = Radon activity entering the water distribution system.

[**] Hydraulic detention time range for 82 m^3/day water flowrate, assuming 3.8 cm and 5.1 cm diameter piping.

[++] Runs 1 and 2 = GAC-treated water, Run 3 = untreated water.

N/A Invalid sampling procedure or homeowner not available.

flow and thus probably represented the shortest detention time in the system. Though small reductions in radon are anticipated in short distribution systems, the removal rate would vary with water usage. Removal of radon could be higher if the water was retained in the system for a period of time, however, that condition could not be relied upon to consistently deliver water of acceptable quality.

Loss During Storage

A series of dye studies were completed on the flow-through reservoir. Tests were conducted to determine if the tank was well mixed. The data indicate that the hydraulic detention time at the high flowrate ($Q = 4.5\text{-}4.8$ m^3/day) was very close to the theoretical time of 8 hr (8.27 ± 0.40 hr) Figure 95). There was no stratification within the tank as a function of T°C or dye concentration at this flowrate. At the low flow condition ($Q = 1.2\text{-}1.4$ m^3/day), however, the calculated detention time from the dye study was 23.3 ± 6.72 hr, as contrasted to the theoretical time of 30 hr (Figure 96). The detention time for the three separate dye studies ranged from 18.9 to 30.3 hr. Though the water volume of the tank remained constant (2,043 L), the T°C and dye concentrations varied with depth in the tank, indicating that flow was short circuiting across the surface of the tank. In addition, the 1.3 m^3/day flow controller tended to clog with fine particles even when a sediment trap was used. As a result, the flow slowly decreased with time. Possibly account for the larger detention time during the second trial. This implies that the data obtained during subsequent studies for the reservoir using the 1.3 m^3/day flow controller probably underestimated the removal attainable at a true detention time of 30 hr. A dye study was not run using the 3.3 m^3/day flow controller, however, it was observed during the loss studies to respond similarly to the 4.9 m^3/day controller therefore, it is assumed that no problems with clogging or short-circuiting occurred at this flow condition.

The data collected during the pilot-scale flow-through reservoir studies at Derry are summarized in Figures 97, 98 and 99. The radon removals achieved at the different detention times were significantly different for all conditions ($\alpha = 0.05$ & 0.01, StT) (i.e., removal always significantly higher with increased detention time) except for the venturi. However, results from the different flowrates should not be directly compared because of the potential problems with short-circuiting at the 1.3 m^3/day flowrate. The reduced removals with the venturi may have been a function of the fact it was designed for laboratory use as an aspirator. In addition, the flowrates tested were probably too low to generate good venturi action (aeration), preventing better removals from being observed in the venturi runs. Testing should be performed at higher flowrates with a commercially-available device to thoroughly evaluate the venturi's ability to enhance radon removal.

Losses for the bottom entry of the influent into the reservoir were very similar to the removals expected as a result of decay and volatilization of radon. However, in all cases where the influent had a 0.61 m free fall upon entry into the tank, radon removals were much greater than those predicted from decay alone or bottom entry without aeration. (N.B., statistical analyses of the data were not warranted because the influent radon activity changed among some of the experiments at a given flowrate from 58,641 to

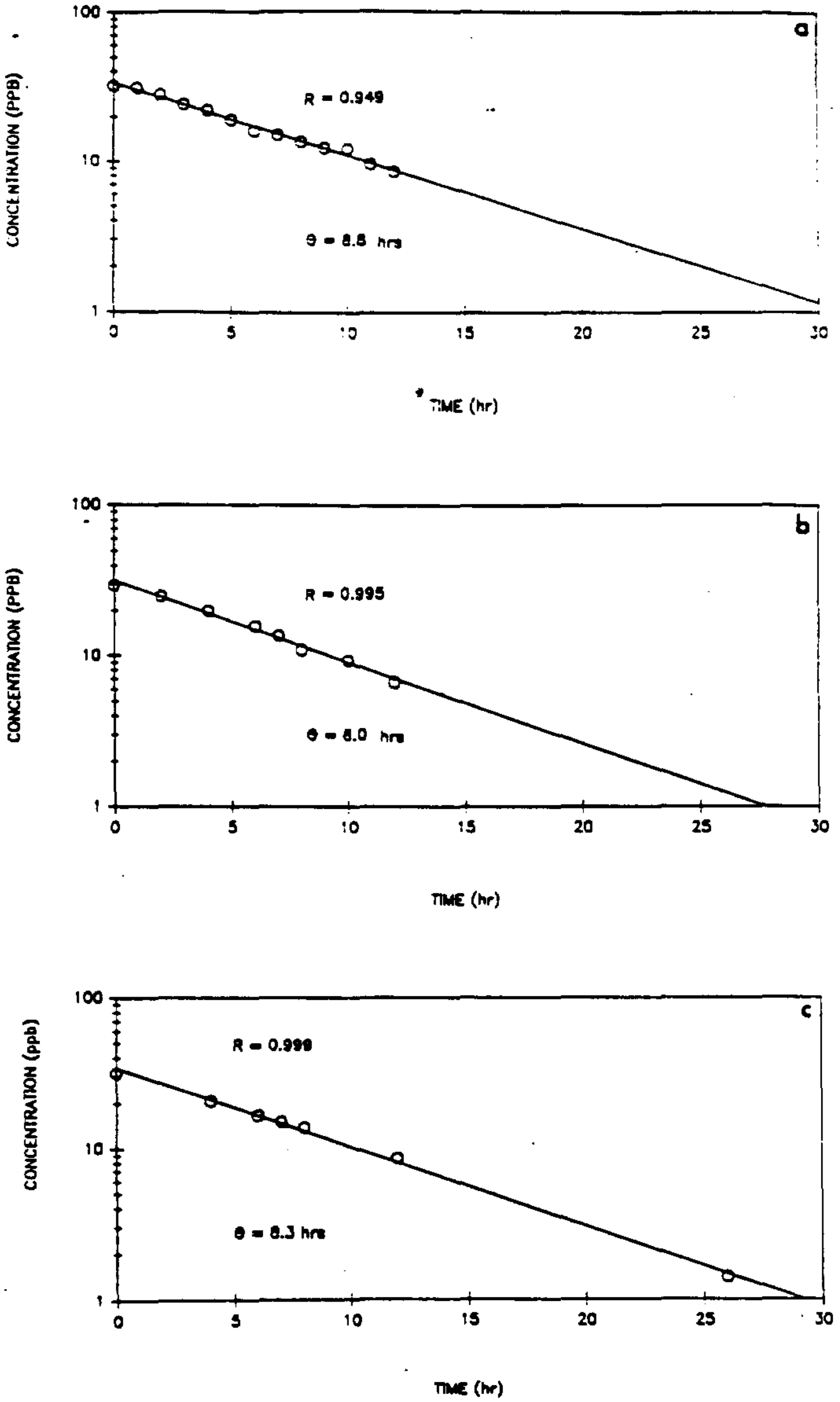

Figure 95. Low technology alternatives - Derry, NH. Dye concentration in pilot-scale atmospheric storage tank with free fall as a function of time. Runs 1 (a), 2 (b), and 3 (c). (Water flowrate = 4.9 m^3/day).

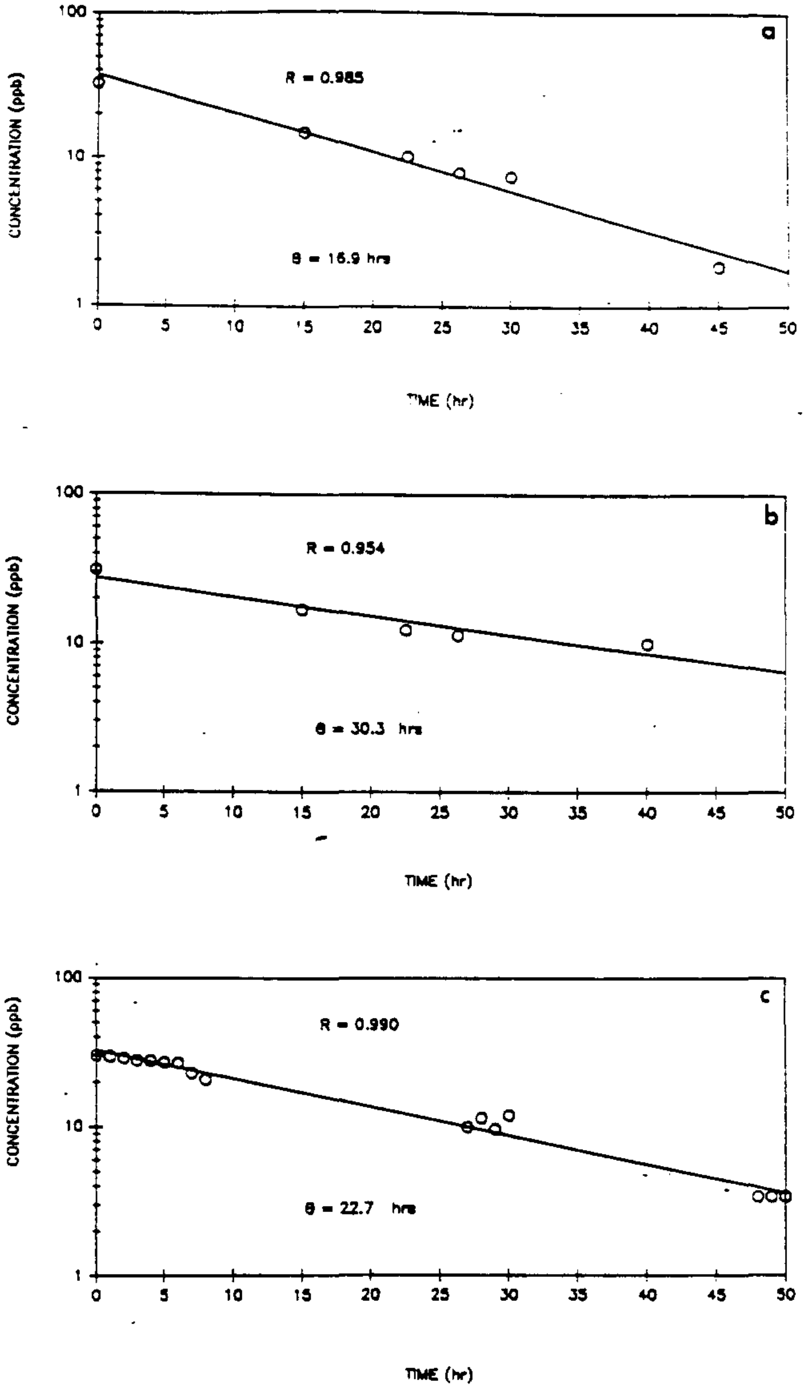

Figure 96. Low technology alternatives - Derry, NH. Dye concentrations in the pilot-scale atmospheric storage tank with free fall as a function of time. Runs 1 (a), 2 (b) and 3 (c). (Water flowrate = 4.9 m^3/day).

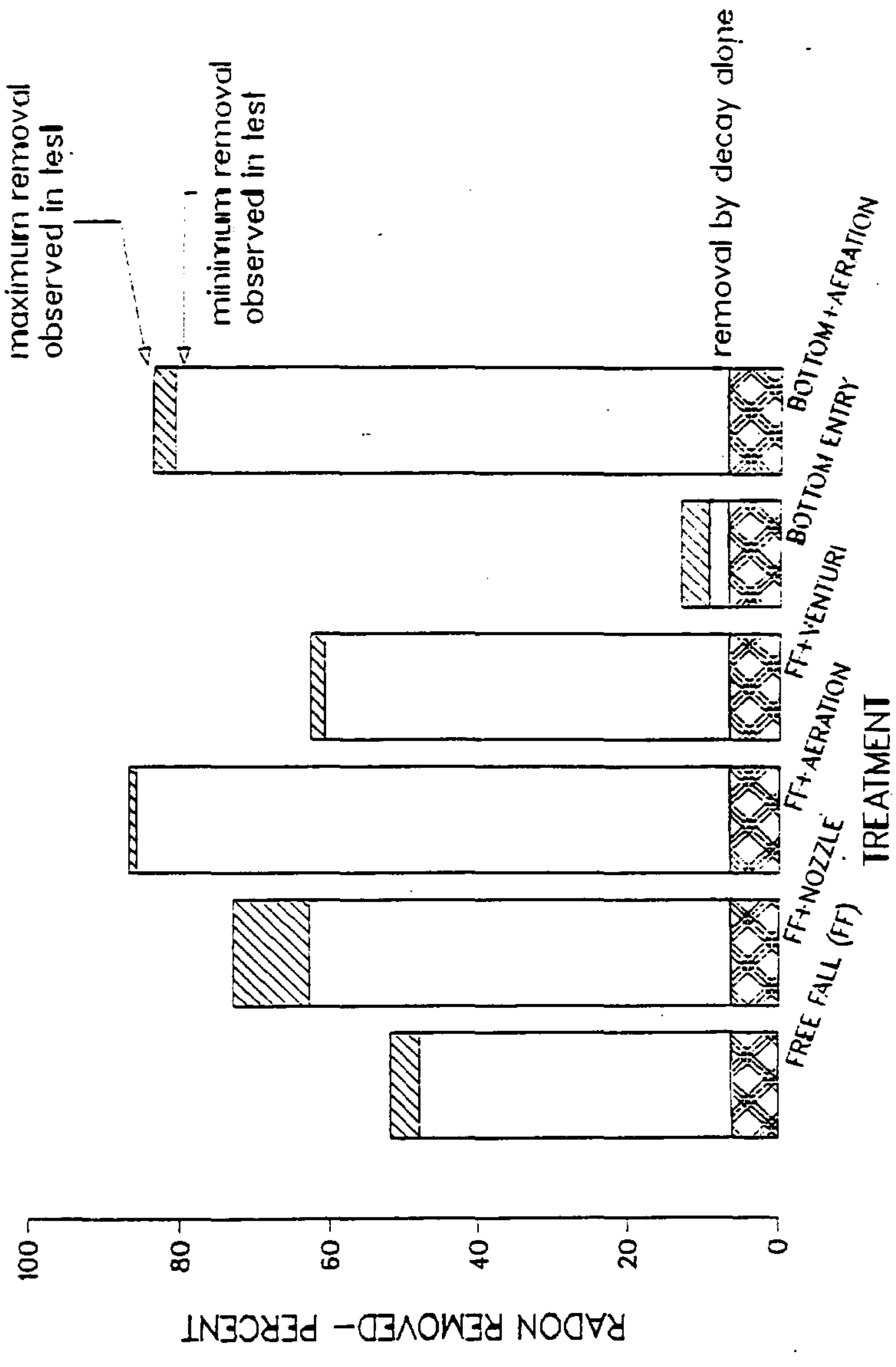

Figure 97. Low technology alternatives - Derry, NH. Percent radon removed for different conditions. (Water flowrate = 4.9 m^3/day).

