Appendix J – Bench and Pilot Testing Report

Colorado Department of Health and Environment
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Colorado Radionuclide Abatement and Disposal Strategy (CO-RADS)

Bench and Pilot Testing Report

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B. Water Quality Results from all Bench and Pilot-Scale Tests
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1. Introduction

1.1. Introduction

For many years, radionuclides have been a major concern for drinking water utilities; they occur in low concentrations throughout the United States. The EPA has been regulating these contaminants for 31 years now, with the latest and most strict regulations being passed in 2000. Since the passing of the 2000 Radionuclide Rule, many water utilities across the nation have had difficulties adhering to the new regulations.

Extensive research in the 1980s led to discovery of many effective treatment processes; however most of these are not commonly used drinking water processes. This study focuses on radionuclides occurrence, concentration levels, regulations governing radionuclides in drinking water, effective treatment technologies for removal of radionuclides from drinking water, and disposal of radioactive materials generated during the treatment processes. The current chapter provides background information about occurrence and technologies for removal of radionuclides from water.

1.2. Radionuclide Classifications

Different classifications exist for radionuclides:

1. The type of radiation that the radionuclide emits
2. Whether the radionuclide is man-made or naturally-occurring
3. The decay series that the radionuclide is a part of

1.2.1. Type of Decay

The basic characterization of radionuclides begins with the type of radiation emitted. There are three types of radiation: alpha, beta, and gamma. These three forms of radiation vary in penetration strength, size of the emitted particle, and how often the particles are emitted.

Alpha decay occurs as the result of an emission of an alpha particle (two protons and two neutrons) from an atom. The resulting daughter isotope is four less in atomic weight and two less in atomic number than the parent. Alpha particles are unable to penetrate skin and represent the least active of the three forms of radiation. However, if ingested with drinking water, alpha-emitting radionuclides are very hazardous to health as they can damage internal organs.

Beta decay occurs as the result of a neutron transforming into a proton and an electron, after which the electron is emitted. The resulting daughter isotope has the same atomic weight and its atomic number is one unit higher than the parent isotope. Beta particles are 100 times more active than alpha particles; they can penetrate skin but can be stopped by an aluminum foil. Beta radiation can cause adverse health effects as an external source and as an internal source (ingestion).

Gamma radiation occurs as the result of a decay process (either alpha or beta); leaving a newly formed daughter isotope in an excited state. In order to get down to a lower energy state, a photon of electromagnetic energy is emitted. The resulting daughter isotope has the same atomic weight and atomic mass. Gamma (photon) radiation, the most penetrating of the three forms of decay, is 100,000 times more active than alpha radiation. Gamma radiation is so strong that it takes a concrete wall or lead shielding to be stopped.

The half-life of a radionuclide impacts its importance in drinking water. The half-life is the amount of time that it takes to reduce a radionuclide to 50% of its original activity. Half-lives of radionuclides can range from less than a second (1.4×10^{-4} s for Po-214) to billions of years (4.49×10^{9} yr for U-238). A radionuclide with a very short half-life has a higher activity per mass but tends to be found at very low concentrations due to its rapid decay rate. Radionuclides with a long half-life have a low activity per mass but due to their slow decay, they are found at higher concentrations in the environment. The radionuclides of greatest concern for drinking water are those that have half-lives that are longer than the time it takes water to travel through drinking water distribution systems and be ingested. Therefore, radionuclides with half-lives less than 1 hour are not contaminants of concern in drinking water systems.

1.2.2. Man-Made vs. Naturally-Occurring Radionuclides

Radionuclides can be classified by the way they are originated. Radionuclides can either occur in the environment as a result of natural process or as a result of man-made processes. This paper focuses on naturally occurring radionuclides; however, it is important to note the differences in these two different groups.

Naturally occurring radionuclides refer to uranium-238, thorium-232, uranium-235, and the decay products of these radionuclides, which are found in trace amounts in the earth's crust. When certain rocks weather, radionuclides are mobilized into sediments in streams and into clays in aquifers, and subsequently enter many water supplies in low concentrations. The most common naturally occurring radionuclides are alpha emitters.

Man-made radionuclides are released into the environment as a result of human use of radioactive materials. Common sources include industries involved in the production of electricity, nuclear weapons, medicines, commercial products (televisions, smoke

detectors), and research. Contrary to naturally occurring radionuclides, the most common man-made radionuclides are beta and gamma emitters.

1.2.3. Decay Series

A third classification of radionuclides relates to their decay series. A decay series is a sequence of unstable isotopes decaying to other unstable isotopes, until reaching a stable isotope. There are three main decay series, which are characterized by the naturally-occurring radionuclide that begins the sequence of decays. These three series are the Uranium, Thorium, and Actinium Series that start with uranium-238, thorium-232, and uranium-235, respectively.

The Uranium Series is initiated by the alpha decay of naturally occurring uranium-238. The series follows a sequence of fourteen decay steps until it reaches the stable lead-206 isotope (Figure 1-1). The Uranium Series is the series of greatest concern due to the number of long-lived alpha emitters and because uranium-238 naturally occurs at concentrations well beyond the naturally occurring concentrations of thorium -232 and uranium-235. The constituents of main concern for drinking water regulations from the Uranium Series are uranium-238, uranium-234, radium-226, and radon-222.

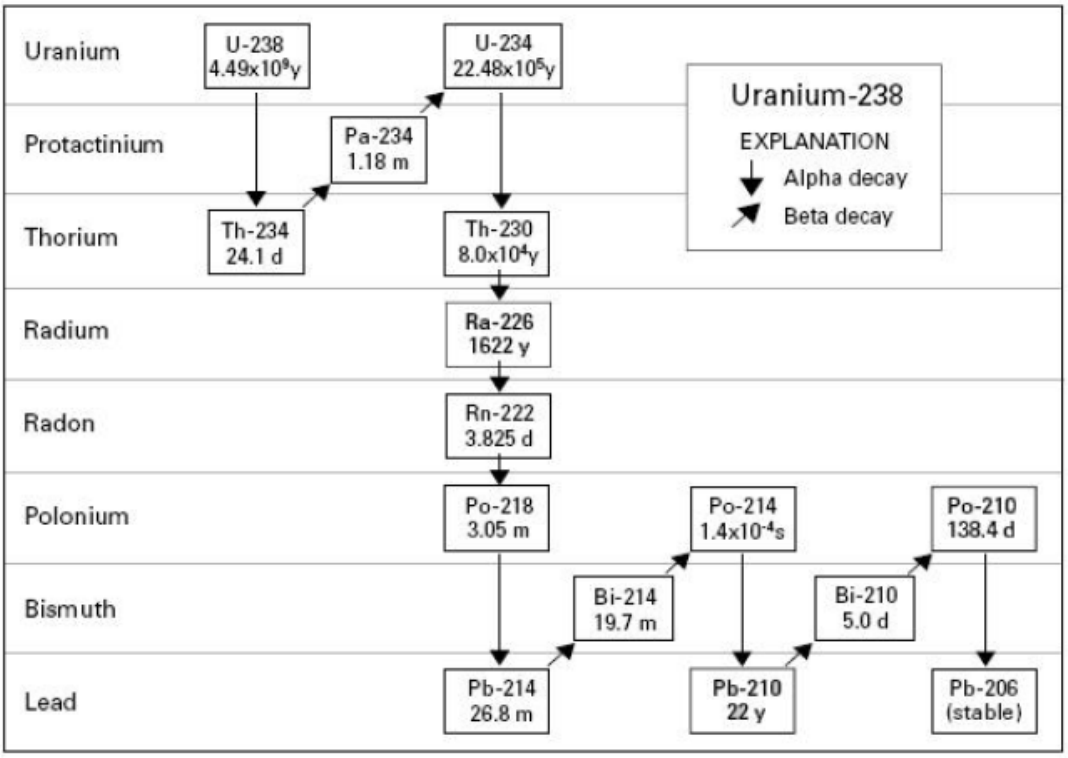

Figure 1-1: Uranium Decay Series [1]

The Thorium Series is initiated by the alpha decay of naturally occurring thorium-232. The series follows a sequence of ten decay steps until it reaches the stable lead-208 isotope (Figure 1-2). The constituent of main concern for drinking water regulations from the Thorium Series is radium-228. An important difference between the Thorium Series and Uranium Series is that the majority of decay steps in the Thorium Series are short-lived decays (half-life less than one day), whereas the majority of the decay steps in the Uranium Series are long-lived decays (half-life greater than one day).

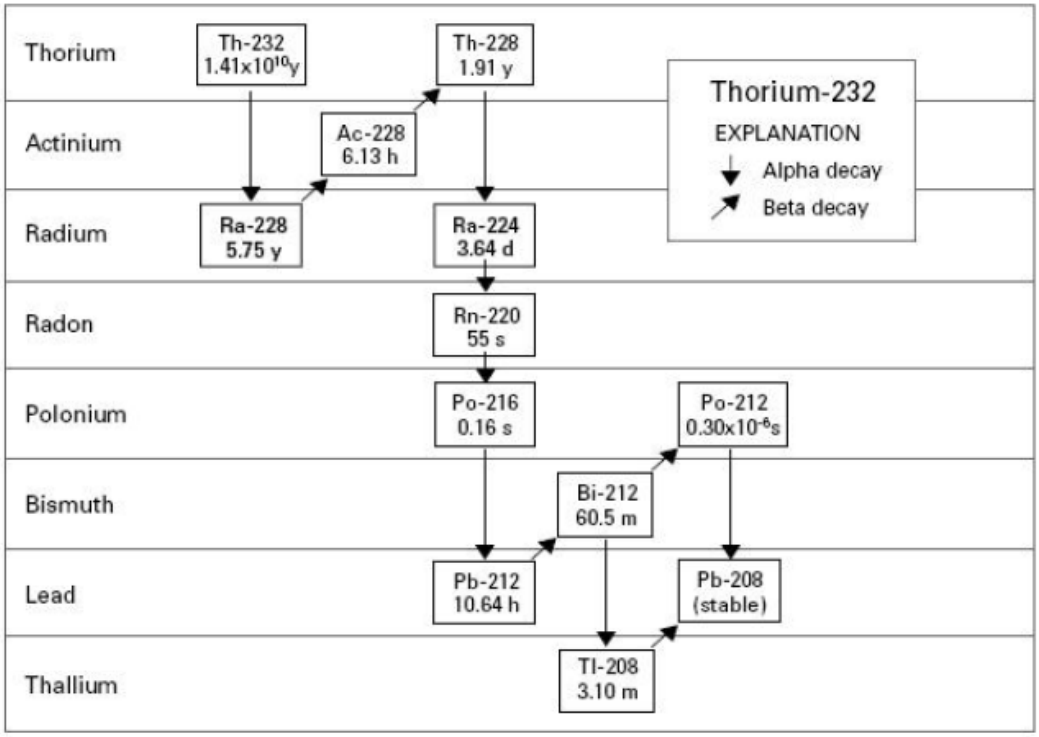

Figure 1-2: Thorium Decay Series [1]

1.2.4. Radionuclide Units

There are a number of different units by which radionuclides can be quantified. The most common unit used to quantify radionuclides in drinking water is the curie. One curie (Ci) is equivalent to 3.7 x 10^{10} transformations per second. Another unit used for radionuclides is rem or roentgen equivalent in man. This measures the whole body dose of radiation that a person is exposed to. Lastly, radionuclides are sometimes quantified by a mass per volume concentration such as µg/L.

1.3. Drinking Water Regulations Pertaining to Radionuclides

The first regulations concerning radionuclides in drinking water were introduced in the 1976 Radionuclides Rule. This rule set maximum contaminant levels (MCLs) for three radionuclide constituents, including combined radium, gross alpha, and beta particle/photon emitters. The combined radium MCL was set at 5 pCi/L and the expected risk was calculated as one excess death in 5,000-20,000. The combined radium MCL included both radium-226 (an alpha emitter) and radium-228 (a beta emitter). The gross alpha MCL was set at 15 pCi/L and the risk associated with it varied based on differing concentrations of various alpha emitters. Gross alpha particles include all alpha-emitting radionuclides except for uranium and radon. Lastly, the MCL for beta particle and photon emitters was set at 4 mrem/year with a perceived risk of one excess death in 15,000. The two radionuclides that are included in the beta particle and photon emitters dose calculation are tritium and strontium-90. The concentrations of these radionuclides that will result in a whole body dose of 4 mrem/year are summarized in Table 1-1.

Table 1-1. Average annual concentration resulting in a whole body dose of 4 mrem [2]

Radionuclide	Critical Organ	Concentration (pCi/L)
Tritium	Total Body	20,000
Strontium-90	Bone Marrow	8,000

In 1991, the EPA issued the 1991 Proposed Radionuclides Rule as a proposal to alter the MCLs that were set in 1976. The proposed modifications included changing the radium MCL from 5 pCi/L combined (radium-226 and radium-228) to 20 pCi/L for radium-226 and 20 pCi/L for radium-228, removing radium-226 from the alpha emitters measured in the gross alpha MCL, increasing the beta and photon emitters MCL by four times the original value of 5 mrem/yr, and implementing an MCL of 30 pCi/L for uranium. When the Final Radionuclide Rule promulgated in 2000, only one of the 1991 proposed changes was adopted. An MCL of 30 µg/L was implemented for uranium instead of the 1991 proposed MCL of 30 pCi/L. The current EPA regulations governing radionuclides are summarized in Table 1-2.

Table 1-2. MCLs for regulated radionuclides in drinking water [2]

Contaminant	MCL (year promulgated)	MCLG
Combined Radium (226 & 228)	5 pCi/L (1976)	Zero
(Adjusted) Gross Alpha	15 pCi/L (1976)	Zero
Beta Particle and Photon Radioactivity	5 mrem/yr (1976)	Zero
Uranium	30 µg/L	Zero

The 2000 Final Radionuclides Rule also implemented a more stringent program for utilities to monitor for radionuclides. The new monitoring protocol requires four consecutive quarters of initial monitoring for gross alpha, combined radium, and uranium. If the average of these results is below the detection limit, sampling for

radionuclides will only be required once every nine years. If the average of these results is above the detection limit but less than one-half the MCL, sampling will be required once every six years. If the average of these results is above one-half the MCL but below the MCL, sampling will be necessary once every three years. And lastly, if the average of these results is above the MCL, sampling must be done every quarter until there have been four consecutive quarters below the MCL. Additionally, if any sample after the initial monitoring is above the MCL, sampling must be done every quarter until there has been four consecutive quarters below the MCL.

Minnesota is one state in the United States that has a stricter radionuclide regulation than the EPA's regulation. The gross alpha MCL in Minnesota is set at 5 pCi/L as opposed to the 15 pCi/L MCL set by the EPA. All other states defer to the MCLs set by the EPA. Technologically-enhanced naturally occurring radioactive material (TENORM) disposal, which the EPA has not released specific regulating rules, is mainly a state issue and will be discussed in Section 1.6.

1.4. Occurrence

Radionuclides are found in very low concentrations in most water supplies across the United States. The National Inorganics and Radionuclides Survey (NIRS), which was carried out in the early 1980's was the first comprehensive study into the occurrence of radionuclides in groundwater. Groundwater samples were collected at 1,000 sites ranging from very small to very large public water systems and at least one site in 48 different states. The average combined radium concentration found in the study was 2.31 pCi/L, well below the MCL for radium [3]. Likewise, the average uranium measured was 0.82 µg/L, also below the MCL. Average concentrations of radium-226, radium-228, radon-222, and uranium-238 for Colorado and for the United States are summarized in Table 1-3.

Table 1-3. Average radionuclide concentrations in groundwater [3]

	Ra-226 (pCi/L)	Ra-228 (pCi/L)	Rn-222 (pCi/L)	U (µg/L)
Colorado Average	0.199	1.00	329.9	6.81
US Average	0.905	1.41	249.0	0.82

According to the Colorado Department of Public Health and the Environment (CDPHE), in 2006 there were 69 water utilities in Colorado that exceeded at least one radionuclide MCL [4]. The largest water utility that violated a radionuclide regulation was the city of Sterling (population of 13,000). The majority of affected water systems, however, served populations of less than 200 people. Most of the affected counties are in the central and eastern part of the state. Otero County in southeastern Colorado had 19 out of compliance water systems, the largest number of any other county in Colorado. Counties which had radionuclide violations and the populations affected are illustrated in Figure 1-3.

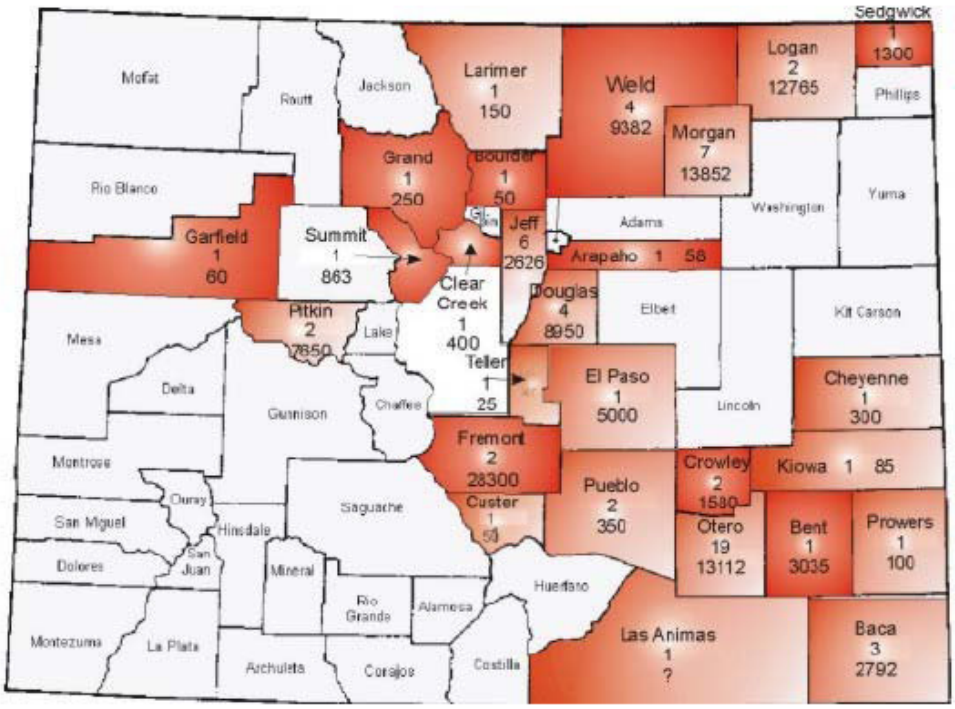

Figure 1-3: Counties with water utilities that exceeded radionuclide MCL and the population affected [4]

Out of the radionuclides that are regulated, the most commonly violated MCLs in Colorado are combined radium and the uranium. Radium is found primarily in groundwater supplies; occurrence of radium in surface water has been directly linked to groundwater discharges [5]. The region in the United States with the most elevated levels of radium includes the north-central states, including Illinois, Wisconsin, Missouri, Minnesota, and Iowa [6]. In aquifers with high total dissolved solid concentrations, such as in the north-central states and coastal areas with saltwater intrusion or brackish water, radium solubility is increased by the common-ion effect, resulting in higher radium concentrations in groundwater [7].

Uranium is found in both groundwater and surface water. Uranium occurs in elevated levels in the western United States [8]. Uranium is particularly of local concern in Colorado. It was found that the an average uranium concentration in Colorado is eight times greater than the national average and second highest among states in the NIRS study [3]. Additionally, a uranium study in Jefferson County, Colorado in the 1980's showed wells with uranium concentrations ranging from 20-90 µg/L [9].

1.5. Treatment

When considering removal of radionuclides from drinking water, there are four main processes listed as best available technologies (BATs) by the EPA. These include ion exchange, reverse osmosis, lime softening, and coagulation/filtration. BATs for each radionuclide contaminant as assessed by the EPA are summarized in Table 1-4.

Table 1-4. Best available technologies for the treatment of radionuclides [2]

Contaminant	Best Available Technologies
Combined Radium	Ion Exchange, Reverse Osmosis, Lime Softening
Uranium	Ion Exchange, Reverse Osmosis, Lime Softening, Coagulation/Filtration
Gross Alpha	Reverse Osmosis
Beta Particle and Photon Emitters	Ion Exchange, Reverse Osmosis

1.5.1. Ion Exchange

Ion exchange is effective for treating water contaminated with radium, uranium, or both. Radium, which is commonly found as the Ra^{2+} cation, can be removed with strong acid cation (SAC) resins. Uranium, which is commonly found as the multivalent complex anion $UO_2(CO_3)_3^{4-}$, can be removed with strong base anion (SBA) resins. The separation factor of a resin defines the exchange preference of one ion over another reference ion. For example, the separation factor of (Cl^-) for $UO_2(CO_3)_3^{4-}$ is 3,200. This means that the $UO_2(CO_3)_3^{4-}$ ion is preferred over the Cl^- ion 3,200 to 1. Separation factors for commonly encountered ions in drinking water are summarized in Table 1-5. Data in Table 1-5 demonstrates that because uranium and radium ions are the most preferred ions for their respective resins, ion exchange is a very effective treatment process for their removal.

Table 1-5. Separation factors for SAC resins and SBA resins [10]

Strong Acid Cation Resins*		Strong Base Anion Resins**	
Cation	α^i_{Na+}	Anion	α^i_{Cl-}
Ra^{2+}	13.0	$UO_2(CO_3)_3^{4-}$	3200
Ba^{2+}	5.8	ClO_4^-	150
Pb^{2+}	5.0	CrO_4^{2-}	100
Cu^{2+}	2.6	SeO_4^{2-}	17
Ca^{2+}	1.9	SO_4^{2-}	9.1
Fe^{2+}	1.7	HSO_4^-	4.1
Mg^{2+}	1.67	NO_3^-	3.2
K^+	1.67	Br^-	2.3
Mn^{2+}	1.6	NO_2^-	1.1
NH_4^+	1.3	Cl^-	1.0
Na^+	1.0	HCO_3^-	0.27
H^+	0.67	F^-	0.07

* SAC resin is polystrene divinylbenzene with sulfonate functional groups
** SBA resin is polystyrene divinylbenzene with $-N+(CH_3)_3$ functional groups

Ion Exchange for Uranium

Bench-scale and full-scale ion exchange experiments have demonstrated ion exchange potential for uranium [9, 11] . One small full-scale ion exchange study in Jefferson County, Colorado exhibited 99.8% removal of uranium and greater than 85% removal of gross alpha for bed volumes ranging from 2,260 to 9,750 by an SBA resin [9]. Results from another ion exchange study examining three different SBA resins for breakthrough of uranium are summarized in Table 1-6 [11]. As shown by both studies, large volumes of water with elevated uranium can be treated before uranium breakthrough occurs. The advantage of these long run times is that it will reduce operation and maintenance required and will lower chemical cost; however, it will result in a highly-loaded resin when breakthrough does occur.

