



Final Report to:
Albuquerque Bernalillo County Water Utility Authority

Independent Review of Key Water Quality and Treatment Issues for the San Juan-Chama Drinking Water Project Water Treatment Plant

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List of Abbreviations

ABCWUA	Albuquerque Bernalillo County Water Utility Authority
AWWARF	American Water Works Association Research Foundation
BAT	best available technology
BOD	biochemical oxygen demand
CCL	contaminant candidate list
COD	chemical oxygen demand
DBP	disinfection by-product
DBPR	disinfection by-product rule
DO	dissolved oxygen
DOC	dissolved organic carbon
DWEL	drinking water equivalent level
EPA	United States Environmental Protection Agency
GAC	granular activated carbon
HAA	haloacetic acid
ICPMS	inductively-coupled plasma mass spectrometry
NOM	natural organic matter
LANL	Los Alamos National Laboratory
LKI	Larson-Skold Index
LSI	Langelier Saturation Index
MCL	maximum contaminant level
mgd	million gallons per day
MRDL	maximum residual disinfectant level
MWRA	Massachusetts Water Resources Authority
NMED	New Mexico Environment Department
NOM	natural organic matter
NPDES	National Pollutant Discharge Elimination System
NRC	National Research Council
NTU	nephelometric turbidity units
NWQD	USGS National Water Quality Database
PPCP	pharmaceuticals and personal care products
RfD	reference dose
RSI	Ryznar Stability Index
SDWA	Safe Drinking Water Act
SOC	synthetic organic chemical
SUVA	specific ultraviolet absorbance
SWRP	Southside Water Reclamation Plant
THM	trihalomethane
TOC	total organic carbon
TSS	total suspended solids
UNM	University of New Mexico

USGS United States Geological Survey
UV₂₅₄ ultraviolet light absorbance at a wavelength of 254 nm
WTP water treatment plant
WWTP waste water treatment plant

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Introduction

The Albuquerque Bernalillo County Water Utility Authority (ABCWUA) has completed construction and startup of a water treatment plant to treat San Juan-Chama Project water diverted from the Rio Grande. The San Juan-Chama Drinking Water Project Water Treatment Plant (WTP) is a state-of-the-art facility that will deliver treated water to a distribution system that has historically supplied local groundwater to the metropolitan area. The overall project is one of the largest public works projects in the history of New Mexico and will supply up to 90 percent of the metropolitan area's future water. The implementation of a large public works project, coupled with the introduction of a new source water into a community, can present a number of challenges. The most important of these is to insure that public health and safety is sustained during the transition from one water supply to the other.

The new treatment facility was designed by one of the leading environmental engineering design firms in the United States. The selection of the process train and design criteria for the plant was essentially complete in 2003. Since then, regulatory activity has continued, the Water Utility Authority has operated a pilot plant to generate treatment information, and additional sampling of river water for contaminants has taken place. To assess whether any of this recent activity or information provides new insight on the ability of the San Juan-Chama WTP to meet all Safe Drinking Water Act (SDWA) regulations, the Water Utility Authority requested an independent third-party review of key water quality and treatment issues. This report is the outcome of that review. The issues addressed in this report include the following:

1. The potential for disinfection by-product formation in the distribution system.
2. The impact of upstream wastewater discharges on source water quality.
3. The occurrence and treatability of radiological contaminants in the source water.
4. The occurrence and treatability of perchlorate in the source water.
5. The potential for river water to cause corrosion in the distribution system.

For each issue, the report reviews existing regulatory requirements for the contaminants of concern, known information about the presence of the contaminants (or chemical precursors) in the source water, and capabilities for the treatment plant to remove (or prevent the formation) of the contaminants. Each section ends with a summary and recommendations for addressing the issue, if necessary.

Disinfection By-product Formation

Disinfection by-products (DBPs) are formed when disinfectants react with natural compounds in source waters. DBPs have been regulated in federal drinking water regulations for nearly 30 years. The San Juan-Chama WTP will be required to meet the current DBP regulations. This section of the report summarizes the regulatory requirements for DBPs in the SDWA, existing information on the presence of DBP precursors in Rio Grande water, and the ability of the new plant to effectively prevent the formation of DBPs.

Drinking Water Regulations for Disinfectants and DBPs

Drinking water distributed by public water systems is regulated by the United States Environmental Protection Agency (EPA) and by the New Mexico Environment Department (NMED) in the state of New Mexico. Federal regulations for disinfection by-products have evolved over the past 30 years, starting with a maximum contaminant level (MCL) for trihalomethanes that was promulgated in 1979. However, balancing the risks between adequate disinfection to prevent acute pathogenic illness and minimizing the formation of DBPs has been challenging, particularly since knowledge about both microbial disease and DBPs has continued to grow the entire time. The Stage 1 Disinfectants and Disinfection By-Products Rule (Stage 1 DBPR) was promulgated in December 1998 and the Stage 2 DBPR was promulgated in January 2006 (U.S. EPA, 1998; U.S. EPA, 2006). The current maximum residual disinfectant level (MRDLs) for disinfectants and MCLs for DBPs are shown in Table 1. In addition, treatment facilities must meet a treatment technique called enhanced coagulation for the removal of DBP precursors. The treatment technique can be met a variety of ways but the main way is by achieving specific removal of total organic carbon (TOC). The required TOC removal depends on the source water TOC concentration and the alkalinity, as shown in Table 2.

Of the DBPs listed in Table 1, the San Juan-Chama WTP will have to meet the requirements for trihalomethanes (THMs), haloacetic acids (HAAs), and bromate. Additional monitoring will be required for the TOC removal treatment technique. The initial sampling requirements are shown in Table 3. Reduced monitoring requirements apply if certain criteria are met. Research suggests that systems with a raw water bromide concentration less than 0.05 mg/L are not expected to violate the bromate MCL. As a result, utilities can reduce bromate monitoring from monthly to quarterly if the average annual raw water bromide concentration is less than 0.05 mg/L.

Existing DBP Concentrations in the Distribution System

The current Water Utility Authority water supply system relies on wells. Well water is chlorinated, stored in reservoirs, and delivered to the distribution system. Storage provides reaction time for DBP formation. The THM and HAA concentrations from the three most recent Water Quality Reports are shown in Table 4. The DBP concentrations are well below the MCL values, so the utility has had no difficulty meeting the DBP regulations during the past 3 years.

Table 1 – Current Disinfectant and DBP Regulations.

Disinfectants	Disinfectant MRDL (mg/L)
Chlorine	4 as Cl ₂
Chloramines	4 as Cl ₂
Chlorine dioxide	0.8
Disinfection By-Products	Disinfection By-Product MCL (µg/L)
Total trihalomethanes (THMs)	80
Haloacetic acids (HAA ₅)	60
Bromate	10
Chlorite	1000

Ref: (U.S. EPA, 1998)

Table 2 – Step 1 Required Percent Removal of TOC.

Source water TOC (mg/L)	Source Water Alkalinity, mg/L as CaCO ₃		
	0-60	> 60 - 120	> 120
> 2.0 to 4.0	35.0 %	25.0 %	15.0 %
> 4.0 to 8.0	45.0 %	35.0 %	25.0 %
> 8.0	50.0 %	40.0 %	30.0 %

Ref: (U.S. EPA, 1999)

Table 3 – Monitoring and Compliance Requirements for DBPs.

Parameter	Frequency	Locations	Compliance method
THMs and HAAs	Quarterly	Distribution System	LRAA
Bromate	Monthly	EPTDS	RAA
TOC	Monthly	Source water and filter effluent	Monthly
Alkalinity	Monthly	Source water	N/A

Ref: (U.S. EPA, 2006)

Table 4 – THM and HAA Concentrations in the Existing Water Supply System.

Parameter	THM concentration (µg/L)	HAA concentration (µg/L)
MCL	80	60
Average measured concentration		
2005	8	3
2006	11	4
2007	11	2
Maximum measured concentration		
2005	18	18
2006	27	5
2007	32	5

Ref: (ABCWUA, 2007)

Treatment for TOC Removal and Prevention of DBP Formation

The San Juan-Chama WTP was designed with several processes to minimize DBP formation. First, the plant is capable of enhanced coagulation, which is listed by the EPA as the best available technology (BAT) for DBPs. The enhanced coagulation system is capable of coagulating the water with ferric chloride at a pH of about 6, which is in the range of pH for optimal TOC removal. Preliminary studies by the design engineer, CH2M-Hill, showed that enhanced coagulation was able to achieve 35 percent TOC removal at an alum dose of 40 mg/L as alum and a pH = 6 (CH2M Hill, 2003). Based on source water alkalinity and TOC measurements shown later in this report, the maximum TOC removal that the San Juan-Chama WTP will be required to achieve with enhanced coagulation should be 35 percent. The most challenging water treated during the CH2M-Hill treatability testing had raw water TOC of 4.59 mg/L. When the TOC concentration is higher, it is expected that more of the TOC will be in particulate form and easy to remove by coagulation/filtration, so the TOC percent removal should be higher when the raw water TOC is higher.

The second process designed into the San Juan-Chama WTP to remove DBP precursors is the ozone/biofiltration process. The plant includes ozone with a maximum applied dose of 1.5 mg/L, a contact time in excess of 10 minutes, and an ozonation pH of around 6. The plant also includes granular activated carbon (GAC) filters that act as biological filters. In addition to providing disinfection and adsorption capabilities in the plant, these processes can also provide additional TOC removal. Studies have shown between 15 and 35 percent TOC removal by ozone and biological filtration (Hozalski et al., 1995; Wang et al., 1995; Emelko et al., 2006). The CH2M-Hill Design Analysis Report indicated that the El Paso WTP achieves 25 percent TOC removal through the ozone and biological filtration processes. As a result, it should be possible for the San Juan-Chama WTP to achieve 50 percent TOC removal or greater with the existing combination of treatment processes.

Raw and Treated Water Quality Analyses

The raw water quality parameters that are most significant with respect to the formation of DBPs are total organic carbon for the formation of THMs and HAAs and bromide for formation of bromate. The raw water alkalinity is necessary for determining the required TOC removal in the enhanced coagulation regulations.

Total organic carbon in source water. TOC data were retrieved from the on-line United States Geological Survey (USGS) National Water Quality Database (NWQD) for several sampling locations between Cochiti Reservoir and Albuquerque (USGS, 2008). The results of over 100 analyses from this data set are shown in Figure 1. The long-term trend is that the TOC concentration in the river typically ranges between 2 and 6 mg/L, and only seven samples above 8 mg/L were reported. No samples were reported as being below the detection limit. Four samples from the USGS database are not shown in Figure 1 because they exceed the range of the graph. Three of the samples not shown were taken at the Alameda Bridge sample location, including a sample on 15 August 2006 that was reported as 92.9 mg/L. It is likely that the high TOC samples are associated with high suspended sediment concentrations in the river, and that the excess organic carbon is particulate rather than dissolved. All three high TOC samples at the Alameda Bridge had suspended sediment concentrations above 1,000 mg/L, and the sample on

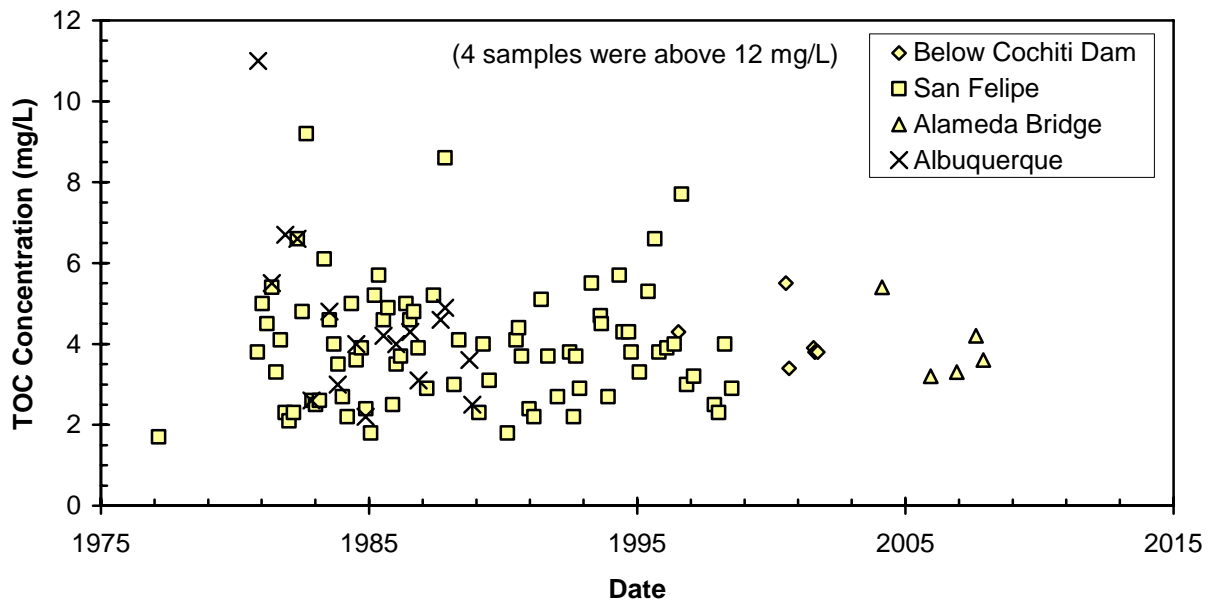


Figure 1 – Total Organic Carbon in Rio Grande Water (Historical Data from the USGS Water Quality Database).

15 August 2006 had a suspended sediment concentration of 17,400 mg/L. The suspended sediment concentrations support the conclusion that the excess organic carbon detected in these samples is probably in particulate form. Particulate TOC can effectively be removed by the coagulation/filtration process, and the range of 2 to 6 mg/L of TOC can be considered the typical range for water entering the treatment plant. Excluding the values above 8 mg/L, the average measured TOC in the USGS data set was 3.9 mg/L.

The Water Utility Authority operated a pilot plant during 2007 to confirm that the treatment process could successfully treat the river water and meet all federal and state drinking water regulations and to produce water for bottling as part of a public awareness campaign. All SDWA water quality parameters were measured for every run of the pilot that produced water for bottling operations, producing an extensive database of raw and finished water quality data with over 30 sampling episodes spread over the course of an entire year (ABCWUA, 2008b). Total organic carbon was measured during each episode, as shown in Figure 2. The TOC concentration in the source water generally ranged from 2 to 4 mg/L during 2007. The data shows a distinct annual trend, with TOC at the lower end of the range during winter months and at the higher end of the range in about June. In addition, a couple of the samples had TOC concentrations substantially higher than the general trend. It is possible that these spikes were associated with higher turbidity in the river, as noted for the USGS data. The turbidity for the two samples in March with high TOC are reported as “>100 NTU” (Nephelometric Turbidity Units) in the pilot data, indicating that it is unknown how high the turbidity was, but that it was clearly higher than in many of the other pilot runs. Thus, the TOC results during the pilot study operation were generally consistent with the long-term data record, although 2007 may have had somewhat lower TOC concentrations than the historical average.

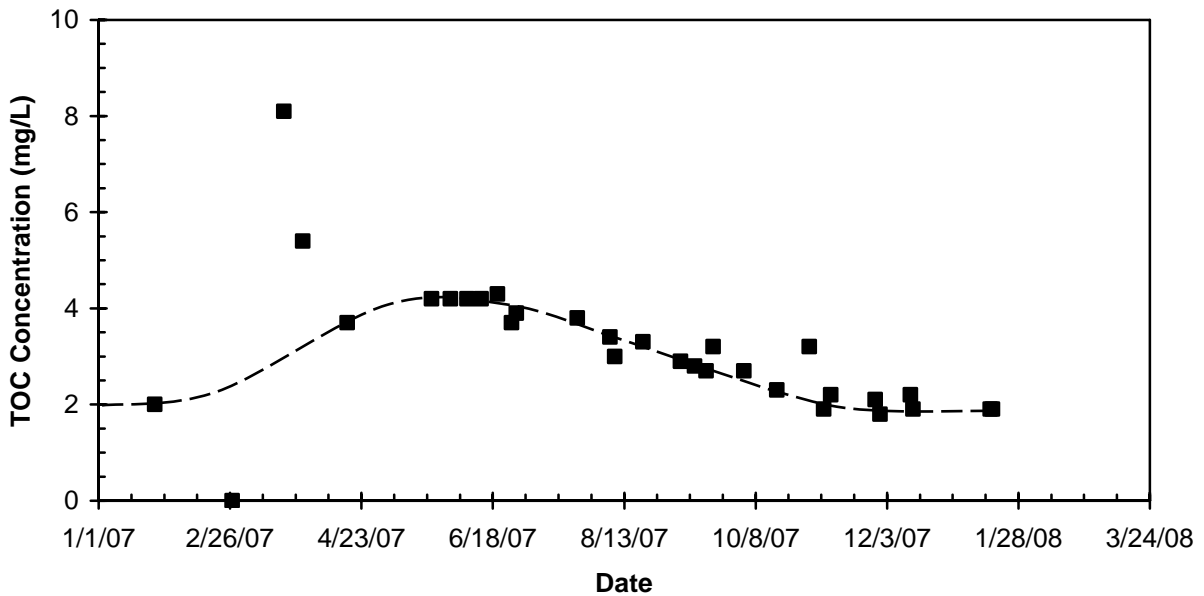


Figure 2 – Total Organic Carbon in the ABCWUA Pilot Plant Source Water.