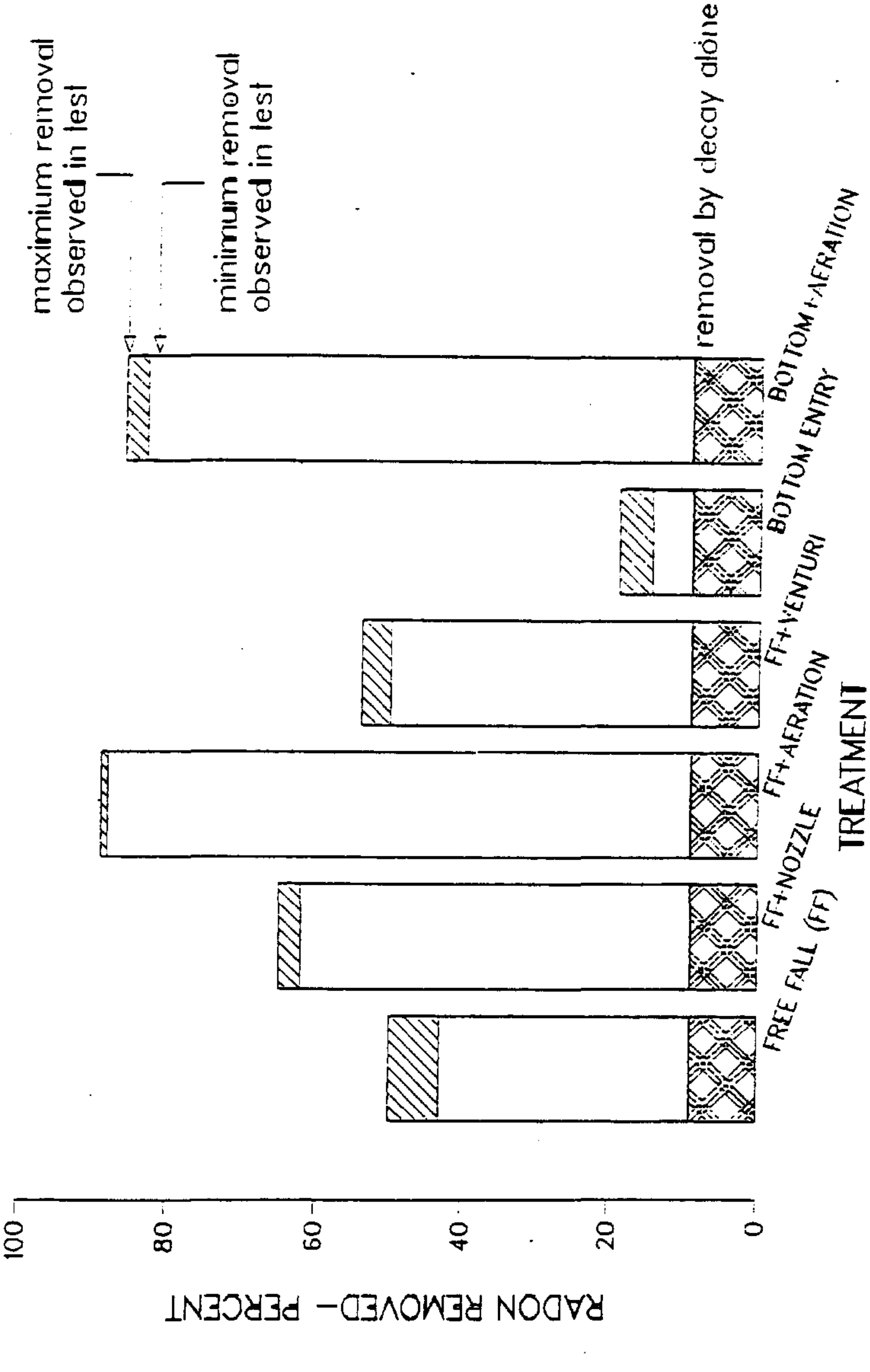

Figure 98. Low technology alternatives - Derry, NH. Percent radon removed for different conditions (Water flowrate = 3.3 m^3/day).

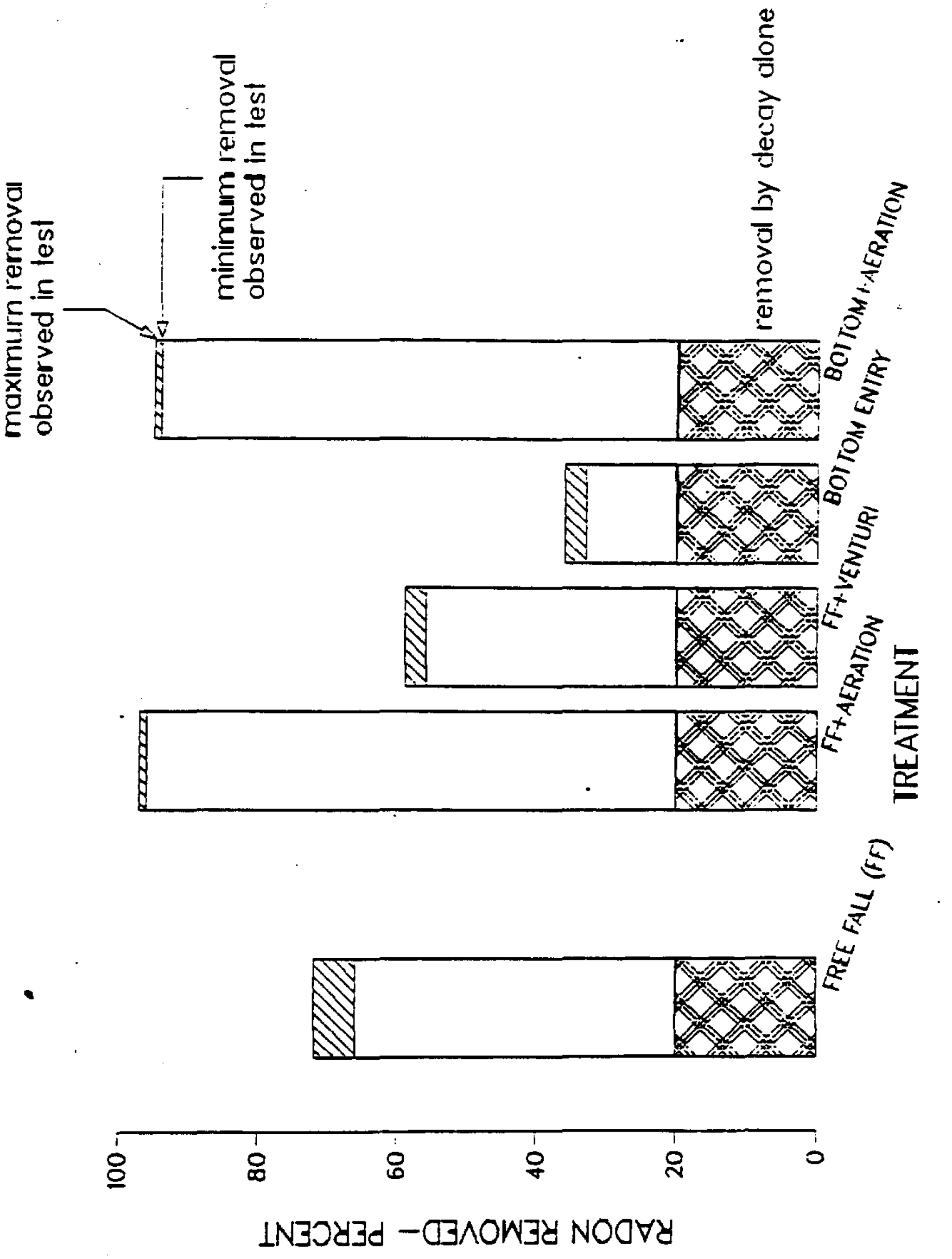

Figure 99. Low technology alternatives - Derry, NH. Percent radon removed for different conditions (Water flowrate = 1.3 m^3/day).

95,704 pCi/L.) While removals with the 0.61 m free fall entry alone were 50% (3.3 and 4.9 m^3/day) to 70% (1.3 m^3/day), they were enhanced by the use of the spray nozzle (60-70%, 3.3 to 4.9 m^3/day) and the coarse bubble aeration system (86-89%, 3.3 to 4.9 m^3/day; 96%, 1.3 m^3/day). The radon removal was also high using the coarse bubble aeration system in conjunction with the bottom entry to the reservoir; removals ranged from 83% (4.9 m^3/day) to 95% (1.3 m^3/day). The data collected in this phase of the study showed that addition of a coarse bubble aeration system to a storage tank may be very effective for radon reduction and might be easily applied in small communities which have existing atmospheric storage tanks. However, pilot scale testing would be needed because sidewall depth could affect removals. In the reservoir tested, the sidewall depth was only 0.61 m. Existing storage tanks may be much deeper causing poorer mass transfer (i.e., bubbles to be saturated with radon before they reach the surface) thus requiring greater air flowrates (A:W ratios) to achieve effective radon removal.

Shortly after our pilot-scale study at Derry was completed, the local water authority decided to add a coarse bubble aeration system to the existing atmospheric storage tank. Two 6 m long PVC pipes (5 cm I.D.) were installed 81 cm apart and 46 cm off the bottom of the tanks. The bottom part of each pipe contained a series of 3 mm diameter holes drilled every 5 cm for the first 4.6 m of length and at 2.5 cm intervals for the remaining 1.5 m. Raw water was pumped from the wells to the terminal end of the tank (6.7 m) before exiting the influent pipe a minimum of 0.30 m above the water's surface. A compressor delivered 0.57 m^3/min of air to the tank through the perforated pipes and was run continuously. The average detention time of the water in the tank was 5.3 hr. Two 15 cm diameter vent pipes were welded on top of the tank for ventilation. Radon removal efficiencies ranged from 80 to 88%, the latter occurring after approximately 6-8 hr of contact (overnight).

Several potential problems may arise with these kinds of coarse bubble aeration techniques. No gas phase monitoring was conducted during the study so the fate of the radon release into the ambient air remains unknown. However, based on the diffused bubble and packed tower aeration data, this may be a concern and should be monitored. In addition, water supplies which contain significant amounts of reduced iron and manganese may experience problems with precipitate formation and deposition in the storage tank and/or the distribution system.

It is important to note that in the case of this water supply the reductions of 80% to 96% observed for the coarse bubble aeration system would not be sufficient treatment to attain the effluent activity of 200-1,000 pCi/L which may be required by the MCL (influent activity = 58,641 to 95,704 pCi/L). In addition, extrapolation of the performance data to design systems which can achieve these required levels for this water supply is not advisable based on the importance of mass transfer limitations found in the diffused bubble and packed tower aeration studies at low radon activities. Additional pilot-scale testing at low radon activities would be required to determine if the coarse bubble aeration system could attain effluent activities of 200-1,000 pCi/L at Derry, NH.

ECONOMICS

Several low technology systems were evaluated for radon removal in small communities. However, since the major costs of these low technology systems involve retrofitting and system specific designs, it is not possible to provide a detailed estimate of these system costs. Therefore, a simple cost example of one type of low technology treatment will be presented for illustrative purposes.

The cost example is based on the low technology coarse bubble aeration system used by the water authority at Derry, NH. Cost estimates for this low technology system are shown in Table 25. The techniques used to estimate the costs of the low technology system are analogous to those described in Section 4 with the following exceptions: 1) support equipment costs were minor and thus combined with process equipment costs; 2) indirect costs were considered negligible and thus omitted from the analysis; 3) O&M costs were based solely on power and maintenance costs; and 4) pretreatment costs were not considered since it was assumed that construction of a sophisticated pretreatment system for iron and manganese would defeat the purpose of using a low technology system for radon removal. Based on these assumptions, the total production cost estimate of retrofitting the full-scale low technology system used at Derry, NH was $0.15/1,000 gallons. It should be noted that the cost estimate assumes no treatment of the radon contaminated vent gas other than discharging at the average home roof line (i.e., 10.7 m) is required.

TABLE 25. COST ESTIMATE FOR LOW TECHNOLOGY TREATMENT AT DERRY, NH

Item	Total Cost (2nd Quarter 1989 dollars)[*]
CAPITAL COSTS	
12.2 m (5.08 cm I.D.) PVC aeration system (Assembled and Installed)	$ 208
0.5 Hp air compressor	630
15.2 cm diameter air vent system (Assembled and Installed)	446
TOTAL CAPITAL COST	$ 1,284
AMORTIZED CAPITAL COST (Annual)	$ 141
ANNUAL O&M COSTS	
Power	332[+]
Maintenance	
Mechanical	63
Non-mechanical	26
TOTAL ANNUAL O&M COST	$ 421
TOTAL ANNUAL COST	$ 562
TOTAL PRODUCTION COST (Design flow 37.85 m^3/day)	$0.15/1,000 gallons

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[*] Based on ENR Construction Cost Index, Base Year 1967 = 100.

[+] Includes required input power of 0.5 Hp for the compressor, operating period of 24 hr/day, and an electric rate of $0.10157/kW-hr.

SECTION 8

EVALUATION OF THE LIQUID SCINTILLATION TECHNIQUE FOR RADON ANALYSIS

INTRODUCTION

As part of the maximum contaminant level (MCL) regulation for radon in drinking water, EPA must recommend sampling and analytical techniques to insure uniformity and comparability in monitoring and compliance. At a minimum, the method(s) recommended must be reliable, available and have an acceptable practical quantification level. In addition, easy operation and moderate cost are desirable characteristics. There have been several techniques developed to measure radon in water. In the Lucas cell method (Lucas, 1957), radon is de-emanated from a water sample into a zinc-sulfide phosphor-coated chamber where the alpha scintillations occurring as a result of radon and its progeny's radioactive decay are counted by a photomultiplier detection system. Though the procedure is labor and equipment intensive, it is the most sensitive of all of the radon techniques available (0.2 pCi/L for a 60 mL sample) (Yang, 1987). Lucas (1964) developed a faster method, later modified by Irfam and Read (1981), which uses gamma spectroscopy for monitoring radiation from the radon progeny, polonium-214 and bismuth-214. The procedure involves minimal sample preparation, but requires large sample volumes (1 L) to have a detection limit in the range of 10-100 pCi/L. Alpha track detector cups, without tops, have also been used for water storage tanks. Radon released from the water is trapped in the inverted cup and energy from alpha decay causes sub-microscopic tracks in the plastic material in the cup. The tracks are made visible by etching the plastic in a caustic soda solution. The number of tracks is counted microscopically. However, the method is difficult to calibrate because it depends on how often the toilet is flushed (P.B. Hahn, USEPA, Environmental Monitoring System Laboratory - Las Vegas, personal communication, 1989). Orme and Eaton (1986), adapted a method used in oceanography, where radon, purged from water, is concentrated on activated carbon and subsequently desorbed into a Lucas cell. The method has a minimum detection limit of 2.0 pCi/L.

Perhaps the most promising method for widespread monitoring because of its reliability, simplicity and speed (~1 min preparation time) is the liquid scintillation technique originally developed by Pritchard and Gesell (1977). In this procedure, a water sample (5 or 10 mL) is injected into a glass vial containing a liquid scintillation cocktail. The cocktail consists of a high molecular weight organic compound (fluor) and a solvent. The solvent originally used was toluene because radon is 12.7 times more soluble in toluene than in water and therefore can be readily extracted from a water sample into the cocktail (Pritchard and Gesell, 1977). A period of 4 hr after injection is required before the samples can be analyzed to allow secular equilibrium to develop between radon and its short-lived progeny. When alpha and beta particles created during the decay of these radionuclides strike the fluor molecules in the cocktail, they emit light pulses which are counted by photomultipliers located in the automated liquid scintillation counter. The decay series of radon-222 through polonium-214 yields three alpha particles (5.49, 6.0, and 7.69 MeV) and two beta particles with end points ranging from 0.69 to 3.26 MeV. Liquid scintillation techniques based on measurement of these kinds of energetic particles are approximately 100% efficient. Detection limits as low as 10 pCi/L (40 min counting time) have been reported for this technique (Pritchard and Gesell, 1977). However, recent research by Hahn (USEPA Environmental Monitoring Systems Laboratory - Las Vegas, personal communication, 1989) indicates that in order to have counting uncertainties (2σ) of $\leq$ 10% when measuring samples with low activity ($\leq$ 500 pCi/L) counting times >> 100 min are required (background counting rate ~100 cpm). If the MCL for radon in drinking water is set in the range of 500 to 1,000 pCi/L, counting times of approximately 60 to 20 min, respectively, would be required to achieve a 10% counting uncertainty (2σ) (P.B. Hahn, USEPA Environmental Monitoring Systems Laboratory - Las Vegas, personal communication, 1989).

The liquid scintillation technique is the only method allowing relatively rapid processing of samples and requiring small sample volumes. These advantages are important because compliance with the radon regulation will probably necessitate analyzing millions of water samples each year. The method has one disadvantage in that it lacks specificity. It will detect any alpha or high energy beta-emitting radionuclide soluble in the liquid scintillation cocktail. However, this drawback is considered insignificant in most groundwater supplies because radon is the primary radioactive contaminant soluble in the two-phase cocktails used.

EPA (1978) has used the liquid scintillation technique in its national monitoring programs to determine the distribution of radon concentrations in water supplies. Their procedures specify mineral oil as the cocktail solvent because toluene cannot be mailed without special handling and can present a problem with respect to laboratory safety and disposal. In addition, EPA collects samples in volatile organics bottles (VOA) and processes them in the laboratory (i.e., injection into the vial containing the cocktail). Though the EPA procedure adds an extra step in processing, it avoids widespread distribution of syringes.