In addition to using SBA resins for uranium removal, there is also the option of using a resin that selectively removes uranium, such as Dow's DOWEX 1. While selective resins will be able to process larger volumes of water than SBA resins, the resulting loaded resin will be so tightly bound that regeneration becomes unfeasible and disposal is required.

Table 1-6. Uranium removal with anion exchange treatment [11]. Resin C is DOWEX 21K, Resin B is DOWEX SBR-P, and Resin D is IONAC A641.

Resin Test Cycle	Influent Uranium Concentration µg/L	Bed Volumes Treated until Uranium Breakthrough
Resin C		
1	200	8,020
2	300	17,375
4	300	13,365
Average		12,920
Resin B		
1	200	18,712
2	300	16,038
4	300	10,024
Average		14,925
Resin D		
1	200	10,024
2	300	18,043
4	300	20,048
Average		16,038

Ion Exchange for Radium

The ion exchange process used in the removal of radium is identical to that of household softening of tap water. In the process of treating water with SAC resins, hardness and radium is adsorbed onto the resins. Because radium is preferred over calcium and magnesium, radium will breakthrough after hardness breakthrough. A field-study in Lemont demonstrated that hardness breakthrough took 300 bed volumes for a virgin SAC

resin for a groundwater with a hardness of approximately 200 mg/L as $CaCO_3$ [12]; the radium, with an approximate influent concentration of 20 pCi/L, broke through after 1,200 bed volumes. Following the first run, radium broke through at the same time as hardness for the subsequent runs. Therefore, this sequence suggests that to appropriately remove radium from groundwater using an SAC ion exchange system, the resin should be regenerated at hardness breakthrough.

There are also cation exchange resins that are selective for certain cations. An example of this type of resin is the DOWEX RSC (radium selective complexer). A study in Red Hill Forest, CO demonstrated the large capacity of the DOWEX RSC for removal of radium from groundwater, resulting in longer runs than SAC resins [13]. However, like the DOWEX 1 for uranium, disposal of the DOWEX RSC resins becomes a problem due the high concentration of radium on the spent resins.

Ion Exchange for Uranium and Radium
Because different types of ion exchange resins absorb uranium and radium, it brings up the question of how to treat water with elevated levels of both uranium and radium. Clifford and Zhang [14] demonstrated that by blending SBA and SAC resins into one ion exchange unit, significant reductions in both constituents were achieved. The greatest combined reduction resulted from a 90% SAC and 10% SBA non-stratified ion exchange unit.

1.5.2. High Pressure Membrane Processes

Like ion exchange, high pressure membranes (reverse osmosis and nanofiltration) have been thoroughly studied for removal of radionuclides [15-17]. A pilot plant study in Florida examined the removal efficiency of uranium from a groundwater source containing 300 µg/L [16]. Four different reverse osmosis membranes were tested in the study and all of them yielded removal of greater than 99%. A uranium study involving four different nanofiltration membranes showed uranium removals greater than 90% [17].

A pilot plant study in Illinois evaluated radium removal with three different reverse osmosis membranes operated at different pressures [15]. The membranes that were operated at 125 and 350 psi had greater than 99% removal of radium from an elevated radium water source, while the membrane operated at 70 psi rejected only 91% of the radium. In these three systems, radium rejection slightly exceeded hardness rejection, suggesting that hardness could be used as a surrogate for monitoring radium removal.

1.5.3. Lime Softening

Lime softening is another proven technology for the removal of both radium and uranium. Jar tests studies demonstrated radium removal ranging between 80-92% and uranium removal ranging between 85-99% [18]. Removal was most effective when magnesium hydroxide is formed, either from natural magnesium in the water source or

through the addition of magnesium prior to lime softening. Removal was also optimized when the pH is greater than 10.6.

1.5.4. Coagulation/Filtration

Coagulation and filtration is a commonly utilized drinking water process that has been shown to remove uranium. Jar testing experiments demonstrated that coagulation and filtration can remove greater than 80% uranium [18]. Removal was optimized at pHs of 6 and 10. Lower removal was observed at pHs of 4 and 8. A study into the removal efficiencies of uranium by coagulation and filtration in actual operating drinking water plants across the United States revealed that uranium is not effectively removed by these processes when the uranium concentration of the feed water is less than 15 µg/L [19]. Figure 1-4 illustrates the uranium removal based on alum dose and pH.

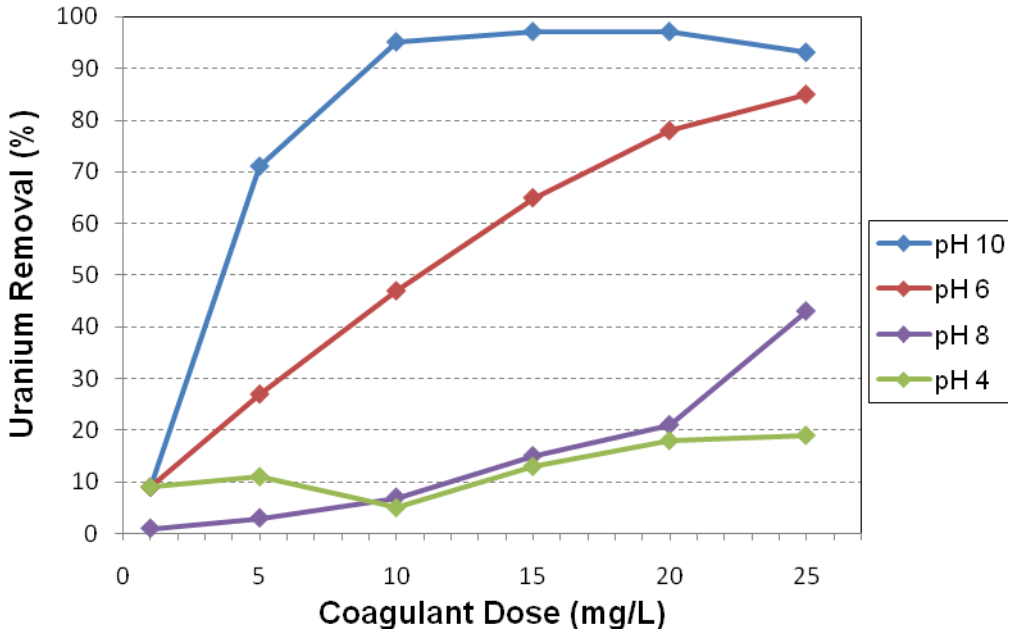

Figure 1-4: Uranium removal based on alum dose and ph via coagulation/filtration (Adapted from Lee and Other, 1982) [18]

1.5.5. Hydrous Manganese Oxide Filtration

Hydrous manganese oxides (HMOs) are known to be good adsorbers of radium. This treatment process involves dosing of a solution containing pre-formed MnO_2 followed by mixing and filtration through granular media. HMO treatment has been suggested as a cost-effective treatment alternative due to the ability to formulate HMO solutions on-site (with relatively low chemical costs) as well as taking advantage of filters that may

already be installed at treatment facilities. [20] If filters are already installed at a treatment facility, additional costs associated with employing HMO include chemical cost of producing or purchasing pre-made HMO, the addition of a pump and mixer to dose the HMO solution, and potential upgrades to the backwash system. There are several different ways to produce HMO solutions, however, the reaction governing the most common one is [20]:

$$3\ MnSO_4 + 2\ KMnO_4 + 2\ H_2O \rightarrow 5\ MnO_2 + 4\ H^+ + 2K^+ + 3SO_4^{2-} \qquad \text{Eq. (1.1)}$$

Batch testing has demonstrated that doses of 0.5 and 1.0 mg/L as Mn have resulted in radium removals of approximately 80% and 90%, respectively [20]. Pilot testing has shown radium removals ranging from 80 to 95% for different doses of 0.5 to 1.0 mg/L as Mn [10, 20]. It has been suggested that hardness competes with radium for adsorption onto the HMO, but hardness levels above 200 mg/L as $CaCO_3$ had negligible effect on radium removal [20].

1.5.6. Other Technologies

In addition to the technologies previously discussed, there are several other treatment processes which have shown ability to remove radionuclides from drinking water. These processes have not been intensively studied for radionuclide removal and may have deficiencies related to waste disposal. The removal efficiencies of several processes that have not been thoroughly studied for radionuclide treatment are summarized in Table 1-7.

Table 1-7. Removal efficiencies of lesser used treatment processes [16]

Radium		Uranium	
Method	Removal Efficiency (%)	Method	Removal Efficiency (%)
Electrodialysis	90	Activated Alumina	90
Greensand Filtration	25 - 50	Activated Carbon	5 - 90

1.6. Disposal of Treatment Residuals

In addition to evaluating removal efficiencies when considering potential technologies, it is also important to consider the impact of the waste generated in each process. None of the technologies available for radionuclide removal actually eliminate the radioactive materials altogether; these technologies merely transfer the radionuclides from the water to a more concentrated phase. For example, when breakthrough occurs in ion exchange, there are two options: to regenerate the resins or to dispose of the spent resins and start with new resins. In the first case, the process of regenerating the resins will result in brine and backwash water that are highly concentrated in radionuclides. In the latter case, the spent resins must be disposed of as solid waste that is also concentrated in radionuclides. Membrane processes (e.g., RO, NF, and electrodialysis) will generate brine highly

concentrated with radionuclides and the spent membranes might require special disposal. Adsorbent processes (e.g., activated alumina and activated carbon also result in spent media and spent filter backwash water. Filtration processes (e.g., coagulation/filtration, greensand filtration, HMO filtration) will have to dispose of spent media, sludge, and backwash water.

In order to deal with the issues associated with residuals generation, the CDPHE released draft regulations in 2007 pertaining to TENORM [4]. The disposal regulations for radium and uranium are summarized in Table 1-8. Solid residuals with less than 10 pCi/g combined radium and less than 100 pCi/g of uranium may be disposed of at municipal solid waste landfills. Solid residuals with up to 50 pCi/g combined radium and up to 300 pCi/g of uranium must be disposed of in an industrial landfill. Solid residuals with up to 400 pCi/g combined radium and up to 0.05% by weight uranium must be disposed of in a RCRA C Hazardous Waste Landfill. Any residuals that have a higher concentration than those listed must be disposed of at the low level radioactive waste facility in Richland, Washington. These disposal regulations could present difficulties to water utilities because there are only one industrial landfill and one RCRA C Hazardous Waste Landfill in Colorado. Depending on the concentration of liquid residuals, there are three possible disposal options: discharge to surface water (only for low concentrations), discharge to sanitary sewers, or evaporate the liquid and dispose of as a solid waste.

Table 1-8. Disposal regulations for TENORM residuals in Colorado [4]

Landfill	Radium	Uranium
Municipal	< 10 pCi/g	< 100 pCi/g
Industrial	< 50 pCi/g	< 300 pCi/g
RCRA C Hazardous Waste	< 400 pCi/g	< 0.05% by weight

1.7. Selection of Treatment Alternatives

Many factors affect the selection of treatment for source waters with elevated radionuclides. Therefore, selection of treatment processes for radionuclide removal should be evaluated on a case-by-case basis. Cost is likely the most important criterion to systems and other criteria include operational complexity, worker safety issues, and other water quality issues the technology can address.

System size also influences which technologies are best suited for the system. In Colorado, the majority of water systems with radionuclide violations are very small, serving 200 people of less. For these water systems, the most cost effective treatment process is one that minimizes chemicals and required operation and maintenance. Smaller systems typically have limited; therefore, implementing a treatment process that is less demanding of the operators is recommended.

Residuals disposal options for treatment systems should also be considered. Some treatment processes will result in larger volumes or more concentrated waste residuals than others. This will affect cost, because disposal cost is directly related to the concentration and volume of the waste. There is only one industrial landfill and one RCRA C landfill in Colorado; therefore, the distance of the water system from these landfills may rule out certain treatment processes due to the cost of transporting the waste. Lastly, the concentration of radionuclides in the source water is an important parameter in selecting a treatment technology. If the radionuclide concentration in the source water is only slightly above the MCL, any of the previously discussed processes could be utilized. However, if the source water has elevated concentration of radionuclides, selecting one of the proven treatment technologies (e.g., ion exchange or reverse osmosis) could reduce demonstration testing requirements by CDPHE.

1.8. Objectives of this study

The Colorado Radionuclide Abatement and Disposal Strategy (CO-RADS) project began in 2007 as a joint effort between CDPHE, Malcolm Pirnie, and Colorado School of Mines. The goal of CO-RADS is to develop recommendations for water utilities in Colorado that are out of compliance with radionuclide MCLs. As part of that project, the objective of the current study was to evaluate several treatment alternatives at the bench- and pilot-scale level for removal of radionuclides. Using water from four representative water sources, several treatment alternatives were tested to determine removal efficiency and design criteria.

2. Materials and Methods

2.1. Water Systems Selected for Bench and Pilot Testing

Thirty three water utilities with source waters containing radionuclides above the MCLs participated in CO-RADS. The populations served by these utilities range from less than 40 to more than 13,000. Some utilities are located in the mountains of the central part of the state of Colorado, but most are located on the eastern plains of the State. As part of the CO-RADS project, a comprehensive water quality analysis was conducted on one or more water sources of each of these water utilities. This information was used to identify the systems best suited for bench and pilot testing.

Four systems were chosen for bench- and pilot-scale testing of different water treatment technologies. The main criteria for selection of the four representative systems were radionuclide concentrations, population served, and water quality (hardness levels). Information including population served, radionuclide species and hardness for the four source waters are summarized in Table 2-1. Water quality data collected through CO-RADS for all the systems are included in Appendix A.

Table 2-1. Summary of four systems chosen for bench- and pilot-scale testing.

System	Population Served	Hardness	Radium	Uranium	Gross Alpha
Sterling	13,794	650 (Very High)		√	√
May Valley	1,500	350 (High)	√		√
Blue Mountain	300	170 (Moderate)	√	√	√
Red Hill Forest	125	190 (Moderate)	√		√

√: Concentration is above MCL (Uranium MCL: 30 µg/L, Combined Radium MCL: 5 pCi/L, Gross Alpha MCL: 15 pCi/L)

The selection of treatment technologies was based on EPA's BATs for small systems with radionuclides, and literature review. Additionally, because the vast majority of the effected water utilities are very small in size (some are serving less than 100 people), the technologies must be applicable for use in small systems; technologies that do not require full-time operators and require minimal operations and maintenance support. Treatment technologies that were selected include:

- Ion exchange
- Hydrous manganese oxide
- Enhanced coagulation
- Nanofiltration and reverse osmosis membranes (bench scale)

- Nanofiltration (pilot scale)

A summary of each water source and technologies tested are summarized in Table 2-2. The following sections provide detailed information on the technologies, methods and materials applied for each system. Locations of the systems selected are illustrated in Figure 2-1

Table 2-2. Bench- and pilot-scale testing of water treatment technologies for each water source

System	Bench-Scale Membrane	Pilot-Scale Membrane	Enhanced Coagulation	Hydrous Manganese Oxide (HMO)	Ion Exchange (IX)
Sterling		√	√		
May Valley	√	√		√	√
Blue Mountain				√	√
Red Hill Forest				√	√

Figure 2-1: Location of the four water systems that were selected for bench - and pilot-scale testing.

2.1.1. May Valley

May Valley is the 3rd largest CO-RADS System and is located in Wiley, CO, along the Arkansas River in the southeast part of the State. The utility serves a population of 1,500 and currently operates 10 wells. It was chosen to represent the group of utilities that have elevated radium and gross alpha, as well as high hardness, alkalinity, and total dissolved (TDS) in their source waters.

Treatment at each well in includes oxidation by chlorination followed by filtration to remove iron and manganese. A schematic drawing of the May Valley treatment train is illustrated in Figure 2-2. Chlorine is added to the raw groundwater as it is pumped out of the well. After the addition of chlorine, the water is stored in a storage tank that feeds the distribution system. As water leaves the storage tank, it passes through a pressurized sand filter to remove iron and particulate matter before entering the distribution system.

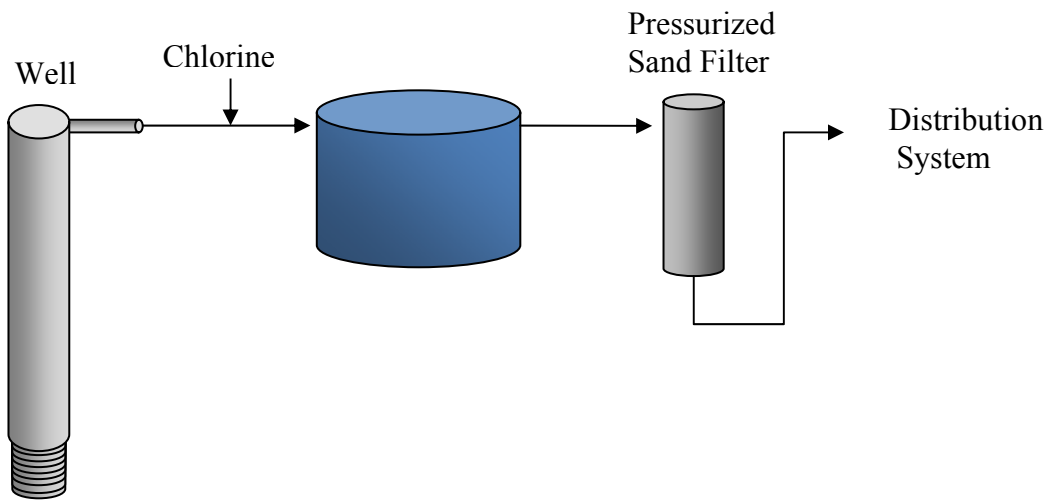

Figure 2-2: Current May Valley treatment train. Groundwater pumped from the wells is chlorinated and stored in a storage tank. Based on demand in the distribution system, the water leaves the storage tank and flow through a pressurized sand filter.

Water quality data from selected wells in May Valley are summarized in Table 2-3. For wells designated with "T" after their names, the data represent water quality after treatment at the wellhead.

Table 2-3. May Valley water quality data collected by Malcolm-Pirnie. Water samples were collected from Wells 2 and 3 prior to treatment (002 and 003) and following treatment (002T and 003T).

Constituent	Units	Well			
		002	002T	003	003T
GAA	pCi/L	58	60	48	32
Combined Ra	pCi/L	27.5	16.8	11.7	11.1
Radon 222	pCi/L	140	270	280	380
Uranium	µg/L	0.5	0.5	0.5	0.5
Alkalinity	mg/L	400	400	400	400
Ca	mg/L	80	80	60	64
Fe	mg/L	1.1	0.04	0.55	0.05
Mg	mg/L	36	36	21	22
Mn	mg/L	0.15	0.005	0.071	0.007
SO4	mg/L	1,200	1,100	1,100	1,100
pH		7.6	7.9	7.6	7.8
TDS	mg/L	2,040	1,970	1,960	1,950
Temp	oC	18.3	19.2	20.3	18
TOC	mg/L	0.5	0.5	0.6	NA
Turbidity	NTU	14.5	0.08	6.38	0.07

Wells at May Valley contained combined radium and gross alpha above the MCLs. The average combined radium concentration measured between 2006 and 2008 was 13.9 pCi/L and the maximum concentration was 21.7 pCi/L. The average gross alpha concentration measured between 2006 and 2008 was 30.7 pCi/L with a maximum concentration of 60.0 pCi/L. Additionally, May Valley has a high hardness (~360 mg/L as $CaCO_3$) and a very high TDS (~2,000 mg/L) in the groundwater.

Three water treatment technologies were tested for May Valley:

- Ion exchange
- HMO
- High pressure membranes

2.1.2. Blue Mountain

Blue Mountain Water District is one of two utilities in the State with elevated levels of radium, uranium, gross alpha, and radon in their groundwater. It is located 10 miles north of Golden, CO, at the foothills of the Rocky Mountains. The utility serves a population of 300 and currently operates 5 wells. Blue Mountain was chosen to represent CO-RADS

systems that had elevated levels of radium and gross alpha, as well as having lower TDS and hardness concentrations compared to other systems.

Water quality data from one well in Blue Mountain is summarized in Table 2-4. Blue Mountain has elevated levels of combined radium, uranium, and gross alpha. The community installed a water treatment system to treat radium, uranium, and gross alpha to comply with the MCL. Additionally, Blue Mountain has a moderately-high hardness (~170 mg/L as $CaCO_3$) and high concentration of iron and manganese.

Table 2-4. Blue Mountain water quality collected by Malcolm-Pirnie.A raw water sample was collected from Well 1.

Constituent	Units	Well 001
GAA	pCi/L	43
Gross Beta	pCi/L	32
Combined Ra	pCi/L	16.6
Radon 222	pCi/L	10,570
Uranium	µg/L	16
Ca	mg/L	44
Fe	mg/L	1.9
Mg	mg/L	14
Mn	mg/L	0.23
SO_4	mg/L	16
TDS	mg/L	170
TOC	mg/L	0.25
Turbidity	NTU	17

A schematic of Blue Mountain's treatment train is illustrated in Figure 2-3. The current treatment train at Blue Mountain includes greensand filtration for removal of iron and manganese, cation exchange for removal of radium, anion exchange for removal of uranium, an air stripping tower for removal of radon and hydrogen sulfide, and chlorination for disinfection.

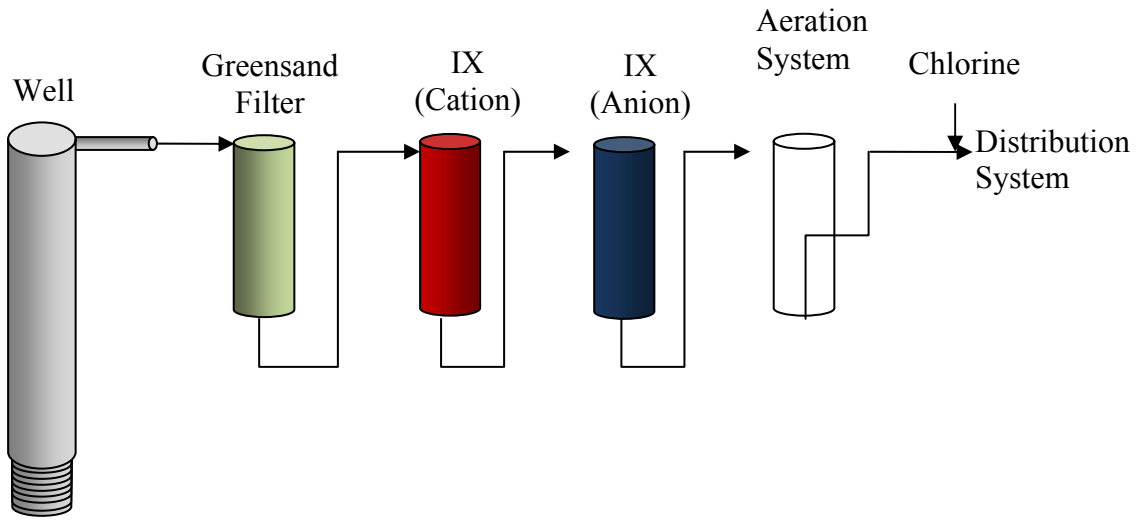

Figure 2-3: Current Blue Mountain treatment train. Water at Blue Mountain undergoes a five-stage treatment process; greensand filtration, cation exchange, anion exchange, air stripping, and chlorination before entering the distribution system.