Total organic carbon in pilot plant finished water. The TOC concentration was also measured in the finished water from the pilot plant during each pilot run. In nearly all cases, the TOC was reported to be below the method detection limit (1 mg/L). Only four samples had detectable TOC. These values indicate that the pilot plant frequently achieved TOC removal greater than 75 percent (source water TOC around 4 mg/L, and finished water TOC below the detection limit of 1 mg/L). However, this performance may not be representative of long-term operation because it may have been influenced by a TOC removal mechanism that may not be active in the full-scale plant—that of TOC adsorption on the GAC filter media. TOC can be removed from water by adsorption, but GAC has a relatively low adsorption capacity for TOC because of the large molecular weight of natural organic matter (NOM) compounds. Once the media is exhausted with respect to TOC adsorption, this removal mechanism is no longer effective. Thus, longer-term operation of the pilot plant may eventually have shown lower TOC removal.

Bromide in source water. Very little historical data exists for bromide concentration in the Rio Grande. Between Cochiti Dam and Albuquerque, only six samples for bromide were identified in the USGS database and all were at or below the detection limit. During pilot plant operation in 2007, bromide was measured on about 30 occasions. Nearly all were below the detection limit (0.02 mg/L), and the remaining samples were reported between 0.02 and 0.03 mg/L. With the low bromide concentration in the source water, this system is not expected to produce bromate above the MCL.

DBP formation during pilot operation. Disinfection by-products were measured in the finished water from the pilot plant during each pilot run. The highest total THM concentration

measured was 5.9 µg/L and the highest HAA concentration measured was 4.74 µg/L. Overall, the measured values are low compared to the MCLs. Bromate was analyzed on 13 occasions with 12 samples below the detection limit and the remaining sample at 5.4 µg/L (below the MCL of 10 µg/L).

Alkalinity in the source water. The raw water alkalinity was measured during pilot plant operation. The alkalinity is shown in Figure 3. Alkalinity ranged between about 90 and 180 mg/L as CaCO₃. Based on the TOC removal requirements presented in Table 2, the San Juan-Chama WTP would be in the second or third column with respect to TOC removal. The removal requirements for alkalinity between 60 and 120 mg/L as CaCO₃ is 25 percent removal if the TOC is below 4 mg/L and 35 percent removal if the TOC is between 4 and 8 mg/L. Lower TOC removal is required if the alkalinity is above 120 mg/L as CaCO₃. Thus, the maximum TOC removal required for the enhanced coagulation requirements for the San Juan-Chama WTP should be 35 percent.

Potential DBP Formation Based on Technical Literature

THM and HAA formation. Disinfection by-products form by chemical reactions between disinfectants and natural organic or inorganic matter in the source water. The pathways for DBP formation can be complex and can depend on the concentration and character of the organic matter, disinfectant concentration, temperature, pH, alkalinity, reaction time, and other factors. Thus, it is not possible to accurately predict the exact DBP concentrations that would be produced in a given situation. Nevertheless, significant research has revealed general relationships between total organic carbon and DBP formation, and models have been developed varying from simple linear regressions (Edzwald et al., 1985) to more complex models

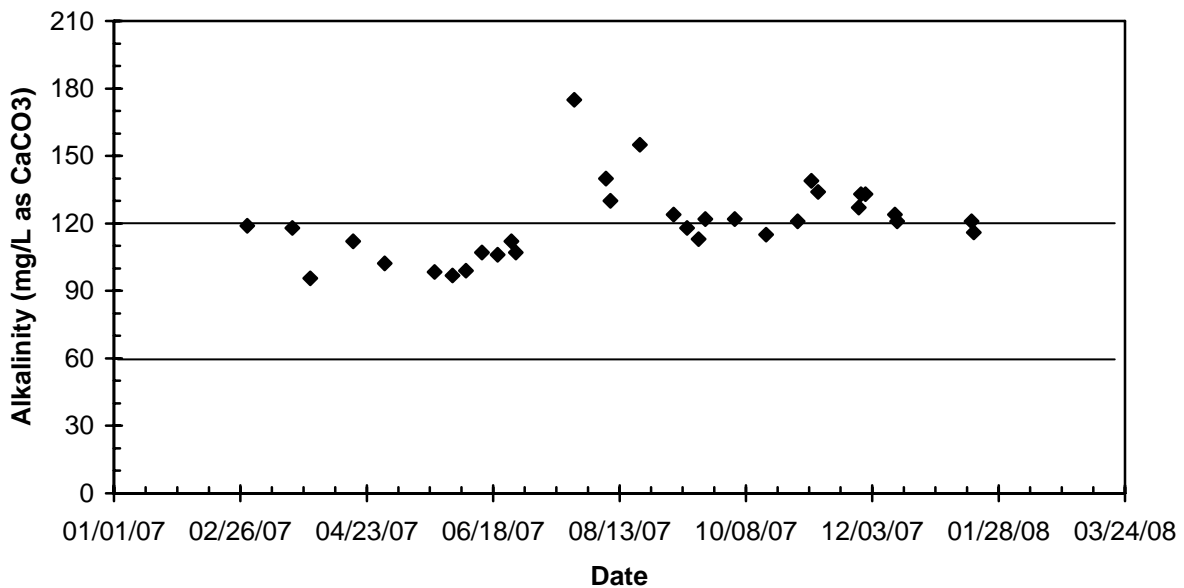


Figure 3 – Source Water Alkalinity during Pilot Plant Operation.

(Harrington et al., 1992). This research suggests that the TOC in the plant source water (typically 2 to 6 mg/L, as noted earlier) is high enough to cause THM and HAA formation to be above the MCLs and that control measures to prevent DBP formation are warranted, such as the use of enhanced coagulation and ozone/biofiltration as designed in the plant.

Bromate formation. Bromate formation is influenced by the bromide ion concentration, pH, ozone dose, temperature, alkalinity, and natural organic matter concentration. Studies have evaluated each of these factors and can be used to estimate the formation of bromate under conditions similar to the San Juan-Chama WTP. For instance, Siddiqui and Amy (1993) studied waters with bromide concentrations up to 1.5 mg/L, dissolved organic carbon (DOC) concentrations between 3 and 7 mg/L, ozone/DOC ratios up to 5 mg/mg, pH between 6.0 and 8.5, and temperatures between 20 and 30 °C. The most significant factors that increase bromate formation are higher pH, higher ozone dose, and higher bromide ion concentrations. In contrast, design and operation of the San Juan-Chama WTP will be characterized by relatively low pH values in the ozone contactor, low ozone doses, and very low bromide concentrations. Because the ozone contactor will be located after enhanced coagulation but before final pH adjustment, the pH during ozonation will be about 6.0. The maximum ozone dose the facility is designed for is 1.5 mg/L. With higher pH and ozone doses, Siddiqui and Amy (1993) found the threshold for detectable bromate formation occurs at about 0.4 mg/L of bromide ion concentration in the raw water. The highest bromide concentration measured in the river near the treatment facility intake was 0.03 mg/L, substantially lower than the concentration needed for formation of bromate that would exceed the MCL. Thus, lack of detectable bromate formation in the vast majority of samples collected during pilot operation is consistent with expectations based on previous literature on bromate formation during ozonation. In addition, the Stage 1 DBP Rule states that it is unlikely that a facility using ozone would exceed the bromate MCL if the source water bromide were below 0.05 mg/L on an average basis and allows reduced sampling if that condition is met. Based on the pilot plant results, not only would the San Juan-Chama WTP be able to meet the bromate MCL without difficulty, but it would also be eligible for reduced bromate sampling because of the low likelihood of bromate formation.

Simulated Distribution System Analyses

Because of the complexities involved in THM and HAA formation, site-specific studies are necessary for predicting the concentrations that will occur in the treated water from new treatment facilities. Since THMs and HAAs can continue to form in the distribution system because of slow chemical reactions, it is important to understand the concentrations that may occur in the distribution system, rather than just the concentration at the effluent from the surface water treatment plant. The simulated distribution system (SDS) test involves analyzing for DBPs after samples have been held in jars for a period of time at conditions (temperature, etc) that are representative of conditions in the distribution system. SDS tests were conducted by CH2M-Hill during the process design phase of the project using raw river water and by the Water Utility Authority in 2007 and 2008 using water produced by the pilot plant.

SDS Testing by CH2M-Hill during the Process Design Studies. CH2M-Hill performed a series of three bench-scale tests during the process design studies for the new treatment plant (CH2M Hill, 2003). CH2M-Hill concluded that the TOC needed to be reduced to about 1.5 mg/L to limit THM and HAA formation to goals of 40 µg/L and 30 µg/L, respectively. The

selection of processes (enhanced coagulation followed by ozone/biofiltration) was designed to meet these goals. In 2003, when the Design Analysis Report was prepared, the Stage 1 DBP Rule was effective but the Stage 2 DBP Rule had not yet been promulgated. Reduction of the MCLs for THMs and HAAs from the then-current levels of 80 µg/L and 60 µg/L to new levels of 40 µg/L and 30 µg/L were one of the actions under consideration in the development of the new regulation. Thus, the goals set in the Design Analysis Report were appropriate at the time. The Stage 2 DBP Rule that was eventually promulgated in 2006, however, did not lower the MCLs for THMs and HAAs, but maintained the 80 µg/L and 60 µg/L MCLs. The rule did require a different compliance calculation, relying on locational running annual averages rather than the previous running annual averages. The resulting regulation effectively applies stricter limits on THM and HAA formation without lowering the MCLs. As a result, the plant was designed to meet DBP goals that are lower than current regulations, providing a degree of conservatism in the design.

A limitation of the bench-scale studies performed by CH2M-Hill is that they used a reaction time of 48 hours. Evidence now suggests that reaction time in the distribution system could be much longer in peripheral areas (Gastian, personal communication). Thus, higher DBP formation may occur during actual full-scale operation than that predicted by the CH2M-Hill study, which may offset some of the conservatism in the process selection.

SDS Testing by the Water Utility Authority. On four occasions during the pilot plant operation in 2007, the Water Utility Authority conducted simulated distribution system (SDS) tests. The results of the four SDS tests are summarized in Table 5. Overall, the measured values are low compared to the MCLs. As noted earlier, pilot plant operation in 2007 achieved greater than 75 percent TOC removal. The high TOC removal may have resulted in low DBP formation. Thus, higher DBP formation may occur during actual full-scale operation than that predicted by the 2007 SDS tests.

To evaluate the impact that adsorption onto the GAC filter media had on DBP formation, the Water Utility Authority conducted additional SDS testing on samples collected from the pilot plant at the end of July 2008. The 2008 testing evaluated DBP formation in the effluent from the pilot plant ozone contactor as well as the pilot plant filters, so the difference in DBP formation could be attributed to changes in water quality through adsorption and biological filtration in the filter process. These results are shown in Figure 4. Over a 2-week period, the THM concentration in the effluent from the ozone contactors rose to about 72 µg/L, about 10 percent below the MCL of 80 µg/L. Over the same 2-week period, the THM concentration in the filter effluent rose to 28 µg/L. In Figure 4, it is evident that THM formation has reached its maximum after two weeks of reaction time.

Table 5 – Summary Results of SDS Testing.

Test	Start date	Maximum duration (hrs)	Maximum measured THMs (µg/L)	Maximum measured HAAs (µg/L)
1	5/23/07	168	7.5	0.6
2	5/31/07	168	10.1	1.3
3	6/6/07	24	10.3	1.5
4	10/31/07	4984	13.2	4.7

Ref: (ABCWUA, 2008a)

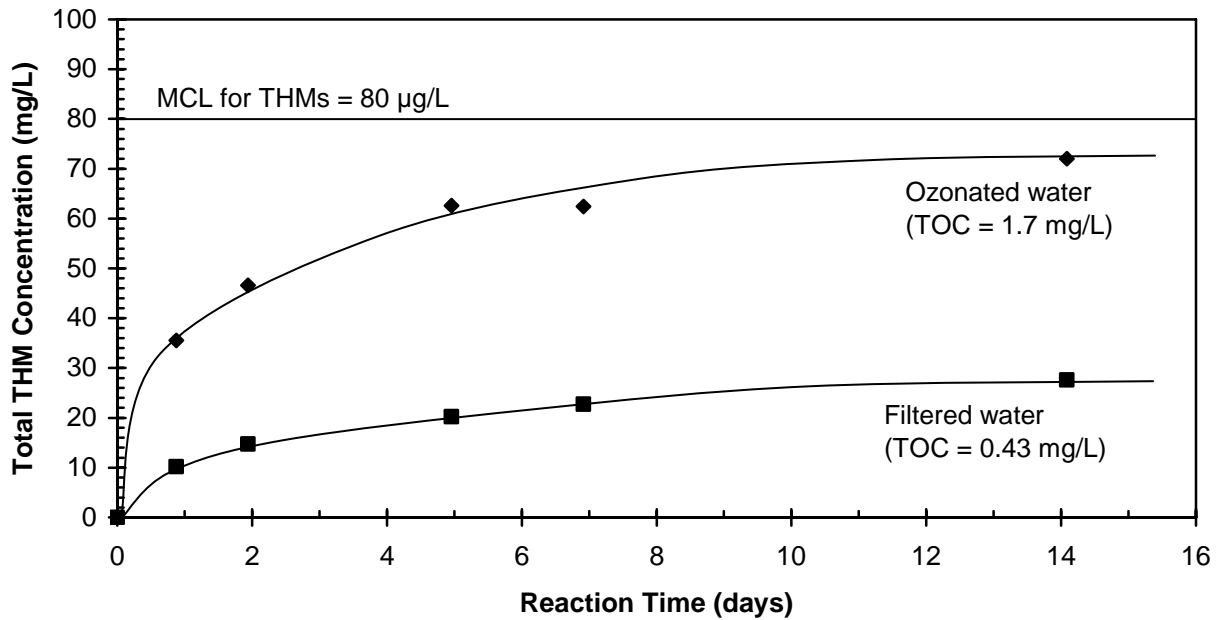


Figure 4 – THM Formation during SDS Tests in July, 2008

When the samples were collected, the measured TOC in the effluent from the pre-sedimentation basins was 3.4 mg/L, which is consistent with the TOC expected in the river during the summer, considering that some TOC removal will occur in the pre-sedimentation basins. The coagulation/sedimentation process achieved 40 percent TOC removal with a ferric chloride dose of 35 mg/L and the ozonation and biological filtration processes achieved an additional 47 percent TOC removal (total 87 percent TOC removal). The TOC removal during coagulation/sedimentation is consistent with previous testing and literature, but the TOC removal during ozonation and biological filtration was higher than observed in previous studies. These results suggest that adsorption as well as biodegradation may have been taking place in the filters. However, the THM concentration in the filter effluent after 14 days in the 2008 tests was higher than what was measured after 200 days of reaction time for a sample collected in 2007, suggesting that the less TOC adsorption was occurring in the pilot GAC filters in 2008 than in 2007, a trend that is consistent with the expected exhaustion of adsorption capacity.

The DBP formation in the full-scale plant and distribution system is expected to fall between the curves shown in Figure 4. In long-term full-scale operation, some TOC removal is expected during biological filtration so the DBP formation would be less than that shown for the ozonated water in Figure 4. Since THM formation in full-scale operation is expected to be lower than the ozonated water results in Figure 4, the new facility is expected to comply with the THM regulations.

Summary and Recommendations for DBP Formation Control

Overall, it is expected that the San Juan-Chama WTP will be able to meet the DBP regulations. Historical data indicates that the TOC in the Rio Grande water is typically between 2 and 6 mg/L

with an average of about 3.9 mg/L. Some higher values of TOC were observed, but they appear to be associated with high turbidity in the river, and therefore should be relatively easy to remove with conventional treatment processes. The San Juan-Chama WTP should be able to achieve 35 percent TOC removal with enhanced coagulation and an additional 15 to 25 percent TOC removal with ozone and biological filtration. Long term SDS testing using treated water from the pilot plant demonstrates that the system can meet the MCL for THMs.

The facility is not expected to have difficulty meeting the bromate MCL. The source water has a low bromide concentration, and the ozone facility is designed to operate at conditions that minimize bromate formation.

Three factors provide a degree of confidence with respect to meeting DBP regulations after the San Juan-Chama WTP is in operation. First, due to uncertainties about future regulatory requirements, the design engineer elected to set DBP goals that were lower than the current regulations. These goals led to the inclusion of ozone and biological filtration as well as enhanced coagulation in the process train. Second, source water measurements of TOC concentrations during the pilot plant operation indicates that the TOC concentration goes through an annual variation, with lower TOC in the winter and higher TOC in the summer. Regulatory compliance is based on a running annual average and the lower DBP formation in the winter months will reduce the average annual value. Third, during the summer when the DBP formation potential of the river water is the highest, the system will benefit from blending between treated river water and low TOC ground water. Blending will be necessary because summer water consumption is greater than the capacity of the plant. Water consumption in 2007 is shown in Figure 5.