There are several ways to calculate the radon present in the original sample. EPA (1978) uses the formula:

$$A = \frac{(C_S - C_B)\ (1000\ \text{mL/L})}{(E)\ (V)\ (D)} \qquad \text{(eq. 13)}$$

where:

A = radon activity in original water sample (pCi/L)

C_S = sample counts per minute (cpm)

C_B = background obtained from counting an equivalent volume of laboratory grade water in the same scintillation cocktail (cpm)

E = efficiency factor (cpm/pCi)

V = volume of sample analyzed (mL)

D = decay correction = $e^{-0.693T/t_{1/2}}$

T = time from sample collection to midpoint of counting time (days)

$t_{1/2}$ = half-life of radon-222 (3.82 days)

The efficiency factor (E) is based on the cpm obtained from counting a standard containing a known amount of radon (dpm or pCi). The standard is prepared by dissolving a known amount of radium-226, the long-lived parent of radon-222, in water. After a period of 20-25 days (i.e., 6 half-lives), the radium-226 is in secular equilibrium with radon so that a constant amount of radon-222 is always present in the vial.

A series of experiments was conducted to evaluate how certain factors affected the liquid scintillation technique results (i.e., radon concentration and variability). Factors investigated included: (i) sampling in the field with a syringe and immediate injection into the liquid scintillation vial versus collection in a VOA bottle and subsequent laboratory processing (Sampling Experiment); (ii) storage in VOA bottles (Storage Experiment); (iii) groundwater source, sample preparation and instrumentation variability (Variability Experiment); (iv) different liquid scintillation cocktails (Cocktail Experiment); and (v) extraction of radon into the cocktail (Extraction Experiment).

METHODS AND MATERIALS

Samples from all of the experiments were collected and analyzed according to the procedure outlined in Section 3 with specific modifications noted below for each experiment. They were all counted for 10 min using a Beckman LS 7000 Liquid Scintillation Counter (Fullerton, CA) with windows set at 0-397 (channel 1) and 397-940 (channel 2). Counts from both channels were used to calculate radon activities. A correction for interference was made for each sample using an internal cesium-137 source. This procedure is called the quench determination and is expressed as an H number (LS 7000 Program Number 6 - Appendix B). A Beckman tritium (hydrogen-3) instrument calibration standard, a radium-226 standard and a field blank (made from double deionized water (DDI)) were counted at the beginning of each analytical run. Duplicate analyses were performed every tenth sample. All glassware used in the experiments was cleaned in warm, soapy tapwater, rinsed three times with deionized water (DI), soaked for 1 hr in 10% Extran (VWR Scientific; Boston, MA), rinsed three times with DI, rinsed once with DDI and air dried.

The radium stock solution (117.5 pCi/mL) was prepared from a radium-226 (4.45 nCi/g) standard supplied in a glass ampule by the USEPA Environmental Monitoring Systems Laboratory (Las Vegas, NV).

The contents of the ampule were emptied into a tared beaker and weighed. They were subsequently transferred to a volumetric flask with several rinses of 0.05M HNO_3 and diluted to 200 mL with 0.05M HNO_3. This 200 mL dilution was used as the stock radium-226 solution. Each radium-226 standard was prepared by pipetting 5 mL of the stock and 5 mL of 0.05M HNO_3 into a 20 mL low absorbance glass liquid scintillation vial to achieve a final activity of 588 pCi. The standards were purged of radon by bubbling nitrogen gas through them for 20 min. 10 mL of liquid scintillation cocktail was slowly added to each vial before capping. Additional details of the radon analytical technique are given in Section 3.

Sampling Experiment

Radon samples were collected from two homes: one in Deerfield, NH (radon activity = 46,600 - 57,500 pCi/L) and one in Lee, NH (radon activity = 1,800-2,700 pCi/L). At each site, aerators on the faucets were removed before sampling and 18 samples were collected: 6 were obtained in amber VOA bottles fitted with teflon septa using the universal faucet connector (EPA, 1978), 6 were collected in amber VOA bottles using free falling water (0.15-0.30 m) directly from the faucet, and 6 were collected from the faucet connector using a 10 mL gas-tight glass syringe fitted with a 13 gauge needle and were injected directly into vials containing a cocktail of Liquifluor (40.3 mL/L) (New England Nuclear; Boston, MA) in a balance of liquid scintillation grade toluene (toluene-based cocktail). For all of the sampling techniques, the water flowrate was set at 927 mL/min at Lee and 800 mL/min at Deerfield. Samples were collected after the water had run for 3 min. The order of sampling was integrated, so that immediately after a free fall sample was taken, one was collected using the faucet connector, followed by one using the syringe. This alternation of sampling methods (free fall, faucet connector, syringe) throughout the collection period avoided confounding (time effects) in the results. Four field blanks of DDI (2 VOA bottles, 2 direct syringe) were collected at the end of each sampling. The entire collection period at a given site was performed within 15 min. After filling, VOA bottles were checked to ensure that air bubbles were not present. Once in the laboratory, 10 mL water samples were removed from each VOA bottle using a 10 mL gas-tight glass syringe and were injected into a toluene-based cocktail. All samples were then analyzed in the liquid scintillation counter as outlined in Section 3.

Storage Experiment

Samples were collected from an abandoned municipal ground water supply in Derry, NH (radon activity = 77,477 ± 6,512 pCi/L) using 60 mL amber VOA bottles filled using the universal faucet connector. A total of 24 water samples and 11 DDI field blanks were collected. In addition, 7 samples were taken using the direct syringe technique. Immediately, after all of the samples were collected (10 min), aliquots from 2 sample VOA bottles and 1 blank VOA bottle were taken and added to scintillation vials containing the toluene-based cocktail. After returning to the laboratory, these 3 samples, as well as the 7 direct syringe technique samples were counted immediately. All other sample and blank VOA bottles were returned to the laboratory and stored at room temperature (18-22°C). Two sample VOA bottles and one blank VOA bottle were processed and analyzed after storage periods of 0.4, 1.0, 3.0, 5.0, 7.0, 9.0, 11.0, 13.0, 15.0, 17.0 and 21.0 days.

Variability Experiment

A hierarchical experiment (Box et al., 1978) was conducted to determine the variation in radon activity associated with a groundwater source, sample preparation and instrumentation. Fourteen samples were collected at the Derry, NH site: 7 using the 60 mL amber VOA bottles and the universal faucet connector with subsequent laboratory processing and 7 using the universal faucet connector and the direct syringe technique. Two subsamples were processed from each VOA bottle. A toluene-based cocktail was used for all samples. The contents of the vials were analyzed in the liquid scintillation counter on 2 consecutive days according to the procedure outlined in Section 3.

Cocktail Experiment

Six radium-226 standards were prepared using a toluene-based cocktail; 3 using a cocktail containing Liquifluor (40.3 mL/L) in a balance of mineral oil (New England Nuclear; Boston, MA) and 3 using an Opti-Fluor O cocktail (Packard; Sterling, VA). All standards were prepared on the same day from the same radium-226 stock solution and were analyzed each day for a period of 29 days, then every 2-3 days for 2 weeks and weekly thereafter for a period of 7 months. In addition, 6 water samples were collected using the direct syringe technique with the universal faucet connector over a period of 20 min using each cocktail at both the Derry and Lee, NH sites. As in the Sampling Experiment, the order of sampling was integrated. Immediately after a toluene-based cocktail sample was taken, one was collected in the mineral oil-based cocktail, followed by one in the Opti-Fluor O. This alternation of cocktails used throughout the sample collection period avoided confounding in the results. All standards and samples were analyzed as outlined in Section 3.

Extraction Experiment

Three identical sets of four radium-226 standards were prepared using each cocktail (toluene-based, mineral oil-based and Opti-Fluor O): Set A was shaken before every analysis; Set B was shaken only on the day the standard was prepared; and Set C was never shaken. In addition, 12 samples were collected using the direct syringe technique with the universal faucet connector over a period of 15 min at both the Derry and Lee, NH sites. Six samples of each cocktail from each site were shaken for 1 min immediately after collection. All samples and standards were analyzed according to the method outlined in Section 3.

RESULTS AND DISCUSSION

Sampling Experiment

Radon samples are usually collected in the field in VOA vials and sent to the laboratory for subsequent processing. This avoids distribution of syringes and volatile scintillation cocktails, and problems with collection of bubbles within the syringe. EPA (1986) recommends filling the bottles with a universal connector attached to a faucet without an aerator. In the sampling experiment, the direct collection of samples using a syringe from a faucet fitted with the universal connector was compared with collection in VOA bottles filled directly from the faucet (free fall) or filled using the universal connector.

The results of samples collected at the Lee and Deerfield, NH sites are presented in Table 26. At the Lee site, analysis of variance (ANOVA) showed that the direct syringe technique yielded significantly higher activity than the free fall and universal connector techniques at $\alpha = 0.05$, but not at $\alpha = 0.02$. There was no significant difference between either VOA sampling method ($\alpha = 0.05$ and 0.10). At the Deerfield site, none of the three methods of sample collection were significantly different at $\alpha = 0.05$ or 0.1.

Numerically, the direct syringe technique always yielded the highest activities of the three methods, while the VOA bottle filled with free falling water yielded the lowest. This is not surprising as less sampling handling prior to injection into the scintillation vial and less agitation during sampling should result in less loss of radon. However, though the VOA collection techniques involve extra handling of the sample, they can produce results statistically similar to the syringe method.

The direct syringe technique may be preferred by researchers because it has the potential to yield results with the least radon loss and sample handling. Its use in compliance monitoring may be limited because of the costs and safety problems associated with widespread distribution of gas-tight glass syringes and volatile scintillation cocktails. In addition, samples taken with syringes must be drawn slowly to avoid cavitation and inclusion of air bubbles. Though plastic disposal syringes may be used, they are not designed for air-tight sampling. Thirteen gauge needles are large enough to limit these problems with air bubbles, but even experienced persons can still have problems completely eliminating them.

Sample collection with the free falling water must be done carefully at low water flowrates (800 to 927 mL/min was used) to insure results comparable to the direct syringe method. Splashing water into the bottles at substantially higher water flows would probably yield markedly lower results. The proper sampling techniques would need to be clearly outlined in any sampling procedure supplied with the VOA bottles. Distribution of universal faucet connectors would be cumbersome and increase the costs of sampling. However, their use insures collection of samples at lower water flowrates. As a result, the samples collected using the connector may be more consistently representative of the radon activity in a water supply and therefore more desirable for compliance monitoring.

TABLE 26. SAMPLING EXPERIMENT RADON ACTIVITIES

Site	Direct Syringe (pCi/L)	Universal Faucet Connector (pCi/L)	Free Fall (pCi/L)
Lee, NH	2,360	2,290	2,072
	2,492	2,578	2,338
	2,360	2,261	2,228
	2,440	2,105	2,415
	2,524	2,360	2,198
	2,468	2,314	2,207
$\bar{X} \pm S^*$	2,486 ± 90	2,318 ± 154	2,243 ± 120
Deerfield, NH	53,349	54,378	46,655
	53,358	52,941	50,313
	54,107	51,252	54,533
	51,441	48,826	46,832
	52,220	51,505	47,985
	57,446	59,591	50,825
$\bar{X} \pm S^*$	53,664 ± 2,084	53,082 ± 3,688	49,524 ± 3,008

*S represents 1 standard deviation.

Storage Experiment

Radon has a relatively short half-life necessitating a minimal storage time, especially for samples with activities near the MCL. Most radon sampling, conducted to ensure compliance with the MCL, will involve collection in VOA bottles, transportation (often by mail) and laboratory processing. The storage experiment was designed to examine the effect of the length of time between collection and laboratory processing on the results obtained. Losses of radon in the VOA bottles are caused by radioactive decay and possible leakage. The effect of decay is easily predicted based on the half-life of radon and the time between sample collection and analysis. The extent of leakage is unknown.

The 7 samples collected initially using the direct syringe technique had a mean radon activity and standard deviation of 65,068 ± 2,941 pCi/L. These samples were used as a basis of comparison for all of the VOA samples. The radon activity measured in the samples held in the VOA bottles decreased exponentially over time as predicted by radioactive decay (Fig. 100). When the results were corrected for the decay which occurred between the time of collection and analysis, only 4 samples (3.0, 5.0, 13.0, 17.0 days) had activities which were significantly different ($\alpha = 0.05$, StT) from those of the direct syringe method. (At $\alpha = 0.02$, only 3.0, 5.0 and 13.0 days were significantly different.) For the 3.0, 5.0, 13.0 and 17.0 day samples, the loss of radon ranged from 8.5 to 16.5%. There was no correlation between the length of storage time and leakage, indicating that leakage was probably a random function of specific bottles and their assembly or variability in sampling rather than storage time (Fig. 101).

The data indicated that loss of radon from VOA bottles could be a factor in some cases, but the loss resulting from radioactive decay has the greatest potential impact on storage time. Within 4 days, 50% of the radon originally present will be lost due to decay. For samples containing high levels of radon, permissible storage times could be substantial provided that the amount remaining at the time of analysis is above the practical quantification level. (For example, a sample containing approximately 7,670 pCi/L could be held up to 10 days, even with a 20% loss due to leakage, and still contain 1,000 pCi/L). However, the amount of radon in a sample is often not known, so the maximum storage time adequate to obtain a valid measurement must be based on the MCL.

Variability Experiment

The total variability and relative contribution of various sources of error within a sampling and analytical procedure can be determined using a hierarchical experiment (Box et al., 1978). In the variability experiment, samples were collected from a single water source over a 20 min period in order to measure the percent relative standard deviation (% RSD) associated with variation in the groundwater source, sample preparation and instrumentation. The hierarchical experiment was conducted using (i) direct syringe sampling and (ii) collection in VOA bottles with subsequent laboratory processing. The variation due to sample preparation could not be separated

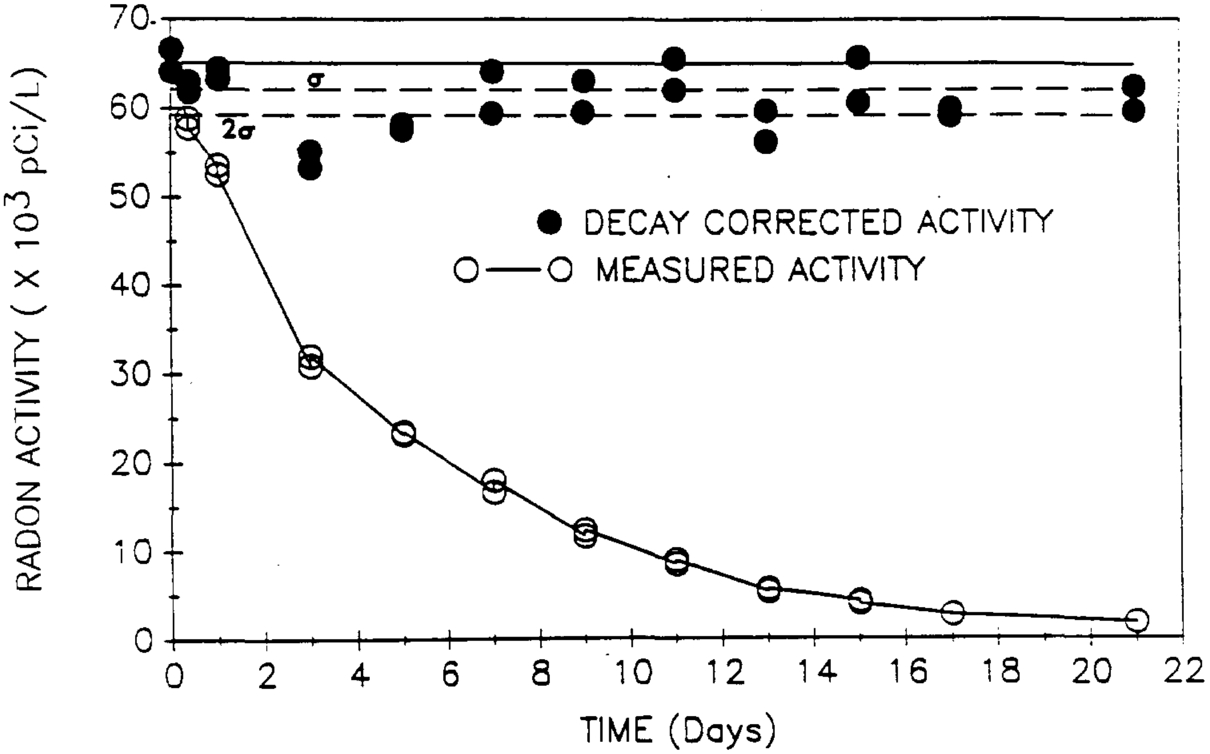

Figure 100. Storage experiment. Radon activity in VOA bottles as a function of storage time. —— Indicates average radon activity at time = 0. Samples taken using direct syringe technique.