Two treatment technologies were tested with water from Blue Mountain – ion exchange and HMO. Blue Mountain's source water differs from May Valley because the hardness in Blue Mountain's water is 50% lower than. Also, Blue Mountain has very high levels of iron and manganese in the water

2.1.3. Red Hill Forest

Red Hill Forest Property Owner's Mutual Water and Cattlemen's Association (Red Hill Forest) serves the Red Hill Forest subdivision, which is located approximately 10 miles southeast of the Town of Fairplay, in the central part of Colorado. The utility serves a population of 150 and currently operates two wells. Red Hill Forest was selected because its water has the highest level of radium and radon of all the CO-RADS systems. Additionally, Red Hill Forest is the utility with the longest history of radionuclides treatment in Colorado, including being the site for a 1990 study that investigated the use of a radium selective complexer, [5] which allowed the team access to additional water quality data and system information. Water quality data from Well 1 in Red Hill Forest is summarized in Table 2-5.

Table 2-5. Red Hill Forest water quality collected by Malcolm Pirnie. A raw water sample was collected from Well 1.

Constituent	Units	Well 001
GAA	pCi/L	159
Gross Beta	pCi/L	87
Combined Ra	pCi/L	35.5
Radon 222	pCi/L	2,300
Uranium	µg/L	8
Alk	mg/L	45
Ca	mg/L	56
Fe	mg/L	3.1
Mg	mg/L	12
Mn	mg/L	0.25
SO_4	mg/L	150
pH		7.65
TDS	mg/L	240
TOC	mg/L	0.6
Turbidity	NTU	17

Red Hill Forest water has moderately-high hardness (~190 mg/L as $CaCO_3$) and elevated concentrations of iron and manganese.

A schematic of the treatment train employed at Red Hill Forest is illustrated in Figure 2-4. The Red Hill Forest treatment begins with an in-well aeration to remove radon. This is followed by addition potassium permanganate, filtration for iron and manganese removal, cation exchange for radium removal, and chlorination before entering the distribution system. On July 26, 2008 it was announced that the Red Hill Forest subdivision was negotiating purchasing of water from the town of Fairplay and pumping it to Red Hill Forest, due to the high costs of operating the described treatment system.

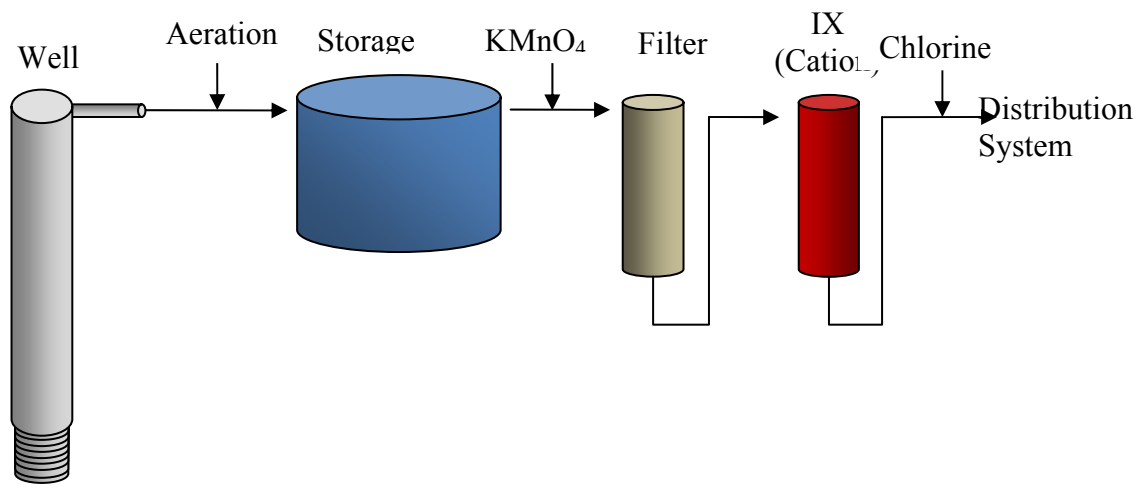

Figure 2-4: Current Red Hill Forest treatment train. Red Hill Forest water is aerated as it is pumped from the well and then stored in a storage tank. The water is then treated to remove iron/manganese (KMnO4 addition followed by filtration) and radium (cation exchange) before being chlorinated and entering the distribution system.

The two treatment alternatives selected for testing on Red Hill Forest water were ion exchange and HMO. Blue Mountain and Red Hill Forest have many similar water quality characteristics (e.g., hardness, TDS, iron, manganese, and TOC); however. Red Hill Forest has much higher levels of radium, gross alpha, and gross beta. Therefore, comparison of the results from the two systems would provide valuable information on the effect of radionuclide concentration on the efficacy of the treatment options.

2.1.4. Sterling

Sterling is the largest utility in the state that is out-of-compliance with regard to radionuclides MCLs. It is located along the South Platte River in the northeast part of the state. The utility serves a population of close to 14,000 and currently operates 16 wells. Sterling was selected to represent water utilities with elevated levels of uranium and gross alpha, as well as having very high concentrations of hardness and TDS. Sterling was also chosen because it represents over 40% of the total population in the CO-RADs study.

During the testing of pilot-scale membrane processes, Sterling water was unavailable due to well shutdown. Therefore, water from the Town of Merino was substituted for the Sterling water. The town of Merino is located 10 miles southwest of Sterling and has a population of 250. Merino draws its groundwater from the same groundwater aquifer and therefore, the water quality of Sterling and Merino are very similar. Water quality data from three wells in Sterling and one well in Merino is summarized in Table 2-6.

Table 2-6. Sterling and Merino water quality collected by Malcolm-Pirnie. Raw water samples were collected from Sterling wells 1, 2, and 22 and Merino well 1.

Constituent	Units	Wells			
		ST 01	ST 22	ST 02	M 01
GAA	pCi/L	26	NA	NA	
Gross Beta	pCi/L	67	37	NA	44
Combined Ra	pCi/L	1	1	1	1
Radon 222	pCi/L	230	320	280	240
Uranium	µg/L	30	44	24	52
Alk	mg/L	290	300	NA	320
Ca	mg/L	140	144	116	164
Fe	mg/L	0.005	0.005	0.05	0.005
Mg	mg/L	36	35	33	53
Mn	mg/L	0.003	0.001	0.018	0.001
SO_4	mg/L	460	840	340	780
pH		7.66	7.53	NA	7.5
TDS	mg/L	1,020	1,220	820	1340
TOC	mg/L	2.2	2.7	1	2.6
Turbidity	NTU	0.19	0.08	0.18	0.09

Sterling is currently out-of-compliance with regard to uranium MCL. The average uranium concentration during 2007 was 43 µg/L. Additionally, Sterling water has very high hardness (~650 mg/L as $CaCO_3$,TDS concentrations and the total trihalomethanes concentration in the water is above the MCL (2007 avg: 0.083 mg/L, MCL: 0.080 mg/L). A schematic of the treatment train in Sterling is illustrated in Figure 2-5.The current water treatment in Sterling includes chlorination at the well, after which the water is pumped to storage tanks and from the tanks to the distribution system.

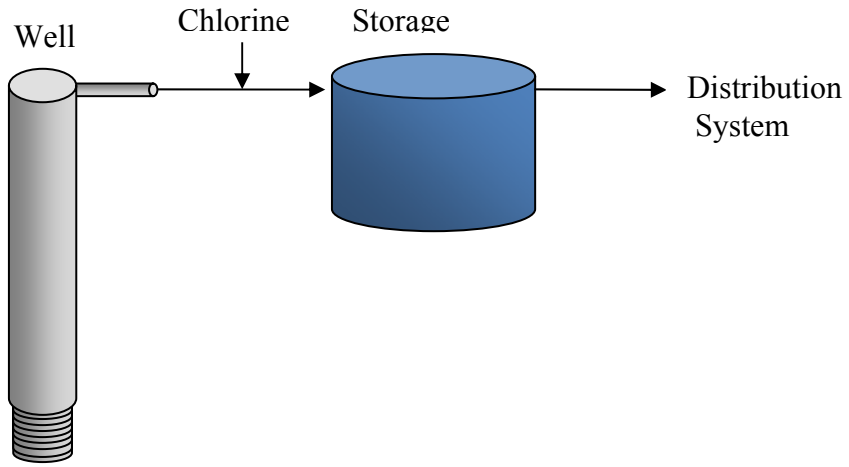

Figure 2-5: Current water treatment trainemployed at Sterling. Sterling water is pumped from 16 wells and chlorinated at the well-heads. The water is then combined in a storage tank before entering the distribution system.

Two treatment technologies were selected for testing on Sterling water: enhanced coagulation and NF high-pressure membranes pilot scale). Enhanced coagulation is a less attractive treatment option for small systems because it requires more operational oversight. However, for a medium sized system like Sterling, enhanced coagulation presents a viable option. High-pressure membranes were also selected because they have shown capacity to remove uranium [6] and organic matter.

2.2. Treatment Technologies Evaluated

A summary of the treatment technologies tested on each source water is provided in Table 2-7. Brief descriptions of each of the treatment technologies tested are provided below. More details can be found in the Task 5 Bench- and Pilot-Scale Testing Plan.

Table 2-7. Bench- and pilot-scale tests conducted for the four selected water sources

System	Ion Exchange (IX)	Hydrous Manganese Oxide (HMO)	Enhanced Coagulation	Bench-Scale Membrane	Pilot-Scale Membrane
Sterling			√		√
May Valley	√	√		√	√
Blue Mountain	√	√			
Red Hill Forest	√	√			

2.2.1. HMO

Pre-formed HMO (prepared in the laboratory at CSM) followed by filtration was tested on two different source waters (Blue Mountain and May Valley). Doses of 0.5 mg/L and

1.0 mg/L were applied to mixed jars for 25 minutes. Waters were then filtered through 0.45 µm filters. Water samples were collected from the permeate for analysis.

2.2.2. High Pressure Membranes

Two types of membranes tests were conducted: flat-sheet membrane tests and pilot tests. Two NF membranes (NF 4040 and TFC-S) and one RO membrane (TFC-HR) test was conducted on the bench-scale to determine the rejection of contaminants. More information is available in literature on radionuclide treatment with RO membranes, therefore, this study focused on NF membranes. Source water from May Valley was used for the flat-sheet membrane tests. Pilot-scale testing was performed with a dual-stage membrane system (one 4040 spiral-wound element in each stage) using a NF membrane. Membrane performance was tested at two recoveries, 30 percent during single pass and 80 percent during circulation. Two sources, May Valley and Sterling, were tested. Operating conditions for the pilot testing at 30% and 80% water recovery are shown in Table 2-8 and 2-9, respectively.

Table 2-8. Operating conditions for membrane pilot operated at 30% recovery

Parameter	Value	Unit
Feed flow rate	8.5	gpm
Combined permeate flow	2.55	gpm
Concentrate flow	5.95	gpm
Membrane surface area	156	ft²
Permeate flux during experiments	23.5 (40)	gfd (lmh)

Table 2-9. Operating conditions for membrane pilot operated at 80% recovery

Parameter	Value	Unit
Feed flow rate	9	gpm
Recycle flow	6.8	gpm
Combined permeate flow	1.75	gpm
Concentrate flow	0.45	gpm
Membrane surface area	156	ft²
Permeate flux during experiments	16 (27.5)	gfd (lmh)

2.2.3. Enhanced Coagulation

Coagulation followed by sedimentation and filtration was tested on Sterling water in 6L jars. Sterling was selected because it is the only CO-RADS system that is large enough to potentially implement coagulation as a compliance option. The coagulation-testing matrix included:

- Two coagulants: alum and ferric sulfate
- Three doses: 5, 10, and 20 mg/L as alum or ferric sulfate
- Two pH conditions: 7.5 and 6.2

Jars were mixed at 200 rotations per minute (RPM) for 2 minutes followed by 40 RPM for 30 minutes. Samples were collected after 20 minutes of sedimentation.

2.2.4. Cation Exchange

Rapid small-scale column tests (RSSCTs) that aim at evaluating adsorption capacity, competitive exchange, fouling by iron and manganese, and breakthrough profiles of radionuclides in the presence of co-ions were used to evaluate cation exchange. RSCCTs were conducted on two source waters, (May Valley and Blue Mountain). Columns were regenerated with an 8% sodium chloride (NaCl) solution and operated to breakthrough six times. Hardness was measured throughout the cycle and radium was measured between each regeneration.

3. Results and Discussion

3.1. May Valley Bench and Pilot Testing Results

3.1.1. Bench-Scale Membrane Tests

Three membranes products (TFC-S, NF 4040 and TFC-HR) were tested at the bench scale for their efficiency to remove radionuclides from May Valley water. Prior to testing, the three membranes were compacted with distilled water at a constant water flux of 30 $l/m^2 \cdot hr$ (LMH). After five hours of operation, the membrane systems stabilized at pressures of 70, 135, and 152 pounds per square inch (psi) for the TFC-S, NF 4040, and TFC-HR membranes, respectively.

During the bench-scale membrane testing, water flux was maintained constant at 30 LMH and pressure was monitored and adjusted periodically. To understand whether membrane fouling was occurring, specific water flux was calculated as the testing progressed. Specific water flux was calculated using Eq 3.1

$$\text{Specific flux} = Q_p/(A_m \cdot P) \qquad (3.1)$$

where Q_p is the permeate flow (L/hr), A_m is the surface area of the membrane (m^2), and P is the transmembrane pressure (kPa). Specific flux is a measure of water permeability through the membrane; if membrane fouling occurs, membrane permeability, and thus specific water flux, decreases. Specific water flux as a function of time is illustrated in Figure 3-1 for the baseline performance monitoring experiments conducted with May Valley water. Initial flux decline was observed, but specific water flux eventually reached steady state. After the experiments, the membranes were visually inspected and very thin layers of fouling were observed on the membranes; thus, longer-term pilot testing is recommended to further evaluate the fouling potential of the May Valley water. The specific flux was relatively constant throughout the 24 hours of bench testing. The Koch TFC-S membrane had the highest specific flux indicating that at water flux of 30 LMH, the TFC-S has the lowest operating pressure. The average operating pressures were 74 kPa (108 psi) for the TFC-S, 95 kPa (138 psi) for the NF 4040, and 125 kPa (181 psi) for the TFC-HR.

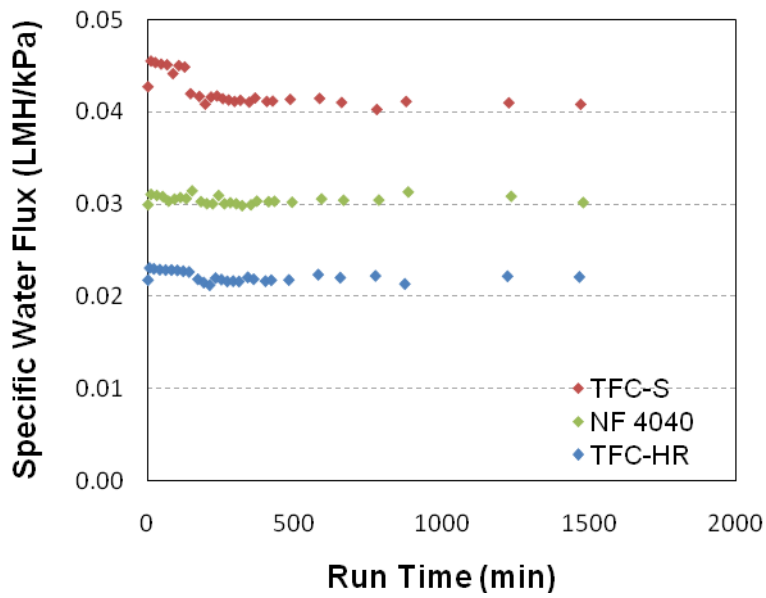

Figure 3-1: Specific water flux as a function of time during baseline performance monitoring experiments with May Valley water

The concentrations and percent rejection of major analytes during the bench-scale membrane experiments are summarized in Tables 3-1 and 3-2. For all experiments, it was assumed that removal of radium 226 and 228 is similar; therefore, samples were analyzed for radium 226 and the radium 228 concentration was calculated using the known radium 226 to radium 228 ratio in the source water.

Table 3-1.
Concentrations and rejection percentages of major analytes in bench membrane experiments for May Valley

Sample	Calcium		Magnesium		Sodium		Chloride		Sulfate	
	Conc mg/L	Rejection %	Conc mg/L	Rejection %	Conc mg/L	Rejection %	Conc mg/L	Rejection %	Conc mg/L	Rejection %
Feed	84.6		38.1		568.8		21.2		1207	
Koch TFC-HR	1.3	98	0.6	99	10.9	98	0.4	98	22.9	98
Koch TFC-S	0.4	100	0.1	100	7.6	99	0.8	96	7.7	99
Dow NF 4040	1.3	99	0.6	98	67.2	88	14.6	31	10.9	99

Table 3-2. Concentrations and rejection percentages of radionuclides in bench membrane experiments

Sample	Ra - 226		Combined Ra		Gross Alpha			Gross Beta	
	Conc pCi/L	Rejection %	MCL pCi/L	Conc pCi/L	MCL pCi/L	Conc pCi/L	Rejection %	Conc pCi/L	Rejection %
Feed	13.1		5	21.0	15	39.0		57.5	
Koch TFC-HR	0.3	98	5	0.5*	15	< 3	> 92	4.5	92
Koch TFC-S	< 0.2	> 98	5	< 0.3*	15	< 3	> 92	< 4	> 93
Dow NF 4040	0.2	98	5	0.3*	15	< 3	> 92	< 4	> 93

* Estimated combined radium concentration based on feed ratio of Ra-226:Ra-228 (1.65:1)

Calcium, magnesium, and radium rejections were greater than 98% and all the membranes reduced gross alpha and gross beta concentrations to below detection limit in all but one case (TFC-HR – gross beta). The TFC-S and TFC-HR membranes rejected greater than 96% of sodium, chloride, and sulfate. The NF 4040 had lower rejections; reducing sodium, chloride, and sulfate concentrations by 88, 31, and 99%, respectively. Feed and permeate water samples were analyzed for TOC; however, all samples were below the detection limit of 0.5 mg/L. Other metal ions and anions were sampled, but concentrations were not high enough to determine meaningful rejection (See Appendix B for complete results from these experiments).

Based on the results from these experiments, any of the membranes tested can be used in small water systems that need to reduce radionuclide concentrations in their water. For all three membranes, high rejection of radionuclides resulted in product water with gross alpha and radium concentrations below the MCL. High rejection of TDS by the three membranes resulted in product water TDS concentration below the secondary standard of 500 mg/L. However, these experiments were operated at a low recovery under conditions that do not reflect operating conditions of larger scale membrane systems. Therefore, pilot-scale experiments were conducted to more closely simulate real operating conditions. The TFC-S NF membrane was chosen for pilot testing because it had high TDS and radionuclide rejections and the lowest operating pressure of the three during bench scale testing, which would result in lower energy costs if this membrane was used in a treatment facility.

3.1.2. Pilot-Scale Membrane Testing

The dual-stage pilot-scale membrane system was deployed and tested for one day at May Valley (Figure 3-2). Feed water was supplied directly from the well (after iron filtration) into a buffer tank that feeds the membrane system. For the TFC-S, two conditions were pilot-tested: 30% water recovery (operating conditions summarized in Table 2-8) and 80% water recovery (Table 2-9). It is desirable to operate membranes at high recovery; thereby, increasing productivity and reducing the volume of the brine stream. However, at higher recoveries, feed concentration increases, and consequently solute diffusion

through RO and NF membranes also increase. Thus, the two recovery conditions tested in the current study were selected to compare radionuclide rejection at a low and a high recovery. The pilot system was adjusted to the predetermined operating conditions and when the system stabilized, the pressure was recorded and samples were taken from the permeate and feed streams.

Figure 3-2: Field testing of the pilot-scale membrane system at May Valley.

Ion concentrations and percent rejection are summarized in Tables 3-3 and 3-4 for May Valley water treated with the pilot TFC-S NF membrane at 30 and 80 percent water recoveries, respectively. Rejection of divalent ions (calcium, magnesium, sulfate) and radionuclides (radium and gross alpha) by the TFC-S membrane were high, removing greater than 94% of the ions. Sodium and chloride rejections were lower (smaller ions). As expected, solute rejection was higher during operation at 30% recovery than during operation at the 80% recovery. As recovery increases, more water diffuses through the membranes; resulting in higher feed concentrations, increased concentration polarization, and more ions diffusing through the membrane.

Table 3-3. May Valley pilot membrane testing: concentration and rejection of major analytes

Sample	Calcium		Magnesium		Sodium		Chloride		Sulfate	
	Conc mg/L	Rejection %	Conc mg/L	Rejection %	Conc mg/L	Rejection %	Conc mg/L	Rejection %	Conc mg/L	Rejection %
Feed	73.5		31.7		460.4		38.4		1,108	
30% Permeate	0.6	99	0.3	99	26.1	94	6.4	83	20.2	98
80% Permeate	1.8	97	0.9	97	80.0	82	17.3	55	68.8	94

Table 3-4. May Valley pilot membrane testing: concentration and rejection of radionuclides

Sample	Ra – 226		Combined Ra		Gross Alpha			Gross Beta
	Conc pCi/L	Rejection %	MCL pCi/L	Conc pCi/L	MCL pCi/L	Conc pCi/L	Rejection %	Conc pCi/L
Feed - May Valley	12		5	19.3	15	12.0		**
30% Permeate	0.2	99	5	0.3*	15	< 3	> 96	< 4
80% Permeate	0.3	98	5	0.5*	15	< 3	> 96	< 4

* Estimated combined radium concentration based on feed ratio of Ra-226:Ra-228 (1.65:1)
** Due to high TDS concentration in the feed water, gross beta analysis could not be performed

The rejections of ions and radionuclides by the TFC-S NF membrane at both bench and pilot-scale are summarized in Figure 3-3. In almost all cases, rejection was highest for the pilot at 30% recovery, and the lowest rejection was for the pilot at 80% recovery. Sodium and chloride, both monovalent ions, removal varied in the tests. Rejection of divalent ions (calcium, magnesium, and radium) was 97-99% rejection for all the tests.

The results for both bench- and pilot-scale membrane tests indicate both RO and NF can remove radium and gross alpha from groundwater to levels below the MCLs. In addition to reducing radium and gross alpha concentrations to below the MCL, the TFC-S NF membrane rejected most of the dissolved solids from May Valley's high TDS water. Results also indicate that NF is likely a better option for treatment of May Valley water than RO due to lower operating pressures (and associated energy costs) compared RO membranes. Additionally, RO membranes remove more salts than NF membranes, typically requiring remineralization of permeate to make the water less corrosive and drinkable.