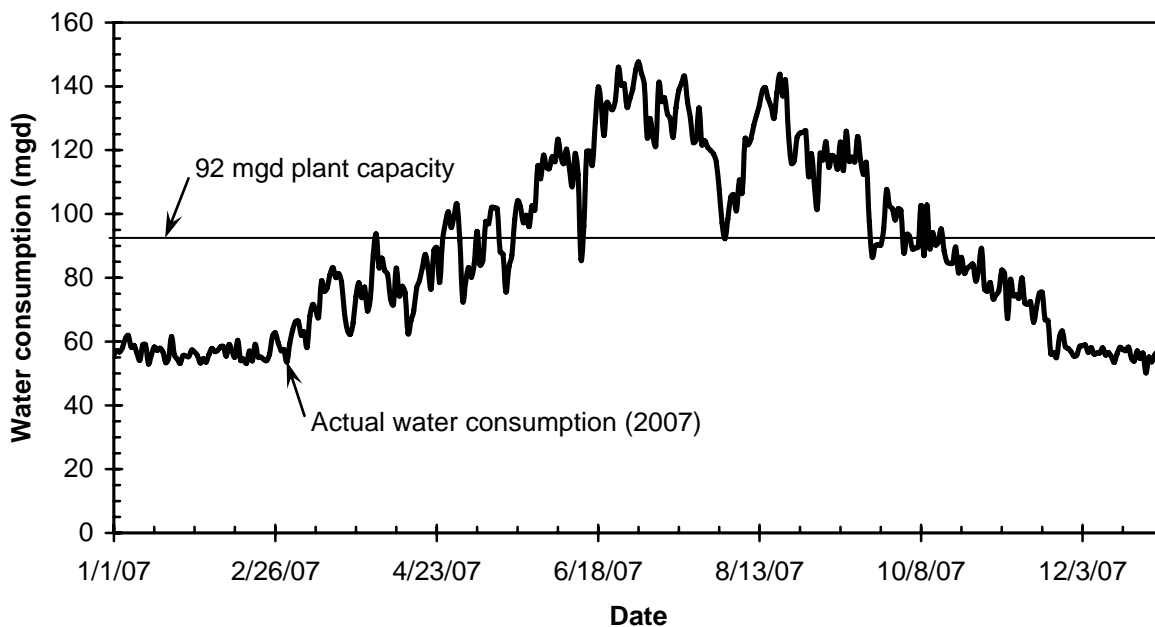


Figure 5 – Water Consumption and San Juan-Chama WTP Capacity.

From about May to September, it will be necessary to supply both treated river water and ground water to meet demand. Blending of the two waters will reduce the DBP concentration in the distribution system compared to that shown in Figure 4. The combination of these factors improves the ability of the system to comply with DBP regulations.

As noted earlier, complex interactions of chemical parameters contribute to DBP formation. Optimization of plant operations to provide the best removal of DBP precursors with the most efficient use of chemicals (coagulants and ozone) can benefit from collection of additional data on TOC removal. This data would improve understanding of the factors that contribute to DBP formation in this specific water source and the ability of the various treatment processes to remove DBP precursors. Additional data collection and analysis could include the following:

- Analysis of both TOC and DOC. Identification of the dissolved and particulate fractions of organic matter will improve the understanding of the nature of organic matter in the Rio Grande. Particulate organic carbon is easily removed by coagulation and filtration and should not contribute to DBP formation in this facility, whereas DOC is only partially removed by enhanced coagulation and ozone/biofiltration. Separate measurements of TOC and DOC will provide improved understanding of the capabilities of treatment processes to remove DBP precursors.
- Measurements of ultraviolet light absorbance at a wavelength of 254 nm (UV₂₅₄) and calculation of specific UV absorbance (SUVA). SUVA values have been correlated with TOC removal and DBP formation, and determination of these parameters will allow application of models to this source water and improve the ability to make predictions of DBP formation in the distribution system as water quality changes.
- Measurement of the above parameters at several points in the process train of the new plant. Ideally, these parameters could be measured (1) in the raw water, (2) immediately before coagulant addition, (3) Actiflo[®] effluent/ozone influent, (4) ozone effluent/filter influent, and (5) filter effluent. Process parameters (flow rate, coagulant dose, etc) should be recorded at the same time. Determination of water quality parameters at several points through the treatment train will improve understanding of DBP precursor removal and will allow optimization of processes individually and in combination. For instance, optimization of pH and coagulant dose can be determined separately from any impact of ozone or biological filtration, and the impact of enhanced coagulation specifically on DBP precursors can be determined (via understanding of the effect of coagulation on the TOC vs. DOC removal and the SUVA value).

Because it is expected that the San Juan-Chama WTP will meet the DBP regulations, no changes to the design or operation of the plant are recommended because of this review of DBP formation potential.

Impact of Upstream Municipal Wastewater Discharges

One aspect of the transition to the use of surface water is that the source water may be susceptible to contamination by discharges of municipal wastewater upstream of the diversion point. The concern from a drinking water perspective is influenced by the level of (or lack of) treatment of the wastewater, the wastewater flow relative to the river flow, and the ability of the water treatment plant to remove constituents that may be in treated or untreated wastewater.

Drinking Water Regulations Related to Wastewater Contaminants

The purpose of wastewater treatment, according to Metcalf and Eddy (2003) is to: “(1) transform (i.e., oxidize) dissolved and particulate biodegradable constituents into acceptable end products, (2) capture and incorporate suspended and nonsettleable colloidal solids into a biological floc or biofilm, (3) transform or remove nutrients, such as nitrogen and phosphorus, and (4) in some cases, remove specific trace organic constituents and compounds.” An additional purpose is to eliminate pathogenic organisms. As a result, the wastewater quality parameters of greatest interest from a regulatory perspective for drinking water include total solids, biodegradable matter, nutrients, and pathogens. The SDWA has regulations related to these parameters for turbidity, TOC regulations in the DBP rules, nitrate, nitrite, and microorganisms.

A previous report prepared by the author and submitted to the Authority in February, 2008, summarized the Safe Drinking Water Act (SDWA) regulations related to microorganisms in surface water (Howe, 2008b). The surface water treatment rules incorporate a multibarrier approach, requiring both filtration (to physically remove pathogens) and disinfection (to inactivate pathogens). Both the treatment requirements and the monitoring requirements are extensive. Additional detail can be found in the previous report. The regulations for microorganisms include a treatment technique for turbidity, requiring finished water turbidity to be below 0.3 NTU.

The removal of organic matter is included under the disinfection by-product rules, since the interaction between disinfectants and organic matter can lead to the formation of by-products that may be harmful to human health. The regulations require a specified amount of removal of TOC depending on several water quality parameters. As noted earlier, the maximum TOC requirement that the San Juan-Chama WTP should have to meet is 35 percent removal. In addition to target removal levels for TOC, the regulations contain maximum contaminant levels (MCLs) for disinfection by-products. With regard to nutrients, the SDWA has MCLs for nitrate at 10 mg/L, nitrite at 1 mg/L, and combined nitrate and nitrite at 10 mg/L. There is not currently an MCL for phosphorus.

Potential for Wastewater to Reach the Water Treatment Plant Intake

Normally, wastewater is treated before being discharged to a receiving water. However, spills of untreated sewage can and have occurred in the Rio Grande upstream of the location for the intake for the San Juan-Chama WTP. For instance, a newspaper article on July 14, 2000 indicated that the City of Rio Rancho released between 400,000 and 500,000 gallons of untreated sewage into the Rio Grande during a mechanical failure at the wastewater treatment plant (WWTP) (AP, 2000).

Once in a river, constituents in wastewater are subject to natural degradation processes and can be reduced to natural background levels. However, natural degradation takes time as contaminants move downstream, so it is relevant how far upstream the discharge may have occurred. In the case of the intake for the San Juan-Chama WTP, the Cochiti Reservoir is about 40 miles upstream. A discharge of wastewater will have different impacts on the San Juan-Chama WTP depending on whether it occurs upstream and downstream of the Cochiti Reservoir.

National Pollutant Discharge Elimination System (NPDES) permits for municipal wastewater discharges into the Rio Grande or its tributaries exist for several facilities upstream of Cochiti Reservoir, including facilities in Santa Fe, Española, Abiquiu, Chama, Taos, Red River, and Los Alamos County. Most, however, are permitted for very small flows or do not reach the Rio Grande. The only flows above 1 million gallons per day (mgd) are the wastewater treatment plants for the City of Santa Fe (13 mgd), City of Española (1.6 mgd) and Los Alamos County Bayo (1.37 mgd) (NMED, 2008). Of these three, only Española’s permit allows discharge directly to the Rio Grande. Los Alamos County and Santa Fe discharge to a canyon and tributary, respectively, and the wastewater flow typically does not reach the Rio Grande.

Wastewater discharges upstream of Cochiti Reservoir would not be expected to have a significant impact on water diversions in Albuquerque because of the effect of dilution and retention in the reservoir. The lake is typically maintained at a water surface elevation that provides a minimum lake surface of 1,200 acres for recreational purposes (Langman and Anderholm, 2004). This elevation results in a minimum lake volume of about 50,000 ac-ft ($2.2 \times 10^9 \text{ ft}^3$) (Langman and Nolan, 2005). The hydraulic residence time of the lake varies significantly with stream flow, but mean annual releases have ranged from 1,361 to 2,355 ft^3/s (Langman and Anderholm, 2004), which correspond to hydraulic detention times ranging from about 10 to 20 days. A 13 mgd discharge of raw wastewater from the Santa Fe WWTP for a 24-hour period would have a total volume of 1.7 million ft^3 , compared to a pool volume of 2.2 billion ft^3 in Cochiti Reservoir. Under these conditions, Cochiti Reservoir provides a 1000:1 dilution factor. This combination of dilution and hydraulic detention time should minimize the impacts of wastewater discharges upstream of Cochiti Reservoir on treatment operations.

Only 3 NPDES permits for municipal wastewater discharges have been issued for the Rio Grande between Cochiti Reservoir and the San Juan-Chama WTP intake (NMED, 2008). The permitted wastewater discharges on this stretch of the river are shown in Table 6, along with the distance upstream of the diversion and the estimated travel time in the river. The travel time is based on streamflow velocity of 1 to 2 miles/hour (Waltemeyer, 1991).

Table 6 – NPDES-Permitted Wastewater Discharges Between Cochiti Reservoir and the San Juan-Chama WTP Intake.

Facility	Permitted discharge (mgd and [ft^3/s])	Distance (river miles)	Travel time (hours)	Dilution ratio (based on 500 ft^3/s river flow)
Rio Rancho WWTP #2	5.5 [8.5]	6.0	3 - 6	60:1
Rio Rancho WWTP #3	0.85 [1.3]	7.9	4 - 8	380:1
Town of Bernalillo WWTP	0.8 [1.2]	10.5	5.2 - 10.5	420:1

The travel time between these discharges and the diversion point are relatively short, ranging from as little as 3 hours to at most about a half of a day. This time is not sufficient for significant degradation to take place. However, dilution can have a significant impact on the concentration of contaminants reaching the intake. The median stream flow in the Rio Grande at the Albuquerque gage has been about 870 ft³/s since the construction of Cochiti Dam. Flows typically range from about 500 ft³/s in August through October and between 3,000 and 4,000 ft³/s during May and June (Langman and Anderholm, 2004). The dilution ratio of each wastewater discharged, based on typical late summer flows in the river, is also shown in Table 6. It should be noted that both river flows and possible wastewater release flows can vary dramatically and the extent of dilution would depend on incident-specific conditions, but the dilution ratio in Table 6 at least provides an indication of the magnitude of such effects.

Considering both travel time and dilution, the wastewater plant with the largest potential for an impact on the source water at the diversion point is the Rio Rancho WWTP #2. A release of wastewater equal to the flowrate permitted in the NPDES permit would be diluted by river water only 60:1 and could reach the San Juan-Chama WTP intake in as little as 3 to 6 hours. Discharges from the other two wastewater plants would only take slightly longer to reach the intake, but the dilution ratios would be significantly greater.

Concentration of Wastewater Constituents Potentially Reaching the Intake

Typical characteristics of untreated wastewater are shown in Table 7. The effluent requirements (30-day average) for treated wastewater from the Rio Rancho WWTP #2 are 10 mg/L biochemical oxygen demand (BOD), 15 mg/L total suspended solids (TSS), 2 mg/L total ammonia, and 100 colonies/100mL for fecal coliform bacteria. A comparison of these values to the values for typical untreated wastewater in Table 7 indicates that substantial degradation takes place in a wastewater treatment plant. While treated wastewater has little potential for impact on water treatment operations after the effluent has been diluted into the receiving water, a release of untreated wastewater may have greater potential for impact. To assess this potential, concentrations of untreated wastewater diluted 60:1 (representing the dilution conditions discussed above) and the average concentration in the Rio Grande for corresponding parameters are also shown in Table 7.

For many general wastewater parameters, a release of untreated wastewater from the Rio Rancho WWTP #2 plant of the magnitude discussed earlier would have relatively little impact on the water quality at the intake. The impact on suspended solids would be negligible, with the wastewater contributing about 3.5 mg/L of TSS compared to the average concentration of suspended sediment in the river of 1,500 mg/L. For other parameters, an impact on water quality may be evident. For instance, measures of organic matter in the wastewater would be around 7.2 mg/L of chemical oxygen demand (COD) and 2.3 mg/L of TOC, compared with values of 29 mg/L and 4.6 mg/L in the river, respectively. For nutrients, concentrations in diluted untreated wastewater and natural river water are also in the same order of magnitude. Organic nitrogen and phosphorus in diluted wastewater would be around 0.3 mg/L and 0.1 mg/L, respectively, compared to average measured values in the Rio Grande of 0.88 mg/L and 0.18 mg/L, respectively.

For pathogens, however, untreated wastewater could have concentrations substantially higher than natural background concentrations in the river, even after dilution. The simple comparison shown in Table 7 indicates fecal coliform concentrations in untreated wastewater after dilution as high as 17,000 colonies/100mL, compared with an average concentration in the Rio Grande at Albuquerque of 400 colonies/100mL. Thus, ability of the water treatment facilities to accommodate the higher concentration of pathogenic organisms would be one of the prime considerations in evaluating the potential for a release of untreated wastewater to have an impact on the drinking water quality.

An important consideration in this discussion is that the water quality parameters that describe wastewater are also naturally present in surface waters and that it is necessary for water treatment facilities to have processes to remove these constituents to levels that meet drinking water standards. The ability of processes in the water treatment facility to remove constituents present in wastewater is discussed in the next section.

Treatment to Remove Wastewater Constituents

One of the primary objectives of a surface water treatment plant is the removal of pathogenic organisms. It was noted above that the primary impact of a release of untreated wastewater to the river upstream of the San Juan-Chama WTP intake would be a higher concentration of pathogenic organisms in the river compared to concentrations naturally present. Concentrations of other common constituents from a release of untreated wastewater are in the same range of magnitude as concentrations normally present in the river.

Table 7 – Characteristics of Typical Untreated Domestic Wastewater Without and With 60:1 Dilution, and Corresponding Parameters Measured in the Rio Grande at Albuquerque.

Constituent	Untreated wastewater ^(A)	Untreated wastewater with 60:1 dilution	Rio Grande at ABQ (average) ^(B)
Total suspended solids (mg/L)	210	3.5	1,500
Biochemical oxygen demand (mg/L)	190	3.2	No data
Chemical oxygen demand (mg/L)	430	7.2	29
Total organic carbon (mg/L)	140	2.3	4.6
Organic nitrogen (mg/L)	15	0.3	0.88
Free ammonia (mg/L)	25	0.4	0.2
Nitrite (mg/L)	0	0	0.06
Nitrate (mg/L)	0	0	0.04
Phosphorus (mg/L)	7	0.1	0.18
Total coliform (colonies/100mL)	$10^7 - 10^9$	$1.7 \times 10^5 - 1.7 \times 10^7$	No data
Fecal coliform (colonies/100mL)	$10^4 - 10^6$	170 – 17,000	400
Cryptosporidium oocysts (no. per 100 mL)	0.1 – 10	0.0017 - 0.17	No data
Giardia lamblia cysts (no. per 100 mL)	0.1 – 100	0.0017 – 1.7	No data

Notes:

(A) Untreated wastewater taken directly from Table 3-15 in Metcalf and Eddy, medium strength wastewater.

(B) Based on data in the on-line USGS National Water Quality Database for Station 08330000 (Albuquerque). The amount of data varied between parameters, ranging from 833 data points for TSS to 2 data points for nitrite. Data below the detection limit was ignored in calculating the average.