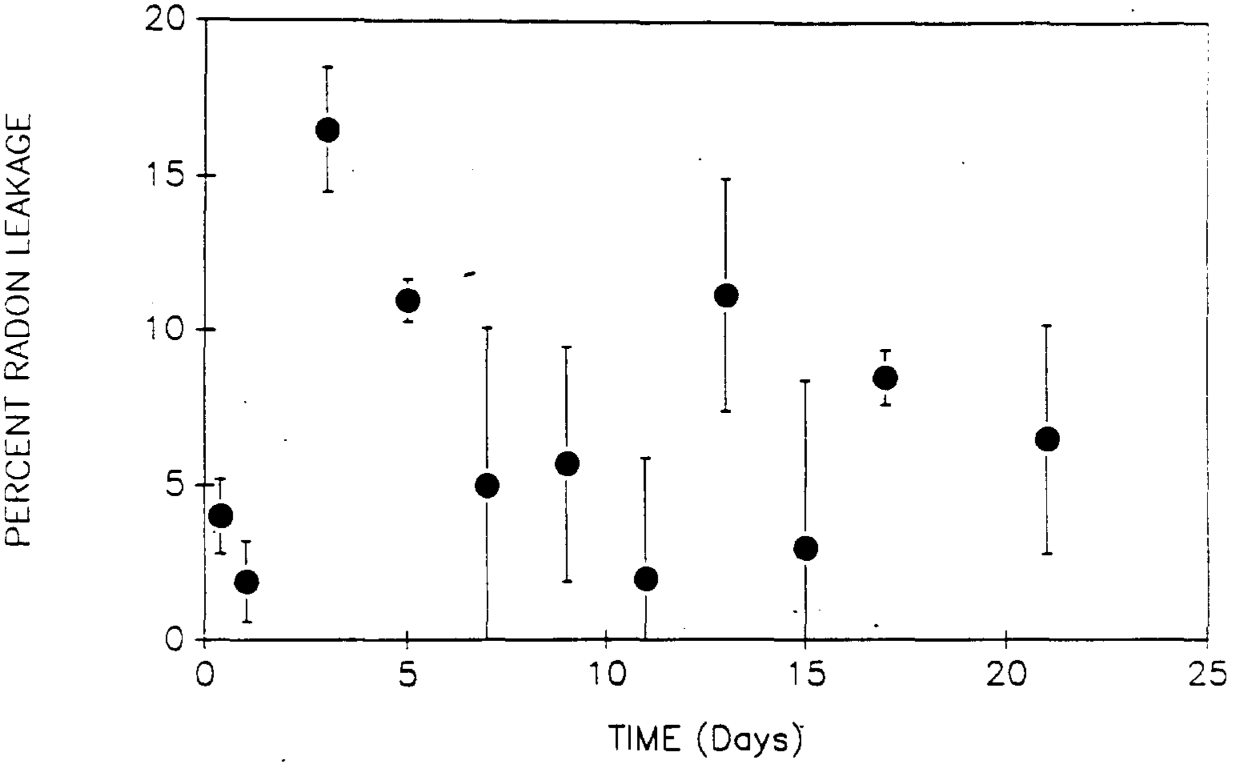

Figure 101. Storage experiment. Percent leakage of radon from VOA bottles as a function of storage time.

from the variation associated with the instrumentation for the direct syringe technique.

Table 27 shows the total variation and that associated with each phase of the process for both sampling methods. There was a significant difference ($\alpha = 0.01$ and 0.05, StT) between the mean radon content obtained by the two methods. The direct syringe technique yielded a significantly higher radon activity. This differed from the findings of the sampling experiment where the two methods yielded similar results. During the variability experiment, the water flowrates were higher which probably resulted in greater loss of radon when the VOA bottles were filled. However, as noted in the sampling section, the direct syringe procedure would require the widespread distribution of syringes which is not desirable and may offset its 5 to 6% higher activities.

The total variability associated with the two methods was not significantly different ($\alpha = 0.05$ and 0.10, F test). Total variation of 4 to 6% as a result of sample handling and instrument variation is not high considering the volatile nature of radon. For example, purgeable organics analysis for water may have standard deviations as high as 10 to 20% (EPA, 1984). In an interlaboratory collaborative study using sealed radium-226 standards, Whittaker et al. (1987) found within-laboratory average repeatability for the liquid scintillation techniques of 3.6 ± 3.0% at 95% confidence (18 laboratories). Their data includes variations due to sample preparation and instrumentation and agrees with the mean %RSD results shown in Table 27.

For many contaminants, the variability in the concentration in the environment represents the largest component of the standard deviation. For radon, very little variation was due to changes in the water source (0-1.6%) over the 20 min sampling period. This may not be true, however, for new wells or those sampled over much longer intervals (F.R. Hall, University of New Hampshire, Department of Earth Sciences, personal communication, 1988). Most of the variability in the direct syringe method (92.1% of total variance) was due to a combination of sample handling and instrumentation. In the VOA technique, the variation (as percent of total variance) due to sample preparation and instrumentation was 55.5% and 44.5%, respectively. Use of an air inlet needle when removing the sample from the VOA bottle and slow injection of the water from the syringe beneath the cocktail's surface are essential to having a small variation associated with sample handling. The air inlet needle prevents a vacuum from forming when the 10 ml sample is withdrawn from the VOA bottle. Tightly capping the scintillation vials and wiping their outside walls clean prior to placement in the counter are also important steps.

Cocktail Experiment

In the original liquid scintillation method developed by Pritchard and Gesell (1977), toluene was chosen as the cocktail solvent. The major problems associated with use of toluene-based cocktails are mailing restrictions, laboratory safety, its volatility and waste disposal. To avoid these problems, EPA (1978) prescribed mineral oil as the solvent for the cocktail

TABLE 27. RESULTS OF THE HIERARCHICAL EXPERIMENT TO MEASURE THE VARIABILITY ASSOCIATED WITH THE GROUNDWATER SOURCE, SAMPLE PREPARATION AND INSTRUMENTATION

Sampling Method	Mean Activity (pCi/L)	Deviation (1σ) (pCi/L)	Mean % Relative Std. Dev.	% of Total Variance*
Direct Syringe	64,977	3,873	5.96	
Groundwater Source		1,087	1.67	7.9
Sample Prep. and Instrumentation		3,717	5.72	92.1
VOA Bottle	61,482	3,693	6.00	
Groundwater Source		0+	0+	0+
Sample Preparation		2,752	4.47	55.5
Instrumentation		2,464	4.01	44.5

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* % of Total Variance = σ_X^2/σ_T^2 x 100.

σ_X^2 = the variance due to the groundwater source, sample preparation or instrumentation.

σ_T^2 = the total variance including groundwater source, sample preparation and instrumentation.

\+ Groundwater source deviation much less than those associated with sample preparation and instrumentation deviations, so it approaches zero in calculations.

because it can be transported through the mail without restriction and does not require special disposal. Recently, Packard (Sterling, VA) began manufacturing Opti-Fluor 0, a cocktail designed for use in radon analysis, which can also be sent through the mail and discarded without special handling.

The results of the cocktail experiment (Fig. 102) indicated that all of the radium-226 standards approached secular equilibrium within 25 days, as predicted by theory. An ANOVA showed that the mean count rates for the mineral oil and Opti-Fluor 0 cocktails were not significantly different ($\alpha = 0.05$ and 0.10), however, the mean count rate of the toluene-based cocktail was significantly less than both of these ($\alpha = 0.05$ and 0.01). In addition, the %RSD associated with the toluene-based cocktail (3.22%) was greater than those of the mineral oil (0.90%) and Opti-Fluor 0 (1.34%). The H numbers (Appendix B) for all of the standards were low (Toluene = 15±1.5, Mineral Oil = 25±2.1, Opti-Fluor 0 = 17±2.8) indicating that quenching was not a major factor for any of the cocktails. All three exhibited good two-phase separation from the water, although the mineral oil and Opti-Fluor 0 had small amounts of emulsion formation at the interface.

When the cocktail experiment was conducted using water samples collected at Derry and Lee, NH similar results were obtained (Table 28). At $\alpha = 0.05$ and 0.01 (ANOVA), the toluene-based cocktail's count rates were significantly less than those of the mineral-based and Opti-Fluor 0, which were not significantly different from each other ($\alpha = 0.05$ and 0.10).

In all cases, the toluene-based cocktail yielded significantly fewer counts (~10%) than the mineral oil and Opti-Fluor 0 cocktails. The increased counts may have come from radium-226 contained in the small emulsions formed at the interface of the water and the Opti-Fluor 0 or mineral oil-based cocktails. To test this hypothesis, an experiment with the cocktail Instagel was performed. Since Instagel forms a single phase gel emulsion with the aqueous phase one would expect to obtain counts from radium-226 present in the water. Radium-226 standards made with Instagel (Fig. 103) had counts approximately 1.6 times greater than the other cocktails. The increased counts resulted from radium-226, lead-210, bismuth-210 and polonium-210 which emit alpha or beta particles, but are not normally soluble in the toluene, mineral oil and Opti-Fluor 0 cocktails. These radionuclides are miscible in Instagel and as a result, this cocktail is exposed to additional radioactive disintegrations compared to two-phase cocktails (Table 29) (Section 3).

The count rates obtained for the samples in the toluene-based cocktail taken at the Lee and Derry, NH sites were also approximately 10% lower than those taken in the mineral oil and Opti-Fluor 0 cocktails. In these samples, radium-226 activity was substantially lower (<10 pCi/L) than in the standards (588 pCi/L). Therefore, it is unlikely that the radium-226 activities in the samples were high enough in the mineral oil-based and Opti-Fluor 0 emulsions to account for the 10% higher count rate. Perhaps, it occurs because they have a higher counting efficiency than the toluene-based cocktail.

The standards containing toluene showed a significant increase in %RSD ($\alpha = 0.05$ and 0.01, F test) over time compared with the other two cocktails

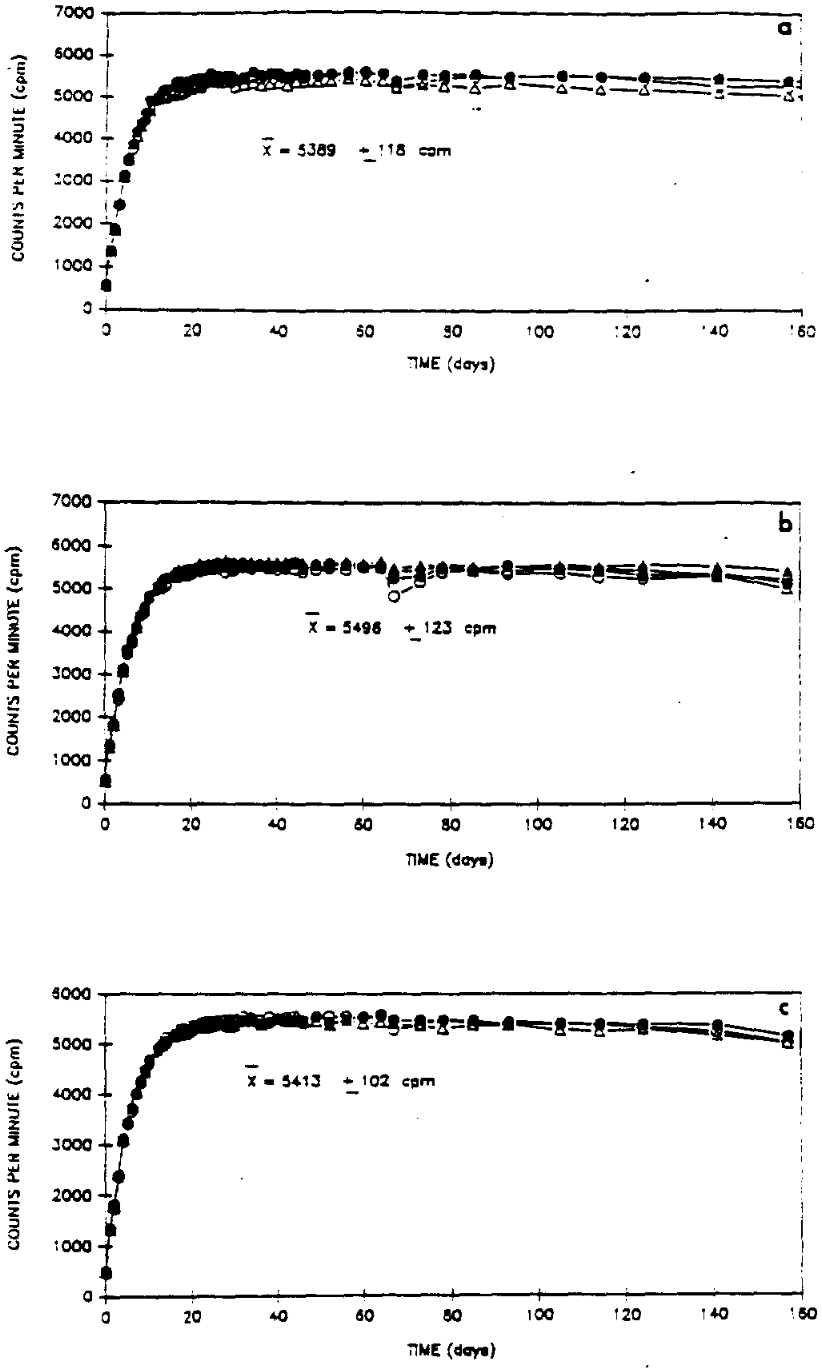

Figure 102. Cocktail experiment. Count rate (cpm) as a function of a time. (a) Toluene-based cocktail, (b) mineral oil-based cocktail, (c) Opti-fluor O.

TABLE 28. COCKTAIL EXPERIMENT COUNT RATES AT LEE AND DERRY, NH

Site	Toluene-Based (cpm)	Scintillation Cocktail Mineral-Oil Based (cpm)	Opti-Fluor O (cpm)
Lee	225±8*	244±9	248±7
Derry	3,028±68	3,426±58	3,483±42

* S represents 1 standard deviation.

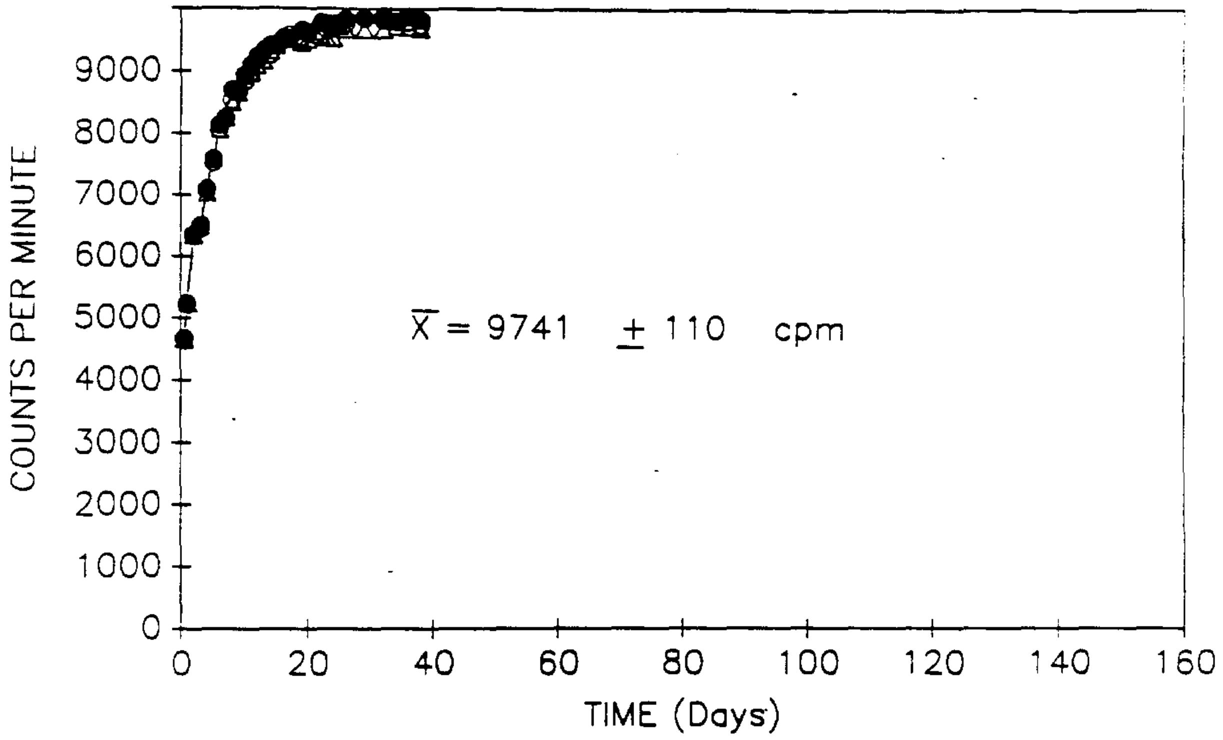

Figure 103. Cocktail experiment. Count rate (cpm) as a function of time for Instagel cocktail.