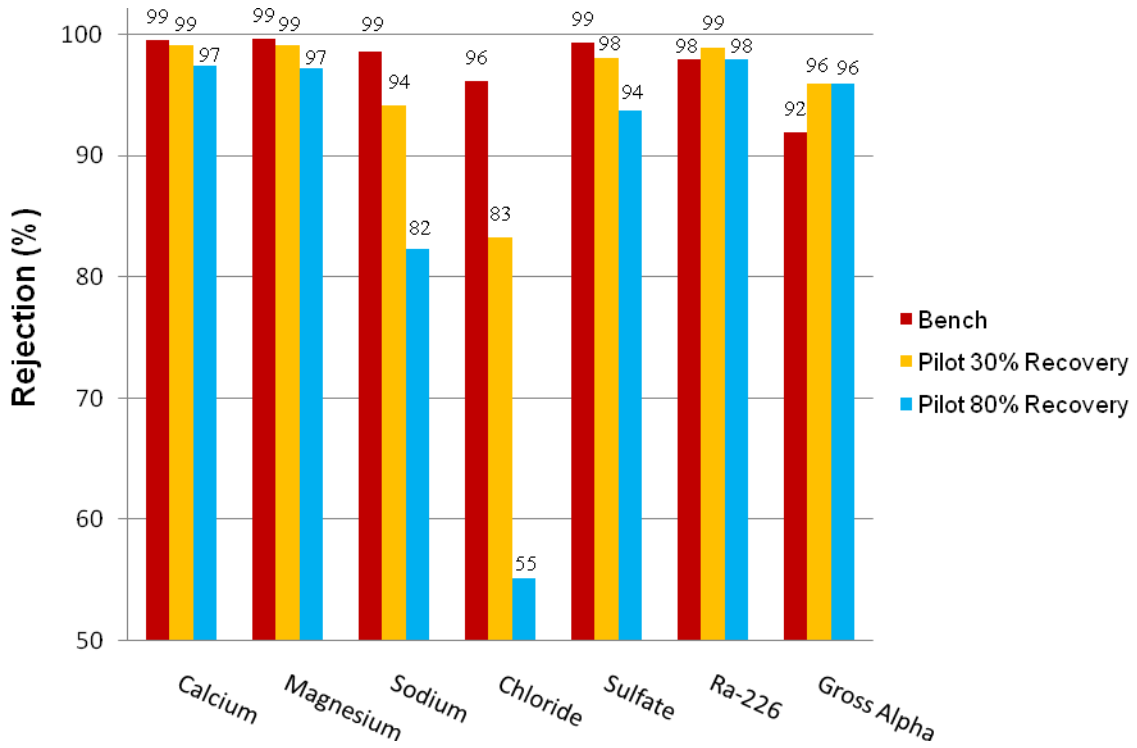

Figure 3-3: Comparison between bench- and pilot-scale test results for NF rejection of major ions and radionuclides from May Valley groundwater using the Koch TFC-S membrane.

3.1.3. Bench-Scale Testing of Hydrous Manganese Oxide (HMO)

Four 2-L jars were filled with May Valley water, which had been passed through an iron filter before sampling. Two jars were dosed with 0.5 mg/L HMO and two jars were dosed with 1.0 mg/L. Operating conditions for all four jars were identical as described in Section 2.2.1. Following the experiment, samples from each jar were analyzed for radium 226 and gross alpha. Results from the HMO treatment of May Valley water are summarized in Table 3-5.

Results indicate radium 226 and combined radium were both reduced to below the MCL with higher removal observed at 1.0 mg/L than 0.5 mg/L. The HMO dose was limited to 1 mg/L because above this concentration the water developed color that would likely be unacceptable from an aesthetic water quality standpoint. Metal ions, uranium, and TOC concentrations were also measured and results are summarized in Appendix B. Almost no change in concentrations of these analytes was observed, however, past research [1] has suggested that slight variations in calcium and magnesium may be due to competitive adsorption in the HMO process.

Table 3-5. Concentrations and percent removal of radionuclides for HMO treatment of May Valley groundwater.

HMO Dose	Ra-226		Combined Ra		Gross Alpha		
mg/L	Conc pCi/L	Removal %	MCL pCi/L	Conc pCi/L	MCL pCi/L	Conc pCi/L	Removal %
Feed	11.4		5	18.3	15	55	
0.5	1.4	88	5	2.2*	15	24	56
0.5	2.1	82	5	3.4*	15	10	82
1.0	0.4	96	5	0.6*	15	6	89
1.0	0.4	96	5	0.6*	15	5	91

* Estimated combined radium concentration based on feed ratio of Ra-226:Ra-228 (1.65:1)

Based on the results from these experiments, HMO has the potential to be an acceptable treatment technology for removal of positively charged radionuclides; however, the variation in concentrations, especially at lower doses of HMO, indicate that further research is needed before recommending HMO as a treatment alternative for water contaminated with radionuclides. Furthermore, the current investigation was conducted with a small batch of groundwater under conditions that allowed one hour of contact time prior to filtration (adopted from previous study by Valentine (1990)); thus, a pilot study is recommended to verify these results on a larger scale and under standard operating conditions. A pilot study would entail inline dosing of an HMO solution followed by 10-30 minutes of contact time and filtration. In the batch study, the filtration was achieved using a 0.45 µm polymeric filter; in a treatment facility (and therefore in a pilot test) the filter would most likely be a granular filter, which may affect the removal efficiency. Furthermore, it would be necessary to investigate the backwashing efficiency of granular filters and evaluate whether radium is permanently adsorbed to the filter media. Accumulation of radium on the filter media might result in additional cost for handling and disposing spent filter media.

3.1.4. May Valley Bench-Scale Ion Exchange

Ion exchange testing was conducted with Purolite C-100E cation resin (Purolite, Bala Cynwyd, PA) with two columns operated in parallel. During operation, hardness concentration in the feed and product water was monitored as a surrogate for radium [2, 3]. Baseline performance experiments were conducted to identify hardness breakthrough times, which were then used to identify radionuclide sample collection times during subsequent performance experiments. Hardness breakthrough curves for cycles 1 through 5 of the performance experiments are illustrated in Figure 3-4.

Hardness breakthrough curves for the baseline performance experiments were nearly identical to the breakthrough curves of the performance experiments. For the five cycles, the columns were operated to 80% hardness breakthrough and then regenerated. Samples for radium and gross alpha analysis were collected in each cycle after 50 BVs were processed through the columns. An arbitrary 40% hardness breakthrough value was

chosen for comparison. During the initial loading (cycle 1) of the resin, 40% breakthrough of hardness occurred after processing approximately 250 bed volumes (BV). Following breakthrough, each column was regenerated and operated for four successive loading cycles. In cycle 2, a decline in resin exchange capacity for hardness was observed. For cycles 2 through 5, 40% breakthrough of hardness occurred after processing approximately 195 BVs. Cycles 2 through 5 showed almost no change in resin hardness capacity. Overall, there was a 22% reduction in BVs processed before hardness breakthrough in cycles 2 through 5 compared to cycle 1.

Hardness, radium, and gross alpha concentrations for cycles 1 through 6 are summarized in Table 3-6. From the results it can be seen that there is a gradual increase in the effluent concentrations with each cycle. This indicates that the system has not achieved steady state conditions. Results from past research [2] indicate that the concentration of column effluents may vary during the first several cycles, but will stabilize after 5 to10 cycles.

Results from the current experiments indicate that cationic exchange resins can remove both hardness and positively charged radionuclides from contaminated groundwater. Concentrations of both radium and gross alpha in the product water were below the respective MCLs.

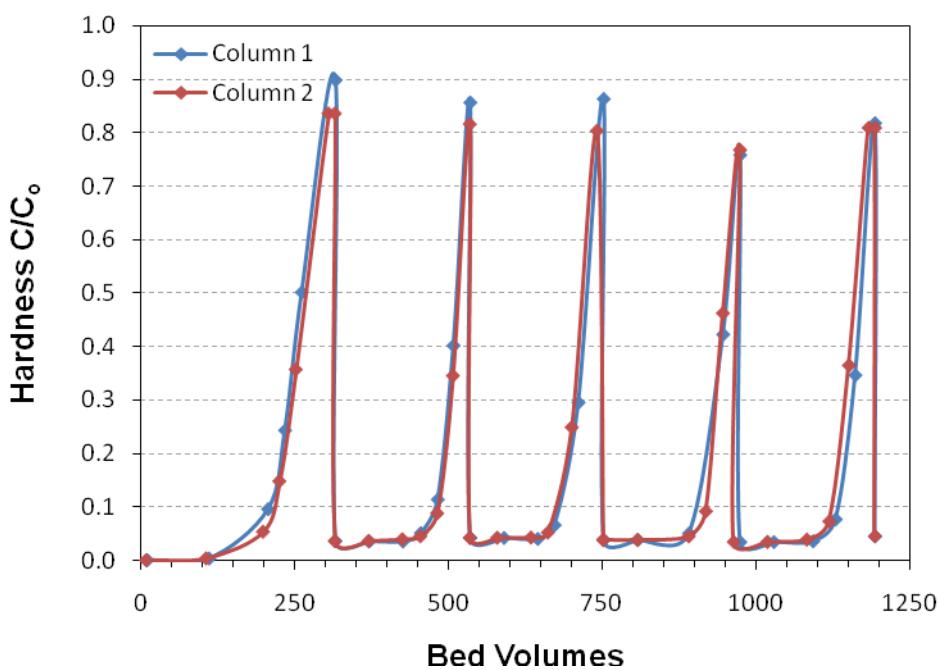

Figure 3-4: Hardness relative concentration as a function of BVs for cycles 1 through 5 during ion exchange testing of May Valley groundwater. Two columns were operated in parallel for the duration of the five cycles. Samples for hardness analysis were collected every 30-50 BVs.

Table 3-6. Feed and column effluent concentrations of hardness, radium, and gross alpha during bench-scale ion exchange experiments with May Valley groundwater. Samples were taken 2 hours after the beginning of each cycle. MCL for radium is 5 pCi/L and for gross alpha the MCL is 15 pCi/L.

Cycle	Hardness (mg/L as $CaCO_3$)		Ra-226 (pCi/L)		Gross Alpha (pCi/L)	
	Feed	Effluent	Feed	Effluent	Feed	Effluent
1	357.2	0.5	12.4	< 0.3	62	3
2		13.2		0.1		< 3
3		15.3		0.1		< 3
4		14.0		0.3		3
5		12.6		0.3		4
6		16.4		0.5		3

Another problem that can affect the operation of ion exchange for this specific groundwater is the potential for iron precipitation and fouling of the resin; raw groundwater in May Valley has elevated concentrations of iron (1.0-1.5 mg/L) that can foul and clog the resin bed. The water in May Valley is treated with iron filters prior to distribution and the iron concentration is reduced to approximately 0.05 mg/L. For the ion exchange experiments performed with May Valley groundwater, post iron filter water was used. During the course of the ion exchange experiments, no significant head loss was observed.

Following hardness breakthrough of the 6th cycle, a resin sample was taken for gamma spectrometry analysis to determine the concentration of radionuclides that had permanently adsorbed to the resin. Following super-regeneration with a 15% NaCl solution at a loading rate of 15 lbs NaCl per ft^3 resin volume (as opposed to 8% NaCl solution used in standard regeneration), another resin sample was taken for gamma spectrometry analysis. Results from the gamma spectrometry analysis are summarized in Table 3-7. These results indicate only a small fraction of radionuclides were removed by regenerating the resin. However, samples collected in these tests indicate radium was removed after the resin was regenerated. Prior research [2] indicates that it takes 5-10 regeneration cycles before the amount of radium adsorbed to the resin during treatment equals the amount eluted during regeneration because the resin capacity for radium is originally very high.

Table 3-7. Radionuclide adsorption on ion exchange resin from testing with May Valley water. The Pre-Regeneration resin sample was taken following hardness breakthrough of the 6th cycle and the Post-Regeneration sample was taken following regeneration after the 6th cycle. Results may be slightly biased due to the small sample size. Detection limits for the two samples were slightly different due to small differences in resin mass sampled.

Test	Pre-Regeneration (pCi/g)	Post-Regeneration (pCi/g)
Potassium-240	< 4.0	< 3.3
Lead-210	< 5.3	< 4.6
Lead-212	2.3	2.4
Lead-214	17	12.6
Radium-226	45	43
Radium-228	30	26
Uranium-235	< 2	< 1.7
Uranium-238	23	27

3.2. Blue Mountain Bench and Pilot Testing

3.2.1. Bench-Scale HMO

Three different HMO conditions were tested for Blue Mountain groundwater: 0.5 mg/L HMO, 1.0 mg/L HMO, and a mixture of 0.5 mg/L HMO and 0.5 mg/L $KMnO_4$ that was dosed 30 minutes prior to HMO dosing. Jars were run in duplicate for each dosing condition. The concentrations of radium, gross alpha, and gross beta before and after HMO treatment of Blue Mountain water are summarized in Table 3-8. Results from the HMO testing show some variability. The radium 226 concentration varied by a factor of approximately 2 for each of the three dosing conditions. It is possible that this variability is a result of errors inherent with the jar testing procedure. The HMO feed solution contains precipitated HMO particles that are constantly being mixed. During dosing of the HMO solution, it is possible that some aliquots of the stock solution will contain higher concentrations of solid HMO than others. Higher radium removal is expected for dose having more solid HMO. The variability could also be the result of sampling or laboratory error but this could not be confirmed. Comparing these results to those from May Valley and Red Hill Forest, the data indicating higher removal are more consistent.

Table 3-8. Concentrations and percent removal of radionuclides for HMO treatment of Blue Mountain groundwater.

HMO Dose	$KMnO_4$ Dose	Ra-226		Combined Ra		Gross Alpha			Gross Beta	
mg/L	mg/L	Conc pCi/L	Removal %	MCL pCi/L	Conc pCi/L	MCL pCi/L	Conc pCi/L	Removal %	Conc pCi/L	Removal %
Feed		35		5	35.5	15	110		29.0	
0.5	0.5	16.7	52	5	17.2*	15	69	37	18	38
0.5	0.5	6.5	81	5	6.7*	15	n/a**	n/a**	n/a**	n/a**
0.5	0	18.1	48	5	18.6*	15	84	24	16	45
0.5	0	8.5	75	5	8.7*	15	n/a**	n/a**	n/a**	n/a**
1.0	0	7.0	80	5	7.2*	15	45	59	12	59
1.0	0	3.9	89	5	4.0*	15	n/a**	n/a**	n/a**	n/a**

* Estimated combined radium concentration based on feed ratio of Ra-226:Ra-228 (70:1)
** only one sample was analyzed

Radium was reduced to below the MCL in only one sample collected in this test and gross alpha concentrations in the treated water were always above the MCL. These results suggest that HMO may not be able to reduce radium to below the MCL in waters with higher radium concentrations, such as in Blue Mountain compared to May Valley source waters).

Results also demonstrate that gross alpha removal with HMO treatment of Blue Mountain water is much lower than the removal seen in May Valley and Red Hill Forest waters. Gross alpha removal was in the 70-90% range for May Valley and Red Hill Forest, compared to 20-60% removal for Blue Mountain; however, Blue Mountain water contains radium and uranium, both of which emit alpha radiation. Metal ions, anions, and TOC concentrations were also measured and are presented in Appendix B. Almost no change in concentrations of these analytes was observed; however, past research has suggested that slight variations in calcium and magnesium may be due to competitive adsorption in the HMO process [1].

Results from the HMO testing also indicate that oxidation of the water with $KMnO_4$ prior to HMO dosing does not substantially increase removal of radionuclides. Results from previous research [2] suggest that dissolved iron could compete with radium for adsorption sites on the HMO particles. The Blue Mountain results indicate that oxidation to precipitate iron out of solution prior to HMO dosing has limited effect on radium removal.

3.2.2. Bench-Scale Ion Exchange

Ion exchange testing was performed with Purolite C-100E cation resin using two columns operated in parallel. Hardness breakthrough curves for cycles 1 through 5 of the performance experiments are illustrated in Figure 3-5 (additional performance results are

included in Appendix C). Hardness breakthrough curves for the baseline performance experiments were nearly identical to the breakthrough curves of the performance experiments. For the five cycles, an arbitrary 40% hardness breakthrough value was chosen for comparison. During the first cycle, 40% breakthrough of hardness occurred after processing approximately 585 BVs. Following breakthrough, each column was regenerated and operated for four successive loading cycles. In cycle 2, a decline in resin exchange capacity for hardness was observed. During cycles 2 through 5 there was no change in hardness capacity; for these cycles, 40% breakthrough of hardness occurred after processing approximately 365 BVs. Overall, approximately 37% decline in the number of BVs processed was observed before hardness breakthrough in cycles 2 through 5 compared to cycle 1.

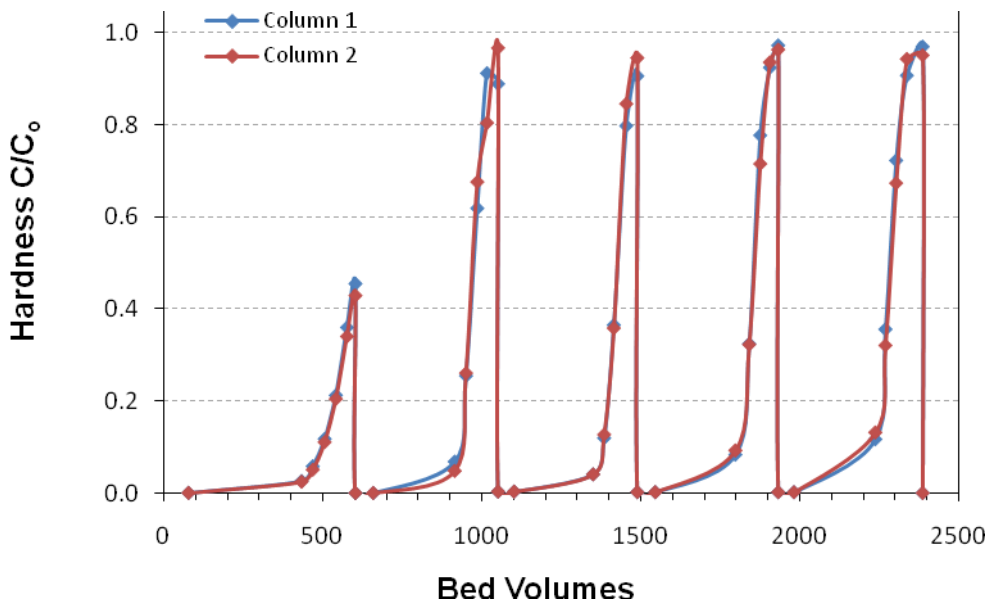

Figure 3-5: Hardness relative concentration as a function of BVs for cycles 1 through 5 during ion exchange testing of Blue Mountain groundwater. Two columns were operated in parallel for the duration of the five cycles. Samples for hardness analysis were collected every 30-50 BVs.

Hardness, radium, gross alpha, and gross beta concentrations for cycles 1 through 6 are summarized in Table 3-9. Results demonstrate that cation exchange is effective for removing radium from Blue Mountain water; reducing the radium concentration in the effluent to below the detection limit in all cases. Results from gross alpha analyses in this experiment support the results from the Blue Mountain HMO experiment; high radium removal from Blue Mountain water does not correlate with high gross alpha removal.

Table 3-9. Feed and column effluent concentrations of hardness, radium, and gross alpha during ion exchange experiments with Blue Mountain groundwater feed. Samples were taken 2 hours after the beginning of each cycle. MCL for radium is 5 pCi/L and for gross alpha the MCL is 15 pCi/L.

Cycle	Hardness (mg/L as $CaCO_3$)		Ra-226 (pCi/L)		Gross Alpha (pCi/L)		Gross Beta (pCi/L)	
	Feed	Effluent	Feed	Effluent	Feed	Effluent	Feed	Effluent
1	203	0.1	3.1	< 0.4	41	30	25	16
2		0.5		< 0.2		37		11
3		0.6		< 0.2		42		12
4		0.4		< 0.1		33		14
5		0.4		< 0.1		38		10
6		0.4		< 0.1		36		12

Following hardness breakthrough of the 6th cycle, a resin sample was taken for gamma spectrometry analysis to determine the concentration of radionuclides, which had adsorbed to the resin during the six cycles. Following super-regeneration, another resin sample was taken for gamma spectrometry analysis to determine the concentration of radionuclides that permanently adsorbed to the resin. Results from the gamma spectrometry analysis are summarized in Table 3-10. When comparing the radionuclide concentrations on the resin before and after super-regeneration, it can be seen that the change is insignificant. Similar to the May Valley ion exchange experiment, due to the short length of this ion exchange experiment, radionuclide concentrations on the resin have most likely not reached high enough to show quantifiable change after super regeneration.

Table 3-10. Radionuclide adsorption on the ion exchange resin from before and after regeneration. (Results may be biased due to the small sample size. Detection limits for the two samples were slightly different due to small differences in resin mass sampled.)

Test	Pre-Regeneration (pCi/g)	Post-Regeneration (pCi/g)
Lead-210	< 4.5	< 3.3
Lead-212	< 0.3	< 2.9
Lead-214	4.2	3.6
Radium-226	20	23
Radium-228	4	2.4
Uranium-235	< 1.6	1.1
Uranium-238	< 3.2	2.2

3.3. Red Hill Forest Bench and Pilot Testing Results

3.3.1. Bench-Scale HMO

Two HMO doses (0.5 mg/L and 1.0 mg/L) were tested in duplicate for Red Hill Forest groundwater in jars. The jar testing procedure was identical for all of the jars, as

described previously. Following the experiment, samples were analyzed for radium 226, gross alpha, and gross beta. The results from this test are summarized in Table 3-11. Results indicate there is a relationship between HMO dose and percent removal of radionuclides, where 70-80% radium removal was observed at a dose of 0.5 mg/L and 80-90% was observed at an HMO dose of 1.0 mg/L. Metal ions, anion, and TOC concentrations were also measured, but no change was observed between feed and product samples.

Results from this experiment and the May Valley HMO are similar; an HMO dose of 0.5 mg/L resulted in approximately 75-85% removal of gross alpha and radium and for a dose of 1.0 mg/L, the removal increased to 85-95%. At these levels of radium removal, compliance will depend on the raw water radium and gross alpha concentrations. Treated May Valley water samples contained radium and gross alpha below the MCL; however, Red Hill Forest waters treated similarly with HMO did not. The raw water radium concentration was 12.4 pCi/L in May Valley water compared to 31 pCi/L in Red Hill Forest water.

Table 3-11. Concentrations and percent removal of radionuclides for HMO treatment of Red Hill Forest groundwater.