The author has previously completed an analysis of the new treatment plant and its ability to remove pathogenic organisms. More detail is available in that report (Howe, 2008b). In summary, that report found that the new treatment facility was capable of removing pathogenic organisms. In accordance with regulations, the plant employs a multibarrier approach consisting of both filtration and disinfection. The Authority's water treatment plant incorporates deep-bed, dual-media filters that have been demonstrated at many surface water treatment plants to provide excellent finished water quality. The disinfection system uses two separate disinfectants: ozone and chlorine. Individually, the disinfectant systems each have sufficient capability to meet surface water treatment regulations. Together, the disinfection systems greatly exceed regulations. As a result, the treatment facility will be highly capable of removing any increased concentration of pathogenic organisms that may be in the source water resulting from a release of untreated wastewater in the river.

The plant also contains processes for the removal of organic matter. The plant will utilize enhanced coagulation, which is listed as a best available technology for removing organic matter prior to disinfection. The plant also employs ozone and biological filtration, which will provide an additional barrier for the removal of biodegradable organic matter in the water.

Nitrate and nitrite are soluble anionic constituents that are not effectively removed by the processes currently present in the San Juan-Chama WTP. Processes for removal of nitrate include biological denitrification, reverse osmosis, and ion exchange. However, the concentrations that may be present in the river after a discharge of untreated wastewater would not exceed the MCL for either nitrate or nitrite. Thus, it would not be necessary for the plant to remove these contaminants in order to meet drinking water standards.

Pharmaceuticals and Personal Care Products in the Rio Grande

The conventional wastewater parameters discussed earlier are only one factor in the discharge of wastewater to the Rio Grande. A topic that has been under investigation in the water treatment industry for a number of years, and more recently identified in the news media, is the possible presence of pharmaceuticals and personal care products (PPCPs) in wastewater.

PPCPs can potentially enter natural waters via the effluent discharge from municipal wastewater treatment plants. PPCPs can enter wastewater by being ingested by humans and then being excreted with waste, or when people flush unused or excess medication down the toilet. Although some PPCPs can be degraded during the wastewater treatment process, some pass through the plant into the wastewater effluent and then into the receiving water. A national reconnaissance of 95 organic compounds in 139 rivers across the United States was conducted by the USGS in 1999-2000 (Kolpin et al., 2002). Modern analytical methods are capable of detecting very low concentrations of organic compounds, which increases the likelihood that contaminants will be found. PPCPs were found in 80 percent of the rivers. Concentrations were generally very low. For instance, the highest measured concentration of ibuprofen (Advil[®]) in that study was 1 µg/L. Lifetime ingestion of water (2 L/day for 70 years) with this concentration would result in a total lifetime intake of 51 mg – about one-fourth of the dose in a single 200-mg Advil[®] tablet. The effect of long-term exposure to trace doses of pharmaceuticals or synergistic effects of multiple pharmaceuticals is not known at this time and is currently being investigated

by the research community. To date, research has not detected an impact on human health from the trace concentrations of pharmaceuticals being found in drinking water.

Some data is available regarding the potential presence of PPCPs in the Rio Grande, including an NMED reconnaissance study, data collected by USGS, and studies conducted by two University of New Mexico (UNM) graduate students. NMED conducted an initial surveillance of the presence of pharmaceutical residues in waters of the state. Samples of river water were collected by NMED staff using accepted protocols, including trip blanks and blind duplicates. Analyses were conducted by the New Mexico Department of Health Scientific Laboratory Division with methods that achieved a detection limit of about 10 ng/L. In general, the frequency of occurrence of drug residues in New Mexico rivers appears to be less than in other locations. NMED detected drug residues in 11 of 15 wastewater effluent samples but only 4 of the 23 surface water samples. The only pharmaceutical detected in the river upstream of the location for the San Juan-Chama WTP intake was the anti-depressant medication amitriptyline, which was detected at Buckman Crossing at 30 ng/L (McQuillan et al., 2002). At a rate of ingestion of drinking water of 2 L/day for 70 years, this concentration corresponds to a lifetime intake of 1.5 mg. As a point of comparison, typical adult doses of amitriptyline for treatment of depression range from 40 to 150 mg daily (MedicineNet.com, 2008). In other words, if the San Juan-Chama WTP were unable to remove amitriptyline, the intake by a resident of Albuquerque over an entire lifetime would be 100 times less than a single day's dose when taken for medical purposes. Caffeine and estrone were detected in the river downstream of Albuquerque.

The USGS has sampled Rio Grande water for numerous organics, including PPCPs, on several occasions and at several locations in recent years (USGS, 2008). Between Cochiti Lake and the WTP intake, the locations have included below Cochiti Dam, San Felipe, and at the Alameda Bridge. At the sampling location below Cochiti Dam, the USGS database includes analyses for 52 organics measured on four occasions. Detection limits ranged from 1 to 50 ng/L, depending on the compound. No compounds were detected. At San Felipe, the analyses included 196 organics, including pesticides, solvents, and other synthetic organic chemicals (SOCs) in addition to PPCPs. No PPCPs or other SOC of any type were detected. At the Alameda Bridge sampling station, the analyses included up to 181 organics sampled on eight occasions. The results of over 1200 individual analyses yielded only one quantified result above the detection limit, a reported concentration for tris(2-butoxyethyl) phosphate, a plasticizer and flame retardant, at the detection limit of 0.2 µg/L. Downstream of the San Juan-Chama WTP intake (at the USGS Albuquerque station), 3 detections were found in 312 samples that were collected (40 ng/L prometon, 10 ng/L metolachlor, and 35 ng/L pendimethalin).

Kathy Brown, a student in the Water Resources Program at UNM, completed a professional project in 2004 that examined the presence of pharmaceuticals in hospital effluent, wastewater treatment plant effluent, and the Rio Grande (Brown, 2004). Samples from 10 sites were analyzed for 39 compounds by the New Mexico Department of Health Scientific Laboratory Division. Seven PPCPs were detected in hospital wastewater and three were detected in the effluent from the Albuquerque Southside Water Reclamation Plant (SWRP). No PPCPs were detected in Rio Grande water upstream of the SWRP and sulfamethoxazole was the only PPCP detected in the Rio Grande downstream of SWRP. The intake for the San Juan-Chama WTP is upstream of the SWRP and near the sampling location for the upstream sample in the study.

Maceo Martinet, a biology graduate student at UNM, also completed a study of PPCPs in the Albuquerque area (Martinet, 2005). In this study, samples were collected from shallow alluvial groundwater adjacent to the Rio Grande upstream and downstream of the SWRP and analyzed for 19 PPCPs. Analyses were done by a reputable commercial laboratory with detection limits that ranged from 5 to 25 ng/L. Seven PPCPs were detected in the groundwater upstream of the SWRP and nine PPCPs were detected downstream. One outcome noted in this study was that the concentrations of several compounds were higher in the shallow groundwater than in the effluent from the SWRP, and it was postulated that some compounds might accumulate in shallow groundwater due to sorption onto sediments.

One reason for the lack of pharmaceutical residues in the Rio Grande is the lack of metropolitan areas upstream of Albuquerque. As noted earlier, the impact of wastewater discharges on the flow of the Rio Grande is relatively small. Another possible reason cited for the low concentrations of pharmaceuticals in New Mexico surface waters is that the combination of shallow rivers and intense sunlight may lead to the photolytic degradation of pharmaceutical residues (McQuillan et al., 2002).

In summary, the information that is available about the presence of PPCPs in Rio Grande water upstream of the intake indicates very little presence of PPCPs in the water column. The potential for contamination is lower than in many rivers. Nationally, PPCPs have been found in many rivers at low concentrations. Human health impacts from exposure at these low concentrations has not been documented but is being studied.

Treatment to Remove PPCPs

PPCPs potentially have a wide range of physicochemical characteristics. As a result, it is not possible to specify a single treatment process that can reliably remove all PPCPs, with the possible exception of high-pressure (i.e., seawater) reverse osmosis. However, recent studies have been collecting data on the treatment efficiency of water treatment processes present in the San Juan-Chama WTP for classes of PPCPs with similar physicochemical properties.

The coagulation/filtration process would not be expected to be very effective for removal of most PPCPs because this process is typically not effective for soluble organic chemicals. A recent study (Snyder et al., 2007) has supported this conclusion. Processes present in the San Juan-Chama WTP process train that may be effective for various PPCPs include application of ozone for oxidation, adsorption with GAC, and final oxidation/disinfection with chlorine.

Snyder et al. (2007) found that GAC is effective for some classes of PPCPs. Charged species like ibuprofen were poorly removed by powdered activated carbon (less than 20 percent removal) and had short bed lives with GAC, but uncharged polar compounds like benzo(a)pyrene were effectively removed. Oxidation with chlorine was not very effective (less than 20 percent removal) for many compounds. Ozone was very effective and achieved greater than 95 percent removal with 2 minutes contact time for the majority of PPCPs (Snyder et al., 2007). The San Juan-Chama WTP will have 12 minutes of ozone contact time at the maximum flowrate and greater contact time at lower flowrates. Other researchers have similarly found that ozonation is very effective for removing PPCPs (Ternes et al., 2002; Westerhoff et al., 2005; Vieno et al., 2007; Kim et al., 2008; Okuda et al., 2008). Okuda, et al. (2008) found that ozone

followed by biologically active carbon was effective at removing all studied PPCPs to below the detection limits.

The PPCP detected in the Rio Grande water above the intake site was amitriptyline. The study reported above did not include that particular PPCP in the list of compounds being examined, so a direct assessment of the removal efficiency for that compound is not possible. However, based on structural characteristics, amitriptyline is most similar to carbamazepine, which was included in the study. Both are tricyclic compounds with two aromatic rings connected by a nonaromatic ring and include some amine functionality. Snyder et al. (2007) found 50 to 80 percent removal of carbamazepine with activated carbon and greater than 95 percent removal with ozone. The San Juan-Chama WTP includes ozone and GAC, so amitriptyline should be effectively removed.

An approach to minimizing the impact of PPCPs on the source water would be for Albuquerque to work with upstream communities to try to minimize the discharge of pharmaceuticals to the Rio Grande with wastewater effluent. One way to do this would be to encourage medication collection programs that discourage people from flushing unused medications down the toilet.

Summary and Recommendations Regarding Upstream Wastewater Discharges

Albuquerque is fortunate to have a surface water source for drinking water that has a relatively small impact from upstream wastewater discharges compared to many other surface water sources used by water utilities throughout the United States. Based on this review, no changes to the San Juan-Chama WTP process train appear to be necessary to cope with upstream wastewater discharges.

Source water protection is an important policy for any water utility, and strategies to minimize contamination of the source water are consistent with the multi-barrier approach to drinking water treatment. The design of the San Juan-Chama WTP contains presedimentation basins that contain about 24 hours of raw water storage capacity, which gives the Water Utility Authority the flexibility to close the intake temporarily in the event of an extreme contamination event in the river. The Water Utility Authority should maintain open lines of communication with the City of Rio Rancho and the Town of Bernalillo, and request to be notified immediately if a discharge of untreated wastewater occurs at their wastewater treatment plants. Upon notification, the Water Utility Authority would then have the flexibility to decide whether it would be prudent to temporarily close the intake.

Another opportunity for source water protection exists with respect to PPCPs. Although the Rio Grande upstream of the intake does not appear to be significantly impacted with PPCPs, it would be prudent to encourage measures to minimize future impacts. It is recommended that the Water Utility Authority encourage the State of New Mexico and/or upstream communities to develop pharmaceutical disposal programs that discourage flushing unused medication down the toilet.

Radionuclides

Radionuclides are chemical compounds that emit radiation. The SDWA has regulations to limit the amount of radionuclides allowed in drinking water. The San Juan-Chama WTP will be required to meet those regulations. This section of the report summarizes the regulatory requirements for radionuclides in SDWA, existing information on the presence of radionuclides in Rio Grande water, and the ability of the San Juan-Chama WTP to effectively remove radionuclides from the source water.

Sources of Exposure to Radiation

Radionuclides are widely present in the earth's crust. The common radioactive chemicals that occur in rock and soil throughout the world include potassium-40 and the decay products of thorium-232 and uranium-238. Because of their wide presence in the environment, these chemicals inevitably enter food we eat, water we drink, and dust in the air we breathe. In enclosed buildings, people can be exposed to radiation from inhalation of radon gas that seeps from the ground. People are also exposed to natural radiation from cosmic rays. Although natural radionuclides occur throughout the world, the distribution is not even and people in some areas are exposed to higher levels of radiation than people in other areas.

In addition to natural radionuclides, people can be exposed to manmade radiation. Sources of manmade radiation include medical procedures (such as x-rays and CAT scans), consumer products (such as smoke detectors), pollution from industrial processes (such as coal-fired power plants), and fallout from global atmospheric nuclear testing that occurred in the 1940s to 1960s. The environment may also contain higher levels of radiation in areas where discharges have occurred from nuclear power generation, nuclear weapons development or testing, or industrial facilities (such as medical equipment or smoke detector manufacturers).

The three types of nuclear radiation relevant to drinking water are alpha, beta, and photon (gamma) radiation. An alpha particle consists of two protons and two neutrons, and is therefore equivalent to a helium nucleus. Relative to other types of ionizing radiation, the alpha particle is heavy and slow moving. Because of their large mass, alpha particles lose energy rapidly and do not travel far. They travel only a few inches in air and cannot penetrate common objects, including paper or skin. Because alpha particles outside the body are essentially harmless, the greatest potential for impact to human health occurs when alpha particles are inhaled or ingested. Beta particles are essentially high-energy, high-speed electrons that are ejected from the nucleus of an atom during radioactive decay. Beta particles will travel several feet in air before losing their energy. Gamma rays are photons just like x-rays and visible light, but with considerably more energy than visible light and somewhat more energy than x-rays. When gamma ray emission occurs, it is typically in conjunction with alpha or beta particle emission. Like x-rays, gamma rays can travel great distances through air and through most solid objects. Lead or other dense materials are used for shielding to stop gamma radiation.

Drinking Water Regulations for Radionuclides

Regulations for radionuclides in drinking water were first incorporated into the Safe Drinking Water Act in 1976 and updated in 2000. The 2000 Radionuclides Rule (U.S. EPA, 2000)

reaffirmed the MCLs for alpha activity, beta and photon activity, and radium and added a new MCL for uranium. The current MCLs are shown in Table 8.

The possible health effects for alpha, beta, photon, and radium are that some people who drink water in excess of the MCL over many years may have an increased risk of getting cancer. For uranium, the health effects may include kidney toxicity in addition to the increased risk of cancer caused by uranium being an alpha emitter.

Many different natural and manmade radionuclides exist. The vast majority are not listed individually in the Radionuclides Rule. All relevant compounds are either alpha emitters or beta emitters and are therefore included within the applicable MCL. In other words, any radionuclides that may be present in the river water are regulated under either the adjusted gross alpha activity MCL or the beta particle and photon activity MCL.

The units of radioactivity expressed in Table 8 can be a little confusing. The units for alpha activity are picocuries per liter (pCi/L), which is a measure of the rate of radioactive decay corresponding to one decay event about every 30 seconds in a liter of water. Beta and photon radiation are regulated on the basis of equivalent dose. Beta and photon radiation can be produced by 126 different isotopes. The biological effects of each isotope are different, so a regulation based solely on the decay rate of beta and photon emitters does not provide adequate protection of human health. Therefore, the MCL considers biological impacts. The equivalent dose (expressed as millirem/year, mrem/yr) is a measure of the biological impact caused by radiation. The activity (in pCi/L) that will cause a 4 mrem/yr dose of radiation is different for each isotope, so the regulations specify the allowed activity for each isotope with respect to the allowable 4 mrem/yr dose. For instance, the allowable activities are 20,000 pCi/L for tritium and 200 pCi/L for Cesium-137. When multiple isotopes are present, the contribution of each to the total 4 mrem/yr dose must be calculated on a weighted basis.

Existing Rio Grande Water Quality Data

The Rio Grande has been monitored for gross alpha and beta activity and several individual radioactive isotopes by a number of agencies over a significant period. The USGS recently summarized all existing water quality data from the Los Alamos National Laboratory (LANL), USGS, NMED, and the U.S. Fish and Wildlife Service in the Rio Grande for the 20-year period between 1985 and 2005 (Falk et al., 2007). The monitoring sites between Cochiti Dam and Albuquerque that contain data on radionuclides are listed in Table 9. A summary of all radionuclides data at those sites is shown in Table 10. Table 10 also presents the relevant regulatory limit or target screening value for each constituent, and compares the maximum

Table 8 – Maximum Contaminant Levels Established in the Radionuclides Rule.

Contaminant	MCL
Adjusted gross alpha activity (not including radon or uranium)	15 pCi/L
Beta particle and photon activity	4 mrem/yr
Radium-226/228 (combined)	5 pCi/L
Uranium	30 µg/L

concentration measured in the river to the appropriate target value. For alpha emitters, each constituent is compared to the adjusted gross alpha particle MCL of 15 pCi/L. For beta and photon emitters, each constituent is compared to the allowed activity for that isotope with respect to the 4 mrem/yr limit from information referenced in the Radionuclides Rule.