TABLE 29. RADIONUCLIDES PRESENT IN COCKTAILS YIELDING DISINTEGRATION

Instagel	Mineral Oil, Toluene, Opti-Fluor O
Radium-226	Radon-222
Radon-222	Polonium-218
Polonium-218	Lead-214
Lead-214	Bismuth-214
Bismuth-214	Polonium-214
Polonium-214	
Lead-210[+]	
Bismuth-210[+]	
Polonium-210[+]	

[*] Lead-210 low energy beta disintegrations will be counted at a 50-90% efficiency in a lower energy window (P.B. Hahn, USEPA Environmental Monitoring Systems Laboratory - Las Vegas, personal communication, 1989).

[+] These radionuclides are not in secular equilibrium with radium-226.

possibly because of volatilization of the cocktail. A 1-2 mL decrease in volume was observed in some of the scintillation vials stored in the laboratory over a period of 100 days in spite of the fact that they were tightly capped. This represents a loss of 10-20% of the cocktail indicating that toluene-based standards may have a limited shelf life. Others have reported that the mineral oil-based cocktail also has a limited life because by chemical reaction or some other means it degrades to a yellow color which in turn causes photodegradation and thus a reduction in light quality (R. Velten, USEPA Environmental Monitoring Support Laboratory - Cincinnati, 1987).

The choice of a cocktail may be influenced by other factors such as cost, disposal and mailing restrictions. Opti-Fluor O is the least expensive ($6.50/L), followed by the toluene-based ($8.50/L) and the mineral oil-based ($31.00/L) cocktails. The overall cost of the toluene-based cocktail is higher (~$12.00/L) when the disposal fees and special mailing requirements are included because toluene is an EPA listed hazardous material. Opti-Fluor O appears to provide the highest efficiency with the lowest variability and cost.

Extraction Experiment

The EPA procedure (1978) for analyzing radon in water requires that the capped vial containing the cocktail and water sample (or standard) be shaken for 1 min. This step is used to speed the extraction of the radon from the water into the cocktail while radionuclides, such as radium and uranium remain in the aqueous phase. The vial is held for 4 hr prior to counting to allow the short-lived progeny of radon to come to secular equilibrium so that a constant number of disintegrations will be occurring during analysis. The extraction experiment was conducted with each cocktail, to test the need for shaking, however the results from different cocktails should not be compared as they were prepared at different times.

The results for the standards with all of the cocktails had the expected increase in count rate over time as radon and its progeny came to secular equilibrium with radium-226 (Figures 104, 105 and 106). In all cases, there were only small differences in the count rates obtained for the standards regardless of whether or not they were shaken daily. If the extraction procedure was correct, the standards never shaken or shaken only on the first day should have had count rates approaching background. The data suggest that transfer of radon to the cocktail is continuous. The same pattern was observed for the samples taken at Lee and Derry (Table 30).

Though there were not large discrepancies between the count rates of the standards and samples as a result of shaking, there were statistically significant differences (Tables 31 and 32). However, there was no clear pattern emerging from these data. For the standards, the differences observed for each cocktail may be due to variations during preparation. The greatest contributors to variation in standards are pipetting, pipette rinsing and air stripping. During each of these steps, even with proper radiological techniques, small amounts of radium may be lost leading to variation in the count rates obtained. This is especially true for the bubbling of the standard which is difficult to regulate properly and consistently. Use of

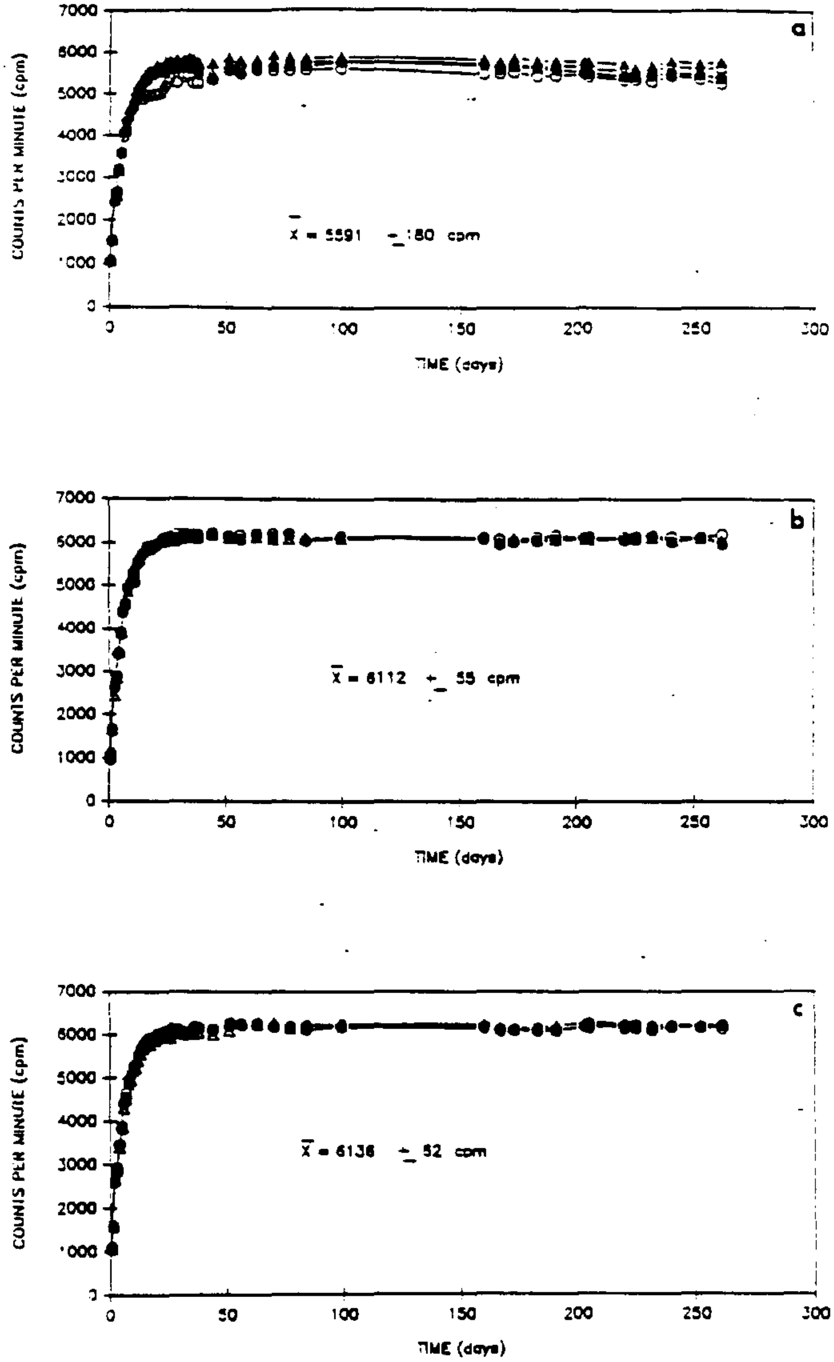

Figure 104. Extraction experiment. Count rate (cpm) for radium-226 standards using a toluene-based cocktail as a function of time. (a) Shaken daily, (b) shaken initially, (c) never shaken.

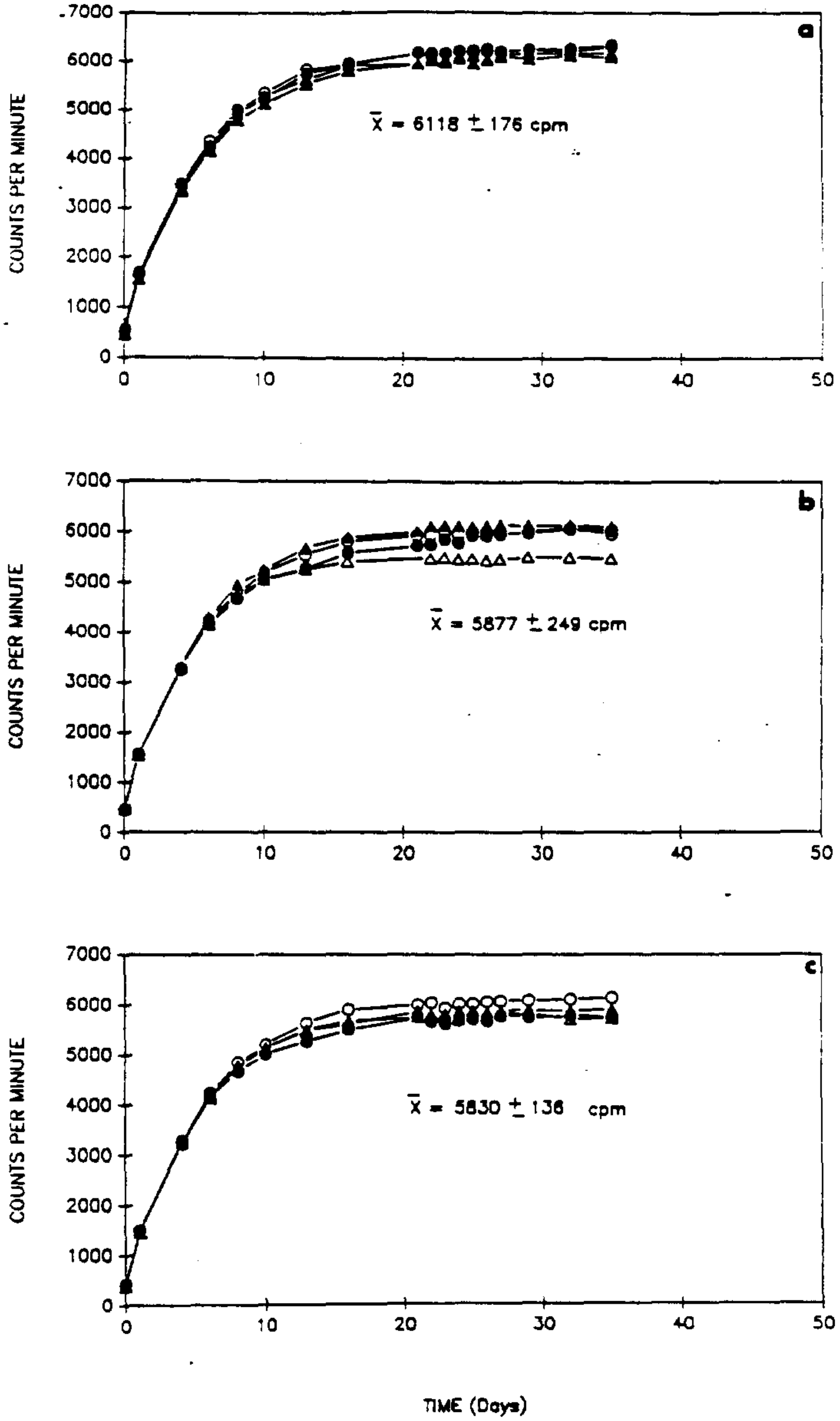

Figure 105. Extraction experiment. Count rate (cpm) for radium-226 using a mineral oil-based cocktail as a function of time. (a) Shaken daily, (b) shaken initially, (c) never shaken.

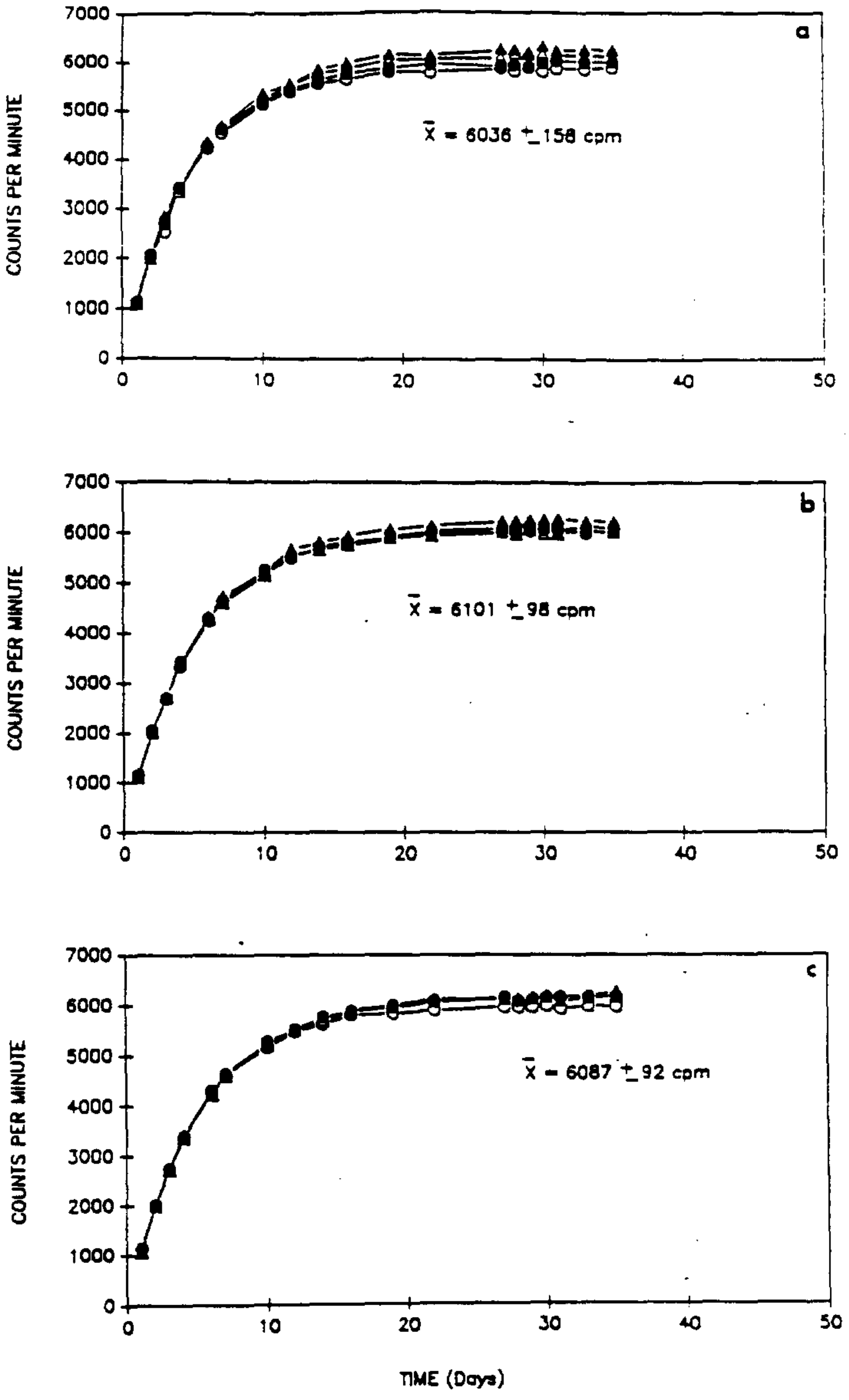

Figure 106. Extraction experiment. Count rate (cpm) for radium-226 using Opti-fluor O as a function of time. (a) Shaken daily, (b) shaken initially, (c) never shaken.