HMO Dose	Ra-226		Combined Ra		Gross Alpha			Gross Beta	
mg/L	Conc. pCi/L	Removal %	MCL pCi/L	Conc. pCi/L	MCL pCi/L	Conc. pCi/L	Removal %	Conc. pCi/L	Removal %
Feed	31		5	33.3	15	97		49.0	
0.5	7.7	75	5	8.3	15	21	78	10	80
0.5	7.1	77	5	7.6	15	20	79	14	71
1.0	4.4	86	5	4.7	15	15	85	10	80
1.0	3	90	5	3.2	15	8	92	7	86

* Estimated combined radium concentration based on feed ratio of Ra-226:Ra-228 (13.5:1)

3.3.2. Bench-Scale Ion Exchange

Ion exchange testing was performed with Purolite C-100E cation resin with two columns operated in parallel. While in operation, hardness concentration in the column effluents was monitored as a surrogate for radium concentration. Hardness breakthrough curves for cycles 1 through 5 of the performance experiments are illustrated in Figure 3-6. Hardness breakthrough curves for the baseline performance experiments were nearly identical to the breakthrough curves of the performance experiments. For the five cycles, an arbitrary 40% hardness breakthrough value was chosen for comparison. During the first loading cycle, 40% breakthrough of hardness occurred after processing approximately 505 BVs.

Following breakthrough, each column was regenerated and operated for four successive loading cycles. In cycle 2, a decline in resin exchange capacity for hardness was

observed. During cycles 2 through 5 there was no change in hardness capacity; for these cycles, 40% breakthrough of hardness occurred after processing approximately 365 BVs. Overall, approximately 28% decline in the number of BVs processed was observed before hardness breakthrough in cycles 2 through 5 compared to cycle 1.

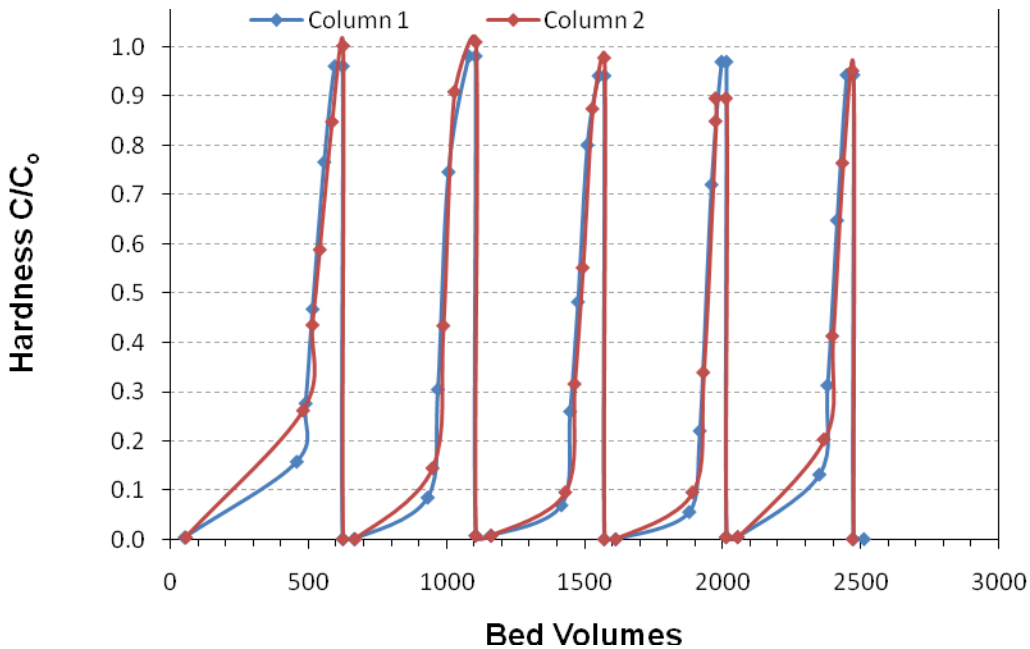

Figure 3-6: Normalized hardness concentration as a function of BVs for cycles 1 through 5 during ion exchange testing of Red Hill Forest groundwater. Samples for hardness analysis were collected every 30-50 BVs.

Hardness, radium, gross alpha, and gross beta concentrations for cycles 1 through 6 are summarized in Table 3-12.

Table 3-12. Feed and column effluent concentrations of hardness, radium, gross alpha, and gross beta during ion exchange experiments with Red Hill Forest groundwater feed. Samples were taken 2 hours after the beginning of each cycle. MCL for radium is 5 pCi/L and for gross alpha the MCL is 15 pCi/L.

Cycle	Hardness (mg/L as $CaCO_3$)		Ra-226 (pCi/L)		Gross Alpha (pCi/L)		Gross Beta (pCi/L)	
	Feed	Effluent	Feed	Effluent	Feed	Effluent	Feed	Effluent
1	204	1.0	31	< 0.3	97	< 3	49	< 4
2		0.4		< 0.2		< 3		< 4
3		1.8		< 0.2		3		< 4
4		0.3		< 0.2		< 3		< 4
5		1.2		< 0.2		3		< 4
6		0.4		< 0.2		3		< 4

Results demonstrate that cation exchange is effective for radium, gross alpha, and gross beta removal from the Red Hill Forest water, lowering most radionuclide concentrations to levels below detection limits. The results from this experiment in conjunction with results from the May Valley cation exchange testing indicate that a cation exchange process treating source waters with elevated radium and gross alpha (but not uranium) can successfully lower the concentrations of radium and gross alpha below the MCLs. Radium was also removed from the Blue Mountain water by cation exchange; however, gross alpha was not reduced to below the MCL, indicating cation exchange alone may need be able to produce water in compliance with the MCL if uranium is also present in the water.

Following hardness breakthrough of the 6th cycle, a resin sample was taken for gamma spectrometry analysis to determine the concentration of radionuclides that adsorbed to the resin during the six cycles. Following super-regeneration, another resin sample was taken for gamma spectrometry analysis to determine the irreversible adsorption of radionuclides to the resin. Results from the gamma spectrometry analysis are summarized in Table 3-13. As a result of super-regeneration, approximately 16% of both Ra-226 and Ra-228 were removed from the resin. The other radionuclides that were tested showed either no change or an increase of concentration on the resin. Increase in uranium concentration on the resin after regeneration is most likely an analytical outlier, resulted from low volume of resin available for analysis.

The combined results on the three waters demonstrate minimal radium removal by regeneration after the 6th run to hardness breakthrough. This could be due to a couple factors:

- Tests have shown cation exchange resin has a very high initial capacity for radium; resulting in a portion of the resin sites containing radium that is not removed by regeneration. The remaining sites then adsorb and radium and are regenerated when the resin reaches steady state.
- The resin samples collected were very small, resulting in a larger margin of error when the results are compared.

Table 3-13. Radionuclide adsorption on ion exchange resin from testing with Red Hill Forest water. The Pre-Regeneration resin sample was taken following hardness breakthrough of the 6th cycle and the Post-Regeneration sample was taken following regeneration after the 6th cycle. Results may be biased due to the small sample size.

Test	Pre-Regeneration (pCi/g)	Post-Regeneration (pCi/g)
Lead-212	0.6	< 0.3
Lead-214	26.3	34.6
Radium-226	168	139
Radium-228	14.4	12.1
Uranium-235	2.6	2.9
Uranium-238	11.3	19.6

3.4. Sterling Bench and Pilot Testing Results

3.4.1. Pilot-Scale Membrane

Two NF membranes (Koch TFC-S and Dow NF 4040) were pilot tested for their efficiency to remove radionuclides from groundwater from the Town of Merino. Unlike the membrane testing for May Valley water, no bench-scale testing was performed with water from Town of Merino prior to pilot testing. Therefore, two NF membranes were tested on the pilot level for comparison. Source water from Merino was used for these experiments because the well in Sterling was off-line on the day of testing. Merino source water is from the same aquifer as the targeted Sterling well and was reflective of the Sterling well water quality (Table 2-6). These two membranes were chosen based on their performance during bench scale testing of May Valley water; during the bench-scale experiments, both NF membranes rejected more than 98% of calcium and magnesium while operating at a lower pressure than the reverse osmosis membrane.

For each membrane, two operating conditions were tested: 30% recovery (Table 2-10) and 80% recovery (Table 2-11). The pilot system was adjusted to the predetermined operating conditions and when the system stabilized, the pressure was recorded and samples were taken from the permeate and feed streams. In addition to taking samples from the combined permeate, several samples were taken from the permeate of the first membrane element only (permeate 1).

Concentrations and rejections of major analytes measured during the pilot-scale membrane experiments are summarized in Table 3-14. Removal of divalent ions (i.e., calcium, magnesium, and sulfate) was greater than 94% for all conditions tested. As expected, rejection of monovalent ions (chloride and sodium) was lower. Feed TOC concentration was 3.08 mg/L, and under all operating conditions tested, TOC concentration in the permeate was below the detection limit of 0.5 mg/L. Other metal ions and anions were measured, but did not have elevated feed concentrations. As expected, ion rejections were higher when operating at 30% water recovery and lower

when operating at 80% water recovery. Also from Table 3-14, it can be seen that the two membranes have very similar rejection of most constituents; but monovalent ions were rejected slightly better with the TFC-S membrane. Other metal ions and anions were also measured and are summarized in Appendix B.

Table 3-14. Concentrations and rejections of major analytes during pilot-scale membrane experiments with groundwater at the Town of Merino. 'Comb Perm' samples refers to samples collected from the combined permeate stream from the first and second membrane elements.

Sample	Calcium		Magnesium		Sodium		Chloride		Sulfate	
	Conc. mg/L	Rej. %	Conc. mg/L	Rej. %	Conc. mg/L	Rej. %	Conc. mg/L	Rej. %	Conc. mg/L	Rej. %
Feed - Merino	162		48.9		155		102		654	
TFC-S 30% - Comb Perm	2.8	98	1.0	98	18.8	88	14.4	86	15.2	98
TFC-S 80% - Comb Perm	6.5	96	2.3	95	38.4	75	32.4	68	31.5	95
NF 4040 30% - Comb Perm	3.7	98	0.9	98	28.9	81	29.2	71	3.5	99
NF 4040 80% - Comb Perm	10.3	94	1.9	96	59.1	62	65.4	36	3.1	99

Concentrations and rejection of radionuclides measured during pilot-scale membrane experiments are summarized in Table 3-15. Both membranes rejected uranium and gross alpha well, reducing radionuclide concentrations to below the detection limit in most cases.

Table 3-15. Concentrations and rejections of radionuclides during pilot-scale membrane experiments with groundwater at the Town of Merino. 'Comb Perm' samples refers to samples collected from the combined permeate stream from the first and second membrane elements.

Sample	Uranium			Gross Alpha		
	MCL ug/L	Conc ug/L	Rejection %	MCL ug/L	Conc pCi/L	Rejection %
Feed - Merino	30	44		15	14	
TFC-S 30% - Comb Perm	30	< 1	> 98	15	< 4	> 71
TFC-S 80% - Comb Perm	30	2	95	15	< 3	> 79
NF 4040 30% - Comb Perm	30	< 1	> 98	15	< 3	> 79
NF 4040 80% - Comb Perm	30	< 1	> 98	15	< 3	> 79

NF membranes removed both radionuclides and TDS from the source water. The results from these experiments suggest that RO is a good separation process, but most often it provides treatment level that is more than actually necessary, likely at higher energy cost,

and lower water recovery. NF on the other hand, provides very adequate treatment and removal of both uranium and gross alpha and should be considered by utilities that have elevated uranium and gross alpha concentrations.

3.4.2. Membrane Residuals

One challenge associated with membrane treatment is residual management; if membranes were employed at a larger-scale, a water utility will have to handle the concentrated brine generated during operation; at high water recoveries, the brine will have relatively high concentrations of TDS and radionuclides. A membrane system design software (ROPRO 7.0, KMS, Wilmington, MA) was used to model scaled-up membrane systems and to predict expected permeate and reject water quality and quantity. Description and modeling results are discussed in Section 3.7.

3.4.3. Enhanced Coagulation

Enhanced coagulation testing with Sterling water was performed under the following testing conditions:

- Two coagulants, alum and ferric sulfate
- Three different coagulant doses (5, 10, 20 mg/L Alum and 5, 10, 20 mg/L ferric sulfate) concentrations
- Two pH conditions, 7.5 (ambient) and 6.0 (adjusted)
- Rapid Mix (200 RPM): 2 minutes, Slow Mix (40 RPM): 30 minutes, Sedimentation: 40 minutes

Uranium removal as a function of coagulant dose is illustrated in Figure 3-7. For both pH conditions alum is shown to have better removal of uranium. Alum coagulation of natural pH water consistently achieved high removal ranging from 60 to 90%, with a peak removal at a dose of 10 mg/L. Alum in the pH adjusted water also showed good removal of uranium, but its peak removal was at a dose of 20 mg/L. Because of the upward trend of uranium removal by alum in the pH-adjusted water, further optimization may be achieved at higher doses. The ferric coagulation of both waters showed low or no removal of uranium, and none of the product waters from the ferric testing were below the MCL of 30 μg/L.

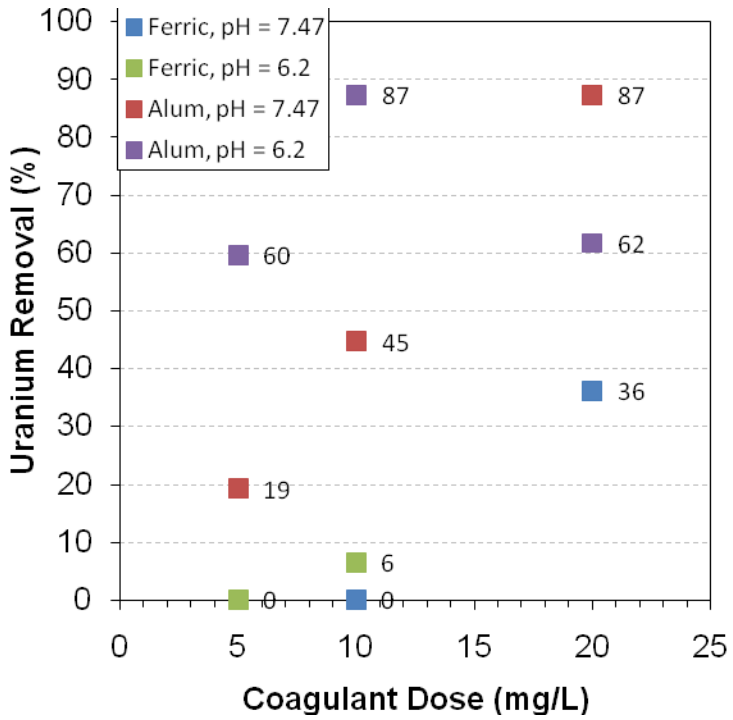

Figure 3-7: Uranium removal as a function of coagulant dose. Alum and ferric sulfate doses of 5, 10, and 20 mg/L were used for two pH conditions. The feed uranium concentration was 47 µg/L.

Table 3-16. Concentration and removal of uranium and gross alpha during enhanced coagulation testing on Sterling water.

Coagulant	Dose mg/L	pH	Uranium		Gross Alpha	
			Conc (µg/L)	Removal (%)	Conc (pCi/L)	Removal (%)
Feed			47		27	
Ferric	5	7.5	51	0	33	0
Ferric	10	7.5	47	0	26	4
Ferric	20	7.5	30	36	23	15
Alum	5	7.5	38	19	32	0
Alum	10	7.5	26	45	20	26
Alum	20	7.5	6	87	5	82
Ferric	5	6.2	47	0	29	0
Ferric	10	6.2	44	6	28	0
Alum	5	6.2	19	60	14	48
Alum	10	6.2	6	87	5	82
Alum	20	6.2	18	62	12	56

Gross alpha removal obtained during the enhanced coagulation testing of Sterling groundwater is illustrated in Figure 3-8. The results for uranium and gross alpha for the coagulation testing show similar trends. In both cases, alum performs better than ferric

sulfate and the optimal removal of uranium and gross alpha corresponds to a dose of 10 mg/L for the natural pH water and 20 mg/L for the adjusted pH.

Similar trends were observed during testing with ferric sulfate coagulation; very low or no removal of gross alpha was achieved, and none of the product waters were below the MCL of 15 pCi/L for gross alpha. Though very inefficient, additional testing is recommended if ferric sulfate is a preferred coagulant for a specific water utility.

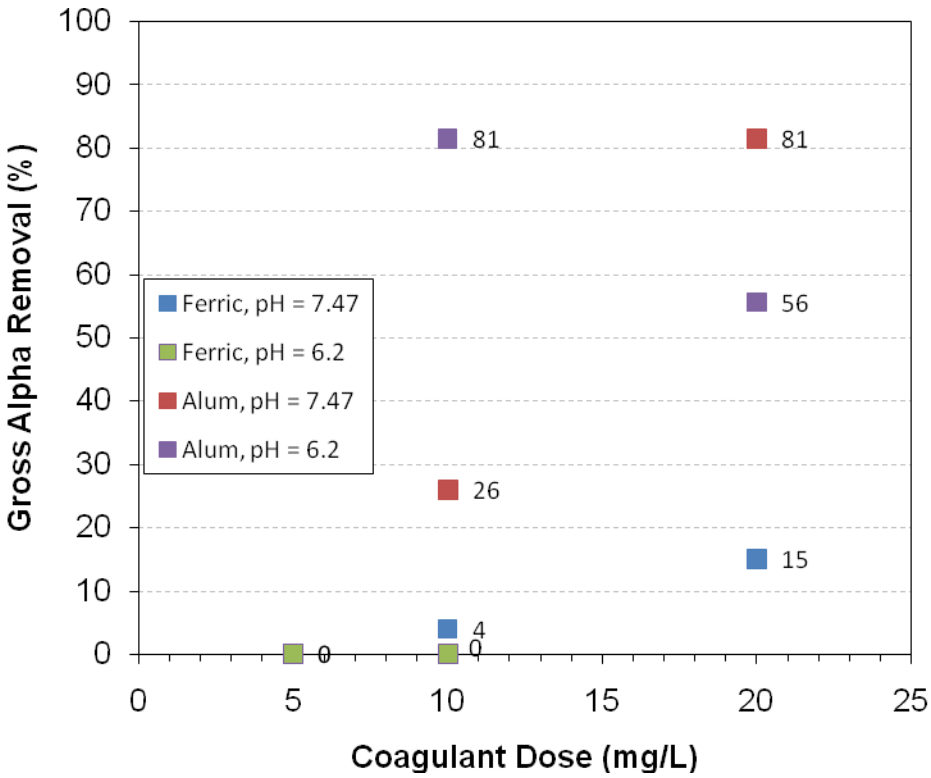

Figure 3-8: Gross alpha removal as a function of coagulant dose. Alum and ferric sulfate doses of 5, 10, and 20 mg/L were used for two pH conditions. The feed gross alpha concentration was 27 pCi/L.

Dissolved organic carbon (DOC) removal as a function of coagulant dose is illustrated in Figure 3-9. The highest DOC removal was achieved at a pH of 6.2 with a ferric sulfate dose of 20 mg/L, reducing the DOC concentration from 3.07 mg/L to 2.43 mg/L. Overall, the highest removal of DOC was achieved at a dose of 20 mg/L for all conditions tested.

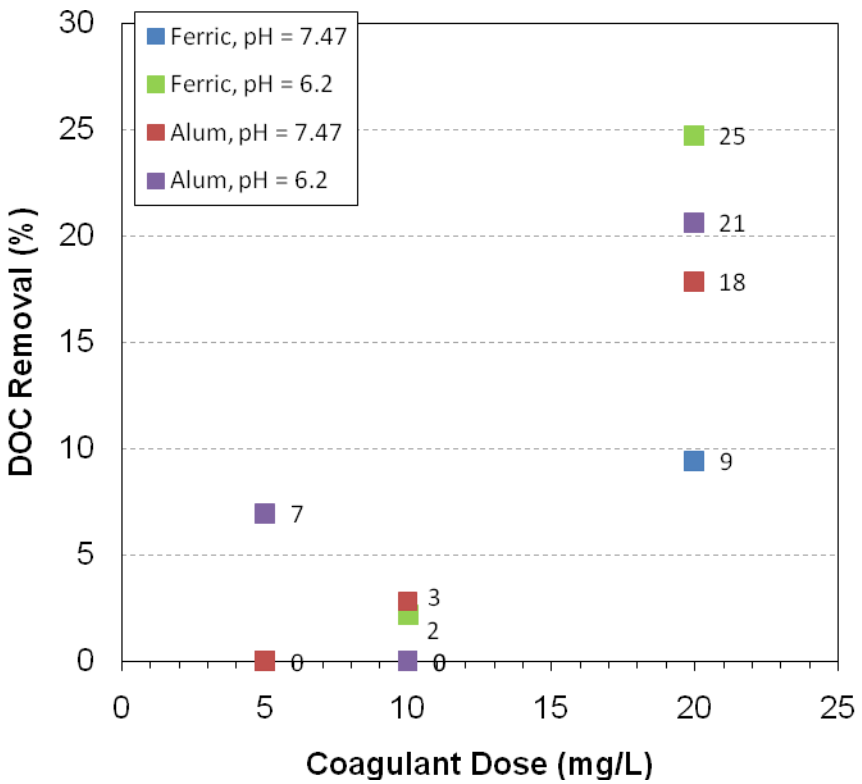

Figure 3-9: TOC removal as a function of coagulant dose. Alum and ferric sulfate doses of 5, 10, and 20 mg/L were used for two pH conditions. The feed TOC concentration was 3.07 mg/L.

The results from the coagulation experiments demonstrate that coagulation followed by filtration are a viable treatment alternative for Sterling water, Because there are numerous variables associated with coagulation optimization, if this technology is being considered for a system, site specific testing should be conducted. Specifically, alum doses of 5, 10, and 20 mg/L at adjusted pH of 6.2 or alum dose of 20 mg/L at natural pH can help Sterling comply with both uranium and gross alpha MCLs. For both pH conditions of alum dosing, no firm dose optimization was achieved. Based on results in Figures 3-7 and 3-8 for alum dosing at a pH of 6.2, the greatest removal of gross alpha and uranium is observed at the dose of 10 mg/L. For the alum dosing at natural pH (7.5), the greatest removal of uranium and gross alpha is at a dose of 20 mg/L. The upward trend, however, suggests that higher doses of alum at natural pH may achieve even higher removals.

3.5. Ion Exchange Modeling

The computer ion exchange modeling program CADIX 6.0 (Dow, Midland, MI) was used to simulate an ion exchange system for a small water utility with a product water flow of 75 gpm. Using May Valley water quality data, CADIX was used to evaluate chemical requirements as well as the volume of residuals subject to disposal.

Design considerations used in the model simulation are summarized in Table 3-17. A product flow rate of 10 gpm was used to simulate ion exchange in a small water system. The regenerant dose used in the simulation was 15 lbs NaCl per ft^3 resin, the same dose that was used in the bench-scale ion exchange testing. Sizing of the ion exchange system and the volume of resin required were based on the assumption that a small system would regenerate the resin twice a week.

Table 3-17. Design considerations used in ion exchange modeling.