Based on the data presented in Table 10, measured radionuclide activity in raw Rio Grande water has never exceeded the regulatory limits for finished water. Thus, even if the San Juan-Chama WTP did not achieve any removal of radionuclides, the finished water would meet the regulated limits. For most of the individual isotopes listed, the measured activity is significantly below the regulatory limit. With the exception of cesium, radium, and strontium, the maximum measured values are less than 10 percent of the regulated limit, and in many cases the maximum measured values are less than 1 percent of the regulated limit. The Water Utility Authority has collected additional samples for individual radionuclides since 2004 and the results are consistent with the USGS summary of earlier data; no individual radionuclides in the raw water exceed the regulatory limits for finished water.

Additional data on radionuclides in raw Rio Grande water was developed during operation of the pilot plant that was used for bottled water production. That data is presented in the next section.

Radionuclide Activities Measured during Pilot Plant Operation

During the pilot plant operation in 2007, the raw and finished water was sampled and analyzed for all regulated parameters in the Safe Drinking Water Act – including radionuclides – every time a bottling operation was conducted. The radionuclide parameters that were measured included gross alpha activity, gross beta activity, radium-226/228, and uranium. The analyses were conducted by the New Mexico Health Department Scientific Laboratory Division. Because of the large number of bottling operations that were conducted over the course of the pilot plant operation, the data provides an indication of the variation in radionuclides in the source water over the period of a year.

Table 9 – Sampling Locations Downstream of Cochiti Lake.

Sampling location	Agency
Below Cochiti Dam	USGS
San Felipe	USGS
At Alameda Bridge	USGS
At Albuquerque	USGS
Below Cochiti Dam	LANL
At Bernalillo	LANL
Below Cochiti Dam	NMED
Below Santa Fe River	NMED
Above Santa Fe River	NMED
At Cochiti south boundary	NMED
At San Felipe north boundary	NMED
At San Felipe Pueblo	NMED
Below San Felipe lagoons	NMED
At and below Angostura Diversion Dam	NMED

Table 10 – Summary of Radionuclide Activity in the Rio Grande between Cochiti Reservoir and Albuquerque for Water Years 1985-2005.

Constituent	Emission type	Sample type	Number of samples	Maximum measured concentration in Rio Grande (pCi/L)	Treated water regulatory limit ^(A) (pCi/L)	Maximum measured as percent of regulatory limit (%)
Gross alpha ^(B)	alpha	total	44	10.9	15	72.7
		dissolved	12	13	15	86.7
Gross beta ^(C)	beta	total	44	32.3	50	64.6
		dissolved	21	6.61	50	13.2
Am-241	alpha	total	45	0.3	15	2.0
		dissolved	11	0.01	15	0.1
Cs-137	beta	total	39	185	200	92.5
		dissolved	1	-0.0389	200	0.0
Pu-238	alpha	total	67	0.188	15	1.3
		dissolved	11	0.0021	15	0.0
Pu-239	alpha	total	21	0.03	15	0.2
Pu-239/240	alpha	total	48	0.075	15	0.5
		dissolved	11	0.00647	15	0.0
Ra-226	alpha	total	28	0.338	5	6.8
		dissolved	14	0.57	5	11.4
Ra-228	beta	total	25	1.4	5	28.0
Ra-226/228	alpha, beta	total	20	1.55	5	31.0
Sr-90	beta	total	30	1.8	8	22.5
		dissolved	11	0.48	8	6.0
Th-228	alpha	total	9	0.134	15	0.9
		dissolved	8	0.31	15	2.1
Th-230	alpha	total	28	0.33	15	2.2
		dissolved	10	0	15	0.0
Th-232	alpha	total	30	0.5	15	3.3
		dissolved	10	0.05	15	0.3
Tritium	beta	total	47	1000	20,000	5.0
U-234 ^(D)	alpha	total	35	1.88	20	9.4
		dissolved	11	0.997	20	5.0
U-235 ^(D)	alpha	total	6	0.0452	20	0.2
		dissolved	5	0	20	0.0
U-235/236 ^(D)	alpha	total	9	0.129	20	0.6
		dissolved	6	0.0394	20	0.2
U-238 ^(D)	alpha	total	35	1.24	20	4.1
		dissolved	11	0.607	20	3.0
Gross gamma ^(E)	photon	total	30	960	N/A	N/A

Notes:

(A) This table presents raw Rio Grande water quality data, but the regulatory limits actually apply to treated water.

(B) Gross alpha values are a summary of data reported using several calibration methods, including Th-230, natural uranium, and Am-241 calibration standards. Measured values for gross alpha are total measured values, but regulatory limits exclude uranium activity.

(C) Gross beta values are a summary of data reported using Cs-137 and Sr-90/Y-90 calibration standards. Measured values for gross beta are total measured values, but the screening level of 50 pCi/L excludes natural potassium-40 activity.

(D) The MCL for uranium is 30 µg/L but measured values are reported in activity (pCi/L). The Radionuclides Rule specifies a conversion rate of 0.67 pCi/µg; therefore, the regulatory limit of 20 pCi/L in this table corresponds to the MCL of 30 µg/L.

(E) Gross gamma does not have a regulatory limit because photon emitters are regulated within the beta emission standard.

Total alpha activity over the duration of the pilot plant operation is shown in Figure 6. Gross alpha activity was analyzed using two different calibration standards; calibration with natural uranium reference standards is shown in the top panel and calibration with Americium-241 reference standards is shown in the bottom panel of Figure 6. The EPA allows both calibration methods in Radionuclides Rule. The natural uranium reference standard is approved for use with co-precipitation and evaporation methods and the Americium-241 reference standard is approved for use with co-precipitation methods for gross alpha activity analysis. However, NMED uses the natural uranium standard for compliance determination. The two calibration methods produced different results, with the natural uranium reference values averaging about 25 percent higher than the Am-241 reference values. Using the higher values, it can be seen that the total gross alpha activity was typically below 7.5 pCi/L, or less than half of the MCL. Total alpha activity of less than half the MCL will qualify the facility for reduced monitoring for alpha activity under the SDWA. Higher alpha activity was observed during July and August, with the peak values in the raw water exceeding the treated water MCL. However, the data demonstrates that the pilot facility was able to effectively remove the total alpha activity, with over 95 percent removal occurring when alpha activity was high. Essentially no increase in alpha activity in finished water occurred when alpha activity in the raw water increased. The maximum total alpha activity measured in the finished water over the duration of the pilot plant operation was 1.6 pCi/L, about an order of magnitude below the MCL.

Total beta activity over the duration of the pilot plant operation is shown in Figure 7, with the cesium-137 reference standard in the top panel and strontium/yttrium-90 reference standard in the bottom panel. The two calibration methods produced nearly identical results, never varying by more than about 5 percent from each other. Total beta activity followed the same trends as total alpha activity. Raw water activity was typically below 10 pCi/L, except for a spike to about 28 pCi/L that occurred in July and August. Because the beta and photon activity MCL can include contributions from as many as 126 different isotopes, each of which decay at a different rate, the regulations include a screening level to determine whether the utility needs to speciate samples to determine the contribution from individual isotopes. The screening level is 50 pCi/L, and the raw water never exceeded about half of this value. As with the alpha activity, the low values of beta activity will qualify the facility for reduced monitoring after the initial monitoring period. The maximum total beta activity measured in the finished water over the duration of the pilot plant operation was 5.4 pCi/L, about an order of magnitude below the screening level.

The Radionuclides Rule indicates that several studies have found that the primary source of beta activity is potassium-40 (K-40) and radium-228, both naturally occurring compounds. Since K-40 is not regulated, the regulations allow the K-40 activity to be subtracted from the total measured beta activity to determine if the screening level has been exceeded. An analysis of the data from the pilot plant reveals that K-40 was the primary contributor to the beta activity in the finished water. Using the calculation method stipulated in the Radionuclides Rule, the average beta activity excluding K-40 activity in the finished water from the pilot plant was 0.65 pCi/L, nearly 2 orders of magnitude below the screening level.

The uranium concentration over the duration of the pilot plant operation is shown in Figure 8. The uranium concentration in the river water fluctuated between 2 and 3 µg/L, substantially below the treated water MCL of 30 µg/L. Uranium was never detected in the finished water

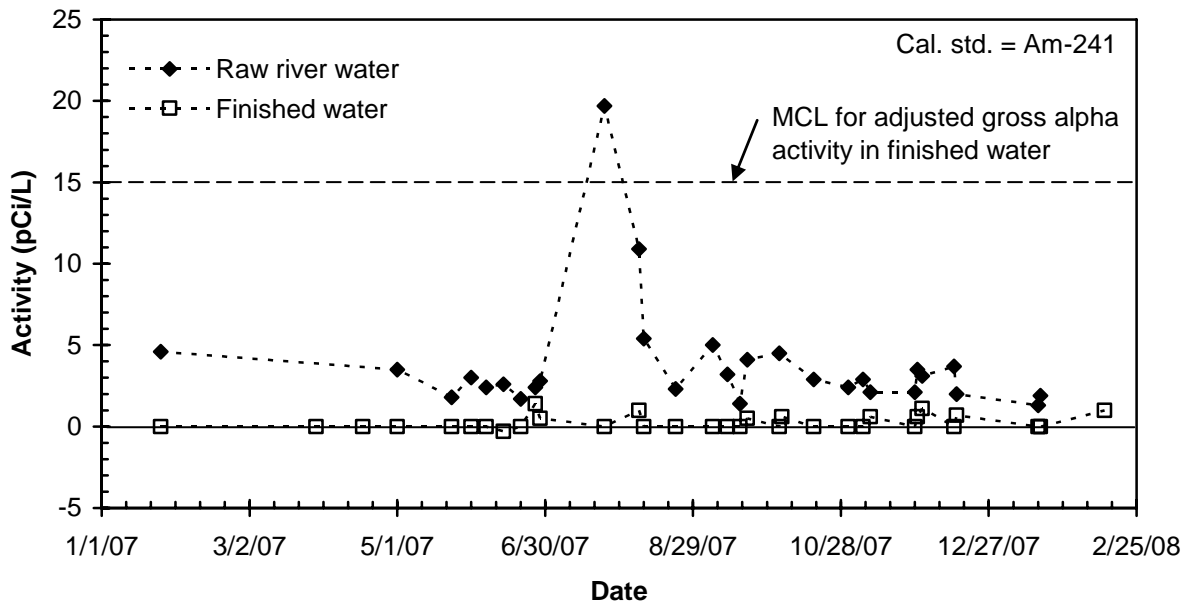
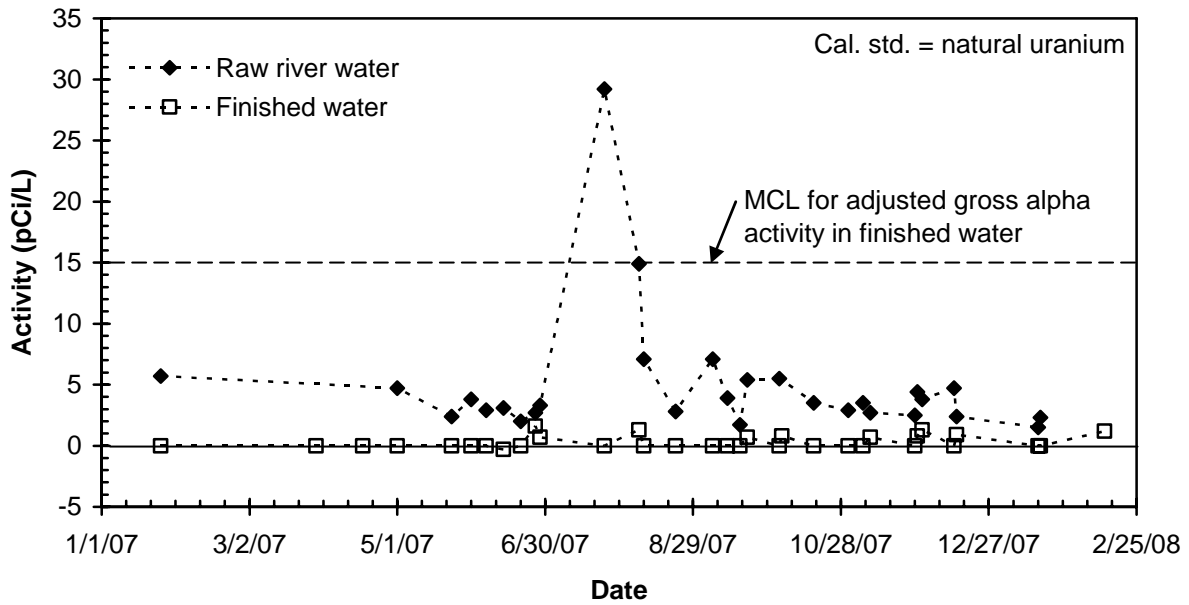


Figure 6 – Total Gross Alpha Activity Measured during ABCWUA Pilot Plant Operation (Reference standard is natural uranium in the top panel and Am-241 in the bottom panel).

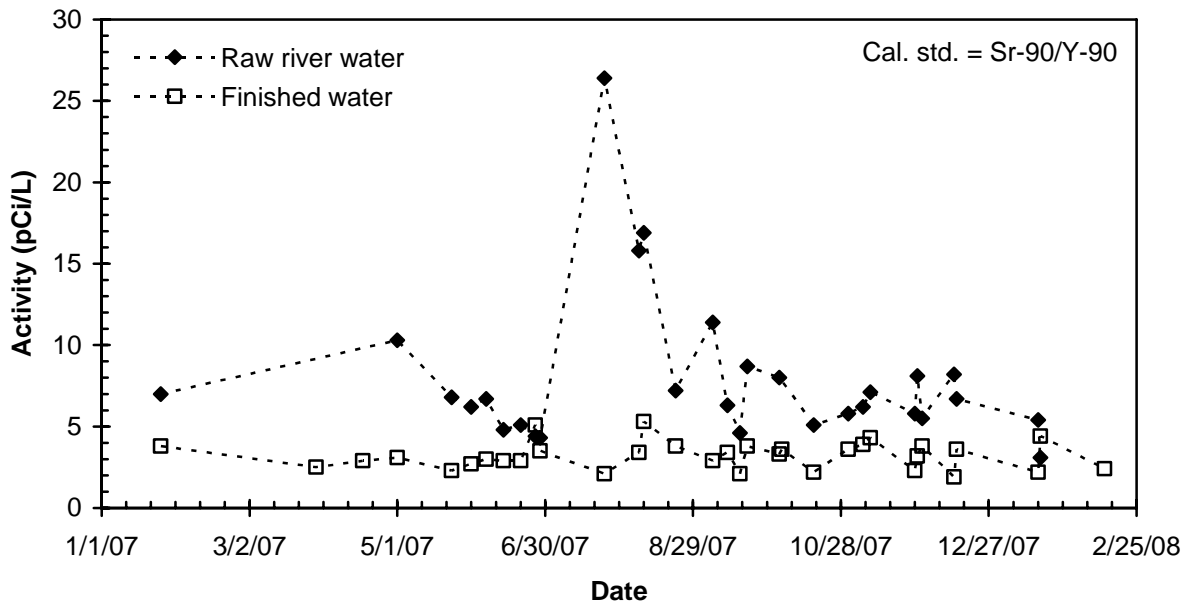
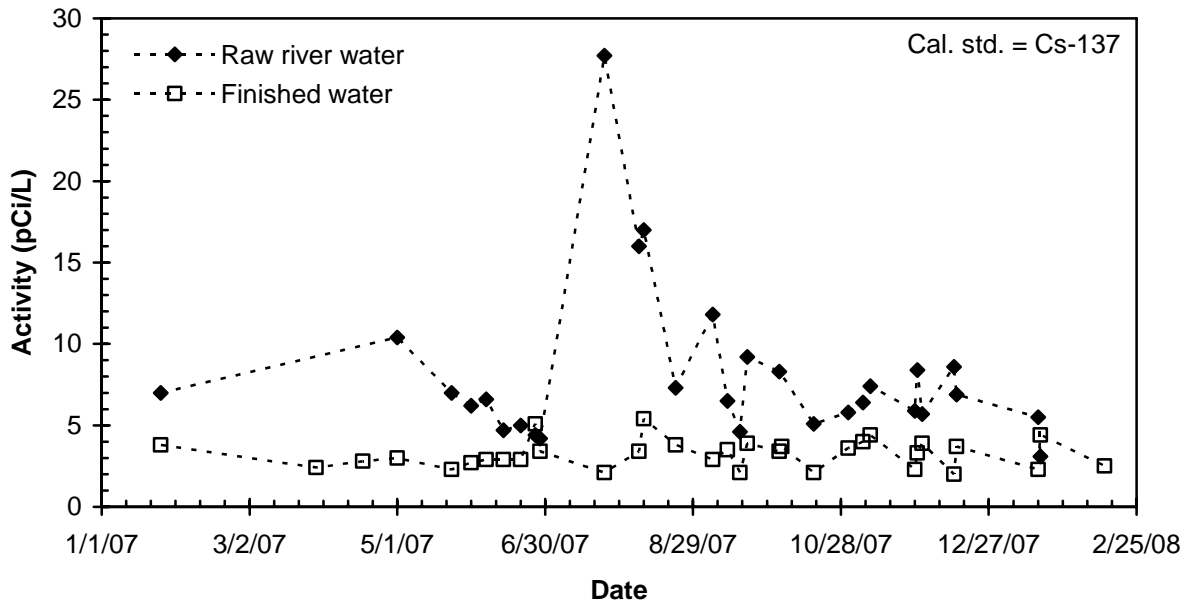


Figure 7 – Total Gross Beta Activity Measured during ABCWUA Pilot Plant Operation (Reference standard is Cs-137 in the top panel and Sr-90/Y-90 in the bottom panel. For comparison, the screening level for water systems designated as “vulnerable” is 50 pCi/L).