TABLE 30. MEAN COUNT RATES OF EXTRACTION EXPERIMENT AT LEE AND DERRY, NH[*]

Site	Toluene-Based (cpm)		Mineral Oil-Based (cpm)		Opti-Fluor O (cpm)	
	Never Shaken	Shaken	Never Shaken	Shaken	Never Shaken	Shaken
Lee,	274±12	289±6	240±7	288±10	254±14	283±11
Derry,	5,684±103	5,649±108	4,178±59	4,380±49	4,212±48	4,439±30

[*] Comparisons should not be made among cocktails as they were sampled on different days.

TABLE 31. STATISTICAL DIFFERENCES BASED ON MEAN COUNT RATES OF STANDARDS FOR EXTRACTION EXPERIMENT

Cocktail	Statistical Relationship[*]	Significance Level α
Toluene-Based	Shaken Daily ≡[+] Never Shaken	0.05
	Shaken Daily < Never Shaken	0.10
	Shaken Daily < Shaken Initially	0.05 & 0.01
	Never Shaken < Shaken Initially	0.05 & 0.01
Mineral Oil-Based	Shaken Initially ≡ Never Shaken	0.05 & 0.10
	Shaken Initially < Shaken Daily	0.05 & 0.01
	Never Shaken < Shaken Daily	0.05 & 0.01
Opti-Fluor O	Never Shaken ≡ Shaken Daily	0.05 & 0.10
	Shaken Daily ≡ Shaken Initially	0.05 & 0.10
	Never Shaken ≡ Shaken Initially	0.05 & 0.10

[*] As assessed by Scheffe analysis which was used because variances were not homogeneous.

[+] ≡: not significantly different at α specified.

TABLE 32 STATISTICAL DIFFERENCES BASED ON MEAN COUNT RATES OF SITES FOR EXTRACTION EXPERIMENT

Site	Cocktail	Statistical Relationship[*]	α
Lee, NH	Toluene-Based	Never Shaken < Shaken	0.05
		Never Shaken ≡[+] Shaken	0.02
	Mineral-Oil Based	Never Shaken < Shaken	0.05 & 0.01
	Opti-Fluor O	Never Shaken < Shaken	0.05 & 0.01
Derry, NH	Toluene-Based	Never Shaken ≡ Shaken	0.05 & 0.10
	Mineral Oil-Based	Never Shaken < Shaken	0.05 & 0.01
	Opti-Fluor O	Never Shaken < Shaken	0.05 & 0.01

[*] As assessed by Student's t test.

[+] ≡: not significantly different at α specified.

multiple point standard curves in the calculation of sample activities will help to overcome the effects of these variations in preparation. An interlaboratory QA/QC program is essential to insure that results obtained at different laboratories are comparable.

The data for the field samples collected at Lee and Derry indicate that, though extraction via shaking is not essential for radon transfer to the cocktail, it does yield significantly higher mean count rates ($\leq$ 20%) for all cocktails. The extraction procedure for samples should be used to maximize the rate of transfer of radon to the cocktail phase.

The extraction experiment results have repercussions with respect to the calculation of the efficiency factor, E, (cpm/pCi) used to determine the radon activity in the samples (eq. 12). Extraction procedure (EPA, 1978) predicts that once radon-222 in the standards is in secular equilibrium with radium-226 in the aqueous phase, a known amount of radon is available to be extracted into the cocktail, but can only be transferred by shaking. In that case, the time between extraction (shaking) and analysis for the standards would determine the amount of radon and its progeny and hence, the number of disintegrations (dpm) occurring in the cocktail at the time of counting.

For example, if a standard containing 587 pCi of radium-226 were in secular equilibrium with radon-222, it would contain 587 pCi of radon. According to the procedure, all of the radon would remain in the aqueous phase until extraction. Immediately, after shaking nearly all of the 587 pCi of radon would be transferred to the cocktail. After 4 hr, the radon in the cocktail would be in secular equilibrium with its short-lived progeny. Based on the data from the extraction experiment, the count rate in the standard at 4 hr would be 5,400 and the net count rate would be 5340 (i.e., 5400 - 60 cpm for the blank). After 4 hrs, 2.98% of the original radon in the cocktail would decay away and not be regenerated from the aqueous phase, so only 569.5 pCi would be present when the analysis was performed. The efficiency factor is therefore 5,340 cpm/569.5 pCi = 9.38 cpm/pCi (equivalent to 5,340 cpm/1,264.3 dpm where 1 pCi = 2.22 dpm).

A similar calculation could be made if the standard were counted 24 hr after shaking. Based on the data from the extraction experiment, the count rate obtained would still be approximately 5,340, but the radon activity remaining in the cocktail according to the theory would have decayed to 489.7 pCi (83.4% of the original 587 pCi). Hence, the efficiency factor would be 10.90.

The effect of this change can be demonstrated by examining eq. 12. According to the extraction procedure, a 10 mL sample yielding a net 89 cpm (sample cpm - background cpm) and counted 12 hr after collection would contain 1,038 pCi/L assuming only 4 hr elapsed between shaking and counting of the standard, whereas the sample would be calculated to contain 894 pCi/L if the standard were counted 24 hr after shaking. However, in reality the actual radon activity would be 1,071 pCi/L based on the findings of the extraction experiment (i.e., the efficiency factor of the standard would remain 5,340 cpm/587 pCi or 9.10 cpm/pCi regardless of when the standard is counted). If the MCL for radon was set at 1,000 pCi/L, such a difference might be crucial.

Thus, if extraction principles are applied to standards, efficiency factors would be inflated leading to an underestimation of actual radon activities in the samples. This error would be especially critical for samples at or near the MCL.

Finally, it is important to analyze radium-226 standards (at secular equilibrium) during every sample run. As shown in Figures 104-106, the count rate obtained from standards analyzed on consecutive days using the same liquid scintillation counter may vary by as much as 5%. Preparation of a calibration curve based on radium-226 standards of various activities analyzed during each liquid scintillation counter run may also provide a better conversion of cpm to pCi. This is especially significant when samples with a variety of radon activities are analyzed.

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APPENDIX A

DAILY AVERAGE ANALYTICAL PRECISION AND ACCURACY

DIFFUSED BUBBLE

Analysis	pH		Alkalinity (mg/L as $CaCO_3$)		Turbidity (NTU)		Mn (mg/L)	Fe (mg/L)	Mn	Fe
Date	S_x*	%RSD+	S_x	%RSD	S_x	%RSD	S_x	S_x	%REC**	%REC
7-28-88	0.03	<1	1.27	2	5.3	40	-	-	-	-
8-24-88	0.04	<1	0.63	1	0.01	8	-	-	-	-
8-25-88	0.02	<1	0.00	0	0.09	13	-	-	-	-
8-26-88	0.01	<1	0.96	3	0.06	2	-	-	-	-
8-29-88	0.01	<1	7.00	8	0.02	11	-	-	-	-
8-30-88	0.02	<1	0.95	3	0.08	1	-	-	-	-
9-1-88	0.03	<1	2.23	4	0.67	25	-	-	-	-
9-2-88	0.01	<1	1.20	2	0.01	4	-	-	-	-
9-15-88	0.05	1	2.33	4	0.03	2	-	-	-	-
9-11-89	-	-	-	-	-	-	0.01	0.02	110	109

PACKED TOWER

8-4-87	0.01	<1	0.00	0	-	-	-	-	-	-
8-7-87	0.01	<1	0.17	1	0.00	<1	-	-	-	-
8-10-87	0.04	<1	0.27	1	0.01	17	-	-	-	-
8-12-87	0.01	<1	0.17	<1	0.01	4	-	-	-	-
8-14-87	0.02	<1	0.54	2	0.01	21	-	-	-	-
8-18-87	0.04	<1	0.54	2	0.05	51	-	-	-	-
8-20-87	0.00	<1	0.00	<1	0.00	0	-	-	-	-
9-1-87	0.10	1	0.06	<1	0.00	0	-	-	-	-
9-3-87	0.02	<1	0.00	<1	0.00	1.2	-	-	-	-
9-4-87	0.07	1	1.43	6	0.02	38	-	-	-	-
9-9-87	0.01	<1	0.00	<1	0.01	25	-	-	-	-
9-10-87	0.02	<1	0.18	<1	0.00	0	-	-	-	-
9-11-87	0.05	<1	0.17	<1	0.00	0	-	-	-	-

(continued)

PACKED TOWER (continued)

Analysis	pH		Alkalinity (mg/L as $CaCO_3$)		Turbidity (NTU)		Mn (mg/L)	Fe (mg/L)	Mn	Fe
Date	S_x^*	%RSD$^+$	S_x	%RSD	S_x	%RSD	S_x	S_x	%REC**	%REC
1-30-88	0.02	<1	0.00	<1	0.06	18	-	-	-	-
2-3-88	0.02	<1	0.23	<1	0.01	8.4	-	-	-	-
2-5-88	0.02	<1	0.37	2	0.01	12	-	-	-	-
2-21-88	-	-	-	-	-	-	0.00	0.00	124	113
2-24-88	-	-	-	-	-	-	0.00	0.00	84	88
6-14-88	-	-	-	-	-	-	0.01	0.01	107	98
6-17-88	-	-	-	-	-	-	0.01	0.01	108	95

GAC SYSTEMS

Analysis	pH		Alkalinity (mg/L as $CaCO_3$)		Turbidity (NTU)		Mn (mg/L)	Fe (mg/L)	Mn	Fe
Date	S_x^*	%RSD$^+$	S_x	%RSD	S_x	%RSD	S_x	S_x	%REC**	%REC
10-9-86	-	-	2.12	9	0.00	0	-	-	-	-
11-30-86	-	-	0.35	1	0.26	14	-	-	-	-
12-4-86	0.01	<1	0.22	1	0.10	3	-	-	-	-
12-11-86	0.02	<1	0.62	3	0.04	8	-	-	-	-
12-18-86	0.04	<1	0.15	1	0.01	13	-	-	-	-
1-7-87	0.01	<1	0.18	1	0.04	13	-	-	-	-
1-16-87	0.02	<1	0.27	1	0.05	18	-	-	-	-
1-24-87	0.02	<1	0.09	<1	0.00	0	-	-	-	-
1-29-87	0.01	<1	0.29	1	0.01	6	-	-	-	-
2-9-87	0.01	<1	0.29	1	0.02	1	-	-	-	-
2-11-87	0.02	<1	0.59	2	0.01	10	-	-	-	-
2-18-87	0.02	<1	0.71	3	0.01	2	-	-	-	-
2-25-87	0.05	1	0.35	2	0.01	15	-	-	-	-
3-4-87	0.02	<1	0.59	2	0.10	18	-	-	-	-
3-11-87	0.03	<1	0.41	1	0.03	13	-	-	-	-
3-13-87	-	-	-	-	-	-	0.01	0.01	136	205
3-16-87	-	-	-	-	-	-	0.01	0.01	110	142
3-17-87	0.00	<1	0.00	0	0.00	0	-	-	-	-
4-14-87	0.04	1	0.00	0	0.01	17	-	-	-	-
5-13-87	0.02	<1	0.00	0	0.01	15	-	-	-	-
6-11-87	0.05	1	0.18	1	0.00	0	-	-	-	-
6-15-87	-	-	-	-	-	-	0.00	0.01	75	90
6-16-87	-	-	-	-	-	-	0.00	0.01	83	90
6-17-87	-	-	-	-	-	-	0.00	0.01	97	90
6-21-87	-	-	-	-	-	-	0.01	0.01	81	106
6-25-87	-	-	-	-	-	-	0.00	0.01	89	80

(continued)

GAC SYSTEMS (continued)

Analysis Date	pH S_x*	pH %RSD+	Alkalinity (mg/L as $CaCO_3$) S_x	Alkalinity (mg/L as $CaCO_3$) %RSD	Turbidity (NTU) S_x	Turbidity (NTU) %RSD	Mn (mg/L) S_x	Fe (mg/L) S_x	Mn %REC**	Fe %REC
7-1-87	0.01	<1	0.25	1	0.00	0	-	-	-	-
7-2-87	0.01	<1	0.53	2	0.00	0	-	-	-	-
7-7-87	0.01	<1	0.14	2	0.00	0	-	-	-	-
7-16-87	0.04	1	0.35	2	0.00	0	-	-	-	-
8-4-87	0.05	1	0.35	2	0.00	0	-	-	-	-
8-6-87	-	-	-	-	-	-	0.01	0.01	98	102
8-11-87	-	-	-	-	-	-	0.00	0.01	75	65
8-13-87	0.04	1	0.36	2	0.00	0	0.00	0.00	-	-
9-14-87	0.04	1	0.00	0	0.00	0	-	-	-	-
9-17-87	0.08	1	0.00	0	0.00	0	-	-	-	-
1-28-88	0.14	2	0.40	1	0.01	15	-	-	-	-
1-29-88	0.03	4	0.50	1	0.01	11	-	-	-	-

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* S_x is the average standard deviation.

\+ %RSD is the average percent relative standard deviation.

** %REC is the average percent recovery.

RADON

The analytical precision, as % relative standard deviation (%RSD), was calculated several times during the course of the project. This was done by counting different blanks and standards three consecutive times in the liquid scintillation counter. The % RSD data are shown below:

Sample	$\bar{X}$ (pCi/L)	S_x (pCi/L)	% RSD
Blank	83.2	3.3	3.9
Blank	99	4.4	4.4
Blank	98	4.0	4.1
Blank	57	2.2	3.9
Blank	56	3.6	6.4
Standard	3,028	168.4	5.6
Standard	4,019	230	5.7
Standard	4,921	58	1.2
Standard	5,195	35	0.7
Standard	4,994	126	2.5
Standard	4,902	159	3.2
Standard	4,902	159	3.2
Standard	4,921	58	1.2

The percent % RSD, as shown in Section 8, for the scintillation counter is usually < 6%. Duplicate samples were also run every tenth sample, however, these samples were taken consecutively and therefore, reflect variability in the source as well as the analysis.

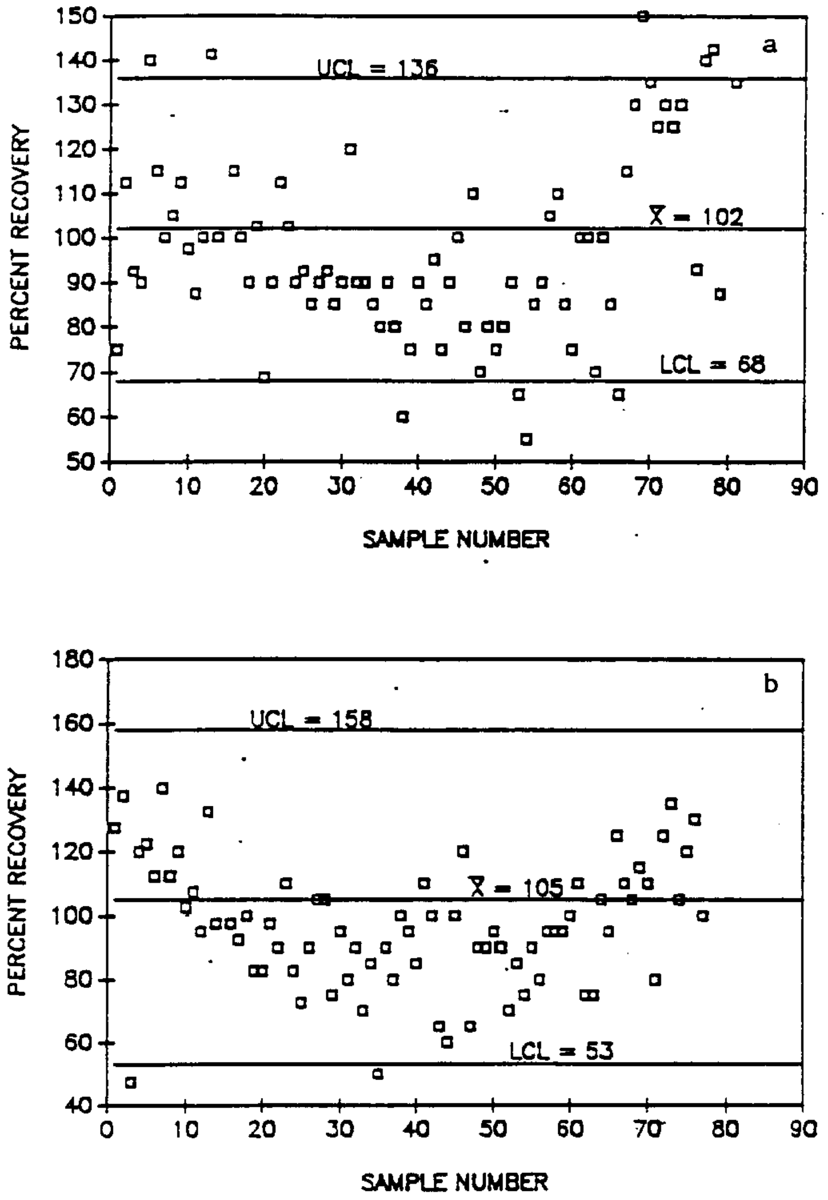


Figure A1. Mont Vernon, NH GAC system. Control charts for accuracy. (a) Iron and (b) manganese.