Parameter	Units	Values
Product Flow Rate	gpm	10
Time between regeneration	days	3.5
Regenerant Dose	lbs/ft^3	15
Regenerant Chemical	% wt. NaCl	10

Results from the modeling simulation are summarized in Table 3-18.. The required resin volume to operate the ion exchange system given the previous stated operating condition is approximately 38 ft^3.

Table 3-18. Simulation results that show volume requirements for resin and vessel

Parameter	Units	Values
Resin Volume	ft^3	37.7
Volume of water treated between regenerations	gal	51,783 (183 BVs)
Vessel Volume	ft^3	65
Vessel Diameter	ft	3.1
Vessel Height	ft	8.6

The volume of water treated by cation exchange in May Valley bench-scale tests between regenerations was 195 BVs, similar to the estimate of bed volumes treated by CADIX. A major consideration for a system that is implementing an ion exchange system is the residual waste generated from the process. The volume of water required for the regeneration processes are summarized in Table 3-19. The total water requirement for one regeneration cycle is approximately 2,800 gallons or 10 bed volumes). Therefore, for this system, which produces 51,783 gal of drinking water, the 2,836 gal that would be wasted in the ion exchange system represents 5.1% of the total water. These model outputs (i.e., bed volumes treated to breakthrough and bed volumes of waste generated)

are similar to those modeled in the CO-RADS system reports developed as part of CO-RADS by Malcolm Pirnie.

Table 3-19. Water requirements for the regeneration processes in the ion exchange process.

Regeneration Process	Volume (gal)
Backwash	735
NaCl Injection 10%	735
Displacement	547
Rinsing	819
Total	2,836

CADIX was also utilized to estimate the chemical composition of the regenerant waste stream. Estimated concentrations of the regeneration stream are summarized in Table 3-20. The regenerant waste stream includes the volumes from the NaCl injection, displacement, and rinsing processes after the regeneration cycle. The total volume of the regenerant waste stream is 2,087 gal. It is important to note that, depending on water quality, the characteristics of ion exchange waste streams may vary.

Table 3-20. Concentrations of different chemical species in the regenerant waste stream. The regenerant waste stream includes the volumes from the NaCl injection, displacement, and rinsing processes of the regeneration cycle.

Chemical Species	Concentration (mg/L)
NaCl	20,648
$CaCl_2$	5,764
$MgCl_2$	3,688

3.6. High Pressure Membrane Modeling

The membrane-modeling computer program ROPRO (KMS, Wilmington, MA) was used to simulate membrane systems for a small water utility. A product flow of 75 gpm was used to simulate a small water system. Radionuclide constituents were incorporated in ROPRO using rejection data obtained from the bench- and pilot-scale experiments of this study. Water quality data from May Valley and Sterling were used in simulating two different systems, one system with elevated radium and gross alpha and one system with elevated uranium and gross alpha. Design considerations used in the simulation are summarized in Table 3-21. Based on recommendations from the manufacturer regarding

treatment of groundwater by membrane systems, the average water flux for these systems was set to an average of 20 GFD (34 LMH). Therefore, because the average water flux and the product flow for all the systems were constant, the same system design was utilized for all cases. A total of 66 Koch TFC-S (4040-element) membranes were required to simulate 75 gpm water production rate with May Valley and Sterling water quality data.

Table 3-21. Design considerations used in membrane modeling.

Parameter	Units	Values
Product Flow	gpm	75
Average Water Flux	GFD (LMH)	20 (34)
Membrane Used	Koch TFC-S (4040 Element)	
Number of Membranes Used		66

Each system was simulated at several different water recoveries in order to compare product and brine concentrations as well as energy demand. Simulations were performed with May Valley water quality data for recoveries of 50, 60, 70, and 75%. The 75% water recovery was the maximum achievable recovery for the system before precipitation of calcium sulfate and subsequent scaling of the membranes. Likewise, simulations were performed with Sterling water quality data for recoveries of 50, 60, 70, and 73%. Operating pressure, capital cost, and energy costs estimates for May Valley and Sterling are summarized in Tables 3-22 and 3-23.

ROPRO was also utilized to estimate the chemical composition of the product and brine streams of the membrane systems. The concentrations of TDS, hardness, chloride, and radionuclides in the brine and product streams are summarized in Table 3-24 and 3-25 for May Valley and Sterling, respectively. Results indicate the concentrations of each constituent in the permeate increase with increasing system water recovery. The concentration of each constituent in the brine also increases with increasing recovery. The results also indicate that for all system recoveries, the radium, gross alpha, and uranium concentrations are reduced to well below the MCLs.

Table 3-22. Operating pressures, brine flow, and energy use estimates from ROPRO for a May Valley system with a product flow of 75 gpm.

Recovery %	Operating Pressure psi	Brine Flow gpm	Energy Used kw-hr/yr
50	247	75	217,232
60	273	50	199,571
70	325	32	184,721
75	349	25	178,824

Table 3-23. Operating pressures, brine flow, capital cost, and energy use estimates from ROPRO for a Sterling system with a product flow of 75 gpm.

Recovery %	Operating Pressure psi	Brine Flow gpm	Energy Used kw-hr/yr
50	210	75	184,865
60	227	50	166,206
70	259	32	162,388
73	270	28	162,591

Table 3-24. Product and brine stream concentrations of TDS, hardness, chloride, radium, and gross alpha for May Valley system with a product flow of 75 gpm.

	Recovery %	TDS mg/L	Hardness mg/L $CaCO_3$	Chloride mg/L	Radium pCi/L	Gross Alpha pCi/L
Product	50	144	5.5	9	0.2	2.5
Brine		4317	691	65	38	118
Product	60	166	6.4	10	0.3	2.9
Brine		5326	860	77	47	145.6
Product	70	209	8.3	13	0.3	3.9
Brine		6943	1140	94	63	191
Product	75	303	8.6	13	0.4	5.0
Brine		8007	1367	108	75	217.2

Table 3-25. Product and brine stream concentrations of TDS, hardness, chloride, uranium, and gross alpha for Sterling system with a product flow of 75 gpm.

	Recovery %	TDS mg/L	Hardness mg/L $CaCO_3$	Chloride mg/L	Uranium µg/L	Gross Alpha pCi/L
Product	50	113	16	13	1.2	1.2
Brine		2896	1227	157	93	59
Product	60	129	19	15	1.4	1.4
Brine		3567	1526	190	115	73
Product	70	155.6	24	18	1.8	1.8
Brine		4651	2017	241	146	96
Product	73	170	27	20	2	2
Brine		5113	2231	261	169	106

4. Conclusions

Tests conducted on four treatment technologies (RO/NF, HMO, IX, and enhanced coagulation) demonstrated removal of radionuclides as described below.

4.1. High Pressure Membranes

Results from bench- and pilot-scale tests demonstrated NF and RO membrane processes are effective for removal of radium, uranium, and gross alpha activity to levels below the respective MCLs for the water quality conditions tested. Other water quality parameters, including calcium, magnesium, sodium, chloride, and sulfate were also significantly reduced. NF also removed radionuclides to below the MCLs with higher water recovery and lower energy requirements compared to RO membranes. NF membranes may also be preferred over RO membranes because a small portion of salts to pass through the membranes, which produces a permeate more suitable for drinking than RO permeate, which requires remineralization post-treatment.

The primary advantages of pressure-driven membrane systems include simple operation and maintenance, small footprint, and the ability to address water quality concerns in addition to radionuclides (e.g., hardness, TDS and sulfate). The main disadvantages of these systems include high capital cost, potential pre- and post-treatment requirements, and more complex residuals management due to the large volumes produced (water recovery can range from 60-85%). Brine generated during treatment has to be properly managed because of the elevated radionuclide concentrations in the water. Disposal options for these liquid residuals can be challenging and expensive; therefore, pressure driven membrane processes should be designed to operate at the highest recovery possible.

4.2. Cation Exchange

Cation exchange is an effective treatment option for waters containing elevated concentrations of radium and gross alpha. Combined radium and gross alpha activity levels were reduced to below the MCLs in waters that did not contain uranium. Gross alpha activity associated with uranium is not removed by cation exchange; therefore, gross alpha activity was not always reduced to below the MCL in these waters, though combined radium was. An ion exchange system comprised of both cation and anion exchange is more suitable for waters containing radium, uranium, and gross alpha activity.

The primary advantages of a cation exchange are minimal maintenance between regenerations and water recovery is relatively high (typically greater than 95%). The main disadvantage of cation exchange is that calcium and magnesium compete with radium for adsorption, reducing the efficiency and increasing the operating cost and volume of liquid residuals produced by the treatment process. Cation exchange may not be an economic solution to treat waters with very high hardness. Furthermore, for systems with elevated iron concentrations in their water (>0.1-0.5 mg/L), pretreatment to remove iron prior to the ion exchange system would be required to iron precipitation and fouling of the resin.

An additional concern with cation exchange was observed during this study was the irreversible accumulation of radium on the cation exchange resin. In this study, only a small portion of the combined radium was eluted from the resin. Previous studies have shown similar results, where initial combined radium accumulation is not completely reversible; however, radium always broke through shortly after hardness in multiple treatment cycles (Subramonian, Clifford et al., 1990). Site-specific tests for cation exchange are recommended for all systems considering this as a compliance option to confirm removal efficiency, and design parameters.

4.3. Hydrous Manganese Oxide

HMO was shown to remove 80-90% of combined radium in waters tested in this study at an HMO dose of 1.0 mg/L. Removal of gross alpha activity ranged from 24-92%. Lower removal of gross alpha activity was observed in waters that also contained uranium because gross alpha activity associated with uranium is not removed by HMO. Doses of 0.5 mg/L and 1.0 mg/L were tested, where when HMO was applied at concentrations above 1.0 mg/L, the water began to discolor; therefore, the dose of HMO that can be applied is limited. HMO could be a relatively inexpensive compliance solution for systems with existing filtration; however, a robust backwash (with air scour) is required to remove the HMO from the filter media. Systems interested in HMO should conduct bench-scale tests to confirm consistent removal of combined radium and gross alpha activity by HMO to levels below the MCLs in their source waters.

To date, HMO has only been employed in a limited number of treatment facilities across the US; therefore, further research should focus on treatment efficiency, operation, maintenance, and potential accumulation of radium on the filter media. Also, disposal of spent filter backwash water from the filters that have radium-laden HMO particles is a concern and should be considered and further studied. Finally, the HMO testing in this study was done in small batches and therefore, the transition to a large-scale continually-operated system may affect the treatment efficiency of this process and should be tested.

4.4. Enhanced Coagulation

Results from this study demonstrated 80-90% removal of uranium and associated gross alpha with coagulation using alum. Coagulation is not a feasible technology for most of the smaller CO-RADS systems, but could be applied at a system like the City of Sterling or other systems that serve more than 10,000 people. Sludge produced from sedimentation will contain elevated levels of uranium and gross alpha activity and will need to be carefully handled and disposed of.

4.5. Technology Comparison

A summary of the testing results, advantages, disadvantages, and additional considerations for each CO-RADS system for each technology are compared in Table 4-1.

Table 4-1. Summary of Testing Results

Treatment Technology	Conditions Tested	Radionuclide Removal Observed in Testing (%)	Advantages	Disadvantages	Additional Considerations for CO-RADS Systems
Nanofiltration	Flat sheet membrane and pilot tests	• Radium-226: 98-99% • Uranium: 95-98% • GAA: 71-96% • Gross Beta:92-93%	• Can address other water quality (and potentially compliance) issues (e.g., TDS, hardness, sulfate) • Operates at higher recovery than RO • Operates at lower pressure than RO (relatively lower energy costs) • Is a physical barrier to radionuclides – treatment disruptions will not impact radionuclide removal	• Large volume of liquid residuals to manage • Low water recovery • May require pre- and post-treatment • Energy intensive	• Significant volume of liquid residuals produced • Requires a Class C operator certification to operate • Site-specific testing is recommended to determine design criteria, potential fouling, and pre- and post-treatment needs.
Reverse Osmosis	Flat sheet membrane tests	• Radium-226: 98% • Uranium: not tested • GAA: 92% • Gross Beta: 93%	• Can address other water quality (and potentially compliance) issues (e.g., TDS, hardness, sulfate) • Is a physical barrier to radionuclides – treatment disruptions will not impact radionuclide removal	• Very large volume of liquid residuals to manage • Low water recovery • Will likely require pre- and post-treatment • Very energy intensive	• Significant volume of liquid residuals produced • Requires a Class C operator certification to operate • Site-specific testing is recommended to determine design criteria, potential fouling, and pre- and post-treatment needs.

Treatment Technology	Conditions Tested	Radionuclide Removal Observed in Testing (%)	Advantages	Disadvantages	Additional Considerations for CO-RADS Systems
Enhanced Coagulation	Alum doses of 5, 10, and 20 mg/L at pH 6.2 and 7.5; Raw uranium concentration = 47 μg/L	• Radium-226: not tested • Uranium: 19-87% • GAA: 0-82% • Gross Beta: not tested	• TOC and particles also removed	• Results in radionuclide accumulation in treatment plant sludge • Requires significant infrastructure if conventional treatment is not already in place • Is not a physical barrier to radionuclides; treatment must be carefully controlled to achieve targeted removal	• Not recommended for systems serving less than 10,000 customers • Site-specific bench-scale tests recommended • Continuous optimization may be required • Requires a Class B operator certification to operate
	Ferric doses of 5, 10, and 20 mg/L at pH 6.2 and 7.5 (20 mg/L dose not tested at pH 6.2) ; Raw uranium concentration = 47 μg/L	• Radium-226: not tested • Uranium: 0-36% • GAA: 0-15% • Gross Beta: not tested	• TOC and particles also removed	• Tests did not show significant removal of uranium for any of the scenarios tested • Results in radionuclide accumulation in treatment plant sludge • Requires significant infrastructure if conventional treatment is not already in place	• Not recommended for systems serving less than 10,000 customers • Site-specific bench-scale tests recommended • Continuous optimization may be required • Requires a Class B operator certification to operate
Cation Exchange	Purolite C-100E resin; Regenerated after hardness breakthrough	• Radium-226: effluent concentration ≤ 0.5 pCi/L in all samples • Uranium: not tested • GAA: ≤ 4 pCi/L in waters with no uranium; little removal in waters with uranium • Gross Beta: not tested	• Hardness also removed	• Not a physical barrier to radionuclides; system can produce water with radionuclides exceeding the MCL if not operated correctly • Pre- and post-treatment may be	• Site-specific bench-scale tests are recommended to confirm design parameters • Requires a Class C operator certification to operate

Treatment Technology	Conditions Tested	Radionuclide Removal Observed in Testing (%)	Advantages	Disadvantages	Additional Considerations for CO-RADS Systems
				required • Radium will accumulate in the resin	
HMO	Doses of 0.5 and 1.0 mg/L	• Radium-226: 52-96% • Uranium: not tested • GAA: 37-92% (GAA associated with uranium not removed in one water) • Gross Beta: not tested	• Can be added as a retrofit to an existing filtration system	• May not reduce combined radium and gross alpha to levels below the MCLs in waters with elevated levels of radium • No installations in CO and few in the US • Sludge produced contains high levels of radium • Removal of radium varies • HMO dose limited at ~1.0 mg/L due to discoloration of water above that dose	• Demonstrating testing likely required • Existing filters may need upgrade to backwash system • Requires a Class B operator certification to operate

5. REFERENCES CITED

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APPENDIX A: WATER SOURCE DATA

Table A.1: Water quality data for water utilities participating in CO-RADS

System Name	Site Desc.	GAA (evap meth)	GAA (precip meth)	Gross Beta	Ra 228	Ra 226	Comb Ra	Radon 222	U	Alk	Ca	DO	Fe	Mg	Mn	pH	TDS	SO4	Temp	TOC	Turb
		pCi/L	pCi/L	pCi/L	pCi/L	pCi/L	pCi/L	pCi/L	µg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L		mg/L	mg/L	°C	mg/L	NTU
Antelope Hills HOA	001	4		38	4.1	2.8	6.9	140	4	220	81	2	3.2	10	0.1	7.2	380	120	16	13.6	39.4
Antelope Hills HOA	002	10		38	5.0	4.5	9.5	140	4	240	83	1.8	1.4	10	0.053	7.11	400	120	19.2	11.2	18
Antelope Hills HOA	005	7		35	4.1	4.1	8.2	110	4	210	81	1.8	1.4	10	0.066	7.18	360	130	19	9.9	28.3
Aspen Park Met Dist	004	9		15	0.5	3.6	4.1	8370	24	140	93	5.5	0.005	7.5	0.021	8.18	200	7	11.6	1	0.39
Buffalo Park Dev	005T									220	85	1.5	0.005	13	0.001	8	410		9.3		
Buffalo Park Dev	008		23	21	0.5	0.5	1.0	7400	91	180	140	6	0.005	13	0.001	7.13	340	16	11.5	1.7	0.06
Buffalo Park Dev	009	2		20	0.5	1.9	2.4	26700	21	160	100	4	0.005	9.2	0.014	7.39	290	17	12	1.6	0.07
City of Sterling	001	26	2	67	0.5	0.5	1.0	230	30	290	350	4.5	0.005	36	0.003	7.66	1020	460	19.4	2.2	0.19
City of Sterling	022			37	0.5	0.5	1.0	320	44	300	360	4	0.005	57	0.001	7.53	1220	840	17.1	2.7	0.08
City of Sterling	002		1		0.5	0.5	1.0	280	24		290		0.05	33	0.018		820	340		1.8	0.18
Eureka WC	001									200		1.9				6.82			23.2		
Eureka WC	003									200		1.9				6.78			23.1		
Eureka WC	007	41	40	36	7.0	8.0	15.0	250	0.5	200	23	4	0.01	9	0.005	7.15	890	610	22.1	1	0.16
Fayette WC	003	24	17	34	11.1	4.9	16.0	71	0.5	190	110	6.5	0.69	30	0.034	6.9	1220	720	19.3	3.7	7.56
Hancock WC	001T		14		6.3	5.8	12.1	220	0.5	190	18	3.0	0.94	7.3	0.01	7.2	930	490	14.5	1	1.9
Hillside TP	001T		25		10.7	8.7	19.4	130	0.5	220	26	7	1.6	11	0.012	7.3	1010	590	18	0.7	13
Holly Town of	001										150		1.8	20	0.064						390
Holly Town of	002	8		16	1.2	2.1	3.3	74	1	220	150	2	0.38	21	0.022	8	420	180	16	0.25	2.67
Las Animas, City of	002		2		0.5	0.5	1.0	660	32		750		0.005	110	0.32		2800	1700		2.8	0.04
Las Animas, City of	002T	1		2	0.5	0.5	1.0	470	0.5	50	0.9	3.0	0.005	0.17	0.001	5.8	5	5	15.7	0.25	0.01
Manzanola Town of	001		11		2.0	1.7	3.7	94	24	280	530	8	0.01	46	0.001	7.74	1490	1100	14.8	1.4	0.06
Manzanola Town of	002		6		2.5	1.2	3.7	240	25	290	570		0.45	50	0.003	7.54	1490	820	16.3	1.5	3.5

Table A.1 (Continued): Water quality data for water utilities participating in CO-RADS

System Name	Site Desc.	GAA (evap meth)	GAA (precip meth)	Gross Beta	Ra 228	Ra 226	Comb Ra	Radon 222	U	Alk	Ca	DO	Fe	Mg	Mn	pH	TDS	SO4	Temp	TOC	Turb
		pCi/L	pCi/L	pCi/L	pCi/L	pCi/L	pCi/L	pCi/L	µg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L		mg/L	mg/L	°C	mg/L	NTU
May Valley WA	002		58		12.0	15.5	27.5	140	0.5	400	200	4	1.1	36	0.15	7.6	2040	1200	18.3	0.5	14.5
May Valley WA	002T		60		7.1	9.7	16.8	270	0.5	400	200	4	0.04	36	0.005	7.9	1970	1100	19.2	0.5	0.08
May Valley WA	003		48		4.9	6.8	11.7	280	0.5	400	150	2	0.55	21	0.071	7.6	1960	1100	20.3	0.6	6.38
May Valley WA	003T		32		3.7	7.4	11.1	380	0.5	400	160	3	0.05	22	0.007	7.8	1950	1100	18		0.07
May Valley WA	007										98		1.5	26	0.013		1490				
May Valley WA	009T										170		0.02	27	0.002		1800				
May Valley WA	010T										200		0.005	32	0.001						2170
Merino Town of	003			44	0.5	0.5	1.0	240	52	320	410	3.5	0.005	53	0.001	7.5	1340	780	17.1	2.6	0.09
Mountain Shadows	001	8		13	2.7	3.1	5.8	460	3	250	160	9.5	0.005	11	0.001	7.57	300	37	14.5	0.9	0.1
Mountain WSD	014	24		77	0.5	0.5	1.0	1930	280	130	78	7	0.04	5.6	0.001	7.29	180	8	15.8	1.7	0.39
Mountain WSD	025	7		9	0.5	0.5	1.0	4990	8	150	97	8	0.005	7.2	0.001	7.5	210	9	13.6	1.8	0.11
Mountain WSD	027	8		9	0.5	0.5	1.0	2900	12	130	110	4	0.02	6	0.004	7.17	220	5	17.1	2.2	0.31
Mountain WSD	029	9		12	0.5	0.5	1.0	8200	18	120	72	7	0.005	7.1	0.001	7.11	190	7	12.2	0.9	0.08
North Holbrook WC	001T		10		3.3	2.5	5.8	90	0.5	240	14	4.0	0.76	6.3	0.007	7.8	830	440	18	0.9	0.55
Park Water Company	005	13	28	25	1.0	0.5	1.5	3500	21	200	210	6	0.005	13	0.001	7.41	490	21	13.9	1.7	0.03
Patterson Valley	001		31		13.2	9.0	22.2	35	0.5	230	94	5	1.5	29	0.043	7.07	1290	1000	25	1.1	0.01
Patterson Valley	001T		23		10.1	7.5	17.6	150	0.5	210	90	4.0	0.06	28	0.044	6.6	1240	1100	24.1	1	0.05
Redhill Forest POW	003T	1		5	0.5	0.5	1.0	3280	0.5	40	4.3		0.06	0.64	0.008	7.37	290	160	9.6	0.6	0.11
Redhill Forest POW	003	159		87	2.9	32.4	35.3	2300	8	45	140	6	3.1	12	0.25	7.65	240	150	9	0.6	17
Sheridan Lake WC	001T	0	5	30	0.5	0.5	1.0	720	33	240	350	8.0	0.02	75	0.001	7.7	1280	780	16.4	4.2	0.07
South Swink WC	003T									280	51	2.4	0.005	18	0.001	6.85			23.8		810
South Swink WC	005	31	39	49	6.0	7.0	13.0	160	0.5	200	72	5	0.005	27	0.001	6.9	1060	670	19.2	0.6	0.02

Table A.1 (Continued): Water quality data for water utilities participating in CO-RADS

System Name	Site Desc.	GAA (evap meth)	GAA (precip meth)	Gross Beta	Ra 228	Ra 226	Comb Ra	Radon 222	U	Alk	Ca	DO	Fe	Mg	Mn	pH	TDS	SO4	Temp	TOC	Turb
		pCi/L	pCi/L	pCi/L	pCi/L	pCi/L	pCi/L	pCi/L	µg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L		mg/L	mg/L	°C	mg/L	NTU
Town of Kit Carson	002T		`		0.5	0.5	1.0	380	34	250	480	0.8	1.8	83	0.22	7.7	2060	1300	20.1	6.7	36.3
Town of Larkspur	003	5		13	2.0	3.0	5.0	600	3	170	60	5	0.33	3.8	0.017	7.46	140	15	22.6	0.25	0.77
Town of Swink	001T	12	13	27	3.0	3.0	6.0	28	2	390	19	6	0.005	7.9	0.001	7.54	950	450	20.4	0.6	0.005
Town of Swink	001			34	0.5	0.5	1.0	540	51	390	800	6	0.005	96	0.001	7.17	2200	1200	17.6	2.2	0.05
Turkey Canyon WD	001	23	16	38	0.5	0.5	1.0	260	39	680	340	7.0	0.09	59	0.001	7	800	80	15.7	0.25	0.025
TV Hills Water LLC	001T			16	0.5	0.5	1.0	6760	43	220	1.3	2.0	0.02	0.24	0.001	7.5	290	25	11.5	0.6	0.1
TV Hills Water LLC	001	10		35	1.7	8.2	9.9	4940	36	220	110	3	0.21	13	0.009	7.6	240	26	10.7	0.6	1.6
Valley WC	002T	13	13	22.5	7.4	3.7	11.1	155	0.5	130	51	3	0.005	14	0.001	7.14	1080	650	20.03	5.9	0.11
Vroman WC	001T	23	25	28	8.0	5.0	13.0	92	0.5	290	57	8	0.04	18	0.015	7.27	1200	850	26.1	0.8	0.06
Wayward Wind	002		5	42	0.5	0.5	1.0	530	49	260	530	0.4	0.005	42	0.001	7.6	1320	760	12.6	1.8	0.08
Whispering Pines	001T	5		24	5.7	1.9	7.6	1390	1	90	86	6	0.005	6.4	0.001	6.93	230		8.3	0.8	0.15
Wiley, Town of	001T		6		2.1	1.8	3.9	28	0.5	220	220	4	0.05	59	0.067	7.4	1230	970	16.5	0.6	0.25

APPENDIX B: WATER QUALITY RESULTS FROM BENCH- AND PILOT-SCALE TESTING

Table B.1 Concentrations of all analytes measured for the May Valley bench-scale membrane performance monitoring experiment. Three samples were taken from each membrane over the course of the 26-hour experiment. 'NF' samples represent samples from the Dow NF 4040; 'TS' samples represent samples from the Koch TFC-S; 'HR' samples represent samples from the TFC-HR.