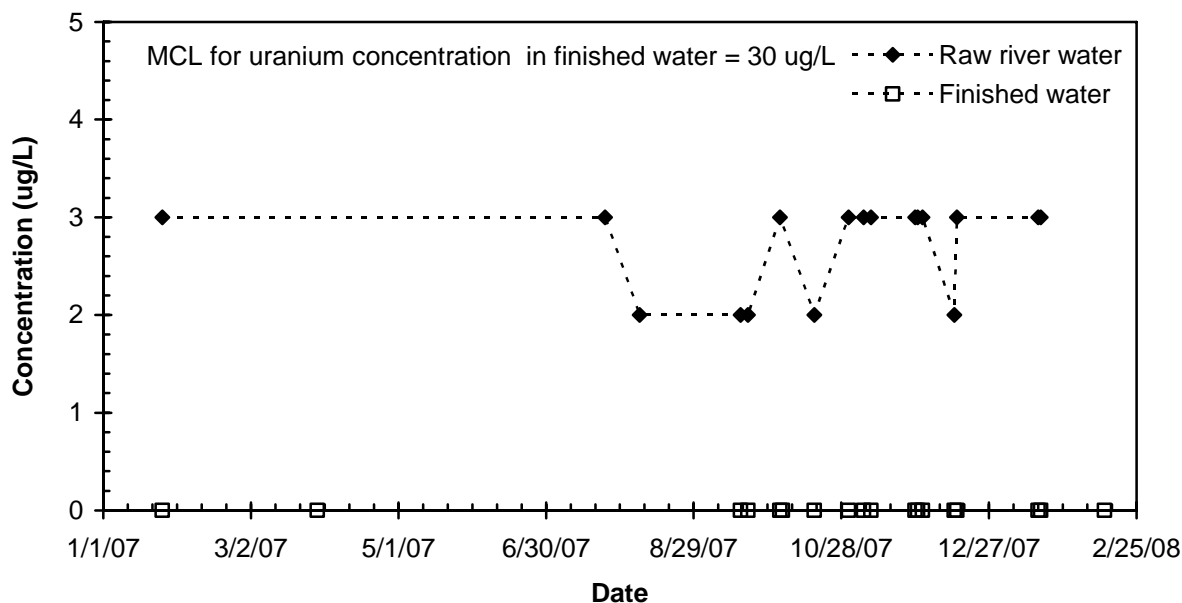


Figure 8 – Uranium Concentration Measured During ABCWUA Pilot Plant Operation.

from the pilot plant. Good removal of uranium by this process is expected since coagulation/ filtration is listed by EPA as a best available technology for uranium.

The total combined radium-226/radium-228 activity over the duration of the pilot plant operation is shown in Figure 9. The raw water activity averaged 0.52 pCi/L and was nearly always below 1 pCi/L, with the exception of a spike to 3.1 pCi/L that occurred in July, the same period as the higher total alpha and beta activity. The MCL for combined Ra-226/228 is 5 pCi/L, so the raw water stayed below the treated water MCL at all times. Good removal of combined Ra-226/228 was achieved by the pilot plant. The maximum combined Ra-226/228 activity measured in the finished water was 0.39 pCi/L, more than an order of magnitude below the MCL.

Treatment for Radionuclides Removal

Radionuclides include a wide variety of chemical species with an equally wide variety of chemical properties, so it is difficult to select treatment processes that are effective for all radionuclides. The Radionuclides Rule (U.S. EPA, 2000) lists reverse osmosis as BAT for all radionuclides and additionally lists ion exchange as BAT for radium, beta particle and photon activity, and uranium. Identification of reverse osmosis as BAT for radionuclides is based on the ability of reverse osmosis to remove nearly all contaminants from water regardless of their chemical properties. However, other treatment processes will often be successful for specific compounds.

The Radionuclides Rule (U.S. EPA, 2000) lists coagulation/filtration as a best available technology for the removal of uranium. Uranium and other actinides adsorb strongly to soil and other inorganic surfaces and can typically be removed from water by removing particles. The

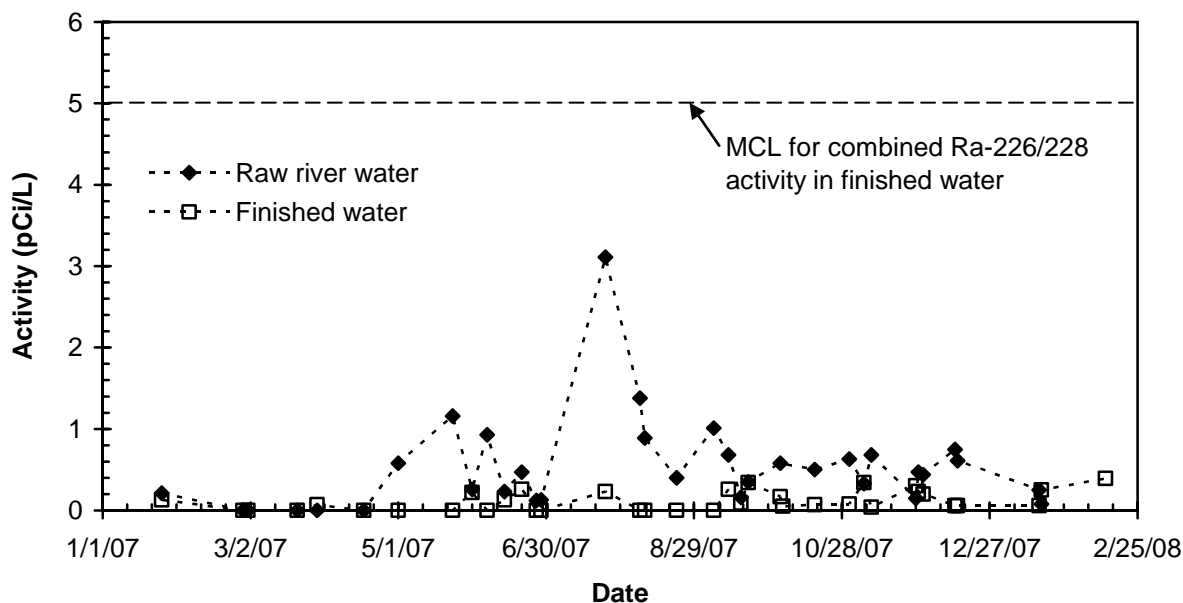


Figure 9 – Total Radium-226/228 Activity Measured During ABCWUA Pilot Plant Operation.

regulations do not distinguish between dissolved and particulate forms for the designation of coagulation/filtration as BAT for uranium. Other actinides, including thorium, plutonium, and americium, have chemical properties similar to uranium and would be expected to be removed by the same processes. The San Juan-Chama WTP would be able to remove these radionuclides using the coagulation/filtration process. Recent studies have demonstrated the effectiveness of the coagulation/filtration process for removing uranium, thorium, plutonium, and americium (Gafvert et al., 2002; Baeza et al., 2004).

Tritium is indistinguishable from H₂O by any water treatment process used at a municipal scale. As a result, none of the technologies in the proposed San Juan-Chama WTP, nor any possible replacement or additional process, will have any affect on the removal of tritium.

With regard to the remaining radionuclides listed in Table 10 (cesium, strontium, and radium), literature suggests that the ability to remove these contaminants by coagulation/filtration depends on the presence of specific clays or sediments that sorb the specific compounds. Evidence from the pilot plant regarding removal of alpha and beta activity suggests that most, if not all, of the radioactivity in the Rio Grande is in a form that can be removed by coagulation/filtration. Other studies of treatment of Rio Grande water suggest the same thing, as reported in Howe (2008a). If coagulation/filtration were not effective for these contaminants, ion exchange or reverse osmosis would be effective treatment processes and could be added to the plant. Both ion exchange and reverse osmosis are listed as BAT for many radionuclides in the SDWA.

Summary and Recommendations for Radionuclides Removal

Historical data of gross alpha and beta activity and selected isotopes indicates that Rio Grande water near Albuquerque is always below applicable regulatory standards for radioactivity. More extensive sampling during the pilot plant operation confirms this observation. The pilot data shows the raw water activity was always below regulated treated water levels for beta activity, uranium, and radium, and was always below the regulated treated water levels for alpha activity with the exception of one sample in July 2007. Activity was higher than average during July and August, but still below the regulatory levels except for the one alpha activity sample. The pilot plant was effective at removing radioactivity. Radioactivity in the finished water was consistently an order of magnitude or more below the regulatory limits. As a result, it appears that the San Juan-Chama WTP will not have difficulty meeting regulations for radionuclides and no changes in plant design or operation are recommended based on this review.

Perchlorate

According to a recent National Research Council (NRC) report (NRC, 2005), perchlorate contamination in groundwater was discovered in California in 1985 and recognized as a more widespread national issue by 1997. Perchlorate is both naturally occurring and a manmade chemical. It is used in explosives, rockets, airbag inflation systems, and fireworks. As a result, it tends to be found in water when explosives have been manufactured or detonated or where firework displays have been staged. Perchlorate has the potential to interfere with the proper function of the thyroid because of its similar chemical behavior to iodide.

Drinking Water Regulations for Perchlorate

Perchlorate is not currently regulated in federal or state drinking water regulations. It is included on the contaminant candidate list (CCL) (U.S. EPA, 2008), which means that EPA may regulate it in the future. The NRC report (NRC, 2005) concludes that current research data supports a reference dose (RfD) of 0.0007 mg/kg per day. The NRC report states that this level is believed to protect the health of even the most sensitive populations. Actual exposure depends on a person's body mass and the amount of water he/she drinks. For a 70-kg (~150 lb) person drinking 2 L of water per day, the concentration in water that corresponds to 0.0007 mg/kg is 24.5 µg/L. Based on this, EPA issued guidance in 2005 setting a Drinking Water Equivalent Level (DWEL) of 24.5 µg/L (U.S. EPA, 2005). The DWEL is merely one reference point for beginning the regulatory process and will not necessarily translate directly to an MCL if regulations are established in the future. It does, however, give a general indication of the magnitude of a potential MCL (i.e., it is not likely that an eventual MCL would be orders of magnitude above or below the DWEL). More recently, EPA published a news release 8 January 2009 setting an interim health advisory level of 15 µg/L (U.S. EPA, 2009). The interim health advisory is guidance for cleaning up local perchlorate contamination sites and is another indication of the range for the MCL if perchlorate is regulated in the SDWA in the future.

At least two states have set MCLs for perchlorate. Massachusetts established an MCL of 2 µg/L in July 2006 and California set an MCL of 6 µg/L in October 2007 (CDPH, 2008; MDEP, 2008). New Mexico does not have a drinking water standard for perchlorate.

Existing Water Quality Data for Perchlorate

The Los Alamos National Laboratory (LANL) has sampled Rio Grande water for perchlorate at numerous locations over the past eight years. Sampling locations have ranged from Embudo on the Rio Grande and Chamita on the Rio Chama, both upstream of LANL, to below Cochiti Reservoir. In all, 36 samples have been analyzed. Twenty-seven of the samples have been reported as below the detection limit (detection limits varied from 0.05 to 4 µg/L). The remaining nine samples have had an average value of 0.064 µg/L and a maximum value of 0.071 µg/L. Detectable samples have been reported both upstream and downstream of LANL, and only one detection occurred downstream of Cochiti Reservoir. No perchlorate data was available in the on-line USGS National Water Quality Database.

The concentrations detected in the river appear to be about two orders of magnitude below the levels at which perchlorate may be regulated in the future based on current regulatory activity. As a result, it appears that it will not be necessary for the San Juan-Chama WTP to include treatment to remove perchlorate.

Treatment for Perchlorate Removal

The chemical formula for perchlorate is ClO_4^- . It is present in water as a monovalent anion and behaves similarly to iodide. Perchlorate salts are relatively soluble in water.

Because perchlorate is an ionic species, ion exchange and reverse osmosis would be effective treatment technologies for removing this contaminant. It has also been shown that perchlorate can be reduced to chloride in biological processes, including biologically active carbon filtration (Brown et al., 2002; Brown et al., 2005). Massachusetts lists ion exchange and California lists biological fluidized bed reactors and ion exchange as best available technology for perchlorate.

The current treatment processes in the San Juan-Chama WTP are not designed to remove soluble monovalent anions, and it is not expected that the treatment plant will be effective at removing perchlorate. The plant contains granular activated carbon (GAC) filters that can support biological activity, suggesting that the capability for perchlorate removal will at least be present in the plant. However, the biological process that converts perchlorate to chloride is a reductive process that is only effective at low oxygen concentrations. The GAC filters in the San Juan-Chama WTP are immediately downstream of the ozone contactor, so it is possible that the low-oxygen conditions necessary for perchlorate removal will not be present in the San Juan-Chama WTP filters.

Summary and Recommendations for Perchlorate Removal

Since the measured concentrations are low compared to the potential regulatory limit for perchlorate, treatment should not be required in the San Juan-Chama WTP. No changes to the design or operation of the San Juan-Chama WTP for perchlorate removal are recommended based on this review. Although currently unnecessary, an additional process such as ion exchange or reverse osmosis could be added if perchlorate became a concern in the future.

Distribution System Corrosion Potential

After treatment, water is transmitted to customers through the distribution system. The water can interact with materials in the distribution system, causing precipitation or dissolution of minerals, formation of biofilms, deposition of solids, or corrosion of metals. The specific interactions that take place depend on water quality. Over long periods of time, the condition of the distribution system can essentially be in equilibrium with the average water quality. Sometimes the condition of the distribution system can become poor (accumulated deposits, precipitates, corroded pipes, etc) but the distributed water quality remains high because equilibrium prevents the poor pipe condition from affecting the water quality. Distribution of water with different quality can upset that equilibrium and interactions between the water and distribution system materials can severely degrade water quality. Different water quality can result from seasonal changes in a source water or a switch to a new source water. Such interactions have happened in other communities. A 1997 survey found that “the most common distribution system problem is corrosion of cast-iron pipe” (McNeill and Edwards, 2001, citing Bray, 1997). As an extreme example, Tucson switched from historic use of native groundwater as a drinking water source to Colorado River water transmitted to the community through the Central Arizona Project in 1992. Shortly after the state-of-the-art surface water treatment plant started operation, residents in certain portions of the service area started complaining of highly colored water. The colored water was caused by release of corrosion products that had formed on the inner surfaces of iron pipe in the distribution system. The utility attempted various combinations of pH adjustment and zinc orthophosphate corrosion inhibitor over the next two years, but was unsuccessful in controlling the problem. In 1994, the plant (a 77 million dollar investment) was shut down (Price and Jefferson, 1997).

It is important to note that distribution system problems are not limited to situations where a water utility puts a new water source into service. Corrosion and other problems are common in distribution systems, and many utilities report periodic, seasonal, or even continuous problems even without evident changes in water quality. No utility is completely immune from customer complaints about perceived water quality problems. Albuquerque has occasionally received customer complaints about perceived water quality in the past (Robinson, personal communication) and is likely to receive complaints in the future regardless of the transition from ground water to surface water. Iron corrosion problems have been noted in the Walker well field and some other wells within the system, with customer complaints in the Cherry Hills area as recently as late 2006 (Olsen, personal communication). The purpose of this section of the report is to review a previous USGS report on water quality, summarize factors that can contribute to corrosion and/or iron release from corroded pipes, summarize differences in water quality between ground water and river water in the Albuquerque area, and provide recommendations for monitoring the water quality in the distribution system after river water is introduced.

Review and Summary of the USGS Study

The scope for this independent review of water quality issues included a review of a study prepared by the USGS for the Water Utility Authority on the differences in water quality between the existing ground water supplies and the new surface water supply (Bexfield and Anderholm, 2008). That study included testing of precipitates within the existing distribution system and equilibrium geochemical modeling of precipitation/dissolution tendencies.