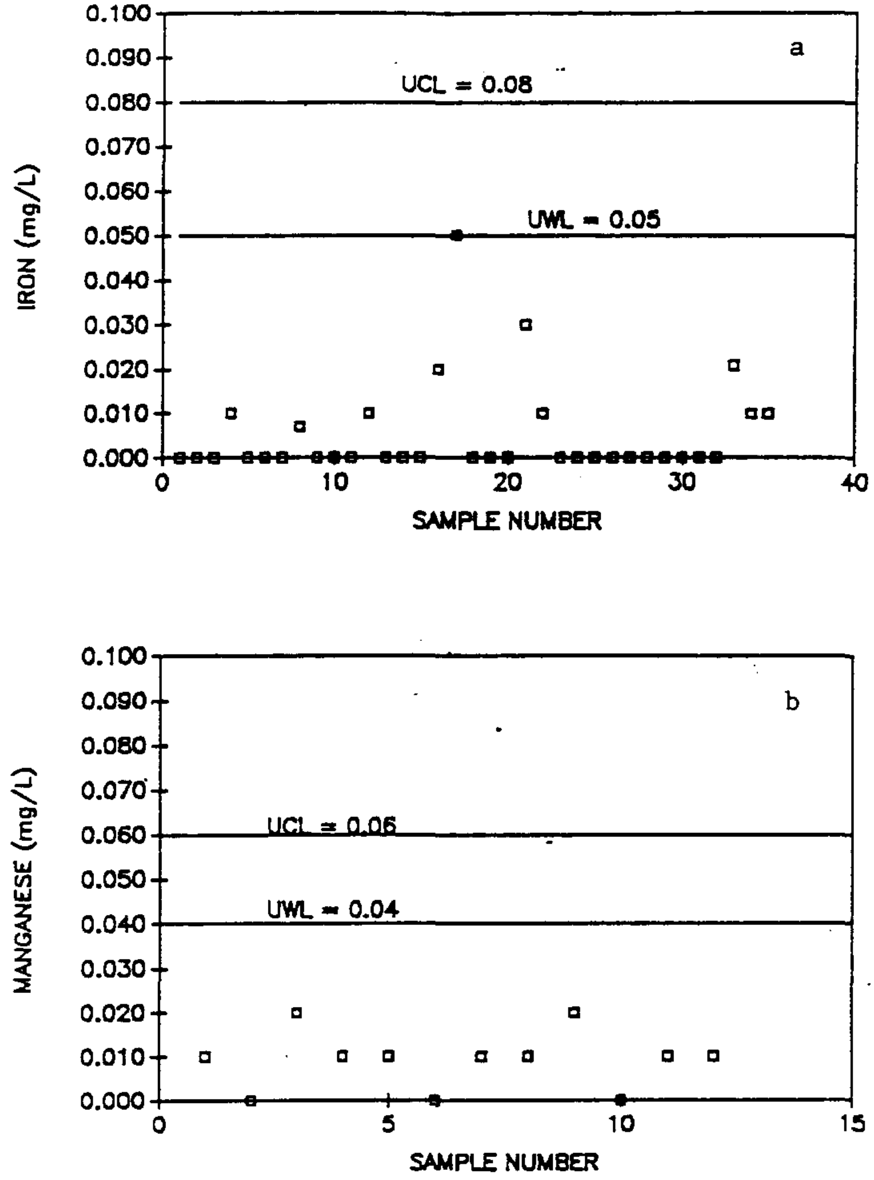


Figure A2. Mont Vernon, NH GAC system. Control charts for precision. (a) Iron and (b) manganese.

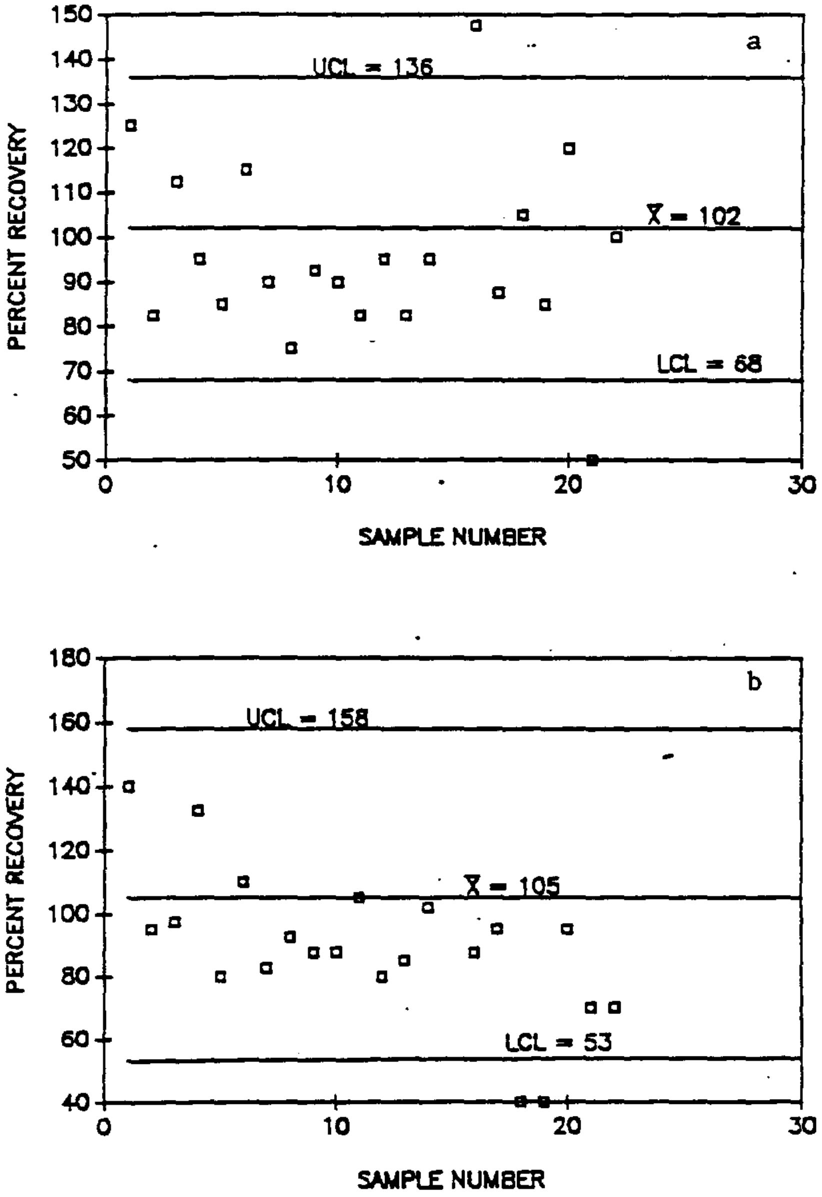


Figure A3. Amherst, NH GAC system. Control charts for accuracy. (a) Iron and (b) manganese.

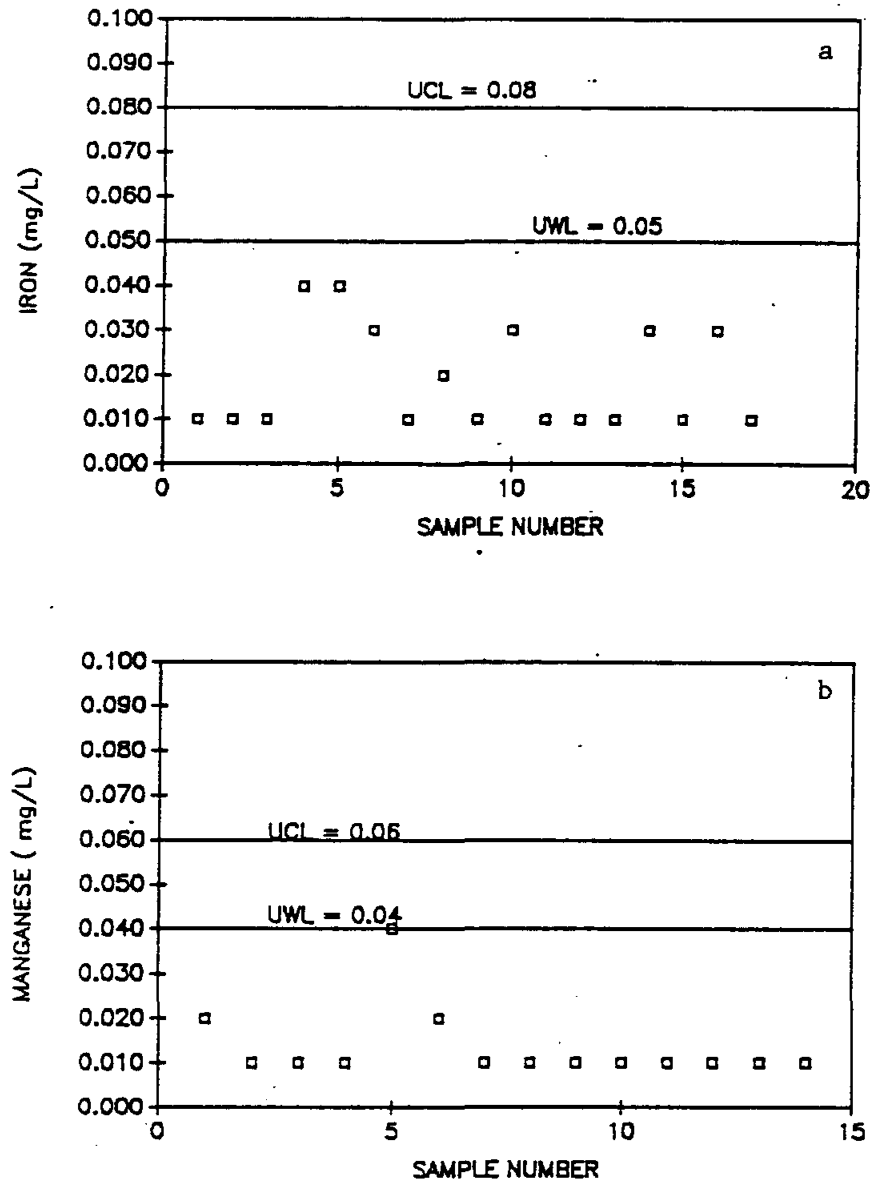


Figure A4. Amherst, NH GAC system. Control charts for precision. (a) Iron and (b) manganese.

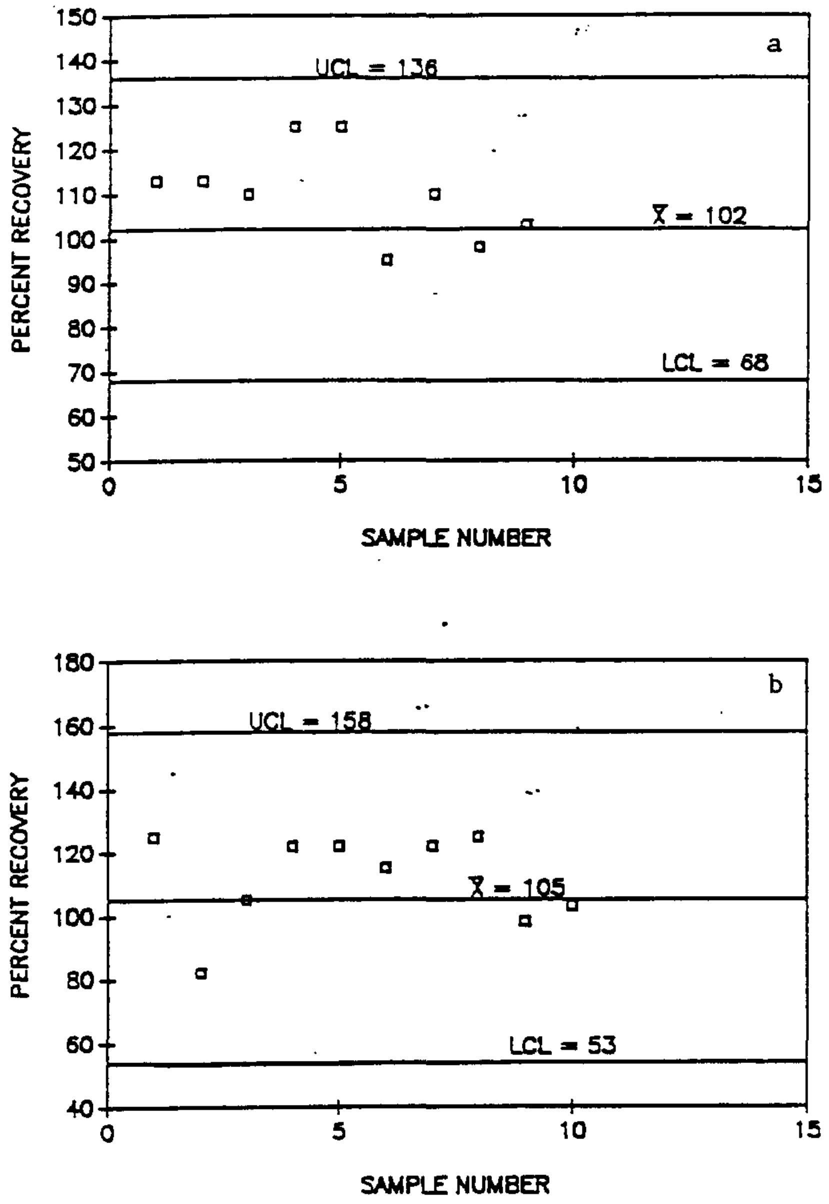


Figure A5. Diffused bubble aeration. Control charts for accuracy. (a) Iron and (b) manganese.