Sample		Feed	NF -1	NF - 2	NF - 3	TS - 1	TS - 2	TS - 3	HR - 1	HR - 2	HR - 3
Date			5/23/08	5/23/08	5/24/08	5/23/08	5/23/08	5/24/08	5/23/08	5/23/08	5/24/08
Length into run	hr		0.5 - 1	4 - 4.5	25 - 25.5	0.5 - 1	4 - 4.5	25 - 25.5	0.5 - 1	4 - 4.5	25 - 25.5
Ag	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.003
Al	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
As	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	0.04	BDL	BDL	BDL
B	mg/L	0.35	0.21	0.21	0.23	0.19	0.19	0.22	0.14	0.13	0.14
Ba	mg/L	0.030	0.001	0.000	0.000	BDL	BDL	BDL	0.001	0.001	BDL
Be	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Ca	mg/L	84.9	1.2	1.1	1.3	0.3	0.3	0.2	1.5	1.1	0.3
Cd	mg/L	0.00	0.00	BDL	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Co	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Cr	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Cu	mg/L	0.004	0.004	0.002	0.003	0.013	0.010	0.006	0.012	0.005	0.006
Fe	mg/L	0.003	0.004	0.002	BDL	BDL	BDL	BDL	0.004	BDL	0.005
K	mg/L	18.1	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Li	mg/L	0.44	0.05	0.05	0.06	0.01	0.01	0.01	0.01	0.01	0.01
Mg	mg/L	38.0	0.6	0.5	0.6	0.1	0.1	0.1	0.6	0.5	0.1
Mn	mg/L	0.167	0.004	0.002	0.003	0.004	0.002	0.001	0.004	0.003	0.000
Mo	mg/L	0.028	0.006	0.004	BDL	BDL	BDL	BDL	0.006	0.005	BDL
Na	mg/L	566.9	59.1	57.3	73.1	7.4	6.5	6.7	11.5	9.3	4.9
Ni	mg/L	0.003	BDL	0.003	BDL	0.003	0.003	BDL	BDL	BDL	0.004
P	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	1.5
Pb	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
S	mg/L	324.9	3.4	2.8	2.1	2.0	1.8	1.4	6.0	4.9	1.8
Sb	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Se	mg/L	0.031	BDL	BDL	BDL	0.035	BDL	BDL	BDL	0.025	0.039
Si	mg/L	4.8	2.9	2.7	3.2	0.2	0.1	0.2	0.7	0.3	0.1
Sn	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Sr	mg/L	1.5	0.019	0.016	0.021	0.004	0.003	0.002	0.023	0.017	0.004
Ti	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
V	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Zn	mg/L	0.019	0.011	0.003	0.001	0.004	0.002	0.001	0.011	0.004	0.004
Sr	mg/L	1.5	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
F	mg/L	1.8	0.3	0.3	0.4	BDL	BDL	BDL	BDL	BDL	BDL
Cl	mg/L	21.4	12.7	13.4	16.9	0.9	0.8	0.7	BDL	0.5	0.4
Br	mg/L	0.18	0.15	0.16	0.17	BDL	BDL	BDL	BDL	BDL	BDL
NO3	mg/L	0.16	0.11	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
PO4	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
SO4	mg/L	1164.8	10.6	9.4	6.7	6.4	6.3	4.6	18.4	14.4	5.3

Table B.2: Concentrations of all analytes measured for the May Valley bench-scale membrane monitoring experiment. Three samples were taken from each membrane over the course of the 26-hour experiment. 'NF' samples represent samples from the Dow NF 4040; 'TS' samples represent samples from the Koch TFC-S; 'HR' samples represent samples from the TFC-HR.

Sample		Feed 1	Feed 2	NF - 1	NF - 2	TS - 1	TS - 2	HR - 1	HR - 2
Date		5/27/08	5/27/08	5/27/08	5/27/08	5/27/08	5/27/08	5/27/08	5/27/08
Length into run	hr			1 - 6.5	6.5 - 12	1 - 6.5	6.5 - 12	1 - 6.5	6.5 - 12
Gross Alpha	pCi/L	36.0	42.0	BDL	BDL	BDL	BDL	BDL	BDL
Gross Beta	pCi/L	56.0	59.0	BDL	BDL	BDL	BDL	5.0	BDL
Ra-228	pCi/L	6.5	8.5	NA	NA	NA	NA	NA	NA
Ra-226	pCi/L	12.6	13.6	BDL	0.2	BDL	BDL	0.3	0.2
TOC	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Ag	mg/L	BDL	BDL	0.003	BDL	BDL	BDL	BDL	BDL
Al	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
As	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
B	mg/L	0.35	0.35	0.25	0.27	0.23	0.23	0.17	0.18
Ba	mg/L	0.016	0.016	BDL	0.000	BDL	BDL	BDL	0.0003
Ca	mg/L	84.3	84.7	1.3	1.5	0.5	0.4	2.1	1.4
Cd	mg/L	0.003	0.003	0.001	0.002	0.001	0.001	0.000	0.000
Co	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Cr	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Cu	mg/L	0.002	0.001	0.003	0.002	0.011	0.010	0.004	0.005
Fe	mg/L	0.005	BDL	BDL	BDL	BDL	0.003	0.003	0.003
K	mg/L	18.6	18.1	0.2	0.6	BDL	BDL	BDL	BDL
Li	mg/L	0.44	0.44	0.05	0.06	0.01	0.01	0.02	0.01
Mg	mg/L	38.1	38.0	0.6	0.7	0.2	0.1	0.9	0.6
Mn	mg/L	0.168	0.166	0.003	0.003	0.001	0.001	0.004	0.003
Mo	mg/L	0.02	0.02	BDL	0.01	BDL	BDL	BDL	BDL
Na	mg/L	570.7	568.2	68.0	78.4	8.7	8.7	16.5	12.3
Ni	mg/L	0.004	0.006	0.001	0.003	0.005	0.003	0.004	0.002
P	mg/L	0.22	0.17	BDL	BDL	BDL	BDL	BDL	BDL
Pb	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
S	mg/L	331.4	329.5	3.6	3.5	2.7	2.2	8.5	5.9
Sb	mg/L	BDL	BDL	0.02	BDL	BDL	BDL	BDL	BDL
Se	mg/L	0.02	0.03	0.05	BDL	BDL	0.03	BDL	0.04
Si	mg/L	4.5	4.4	3.0	3.3	0.1	0.1	0.1	0.0
Sn	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.019
Sr	mg/L	1.50	1.51	0.02	0.02	0.01	0.00	0.03	0.02
Ti	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Zn	mg/L	0.011	0.017	0.003	0.002	0.003	0.002	0.004	0.002
Sr	mg/L	1.5	1.5	BDL	BDL	BDL	BDL	BDL	BDL
F	mg/L	1.75	1.76	0.36	0.44	BDL	BDL	0.05	BDL
Cl	mg/L	21.0	20.9	13.8	16.3	0.8	0.8	0.7	0.7
Br	mg/L	0.16	0.21	0.18	0.18	BDL	BDL	BDL	BDL
NO3	mg/L	BDL	3.698	BDL	BDL	BDL	BDL	BDL	BDL
PO4	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
SO4	mg/L	1203.0	1255.6	109.4	10.9	8.1	7.3	26.7	19.0

Table B.3: Concentrations of all analytes measured for the May Valley pilot-scale testing. The Koch TFC-S membrane was tested at recoveries of 30% and 80%. Samples were taken from the combined permeate ('Perm Com'), the permeate of the first membrane ('Perm 1'), and from the reject ('Reject')

Sample		Feed	30 Perm Com	30 Perm 1	30 Reject	80 Perm Com	80 Perm 1	80 Reject
Date		7/24/08	7/24/08	7/24/08	7/24/08	7/24/08	7/24/08	7/24/08
Gross Alpha	pCi/L	70	< 3	< 3	NA	< 3	< 3	186
Gross Beta	pCi/L	NA	< 4	< 4	NA	< 4	5	NA
Ra-228	pCi/L	7	NA	NA	NA	NA	NA	NA
Ra-226	pCi/L	12	0.1	0.2	NA	0.2	0.3	30.9
TOC	mg/L	BDL	BDL	NA	BDL	BDL	NA	BDL
Ag	mg/L	BDL	0.002	NA	0.001	BDL	NA	BDL
Al	mg/L	0.004	0.005	NA	0.009	0.008	NA	0.014
As	mg/L	BDL	BDL	NA	BDL	BDL	NA	BDL
B	mg/L	0.13	0.24	NA	0.13	0.18	NA	0.12
Ba	mg/L	0.006	0.000	NA	0.009	0.000	NA	0.015
Ca	mg/L	73.5	0.6	NA	104.3	1.8	NA	188.5
Cd	mg/L	0.001	0.000	NA	0.002	BDL	NA	0.003
Co	mg/L	0.001	0.000	NA	0.001	0.000	NA	0.000
Cr	mg/L	BDL	BDL	NA	BDL	BDL	NA	BDL
Cu	mg/L	0.07	0.23	NA	0.08	0.18	NA	0.07
Fe	mg/L	0.071	0.007	NA	0.313	0.007	NA	0.005
K	mg/L	15.4	1.1	NA	21.6	3.4	NA	38.9
Li	mg/L	0.39	0.03	NA	0.55	0.07	NA	0.95
Mg	mg/L	31.7	0.3	NA	44.4	0.9	NA	78.4
Mn	mg/L	0.011	0.003	NA	0.005	0.002	NA	0.002
Mo	mg/L	0.005	0.013	NA	0.010	0.004	NA	0.016
Na	mg/L	460.4	26.1	NA	734.1	80.0	NA	1124.1
Ni	mg/L	0.001	0.001	NA	0.001	0.001	NA	BDL
P	mg/L	0.091	0.001	NA	0.095	BDL	NA	0.225
Pb	mg/L	BDL	BDL	NA	BDL	BDL	NA	BDL
S	mg/L	337.9	6.7	NA	460.7	22.6	NA	815.0
Sb	mg/L	BDL	BDL	NA	BDL	BDL	NA	BDL
Se	mg/L	0.014	0.014	NA	0.009	0.009	NA	0.043
Si	mg/L	4.4	1.1	NA	5.7	2.1	NA	8.4
Sn	mg/L	BDL	0.0014	NA	BDL	BDL	NA	BDL
Sr	mg/L	1.29	0.01	NA	1.85	0.03	NA	3.43
Ti	mg/L	BDL	BDL	NA	BDL	BDL	NA	BDL
V	mg/L	0.000	0.000	NA	0.000	BDL	NA	0.001
Zn	mg/L	0.01	0.03	NA	0.03	0.02	NA	0.02
Sr	mg/L	1.29	0.01	NA	1.81	0.04	NA	3.36
F	mg/L	1.65	0.42	NA	3.81	0.91	NA	2.33
Cl	mg/L	38.4	17.3	NA	74.4	6.4	NA	51.1
Br	mg/L	0.07	0.02	NA	0.05	0.00	NA	0.05
NO3	mg/L	0.48	0.49	NA	0.59	0.24	NA	0.58
PO4	mg/L	0.23668	BDL	NA	BDL	BDL	NA	BDL
SO4	mg/L	1102	69	NA	2675	20	NA	1562

Table B.4: Concentrations of all analytes measured for the May Valley HMO testing. Two doses (0.5 mg/L and 1.0 mg/L) were tested in duplicate.

Sample		Feed	0.5 A	0.5 B	1.0 A	1.0 B
HMO Dose	mg/L		0.5	0.5	1.0	1.0
Date		7/22/08	7/24/08	7/24/08	7/24/08	7/24/08
Gross Alpha	pCi/L	55	24	10	6	5
Ra-228	pCi/L	6.9	NA	NA	NA	NA
Ra-226	pCi/L	11.4	1.4	2.1	0.4	0.4
TOC	mg/L	0.73	0.92	0.84	0.99	0.82
Ag	mg/L	BDL	BDL	BDL	BDL	BDL
Al	mg/L	BDL	0.0201	0.0011	0.0108	0.0224
As	mg/L	BDL	BDL	BDL	BDL	BDL
B	mg/L	0.14	0.12	0.22	0.16	0.14
Ba	mg/L	0.006	0.003	0.003	0.002	0.002
Be	mg/L	BDL	BDL	BDL	BDL	BDL
Ca	mg/L	66.2	74.4	66.1	77.9	70.4
Cd	mg/L	0.001	0.001	0.001	0.001	0.001
Co	mg/L	BDL	BDL	0.0017	0.0004	0.0006
Cr	mg/L	BDL	BDL	BDL	BDL	0.0019
Cu	mg/L	0.08	0.04	0.08	0.07	0.06
Fe	mg/L	0.001	0.003	0.007	0.002	0.001
K	mg/L	12.8	16.1	13.3	17.8	15.2
Li	mg/L	0.35	0.39	0.35	0.41	0.37
Mg	mg/L	28.9	32.5	29.5	34.4	31.0
Mn	mg/L	0.001	0.015	0.009	0.013	0.010
Mo	mg/L	0.008	0.007	0.016	0.008	0.007
Na	mg/L	455	424	438	439	462
Ni	mg/L	0.000	0.004	0.003	0.005	0.011
P	mg/L	0.08	0.08	0.08	0.06	0.02
Pb	mg/L	BDL	BDL	BDL	BDL	BDL
S	mg/L	294	329	307	356	326
Sb	mg/L	BDL	BDL	BDL	BDL	BDL
Se	mg/L	0.016	0.018	0.018	0.009	0.016
Si	mg/L	3.8	4.4	4.0	4.8	4.1
Sn	mg/L	BDL	BDL	BDL	BDL	BDL
Sr	mg/L	1.2	1.3	1.2	1.4	1.2
Ti	mg/L	BDL	BDL	BDL	BDL	BDL
V	mg/L	BDL	BDL	BDL	BDL	BDL
Zn	mg/L	0.014	0.006	0.005	0.010	0.011
Sr	mg/L	1.2	1.3	1.2	1.4	1.2
F	mg/L	1.7	1.7	1.6	1.6	1.6
Cl	mg/L	38.7	38.9	39.1	38.2	38.6
Br	mg/L	0.077	0.088	0.063	0.057	0.059
NO3	mg/L	0.75	0.51	0.53	0.51	0.56
PO4	mg/L	BDL	BDL	0.07	BDL	0.17
SO4	mg/L	1114	1102	1108	1084	1090

Table B.5: Concentrations of all analytes measured for the May Valley ion exchange testing. One sample was taken 2-3 hours into each of the six cycles.

Sample		Feed 1	Cycle 1	Cycle 2	Cycle 3	Cycle 4	Cycle 5	Cycle 5
Date		7/22/08	7/23/08	7/24/08	7/28/08	7/28/08	7/29/08	7/29/08
Gross Alpha	pCi/L	62	3	< 3	< 3	3	4	3
Ra-228	pCi/L	3	NA	NA	NA	NA	NA	NA
Ra-226	pCi/L	12.4	< 0.3	0.1	0.1	0.3	0.3	0.5
TOC	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Ag	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Al	mg/L	BDL	BDL	0.0062	0.0065	BDL	BDL	BDL
As	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
B	mg/L	0.15	0.14	0.14	0.14	0.13	0.13	0.13
Ba	mg/L	0.007	BDL	BDL	BDL	BDL	BDL	BDL
Be	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Ca	mg/L	82.9	0.2	3.9	4.4	4.1	3.8	4.7
Cd	mg/L	0.001	0.001	0.001	0.001	0.001	0.001	0.001
Co	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	0.0010
Cr	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Cu	mg/L	0.11	0.04	0.05	0.05	0.04	0.03	0.05
Fe	mg/L	BDL	0.01	0.00	0.01	0.00	0.01	0.01
K	mg/L	18.8	17.2	7.9	1.4	2.2	2.1	5.8
Li	mg/L	0.44	0.41	0.46	0.44	0.43	0.42	0.49
Mg	mg/L	36.6	0.3	0.9	1.1	0.9	0.8	1.2
Mn	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Mo	mg/L	0.010	0.007	0.008	0.006	0.008	0.006	0.008
Na	mg/L	449	580	645	575	605	574	607
Ni	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
P	mg/L	0.08	0.02	BDL	0.06	0.00	0.03	BDL
Pb	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
S	mg/L	387	313	361	340	339	338	401
Sb	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Se	mg/L	0.03	0.02	0.04	0.02	0.03	0.02	0.02
Si	mg/L	5.0	3.9	4.6	4.4	4.3	4.2	5.0
Sn	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Sr	mg/L	1.46	0.00	0.05	0.07	0.07	0.07	0.08
Ti	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
V	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Zn	mg/L	0.014	0.003	0.007	0.005	0.002	0.001	0.003
Sr	mg/L	1.46	0.00	0.06	0.07	0.08	0.07	0.08
F	mg/L	1.7	1.6	1.7	1.6	1.6	1.6	1.7
Cl	mg/L	38.0	37.8	37.8	38.0	38.2	41.8	38.6
Br	mg/L	0.03	0.02	BDL	BDL	0.02	BDL	0.03
NO3	mg/L	0.50	0.47	0.49	0.44	0.45	0.48	0.45
PO4	mg/L	BDL	0.0407	BDL	BDL	BDL	BDL	0.2347
SO4	mg/L	1096	1090	1102	1084	1096	1096	1120

Table B.6: Concentrations of all analytes measured for the Blue Mountain HMO testing. Three HMO dosing conditions were tested: 0.5 mg/L, 0.5 mg/L (w/0.5 mg/L $KMnO_4$), and 1.0 mg/L

Sample		Feed 1	Feed 2	0.5 A	0.5 B	1.0 A	1.0 B	0.5# A	0.5# B
HMO Dose	mg/L			0.5	0.5	1.0	1.0	0.5	0.5
$KMnO_4$ Dose	mg/L			0	0	0	0	0.5	0.5
Date		6/26/08	6/26/08	6/26/08	6/26/08	6/26/08	6/26/08	6/26/08	6/26/08
Gross Alpha	pCi/L	110	NA	84	NA	45	NA	69	NA
Gross Beta	pCi/L	29	NA	16	NA	12	NA	18	NA
Ra-228	pCi/L	BDL	BDL	NA	NA	NA	NA	NA	NA
Ra-226	pCi/L	33.0	35.9	18.1	8.5	7.0	3.9	16.7	6.5
TOC	mg/L	0.6	0.6	0.6	0.5	0.5	0.5	0.5	0.5
Ag	mg/L	0.038	0.038	BDL	0.004	BDL	BDL	BDL	BDL
Al	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
As	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
B	mg/L	0.16	0.16	0.08	0.01	BDL	BDL	BDL	0.03
Ba	mg/L	0.10	0.10	0.08	0.08	0.05	0.05	0.08	0.08
Be	mg/L	BDL	BDL	0.0008	BDL	BDL	BDL	BDL	BDL
Ca	mg/L	49.4	49.4	49.8	50.9	50.1	51.1	51.3	53.8
Cd	mg/L	0.005	0.005	0.003	BDL	0.002	0.002	0.002	0.002
Co	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	0.0078	BDL
Cr	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Cu	mg/L	0.01	0.01	0.02	0.01	0.01	0.02	0.01	0.01
Fe	mg/L	0.05	0.05	0.00	BDL	BDL	BDL	BDL	BDL
K	mg/L	0.95	0.95	0.81	0.92	1.22	1.01	1.61	3.33
Li	mg/L	0.01	0.01	0.02	0.01	0.01	0.01	0.01	0.02
Mg	mg/L	14.8	14.8	14.7	15.0	14.8	15.2	15.2	15.9
Mn	mg/L	0.10	0.10	0.02	0.05	0.00	0.00	0.06	0.07
Mo	mg/L	0.03	0.03	0.07	0.03	0.02	0.02	0.02	0.02
Na	mg/L	5.2	5.2	3.2	3.3	3.5	3.5	3.3	3.5
Ni	mg/L	BDL	BDL	0.007	0.006	0.008	0.004	0.006	0.009
P	mg/L	2.58	2.58	BDL	BDL	BDL	BDL	BDL	BDL
Pb	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
S	mg/L	15.8	15.8	7.6	7.7	7.8	7.9	7.8	8.2
Sb	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Se	mg/L	0.055	0.055	BDL	BDL	BDL	BDL	BDL	BDL
Si	mg/L	4.2	4.2	4.3	4.2	4.0	4.2	4.2	4.5
Sn	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Sr	mg/L	0.23	0.23	0.23	0.23	0.22	0.23	0.23	0.24
Ti	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Zn	mg/L	0.64	0.64	0.02	0.05	0.03	0.03	0.03	0.03
Sr	mg/L	0.24	0.24	BDL	0.2408	BDL	BDL	BDL	BDL
F	mg/L	2.7	2.7	NA	1.9	3.0	3.0	3.0	2.3
Cl	mg/L	1.4	1.4	1.6	1.4	1.7	1.6	1.5	1.5
Br	mg/L	0.031	0.031	0.029	0.089	0.020	0.023	0.024	0.041
NO3	mg/L	0.007	0.007	0.076	0.089	0.199	0.117	0.084	0.206
PO4	mg/L	0.04	0.04	0.10	0.11	0.22	0.72	0.05	0.08
SO4	mg/L	21.8	21.8	22.1	22.4	22.7	23.0	21.9	22.5

Table B.7: Concentrations of all analytes measured for the Blue Mountain ion exchange testing. One sample was taken 2-3 hours into each of the six cycles.