Composition of solids found in the distribution system. Samples of minerals and/or other deposits with the distribution system were collected from three storage tanks, two segments of iron or steel distribution piping, and one residential water heater. Mineralogy of these deposits was determined by X-ray diffraction and chemical composition was determined by inductively-coupled plasma-mass spectrometry (ICPMS). Mineralogically, most deposits fell into one of three categories: silicon-based deposits (clay, quartz, feldspar, potassium feldspar, plagioclase, and possibly zeolite), calcium-based deposits (calcite, aragonite, and possibly gypsum), and iron-based deposits (goethite, hematite, lepidocrocite, and amorphous iron or possibly amorphous iron oxides). Additional deposits that did not fit one of these three categories included magnesite (MgCO_3), rhodochrosite (MnCO_3), gibbsite ($\text{Al}(\text{OH})_3$), and possibly hornblende (containing Ca, Fe, Mg, and Si). Only one sample from a pipe was analyzed for chemical composition, and that sample was found to be 44 percent iron, less than 1 percent each of seven other elements, and trace amounts of 19 other elements. Less than half of the mass was accounted for by the ICPMS analysis, and the remaining mass could be silicon, carbon, oxygen, sulfur, or hydrogen, none of which is detected by ICPMS (the method detects metals). The iron-based mineral detected in that sample was goethite (FeOOH). FeOOH is 63 percent iron by mass. The iron content of other iron-based minerals that have been found in drinking water distribution systems are hematite (70 percent iron), lepidocrocite (63 percent), amorphous iron hydroxide (52 percent), FeCO_3 (48 percent), and FeS_2 (47 percent). It should be noted that the iron content of each of these compounds would be lower if they contained chemically-bound water.

Some iron-containing minerals were found to contain relatively high concentrations of some metals, including arsenic, chromium, copper, manganese, lead, and zinc. Bexfield and Anderholm indicated that these minerals were not likely to dissolve and release their constituents into the bulk water, based on the calculation of saturation indices. However, other studies have found the release of trace contaminants can occur when dissolution of corrosion scale in iron pipes occurs, and that iron release from corrosion scale is affected by many water quality parameters.

An evaluation of the quantity of solids deposited in various locations in the distribution system was not within the scope of the USGS study.

Modeling results. The results from water quality modeling in the USGS study included two components. First, saturation indices for a wide variety of minerals were compared between representative ground water, surface water, and ground/surface water blends. Saturation indices predict the tendency of individual minerals to either precipitate or dissolve, with a positive value predicting precipitation and a negative value predicting dissolution. The modeling software is strictly an equilibrium-based model and therefore the saturation indices predict changes that would need to occur for the given water chemistry to reach equilibrium. However, USGS supplemented the model results with knowledge and experience about the actual minerals that would likely precipitate or dissolve in this system based on kinetic and geochemical considerations. The second component included in the modeling results was the calculation of corrosion indices for representative ground water, surface water, and ground/surface water blends.

Water quality is not uniform throughout the distribution system, but varies by location because the water quality from individual wells is influenced by local geology. Water quality also changes over time due to changes in pumping rates or other factors that influence the rate and direction of flow through the aquifer. To capture this variation, the USGS analyzed existing water quality data from wells and previous reports, and carefully selected 10 groundwater samples that represented the variation in water quality expected throughout the distribution system. Similarly, they reviewed existing water quality data from the Rio Grande and selected eight sampling dates that represented the variation in water quality expected in the river. In both cases, individual water quality parameters were adjusted if necessary to make them representative of longer-term conditions observed in either ground water or river samples.

Based on saturation indices, the report concluded that the minerals most likely to either dissolve or newly precipitate when the water quality changes from groundwater to surface water were calcium carbonates (particularly calcite and aragonite).

The USGS report concluded that saturation indices for aragonite and calcite in river water are similar in magnitude to those for groundwater. However, for several samples, saturation indices were slightly positive for groundwater and slightly negative for surface water, indicating there may be a tendency for dissolution of scale if water is distributed as modeled in the USGS report. Calcium carbonate saturation could easily be adjusted by delivering water at a slightly higher pH. In fact, the Water Utility Authority has indicated an intent to distribute water at a pH = 8.3 (Chwirka, personal communication), whereas the modeling in the USGS report used a pH = 8.0.

For iron-based minerals, the USGS report suggested that saturation indices in treated surface water are 3 to 4 orders of magnitude higher than they are in the groundwater. Thus, the report modeling results do not reflect any possibility for iron-based minerals found in the distribution system to dissolve and affect water quality. The report noted that the high potential for precipitation of iron-based minerals was due to the addition of ferric chloride at the treatment plant, and noted that if these precipitates formed in the distribution system, the ability of the distribution system to convey water could be reduced over time. It is important to note, however, that the addition of ferric chloride causes the precipitation of ferric hydroxide at the water treatment plant, where it is removed from the water by the sedimentation and filtration processes. Thus, the iron concentrations used in the modeling are not indicative of the conditions that will be present in the distribution system. Furthermore, the source for iron deposits in distribution systems is typically corrosion of iron pipes, not precipitation of iron-based minerals from iron in the source water.

Corrosion indices calculated in the report suggested similarities between surface and ground water. The groundwater had a Langelier Saturation Index (LSI) between -0.6 and 0.5, with only one sample of 10 having a negative LSI value. Positive LSI values indicate a tendency for calcium carbonate to precipitate, whereas negative values reflect the tendency for calcium carbonate to dissolve. Calcium carbonate precipitation is sometimes considered to form a protective coating in the distribution system and prevent corrosion of iron pipe. For surface water, the LSI values ranged from -0.5 to 0.2, with six of eight samples having negative LSI values. Thus, the LSI values indicate that the groundwater has exhibited a slight tendency for calcium carbonate to precipitate and that the surface water may exhibit a slight tendency for

calcium carbonate to dissolve. These results are the same as noted earlier for saturation index values. It should be noted that the deviation from zero in this study was relatively small, and the LSI value can be adjusted by changing the pH of the water.

The Ryznar Stability Index (RSI) is also a corrosion index that predicts the influence of calcium carbonate dissolution on iron corrosion. The RSI values ranged from 7.2 to 9.4 in the groundwater, and from 7.6 to 9.1 in the treated surface water. Bexfield and Anderholm (2008) cite literature from Kingston Technical Software that states that values greater than 8 are fairly corrosive. The groundwater had two values greater than 8, whereas the surface water had six values greater than 8.

The Larson-Skold Index (LKI) compares the concentrations of chloride and sulfate to those of bicarbonate and carbonate, and therefore is not an index of calcium carbonate precipitation like the LSI or RSI. This corrosion index is based on observations that higher concentrations of chloride and sulfate can exacerbate iron corrosion but that higher concentrations of bicarbonate and carbonate can mitigate that effect. Bexfield and Anderholm (2008) cite information from Kingston Technical Software that states values below 0.8 indicate that chloride and sulfate will not interfere with scale formation, whereas higher values can enhance corrosion by interfering with scale formation. In the groundwater samples, the LKI values ranged from 0.4 to 1.1, whereas in the surface water samples the LKI ranged from 2.1 to 3.5.

Bexfield and Anderholm (2008) concluded that the LSI, RSI, and LKI were only slightly different between ground water and surface water, but that the surface water and groundwater/surface water mixtures could be somewhat more corrosive than the groundwater.

Corrosion Potential in Water Distribution Systems

The USGS study provides valuable insight into possible interactions between the river water and the distribution system, based primarily on a detailed analysis of the precipitation/dissolution tendencies of minerals composed of the various chemical species actually present in the groundwater and river water. However, the possible factors that can degrade water quality in the distribution system are more complex than what was included in the scope of that study. Some factors that contribute to the complexity include (1) corrosion is an electrochemical process and can therefore be caused by electrochemical effects as well as solubility effects, (2) because of the electrochemical effects, ions such as chloride and sulfate that are not strictly involved in precipitation/dissolution reactions can affect corrosion or iron release even beyond their possible participation as complexing ions, (3) tubercles that form on the inside of iron/steel pipes can contain thick layers that have microenvironments that are substantially different from the bulk water, including different oxidation states, (4) microorganisms residing in the tubercles can mediate redox and/or dissolution reactions or change the water chemistry locally, factors that are not reflected in equilibrium geochemistry models, (5) corrosion of iron surfaces and iron release from existing corroded surfaces are distinctly different processes that may respond differently to changes in water chemistry; i.e., a change in a particular water quality parameter may increase iron corrosion but decrease iron release, and (6) corrosion and iron release can be affected by fluid movement, changes in velocity or direction, and periods of stagnant versus flowing water. Furthermore, the Water Utility Authority has the flexibility to adjust some water chemistry

parameters at the treatment facility, so these parameters in the river water could be adjusted to be more similar to the groundwater (a prime example being pH).

The corrosion of iron in water distribution systems and its ability to degrade water quality is a multi-step process. First, corrosion at the pipe wall can convert metallic iron to corrosion products. If the corrosion products are released to the bulk water in either soluble or particulate form, the water quality can be degraded. However, it is common for the corrosion to produce solid products that accumulate at the pipe surface in thick iron-based scale known as tubercles. The loss of metallic iron can reduce the structural integrity of the pipe and cause leaks, and formation of tubercles can increase head loss and reduce the hydraulic capacity of the pipe, but both of these processes can proceed for long periods of time (decades) without degrading water quality. A final step necessary for the degradation of water quality is the release of iron into the bulk water in either soluble or particulate form. As noted above, the water quality that promotes corrosion and tubercle formation and the water quality that promotes iron release from existing tubercles are not necessarily the same. Thus, a transition from water that promotes tubercle formation to water that promotes iron release is perhaps the worst combination from a water quality perspective.

Characteristics of Iron Scale on Corroded Iron Pipe

Burlingame et al. (2006) provide an excellent overview of the nature and composition of iron scale. The following discussion is summarized primarily from that article. Corrosion scale typically contains three distinctly different layers. Throughout these layers, many different amorphous and mineral forms of iron can be present, including goethite, lepidocrocite, magnetite, siderite, maghemite, ferrous hydroxide, ferric hydroxide, and others. The layer closest to the pipe wall is typically anaerobic and composed of ferrous iron (Fe^{2+}) compounds. Oxidants such as chlorine or oxygen typically do not penetrate to this layer. The layer is generally porous and anions such as chloride can diffuse into it, which increases the rate of corrosion by promoting the release of ferrous iron (Fe^{2+}) from the pipe wall in order to maintain electroneutrality. This layer can harbor anaerobic bacteria, including sulfate-reducing bacteria that produce hydrogen sulfide. Sulfate-reducing bacteria can have a significant effect on the rate of corrosion and cause the formation of iron sulfide precipitates. While the observed effect of biological activity on iron release has been mixed in previous literature, the effect is generally unfavorable in most situations.

Above the first layer is a series of one or more shell-like intermediate layers. These layers are composed mostly of ferric iron (Fe^{3+}) compounds. The composition of these layers are affected by the availability of oxidants (oxygen or chlorine) diffusing from the bulk water. If insufficient oxidant concentrations are present, which may occur under conditions of stagnant or low flow of water, the layers will tend to be more porous and allow ferrous iron to diffuse from the deeper layers out into the bulk water, where it can oxidize and form precipitates that degrade water quality. Lack of oxidant will also produce anoxic zones in this layer, allowing ferric iron to convert to ferrous iron that then diffuses into the bulk water and causes water quality problems.

Above the intermediate shell-like layers is an outer surface layer that is in contact with the bulk water. Because of the oxidants present in the bulk water, this layer is composed only of ferric iron compounds. Oxidation by oxygen forms a harder scale than oxidation by chlorine. This

layer is loosely held to the underlying layer, so changes in flow direction or rate can scour particles from this layer and degrade water quality.

Chemical Factors that Influence Iron Release from Corroded Pipes

As noted earlier, the water quality that promotes corrosion and tubercle formation is not necessarily the same as the water quality that promotes iron release. McNeill and Edwards (2001) recently reviewed several hundred peer-reviewed articles that investigated various aspects of iron corrosion in water distribution systems. For the Water Utility Authority's distribution system, iron release is more relevant with respect to introduction of a new water source into the distribution system. McNeill and Edwards found many instances in previous research where a change in a particular water quality parameter did not consistently lead to the same result with respect to iron release. However, factors that generally lead to a decrease in iron release from corroded pipe were an increase in pH, increase in alkalinity, and increase in the presence of siderite or green rust in the tubercles. Factors that had detrimental effects on water quality included an increase in the disinfectant residual, increase in stagnation time, increase in microbial activity, increase in temperature, and increase in natural organic matter. Factors that had mixed results or results that varied between studies included increases in dissolved oxygen, increases in chloride and sulfate concentrations, presence of calcite scale, use of phosphate inhibitors, presence of silicates, and changes in water velocity. McNeill and Edwards made a particular note that indices such as the Langelier Index or Ryznar Stability Index have been "improperly applied as the cure-all for solving corrosion problems," and that they are not sufficient predictors of corrosion problems and should not be relied upon (McNeill and Edwards, 2001).

The reason that corrosion indices are insufficient and that many factors have had results that varied from one study to the next is because of interactions between parameters. For instance, a change in water quality that results in increases in both pH and NOM may find the negative influence of NOM outweighing the positive influence of higher pH, resulting in a degradation of water quality even though improved water quality was expected. Similarly, although increasing disinfectant residual generally increases corrosion rates, if microbial activity is substantially implicated in corrosion, an increase in disinfectant residual may decrease microbial activity and decrease corrosion rates. Thus, it is important to evaluate changes in water quality in aggregate rather than focus on individual parameters, or to consider physical studies.

Other recent research has come to similar conclusions about the impact of various water quality parameters on iron release. Burlingame et al. (2006) discussed the impact of water quality in the context of the composition and structure of tubercles that was summarized earlier. The reason that higher pH reduces iron release is because it decreases the solubility of ferrous iron. Chloride can diffuse through the outer layers, where it can accumulate near the pipe surface and increase the corrosion rate. In contrast, orthophosphates can form insoluble iron-phosphate compounds that reduce the rate of diffusion of ferrous iron from the scale, decreasing iron release.

Several recent studies have studied iron release using pipe loops constructed of actual corroded iron pipes removed from distribution systems. A study funded by the American Water Works Association Research Foundation (AWWARF) used 90-100 year-old unlined cast iron pipe that was removed from the Massachusetts Water Resources Authority (MWRA) distribution system

in Boston (Sarin et al., 2003; Sarin et al., 2004b). This study found that the conditions that reduced iron release included high pH (pH = 9.5), higher alkalinity (58 - 65 mg/L as CaCO₃ compared to 10 - 15 mg/L as CaCO₃), and dosing of orthophosphate (3 mg/L) under specific pH and alkalinity conditions. Varying dissolved oxygen concentrations over the range of 1 to 28 mg/L (high doses associated with ozone disinfection) did not substantially change iron release. It was noted that the reduction in iron release was a slow process, and that continuation of treatment practices over a several-month period was necessary to achieve steady-state iron release results. It was also noted that maintaining a constant pH and alkalinity caused iron release to decrease over time even when pH = 7.6, alkalinity = 30 mg/L as CaCO₃, and no phosphate was used.

A following study using 70-year-old unlined galvanized iron pipe from the Champaign, IL distribution system found that higher oxygen concentration (14 mg/L) decreased iron release during periods of stagnation (Sarin et al., 2004a). In the presence of dissolved oxygen, iron release was greater during period of stagnation than during periods of flowing water. The study demonstrated that the release of iron from corroded pipes was primarily in the form of ferrous iron, which then oxidizes and precipitates to ferric iron and causes red water.

Another similar study, also funded by AWWARF, was conducted using pipe loops constructed of corroded pipe removed from several water distribution systems in the Tampa, Florida area (Imran et al., 2005; Imran et al., 2006; Mutoti et al., 2007). The feed water for this study consisted of blends of water ranging from groundwater, softened groundwater, treated fresh surface water, and desalinated seawater. Apparent color was used as a surrogate for iron release. A multi-variate nonlinear regression analysis indicated that the significant factors that reduced color were higher alkalinity and lower chloride, sulfate, sodium, dissolved oxygen and temperature. Calcium and pH were not identified as significant variables, but that was due to the experimental design (all feed waters had been adjusted to a similar pH, ranging from 7.9 to 8.3). Alkalinity above 80 mg/L as CaCO₃ was identified as the most significant controllable water quality parameter for reducing iron release.

After the Tucson system failure, a study was performed using 50-year-old, 1-inch diameter corroded galvanized iron pipes removed from the Tucson distribution system (Chowdhury et al., 1998; Swanson et al., 1998). This study found that raising the pH to near 8.5 and feeding 3 to 5 mg/L of polyphosphates were effective at reducing iron release from the corroded pipes, with pH adjustment having the most significant effect.

These studies confirm that dissolution of existing iron scale is an important method of iron release from corroded iron pipes. However, the structure and composition of the tubercles can influence how iron is released and what water quality changes will be important. The experience from these previous studies indicates that three factors can lead to corrosion-related degradation of water quality in the Water Utility Authority's distribution system:

1. Presence of tuberculated iron pipe in the distribution system.
2. Changes in water quality that increase the tendency for iron release from tubercles.
3. Changes in hydraulic conditions that physically remove particles from the existing scale.

Each of these factors is examined in detail in the following sections.