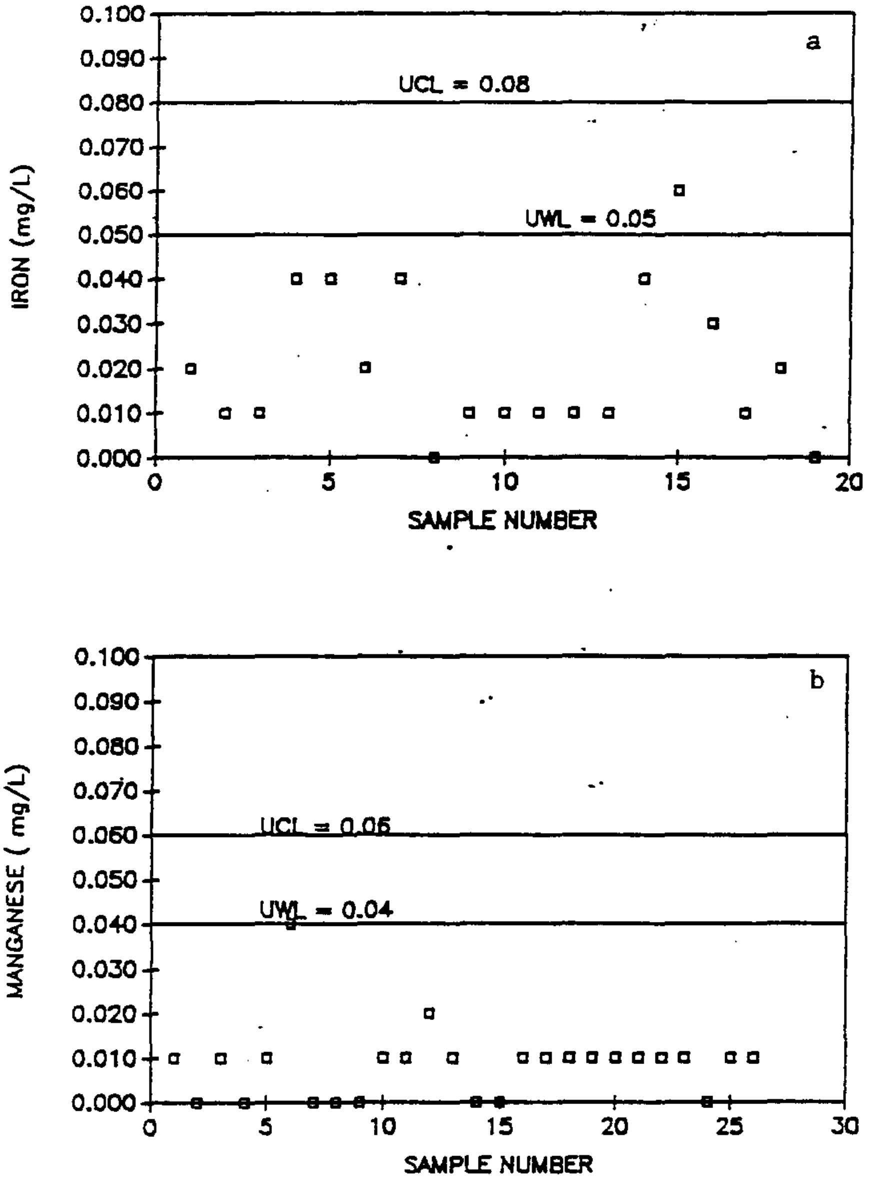
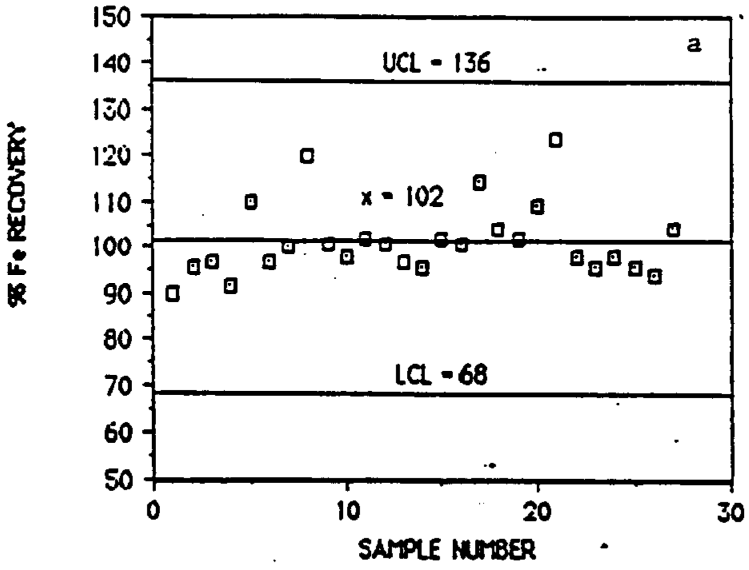


Figure A6. Diffused bubble aeration. Control charts for precision. (a) Iron and (b) manganese.



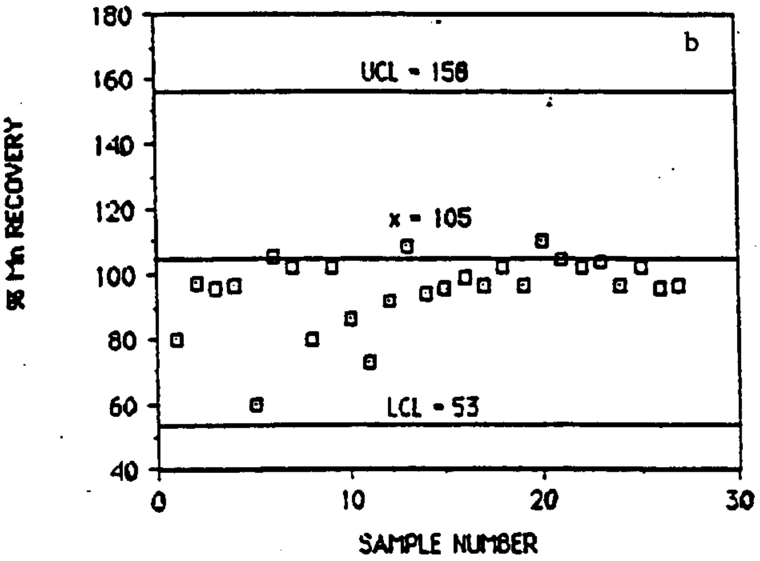


Figure A7. Packed tower aeration. Control charts for accuracy. (a) Iron and (b) manganese.

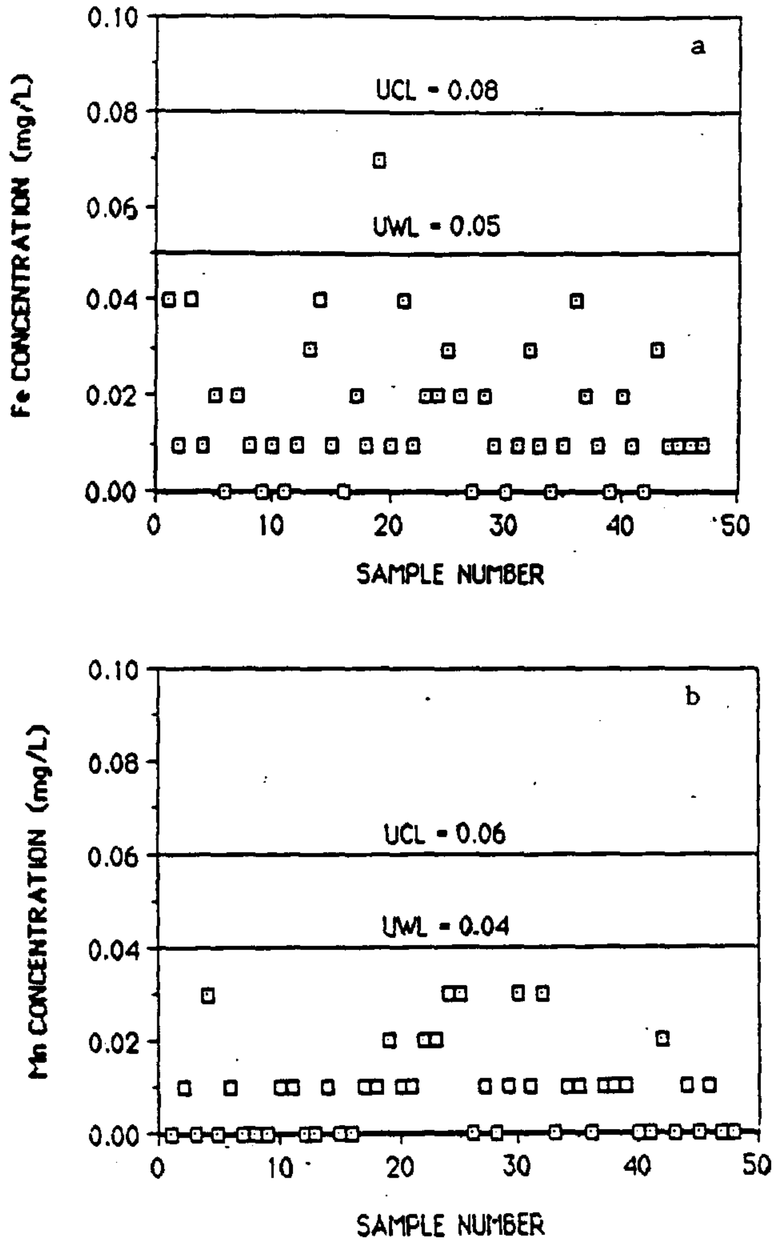


Figure A8. Packed tower aeration. Control charts for precision. (a) Iron and (b) manganese.

APPENDIX B

SUMMARY OF LIQUID SCINTILLATION COUNTER PROGRAM

The program outlined below was obtained from the operations manual for the Beckman (Fullerton, CA) LS 7000 liquid scintillation counter used in this research project. It is library program #6 for this instrument.

Unmodified Count Time — 10 min

Channels	Counted	Lower Limit	Upper Limit
1	2.00σ%*	0	397
2	2.00σ%*	397	940

Sample Channels Ratio	No
H Number Calibration**	1 time (in channels 1 & 2)
Automatic Quench Compensation	Yes
Data Printout Units	cpm

---

[*] 2.00σ% = $200/\sqrt{N}$ where N = total number of counts obtained at the time of calculation.

[**] The H-number quantifies quench and indicates the effect of interferences on the counting efficiency. Automatic quench compensation uses an internal cesium-137 source to generate an H-number for each sample, which the liquid scintillation counter uses to adjust the count rate generated in correspondence to the quenching.

APPENDIX C

METHODS OF CALCULATION OF LEAD-210 ADSORPTION

I. Calculation of Lead-210 activity from gamma counter data using channels 89-96.

$$A' = \frac{\text{Net Peak Counts}}{(t)\ (m)\ (E)\ (I_g)}$$

where A' = activity (dpm/g GAC dry weight)
Net Peak Counts = (Adjusted Peak Counts - Adjusted Compton Counts)

$$\underset{\text{Counts}}{\text{Adjusted Peak}} = \left(\underset{\text{Counts}}{\text{Sample Peak}} - \underset{\text{Counts}}{\text{Background Peak}}\right)$$

$$\underset{\text{Counts}}{\text{Adjusted Compton}} = \left(\underset{\text{Counts}}{\text{Sample Compton}} - \underset{\text{Counts}}{\text{Background Compton}}\right)$$

t = time sample counted (min)

m = mass of sample (g)

E = efficiency factor based on counts obtained from Lead-210 standard sorbed to equivalent volume of virgin GAC (cpm/dpm)

I_g = 0.04 = amount of Lead-210 emissions which are gamma radiation

$$A = A' \cdot \frac{1\ \text{pCi}}{2.22\text{dpm}} \cdot \frac{1{,}000\ \text{g}}{\text{kg}}$$

where $$A = \frac{\text{pCi Lead-210 measured}}{\text{kg GAC (dry weight)}}$$

II. Calculation of theoretical Lead-210 accumulated on the GAC.

Assumptions:

(i) All radon-222 removed by the GAC was completed retained.

(ii) All lead-210 accumulation originates from decay of the radon-222 progeny. Any direct adsorption of lead-210 from the water supply or lead-210 resulting from the decay of other adsorbed species such as radium-226 was considered negligible.

(iii) The measured water flowrate and radon removed by a given volume of GAC were constant over a sampling interval.

A. Calculation of pCi of Radon-222 adsorbed in a given volume of GAC during the entire operating period (up to time of coring)

$$A_{i_{Rn}} = Q_{w_i}(C_{inf_i} - C_{eff_i})t_i$$

where $A_{i_{Rn}}$ = pCi radon adsorbed during sampling interval i

Q_{w_i} = water flowrate during sampling interval i (L/d)

C_{inf_i} = measured influent radon into volume of GAC during sampling period i (pCi/L)

C_{eff_i} = measured effluent radon out of volume of GAC during sampling period i (pCi/L)

$(C_{inf_i} - C_{eff_i})$ = net change in radon activity through the volume of GAC (pCi/L)

t_i = length of time of sampling interval (d)

$$A_{T_{Rn}} = \sum_{i=1}^{n} A_{i_{Rn}}$$

where $A_{T_{Rn}}$ = pCi of radon adsorbed to a given volume of GAC during the entire operating period (up to time of coring)

B. Calculation of mass of Radon-222 adsorbed to a given volume of GAC

$$M_{Rn} = A_{T_{Rn}} \times \frac{6.50 \times 10^{-18}\text{g Rn}}{\text{pCi Rn}}$$

where M_{Rn} = g of radon adsorbed to a given volume of GAC during the operating period (up to the time of coring)

$$\frac{6.50 \times 10^{-18}\text{g Rn}}{\text{pCi Rn}} = \frac{2.22 \text{ dpm}}{1 \text{ pCi}} \times \frac{1}{\lambda_{Rn}} \times \frac{\text{mole Rn}}{6.023 \times 10^{23} \text{ atoms}} \times \frac{222\text{gRn}}{\text{mole Rn}}$$

$$\lambda_{Rn} = \frac{\ln 2}{t_{1/2_{Rn}}} = \frac{0.693}{(3.82\text{d})(24\text{hr/d})(60\text{min/hr})} = 1.26 \times 10^{-4} \text{ min}^{-1}$$

where λ_{Rn} = number of radon atoms disintegrating per total radon atoms present per minute

$t_{1/2_{Rn}}$ = half-life of Radon-222 = 3.82d

C. Calculation of atoms of Radon-222 absorbed to a given volume of GAC.

$$N_{Rn} = M_{Rn} \text{ x } \frac{\text{mole Rn}}{\text{222gRn}} \text{ x } \frac{6.023 \text{ x } 10^{23} \text{ atoms Rn}}{\text{mole Rn}}$$

where N_{Rn} = number of atoms of radon adsorbed to a given volume of GAC

D. Calculation of mass of Lead-210 (Pb) adsorbed to a given volume of GAC

$$N_{Pb} = N_{Rn}$$

where N_{Pb} = number of atoms of Lead-210 adsorbed to a given volume of GAC

$$M_{Pb} = N_{Pb} \text{ x } \frac{\text{mole Pb}}{6.023 \text{ x } 10^{23} \text{ atoms}} \text{ x } \frac{\text{210g Pb}}{\text{mole Pb}}$$

where M_{Pb} = g of Lead-210 adsorbed to a given volume of of GAC

E. Calculation of pCi of Lead-210 (Pb) adsorbed to a given volume of GAC

$$A_{T_{Pb}} = M_{Pb} \text{ x } \frac{\text{pCi Pb}}{1.24 \text{ x } 10^{-14} \text{g Pb}}$$

$$\frac{\text{pCi Pb}}{1.24 \text{ x } 10^{-14} \text{g Pb}} = \frac{\text{pCi}}{\text{2.22 dpm}} \text{ x } \lambda_{Pb} \text{ x } \frac{6.023 \text{ x } 10^{23} \text{ atoms Pb}}{\text{mole Pb}} \text{ x } \frac{\text{mole Pb}}{\text{210g Pb}}$$

where $A_{T_{Pb}}$ = pCi Lead-210 adsorbed to a given volume of GAC

$$\lambda_{Pb} = \frac{\ln 2}{t_{1/2_{Pb}}} = \frac{0.693}{(21\text{yr})(\frac{365\text{d}}{\text{yr}})(\frac{24\text{hr}}{\text{d}})(\frac{60\text{min}}{\text{hr}})} = 6.28 \text{ x } 10^{-8} \text{ min}^{-1}$$

where λ_{Pb} = number of Lead-210 atoms disintegrating per total Lead-210 atoms present per minute

$t_{1/2_{Pb}}$ = half-life of Lead-210 = 21 yr

F. Calculation of pCi of Lead-210 per mass of GAC:

$$\frac{\text{pCi Lead-210}}{\text{kg}} = \frac{A_{T_{Pb}}}{x}$$

where x = kg (dry weight) of GAC per given volume

TECHNICAL REPORT DATA

(Please read Instructions on the reverse before completing)

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16. ABSTRACT

This report presents the results of an evaluation, performed by the University of New Hampshire--Environmental Research Group (ERG), of radon removal in small community water supplies using full-scale granular activated carbon adsorption, diffused bubble aeration and packed tower aeration. Various low technology alternatives, such as loss in a distribution system and addition of coarse bubble aeration to a pilot-scale atmospheric storage tank were also evaluated. The report discusses each of the treatment alternatives with respect to their radon removal efficiency, potential problems (i.e., waste disposal, radiation exposure and intermedia pollution), and economics in smal community applications. In addition, several sampling methods, storage times, scintillation cocktails and extraction procedures currently used in the liquid scintillation technique for analysis of radon in water were compared.

17. KEY WORDS AND DOCUMENT ANALYSIS

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