Sample		Feed	Cycle 1	Cycle 2	Cycle 3	Cycle 4	Cycle 5	Cycle 6
Date		8/10/08	8/11/08	8/12/08	8/13/08	8/14/08	8/15/08	8/18/08
Gross Alpha	pCi/L	41	30.0	37.0	42.0	33.0	38.0	36.0
Gross Beta	pCi/L	25	16.0	11.0	12.0	14.0	10.0	12.0
Ra-228	pCi/L	BDL	NA	NA	NA	NA	NA	NA
Ra-226	pCi/L	3.1	< 0.4	< 0.2	< 0.2	< 0.1	< 0.1	< 0.1
TOC	mg/L	0.55	1.43	0.79	0.79	0.88	0.81	0.86
Ag	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Al	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
As	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
B	mg/L	0.08	0.04	0.17	0.06	0.06	0.06	0.06
Ba	mg/L	0.092	BDL	0.001	0.001	BDL	BDL	BDL
Be	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Ca	mg/L	43.84	0.02	0.12	0.17	0.10	0.09	0.12
Cd	mg/L	BDL	BDL	0.001	BDL	BDL	BDL	BDL
Co	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Cr	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Cu	mg/L	0.26	0.35	BDL	0.35	0.49	0.45	0.54
Fe	mg/L	BDL	0.004	0.004	0.0040	BDL	BDL	BDL
K	mg/L	5.95	BDL	1.04	0.25	BDL	0.32	0.23
Li	mg/L	0.080	BDL	0.004	BDL	0.006	0.002	0.003
Mg	mg/L	26.24	0.00	0.04	0.05	0.04	0.03	0.03
Mn	mg/L	0.030	0.019	0.035	0.015	0.024	0.006	0.015
Mo	mg/L	0.045	0.022	0.106	0.033	0.033	0.038	0.037
Na	mg/L	16.2	78.7	105.8	101.9	102.6	109.5	108.6
Ni	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
P	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Pb	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
S	mg/L	14.7	3.7	9.9	9.8	11.4	11.9	12.9
Sb	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Se	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Si	mg/L	4.3	2.9	3.8	3.5	3.5	3.6	3.7
Sn	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Sr	mg/L	0.713	BDL	0.001	0.001	0.000	0.000	0.000
Ti	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
V	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Zn	mg/L	0.050	0.004	0.006	0.003	0.003	0.004	0.002
Sr	mg/L	0.69	BDL	BDL	BDL	BDL	BDL	BDL
F	mg/L	0.47	0.50	0.50	0.49	0.48	0.47	0.46
Cl	mg/L	1.5	1.6	1.9	1.6	1.6	1.6	1.8
Br	mg/L	0.04	0.03		0.04	0.05	0.03	0.03
NO3	mg/L	0.35	0.47	0.49	0.84	0.54	0.47	0.25
PO4	mg/L	0.07	0.09	BDL	BDL	0.16	BDL	BDL
SO4	mg/L	45.3	11.8	29.4	30.9	37.8	38.0	40.2

Table B.8: Concentrations of all analytes measured for the Redhill Forest HMO testing. Two doses (0.5 mg/L and 1.0 mg/L) were tested in duplicate.

Sample		Feed	0.5 A	0.5 B	1.0 A	1.0 B
HMO Dose	mg/L		0.5	0.5	1.0	1.0
Date		8/30/2008	8/31/2008	8/31/2008	8/31/2008	8/31/2008
Gross Alpha	pCi/L	97	21	20	15	8
Gross Beta	pCi/L	49	10	14	10	7
Ra-228	pCi/L	2.3	NA	NA	NA	NA
Ra-226	pCi/L	31	7.7	7.1	4.4	3.0
TOC	mg/L	0.98	1.26	0.98	1.04	1.13
Ag	mg/L	BDL	BDL	BDL	BDL	BDL
Al	mg/L	0.03	0.05	0.02	BDL	0.04
As	mg/L	BDL	BDL	BDL	BDL	BDL
B	mg/L	0.03	0.03	0.03	0.03	0.03
Ba	mg/L	0.02	0.02	0.02	0.01	0.01
Be	mg/L	BDL	BDL	BDL	BDL	BDL
Ca	mg/L	59.5	61.0	60.9	58.5	60.2
Cd	mg/L	BDL	BDL	BDL	BDL	BDL
Co	mg/L	0.013	0.009	0.008	BDL	BDL
Cr	mg/L	BDL	BDL	BDL	BDL	BDL
Cu	mg/L	0.006	0.009	0.006	0.005	0.007
Fe	mg/L	0.03	BDL	BDL	BDL	BDL
K	mg/L	2.7	3.3	3.4	2.7	3.4
Li	mg/L	0.015	0.016	0.018	0.015	0.016
Mg	mg/L	13.6	13.9	13.8	13.4	13.8
Mn	mg/L	0.40	0.32	0.30	0.21	0.21
Mo	mg/L	0.007	0.007	0.006	0.006	0.006
Na	mg/L	8.0	7.8	11.9	8.0	8.0
Ni	mg/L	0.012	0.013	0.014	0.013	0.015
P	mg/L	BDL	BDL	BDL	BDL	BDL
Pb	mg/L	BDL	BDL	BDL	BDL	BDL
S	mg/L	48.0	47.0	51.6	45.7	48.8
Sb	mg/L	BDL	BDL	BDL	BDL	BDL
Se	mg/L	BDL	BDL	BDL	BDL	BDL
Si	mg/L	3.7	3.7	3.9	3.6	3.7
Sn	mg/L	BDL	BDL	BDL	BDL	BDL
Sr	mg/L	0.36	0.37	0.37	0.36	0.37
Ti	mg/L	BDL	BDL	BDL	BDL	BDL
V	mg/L	BDL	BDL	BDL	BDL	BDL
Zn	mg/L	0.30	0.27	0.23	0.20	0.22
Sr	mg/L	0.37	0.39	0.39	0.37	0.38
F	mg/L	0.22	0.22	0.22	0.22	0.22
Cl	mg/L	11.2	11.4	11.3	11.3	11.2
Br	mg/L	0.17	0.18	0.66	0.19	0.17
NO3	mg/L	0.15	0.20	0.21	0.30	0.11
PO4	mg/L	0.080	BDL	BDL	0.029	BDL
SO4	mg/L	146.3	146.9	150.5	148.1	145.1

Table B.9: Concentrations of all analytes measured for the Blue Mountain ion exchange testing. One sample was taken 2-3 hours into each of the six cycles.

Sample		Feed	Cycle 1	Cycle 2	Cycle 3	Cycle 4	Cycle 5	Cycle 6
Date		8/30/08	9/3/08	9/4/08	9/5/08	9/8/08	9/9/08	9/10/08
Gross Alpha	pCi/L	97	< 3	< 3	3.0	< 3	3.0	3.0
Gross Beta	pCi/L	49	< 4	< 4	< 4	< 4	< 4	< 4
Ra-228	pCi/L	2.3	NA	NA	NA	NA	NA	NA
Ra-226	pCi/L	31	< 0.3	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2
TOC	mg/L	0.96	1.16	0.93	0.90	0.80	0.90	0.82
Ag	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Al	mg/L	0.0272	BDL	BDL	BDL	BDL	BDL	BDL
As	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
B	mg/L	0.03	0.03	0.03	0.03	0.03	0.07	0.06
Ba	mg/L	0.02	BDL	BDL	BDL	BDL	BDL	BDL
Be	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Ca	mg/L	59.4	0.3	0.1	0.4	0.1	0.4	0.1
Cd	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Co	mg/L	0.01	BDL	BDL	BDL	BDL	BDL	BDL
Cr	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Cu	mg/L	0.006	0.007	0.009	0.011	0.010	BDL	BDL
Fe	mg/L	0.03	0.03	0.03	0.03	0.02	0.02	0.02
K	mg/L	2.68	0.71	0.53	0.57	0.39	0.92	0.90
Li	mg/L	0.02	BDL	BDL	BDL	BDL	BDL	BDL
Mg	mg/L	13.56	0.07	0.04	0.02	0.02	0.07	0.02
Mn	mg/L	0.39	BDL	BDL	BDL	BDL	0.001	0.000
Mo	mg/L	0.006	0.004	0.005	BDL	BDL	0.013	0.008
Na	mg/L	7.1	103.1	104.2	105.8	102.5	104.7	103.2
Ni	mg/L	0.0116	BDL	BDL	BDL	BDL	BDL	BDL
P	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Pb	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
S	mg/L	48.3	47.7	47.7	47.5	47.8	47.5	47.6
Sb	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Se	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Si	mg/L	3.64	3.48	3.47	3.53	3.41	3.59	3.45
Sn	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Sr	mg/L	0.36	0.002	0.000	0.000	0.001	0.002	BDL
Ti	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
V	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Zn	mg/L	0.32	0.005	0.007	0.001	0.003	0.004	0.010
Sr	mg/L	0.38	BDL	BDL	BDL	BDL	BDL	BDL
F	mg/L	0.22	0.22	0.22	0.22	0.22	0.21	0.22
Cl	mg/L	11.3	11.7	11.4	11.5	12.5	11.6	11.5
Br	mg/L	0.19	0.18	0.19	0.18	0.17	0.17	0.18
NO3	mg/L	0.14	0.29	0.10	0.25	0.31	0.20	0.06
PO4	mg/L	BDL	BDL	0.10476	BDL	BDL	BDL	BDL
SO4	mg/L	145.7	146.9	148.7	146.3	146.3	147.5	146.9

Table B.10: Concentrations of all analytes measured for the Sterling pilot-scale membrane testing. The Koch TFC-S and Dow NF 4040 membrane was tested at recoveries of 30% and 80%. 'NF' samples represent samples from the Dow NF 4040; 'TS' samples represent samples from the Koch TFC-S.

Sample		Feed	NF 30 - Perm Com	NF 80 - Perm Com	TS 30 - Perm Com	TS 80 - Perm Com	TS 80 - Perm 1
Date		7/25/2008	7/25/2008	7/25/2008	7/25/2008	7/25/2008	7/25/2008
Gross Alpha	pCi/L	14	< 3	< 3	< 4	<3	< 4
Gross Beta	pCi/L	NA	< 4	5	< 2	4	< 2
Uranium	μg/L	44	< 1	< 1	< 1	2	< 1
TOC	mg/L	3.08	BDL	BDL	BDL	BDL	BDL
Ag	mg/L	BDL	BDL	BDL	BDL	BDL	BDL
Al	mg/L	0.01	BDL	BDL	BDL	BDL	BDL
As	mg/L	BDL	BDL	BDL	BDL	BDL	BDL
B	mg/L	0.25	0.23	0.25	0.21	0.23	0.23
Ba	mg/L	0.02	BDL	BDL	BDL	BDL	BDL
Be	mg/L	BDL	BDL	BDL	BDL	BDL	BDL
Ca	mg/L	162.1	3.7	10.3	2.8	6.5	5.4
Cd	mg/L	BDL	BDL	BDL	BDL	BDL	BDL
Co	mg/L	BDL	BDL	BDL	BDL	BDL	BDL
Cr	mg/L	BDL	BDL	BDL	BDL	BDL	BDL
Cu	mg/L	0.04	0.05	0.05	0.06	0.05	0.06
Fe	mg/L	BDL	BDL	BDL	BDL	BDL	BDL
K	mg/L	14.7	2.4	5.1	1.4	3.1	2.8
Li	mg/L	0.05	0.01	0.02	0.01	0.01	0.01
Mg	mg/L	48.9	0.9	1.9	1.0	2.3	1.9
Mn	mg/L	BDL	BDL	BDL	BDL	BDL	BDL
Mo	mg/L	BDL	BDL	BDL	BDL	BDL	BDL
Na	mg/L	155.0	28.9	59.1	18.8	38.4	33.5
Ni	mg/L	BDL	BDL	BDL	BDL	BDL	BDL
P	mg/L	0.14	0.02	0.05	0.03	BDL	BDL
Pb	mg/L	BDL	BDL	BDL	BDL	BDL	BDL
S	mg/L	196.5	1.1	0.8	4.2	9.5	8.2
Sb	mg/L	BDL	BDL	BDL	BDL	BDL	BDL
Se	mg/L	0.02	0.02	0.02	0.02	0.02	0.00
Si	mg/L	14.8	5.1	9.2	1.9	3.6	3.1
Sn	mg/L	BDL	BDL	BDL	BDL	BDL	BDL
Sr	mg/L	1.67	0.04	0.09	0.03	0.07	0.06
Ti	mg/L	BDL	BDL	BDL	BDL	BDL	BDL
V	mg/L	BDL	BDL	BDL	BDL	BDL	BDL
Zn	mg/L	BDL	BDL	BDL	BDL	bb	BDL
Sr	mg/L	1.68	0.04	0.09	0.04	0.07	0.06
F	mg/L	0.77	0.05	0.16	0.06	0.13	0.11
Cl	mg/L	102.5	29.2	65.4	14.4	32.4	27.9
Br	mg/L	0.41	0.14	0.28	0.23	0.26	0.24
NO3	mg/L	27.2	19.6	27.8	10.6	17.7	15.6
PO4	mg/L	0.11	BDL	0.45	BDL	0.02	0.03
SO4	mg/L	654.5	3.5	3.1	15.2	31.5	26.5

Table B.11: Concentrations of all analytes measured for the Sterling bench-scale coagulation testing. Three doses of two different coagulants were tested on feed water at natural pH (7.5)

Sample		Feed	Ferric A	Ferric B	Ferric C	Alum A	Alum B	Alum C
Dose	mg/L		5	10	20	5	10	20
Date		6/26/2008	6/27/2008	6/27/2008	6/27/2008	6/27/2008	6/27/2008	6/27/2008
Gross Alpha	pCi/L	27	33	26	23	32	20	5
Uranium	μg/L	47	51	47	30	38	26	6
TOC	mg/L	3.07	3.38	3.48	2.78	3.22	2.98	2.52
Ag	mg/L	BDL	BDL	BDL	0.005	0.004	0.005	0.004
Al	mg/L	0.044	BDL	BDL	BDL	0.091	0.065	0.027
As	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
B	mg/L	0.32	0.19	0.18	0.21	0.19	0.18	0.19
Ba	mg/L	0.038	0.033	0.031	0.030	0.035	0.031	0.031
Be	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Ca	mg/L	161	143	141	142	142	141	142
Cd	mg/L	0.003	0.003	0.002	0.003	0.003	0.002	0.002
Co	mg/L	BDL	BDL	0.007	0.006	BDL	BDL	BDL
Cr	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Cu	mg/L	0.008	0.011	0.010	0.009	0.011	0.014	0.013
Fe	mg/L	BDL	0.007	0.037	0.158	0.003	BDL	0.010
K	mg/L	17.8	10.2	9.0	12.4	11.5	9.7	10.4
Li	mg/L	0.051	0.042	0.040	0.043	0.043	0.041	0.040
Mg	mg/L	60	54	53	53	53	53	53
Mn	mg/L	0.001	0.019	0.025	0.027	0.045	0.059	0.094
Mo	mg/L	0.011	0.022	0.017	0.014	0.014	0.015	0.011
Na	mg/L	159	142	140	142	141	140	140
Ni	mg/L	0.006	0.007	0.008	0.008	0.010	0.007	0.014
P	mg/L	0.354	BDL	BDL	1.37	0.51	0.17	0.26
Pb	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
S	mg/L	226	207	210	232	214	220	241
Sb	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Se	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Si	mg/L	9.4	7.8	7.3	7.0	7.3	6.8	6.9
Sn	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Sr	mg/L	1.6	1.4	1.4	1.4	1.4	1.4	1.4
Ti	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
V	mg/L	0.007	BDL	BDL	BDL	0.004	0.005	0.003
Zn	mg/L	0.031	0.020	0.021	0.072	0.040	0.023	0.037
Sr	mg/L	1.6	1.4	1.4	1.4	1.4	1.4	1.4
F	mg/L	0.85	0.80	0.00	0.78	0.06	0.44	0.20
Cl	mg/L	85.8	86.2	85.4	83.8	85.4	85.8	85.4
Br	mg/L	0.59	0.57	0.61	0.56	0.59	0.59	0.59
NO3	mg/L	7.9	8.0	8.3	8.0	8.0	7.9	8.0
PO4	mg/L	0.33	0.09	0.10	0.04	0.21	0.10	0.05
SO4	mg/L	612	624	642	679	648	679	733

Table B.12: Concentrations of all analytes measured for the Sterling bench-scale coagulation testing. Three doses of two different coagulants were tested on feed water with an adjusted pH (6.2)

Sample		Feed	Ferric A	Ferric B	Ferric C	Alum A	Alum B	Alum C
Dose	mg/L		5	10	20	5	10	20
Date		6/26/2008	6/30/2008	6/30/2008	6/30/2008	6/30/2008	6/30/2008	6/30/2008
Gross Alpha	pCi/L	27	29.0	28	NA	14	5	12
Uranium	μg/L	47	47	44	NA	19	6	18
TOC	mg/L	3.07	3.08	3.00	2.31	2.86	3.37	2.44
Ag	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Al	mg/L	0.044	0.044	0.021	0.029	0.043	0.024	0.428
As	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
B	mg/L	0.32	0.24	0.24	0.24	0.26	0.24	0.24
Ba	mg/L	0.038	0.037	0.035	0.034	0.038	0.035	0.036
Be	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Ca	mg/L	161	158	146	150	159	144	148
Cd	mg/L	0.003	0.003	0.003	0.002	0.003	0.003	0.003
Co	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Cr	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Cu	mg/L	0.008	0.466	0.450	0.447	1.202	0.574	0.493
Fe	mg/L	BDL	0.003	0.006	0.014	BDL	0.003	0.015
K	mg/L	17.8	14.9	13.0	13.9	15.3	12.7	13.5
Li	mg/L	0.051	0.048	0.044	0.046	0.049	0.044	0.045
Mg	mg/L	60	57	53	55	58	52	54
Mn	mg/L	0.001	0.021	0.066	0.144	0.005	0.006	0.005
Mo	mg/L	0.011	0.009	0.006	BDL	0.010	0.008	0.004
Na	mg/L	159	161	150	154	163	151	151
Ni	mg/L	0.006	0.008	0.005	0.008	0.246	0.018	0.006
P	mg/L	0.354	BDL	BDL	BDL	BDL	BDL	BDL
Pb	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
S	mg/L	226	212	198	224	227	215	239
Sb	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Se	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Si	mg/L	9.4	9.6	8.9	8.5	9.5	8.5	8.6
Sn	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Sr	mg/L	1.6	1.6	1.5	1.5	1.6	1.5	1.5
Ti	mg/L	BDL	BDL	BDL	BDL	BDL	BDL	BDL
V	mg/L	0.007	BDL	BDL	BDL	0.004	0.002	BDL
Zn	mg/L	0.031	0.029	0.040	0.048	0.221	0.140	0.041
Sr	mg/L	1.6	1.6	1.5	1.5	1.6	1.5	1.5
F	mg/L	0.85	0.77	0.80	0.74	0.50	0.30	0.40
Cl	mg/L	85.8	160	164	164	167	167	167
Br	mg/L	0.59	0.59	0.62	0.61	0.60	0.60	0.62
NO3	mg/L	7.9	8.4	9.1	8.2	7.8	8.0	7.9
PO4	mg/L	0.33	BDL	BDL	BDL	BDL	BDL	BDL
SO4	mg/L	612	618	623	685	648	697	727

APPENDIX C: ION EXCHANGE PERFORMANCE MONITORING RESULTS

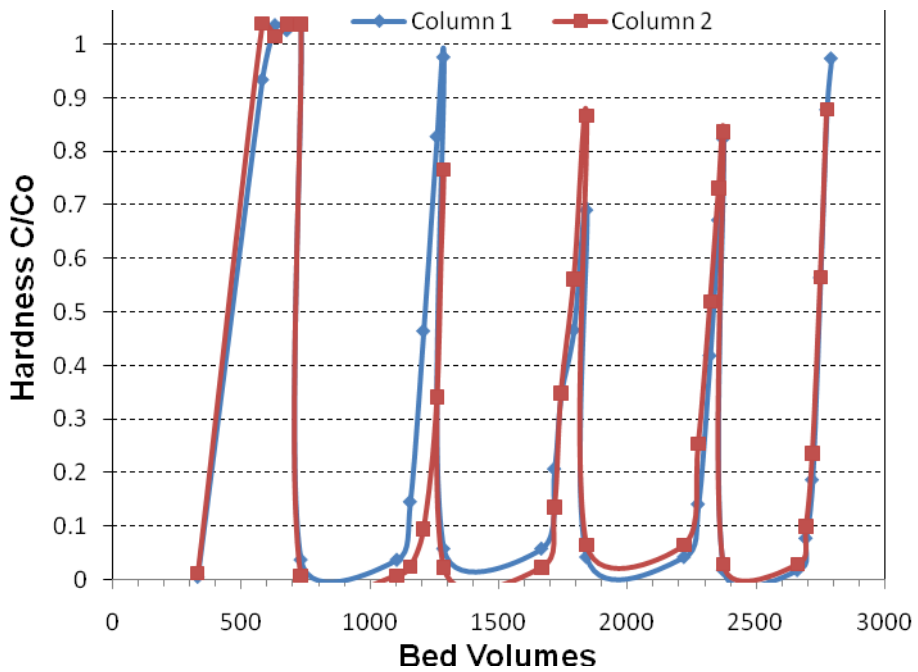

Figure C.1: Hardness relative concentration as a function of BVs for cycles 1 through 5 during ion exchange performance monitoring testing of Blue Mountain groundwater. Two columns were operated in parallel for the duration of the five cycles. Samples for hardness analysis were collected every 30-50 BVs.