Composition of Pipes in the Water Utility Authority’s Distribution System

The distribution system managed by the Water Utility Authority contains nearly 3,000 miles of pipe. A breakdown of pipe materials is shown in Table 11. The vast majority of the pipe in the distribution system is cast or ductile iron (40%), PVC (36%), or cement/concrete cylinder pipe (21%). Pipe material that has a potential for corroding and affecting water quality is unlined pipe composed of iron or steel. Although the most prevalent material in the distribution system is cast iron, most of the cast iron pipe installed in municipal distribution systems in the United States since the early 1950s is mortar lined (DIPRA, 2008). The mortar lining effectively prevents corrosion. Before the 1950s, gray iron pipe was sold with a hot-dip bituminous coating and lining; pipe of that vintage may be more susceptible to corrosion. Water authority personnel have indicated that their experience is that most if not all of the cast iron pipe in the Albuquerque distribution system is mortar lined, with the possible exception of cast-iron pipe less than 4 inches in diameter (Stomp, Robinson, and Olsen; personal communication). Steel pipe is mostly lined except some pipe installed before 1945, and the galvanized steel pipe is unlined.

The 3 miles of unlined galvanized steel pipe is probably the most susceptible to corrosion of any material in the distribution system, along with any portions of the cast iron or steel pipe that may be unlined. While these pipes appear to represent less than 1 percent of the distribution system, the potential for water quality changes to occur in these areas does exist. Small diameter pipe generally serves individual neighborhoods, so red water problems may become evident in individual neighborhoods served by old galvanized steel pipe. Susceptibility to corrosion is possible regardless of whether ground water or surface water is being used. Because of this, the Water Utility Authority already has an ongoing program to remove old steel pipe and replace it with PVC pipe.

Differences in Water Quality between Groundwater and Surface Water

As noted earlier, water quality varies throughout the distribution system. The USGS conducted a detailed study of the groundwater and concluded that the aquifer could be divided into 5 distinct

Table 11 – Pipe Materials and Estimated Amount in the Albuquerque Water Distribution System.

Pipe material	Length of pipe (miles)
Cast Iron	1,082
PVC	1,073
Asbestos Cement	400
Concrete Cylinder	199
Ductile Iron	94
Steel	82
Linear Wrapped Steel	7
Reinforced Concrete	6
Galvanized Steel	3
Copper	0.06
Corrugated Metal	0.04
Total	2,945

Ref: (ABCWUA, 2008c)

Table 12 – Median Values for Water Quality Parameters in Albuquerque-area Groundwater and Surface Water

Parameter	Groundwater					River water	
	West	Central	East Mesa	North-east	Mountain Front	Pilot raw	Pilot treated
pH	8.7	7.8	7.8	7.6	7.8	8.2	8.2
Alkalinity (mg/L as CaCO ₃)	121	104	89	107	90	120	92
Bicarbonate (mg/L)	142	126	108	130	109	127	93
Carbonate (mg/L)	2.95	0.33	0.28	0.21	0.29	1.7	0.94
Chloride (mg/L)	9.9	12.4	16.6	56.9	21.4	10.0	48.0
Sulfate (mg/L)	67.9	67.1	33.6	31.7	20.5	50.1	48.6
Sodium (mg/L)	104	48.8	23.9	44.1	28.3	22.1	24.7
Calcium (mg/L)	3.9	29.4	41.1	52	39.7	41.2	47.6
Silica (mg/L as SiO ₂)	32.4	66	34	34.6	28.6	30.7	16.2
Conductivity (µS/cm)	484	432	357	473	331	342	418
TDS (mg/L)	324	308	231	300	210	276	280
Larson-Skold Index	0.70	0.85	0.66	1.06	0.57	0.64	1.55

Refs: (Bexfield and Anderholm, 2002; ABCWUA, 2008b; USGS, 2008)

zones, each with different water quality (Bexfield and Anderholm, 2002). The median concentration of key water quality parameters from each of these zones, based on well data from 1988 to 1997, is summarized in Table 12. In addition, median water quality data of raw and treated water from the pilot plant operated by the Water Utility Authority in 2007 is also shown in Table 12 (ABCWUA, 2008b). Concentrations of individual parameters in surface water delivered to the distribution system will change in response to addition of chemicals at the treatment plant, as follows:

- Addition of ferric chloride for coagulation will increase the chloride concentration and decrease the pH and alkalinity.
- Addition of sulfuric acid for pH adjustment will increase the sulfate concentration and decrease the pH and alkalinity.
- Aeration at low pH for removal of dissolved oxygen will increase the pH and decrease dissolved inorganic carbon (carbonate and bicarbonate) concentrations.
- Addition of hydrated lime for pH adjustment will increase the pH, alkalinity, and calcium concentration.

pH, alkalinity, and dissolved inorganic carbon: Groundwater in Albuquerque typically has a median pH of about 7.8, except on the west side, where the median pH is 8.7. Median alkalinity ranges from 89 mg/L as CaCO₃ to 121 mg/L as CaCO₃. The river water has a slightly higher pH

and similar alkalinity. Bicarbonate and carbonate concentrations are relatively similar between area groundwater and surface water. However, it is important to note that the values for these parameters are impacted by water treatment practices. pH is impacted by the addition of ferric chloride, sulfuric acid, and hydrated lime, by aeration, and to a minor extent by the addition of sodium hypochlorite and fluosilicic acid. Alkalinity is impacted by acid or base chemical addition, and the bicarbonate and carbonate concentrations are impacted by aeration. Addition of 30 mg/L of ferric chloride to the river water would decrease the pH to about 6.8 and the alkalinity by 28 mg/L as CaCO₃. Addition of sulfuric acid to decrease the pH to 6.0 would reduce the total alkalinity further, resulting in an alkalinity of about 30 mg/L as CaCO₃. Aeration in the ozone contactor will decrease the bicarbonate and carbonate concentrations, but the extent of the reduction will depend on the pH of the water in the ozone contactor. Aeration at pH = 6.0 will strip significantly more carbonate from solution than would aeration at pH = 6.8.

pH and alkalinity are important parameters in corrosion and iron release. The Water Utility Authority has indicated intent to maintain the pH in the surface water delivered to the distribution system at 8.3 (Chwirka, personal communication). Higher pH has generally been shown to minimize iron release from corroded pipes. Nevertheless, the alkalinity and carbonate concentration of the delivered water also needs to be considered. Sarin et al. (2003) found that a reduction in alkalinity and carbonate caused an almost immediate increase of iron release of 50 to 250 percent, even when pH was held constant.

Chloride and sulfate: Chloride concentrations are generally low in the groundwater, with median concentrations ranging from 10 to 20 mg/L throughout the city except for a median value of 56.9 mg/L in the northeast. Native Rio Grande water has a lower chloride concentration than all areas of the city (median concentration 10 mg/L), but the addition of treatment chemicals in the pilot plant increased the median chloride concentrations to 48 mg/L. Thus, although the median chloride concentration in the river is lower than in all regions of the city groundwater, after treatment the chloride concentration will be higher than in the groundwater in all areas except the northeast.

Median sulfate concentrations are around 67 mg/L in the west and central portions of the city and between 20 and 30 mg/L in the remainder of the city. The native median sulfate concentration in the river is between the median concentrations in the groundwater. Although sulfuric acid was not used during the pilot plant operation, it is a treatment option in the full-scale facility. The use of sulfuric acid for enhanced coagulation would lead to an increase in the sulfate concentration. The sulfate concentration of the delivered water will depend on the acid dose.

Increases in chloride and sulfate have been shown to increase iron release from corroded pipe, so the increase of chloride concentration in most areas of the city may cause an increase in the potential for iron release.

Calcium: Calcium concentrations in groundwater throughout the city typically range from 30 to 50 mg/L with the exception of the western area, which has a median calcium concentration of less than 4 mg/L. The raw river water has a median calcium concentration of about 40 mg/L, and the concentration in surface water delivered to the distribution system would be somewhat higher because of lime addition to the water for pH adjustment.

The impact of calcium on corrosion and iron release is usually discussed in the context of corrosion indices such as the Langelier Index or the Ryznar Stability Index. While higher calcium concentration or positive values of corrosion indices have in some instances had beneficial effects, numerous studies have shown no correlation between these indices and iron release. As a result, many corrosion experts believe that the use of calcium-based corrosion indices should be abandoned.

Silica: The median silica concentration throughout the city is about 30 mg/L as SiO₂, except in the central region, which has a median concentration of 66 mg/L as SiO₂. The raw river water has a similar silica concentration to the groundwater. The silica concentration in the treated water from the pilot plant had a lower concentration, possibly from the removal of particulate or colloidal silica during the coagulation/filtration process. Silica has been observed to decrease the oxidation of ferrous iron and therefore reduce the rate of iron release from corroded pipes. However, it has also been reported that the beneficial effect ceases if silica feeding stops. Thus, a reduction in the silica concentration resulting from transition from ground water to surface water has the potential to increase iron release from corroded pipe.

TOC: The groundwater in the Albuquerque area is reported to have a total organic carbon concentration of less than 1 mg/L. The TOC of Rio Grande water is considerably higher but will be reduced during enhanced coagulation and ozone/biofiltration at the treatment plant, as noted earlier in this report. Nevertheless, the TOC concentration in the delivered surface water will likely be higher than in the groundwater, particularly after the adsorption capacity of the GAC has been exhausted. Organic carbon can complex metal ions, which may lead to an increase of iron release from corroded pipes.

Dissolved oxygen: The dissolved oxygen (DO) of the delivered surface water will be near the saturation concentration (around 8 mg/L) after being oxygenated in the ozone contactor. Plummer, et al. (2004) report that the DO concentration in the groundwater is generally above 5 mg/L near the Sandia Mountains, between 0.5 and 5 mg/L in intermediate regions of the city, and below 0.5 mg/L near the Rio Grande. Studies on the impact of DO on iron release has been mixed, although recent studies with actual corroded pipe from distribution system has indicated either positive or neutral effects from higher DO concentration.

Water Quality Changes in Tucson, Arizona

For comparison to the change in water supply in Albuquerque, groundwater and surface water from Tucson, AZ is shown in Table 13. The water quality shown in Table 13 is from a study reported in 1998, 4 years after the surface water treatment plant had been shut down (Chowdhury et al., 1998). Although the water quality shown here is not necessarily the same as what was in the Tucson distribution system before and after the surface water treatment plant started up, it should be indicative of the basic differences between the two water sources.

In Tucson, the groundwater had a pH of 8.1 and alkalinity of 131 mg/L as CaCO₃, whereas the surface water had a pH of 7.5 and alkalinity of 116 mg/L as CaCO₃. Price and Jefferson (1997) report that the pH of the water delivered to the distribution system varied from about 7.4 to 8.0 with a couple periods of higher or lower pH during the 23 months when the surface water treatment plant was operational. The surface water was considerably harder than the

groundwater, with a total hardness of 317 mg/L as CaCO₃ compared to 81 mg/L as CaCO₃. Hard water can cause calcite scale formation, although the Langelier Index was close to zero for both the groundwater and surface water. The significantly higher hardness, similarity in alkalinity, and similarity in Langelier Index could be interpreted as indications that the surface water should not have been significantly more corrosive than the groundwater. Tucson may be an example where the Langelier Index was not an adequate indicator of the potential for iron release.

Differences in other water quality parameters between Tucson groundwater and surface water are much more significant. The chloride and sulfate concentrations were 18 and 10 mg/L in the groundwater, compared to 121 and 339 mg/L in the surface water. Higher chloride and sulfate concentrations have been implicated in higher iron release from iron tubercles, and are the basis for the Larson-Skold corrosion index. A calculation of the LKI based on values in Table 13 indicates that the value increased by more than an order of magnitude, from 0.27 to 4.52. Based on this information, the substantial increase in chloride and sulfate concentrations may have had a more significant impact on iron release in the Tucson distribution system than the change in pH. Similarly, the silica concentration decreased from 14 mg/L as SiO₂ to 2 mg/L as SiO₂ and TOC increased from 0.3 to 14.5 mg/L when surface water was introduced. These factors may have had a substantial impact on the rate of iron release when the new plant started up.

An examination of Tucson's water quality data contains both encouraging and cautionary messages for the Water Utility Authority. First, it is evident by comparing Tables 12 and 13 that the change in water quality with the delivery of surface water was much greater in Tucson than it will be in Albuquerque. Thus, it might be expected that the potential for iron release from water quality changes should be lower in Albuquerque than it was in Tucson. However, the cautionary

Table 13 – Water Quality in Tucson Groundwater and Surface Water.

Parameter	Tucson groundwater	Tucson surface water
pH	8.1	7.5
Alkalinity (mg/L as CaCO ₃)	131	116
Bicarbonate (mg/L)	158	141
Chloride (mg/L)	18	121
Sulfate (mg/L)	10	339
Sodium (mg/L)	40	145
Calcium (mg/L)	26	64
Magnesium (mg/L)	3.9	38
Hardness (mg/L as CaCO ₃)	81	317
Silica (mg/L as SiO ₂)	14	2
TOC (mg/L)	0.3	14.5
TDS (mg/L)	270	850
Larson Index	0.27	4.52

Adapted from (Chowdhury et al., 1998).

message is that it is evident that many parameters other than pH may have contributed to the problems in Tucson, and a focus on pH as a single parameter for controlling corrosion may overlook other water quality parameters that should be considered.

Changes in Hydraulic Conditions

Water Utility Authority personnel report that the surface water treatment and distribution system has been designed to minimize changes in hydraulic conditions (Chwirka, personal communication). Water from the surface water treatment plant will be piped to existing reservoirs that currently receive water from the Water Utility Authority's wells. From there, surface water will be distributed using identical flow patterns as the existing well water. The only pipes that will experience reverse flow from the current configuration are large transmission mains that will carry water from reservoirs in lower pressure zones to reservoirs in higher-pressure zones. These pipes are large-diameter mortar-lined pipes that are not expected to have significant tuberculation that is characteristic of small-diameter unlined iron pipe. Thus, changes in hydraulic conditions are not expected to have a significant impact on iron release in the Water Utility Authority's distribution system. In contrast, changes in hydraulic conditions have anecdotally been reported to be a factor in the iron release problems in the Tucson distribution system (Chowdhury; personal communication).

Summary and Recommendations Relative to Distribution System Iron Release

The Water Utility Authority commissioned a study by the USGS to evaluate changes in water quality between the current groundwater and new surface water supply in the distribution system. That study focused primarily on the precipitation/dissolution tendencies of minerals based on equilibrium modeling and the calculation of saturation indices. It found that the groundwater and surface water were similar with respect to the minerals that have a tendency to precipitate or dissolve. None of the iron-based minerals found during sampling of deposits in the distribution system were predicted to dissolve when exposed to surface water. The study also calculated three common corrosion indices and concluded that the groundwater and surface water differ only slightly in corrosion potential, but that the surface water is somewhat more corrosive than the groundwater.

The USGS study provided valuable insight into the differences between the former and future water supplies but the scope did not include the evaluation of all impacts of changes in distribution system water quality. Specifically, iron release from corroded pipes can be caused by factors other than precipitation/dissolution chemistry. Factors such as electrochemistry, anaerobic conditions within the iron scale, microorganisms living in the iron scale, and hydraulic conditions can influence iron release. Iron release can significantly degrade water quality and was responsible for the problems in Tucson that led to abandoning a surface water treatment plant two years after it went online.

Three factors are important with regard to iron release in the Water Utility Authority's water distribution system. First, corroded iron pipe must be present. The distribution system contains almost 3,000 miles of pipe. Of this, 82 miles are steel pipe and 3 miles are galvanized steel pipe that may be unlined and corroded. While these quantities are small compared to the total amount of pipe, it is nevertheless sufficient to cause iron release problems in specific parts of the

community. Second, changes in hydraulic conditions can scour loosely-held scale from corroded surfaces. Water Utility Authority personnel have stated that the introduction of surface water into the distribution system has been designed to minimize changes in flow conditions and they do not expect hydraulics to be significant in causing problems in the distribution system. Third, changes in water quality can increase iron release from the existing corroded surfaces. Corrosion indices can provide insight but are not sufficient for evaluating the impact of changes in water quality. Water quality parameters that may tend to minimize iron release when the surface water is introduced into the Albuquerque distribution system include maintenance of pH at or above the current pH and a likely increase in dissolved oxygen. Water quality parameters that might contribute to an increase in iron release include an increase in chloride concentration, decrease in alkalinity, decrease in dissolved inorganic carbon, increase in TOC, and decrease in silica. A release of iron from tubercles in the distribution system may also release trace metals that were detected in iron-based deposits by the USGS, including arsenic, chromium, copper, manganese, lead, and zinc. It is important to note, however, the difference in water quality between the ground water and surface water in Albuquerque is much smaller than it was in Tucson.

The complexity of factors that can contribute to iron release suggest that the Water Utility Authority should proceed cautiously to minimize potential water quality problems associated with iron release. The following actions should be considered as part of a program to address this issue:

1. Evaluate the condition of existing pipe by examining the interior surfaces of galvanized steel or unlined iron pipe whenever it is exposed for maintenance, repair, or replacement. If existing pipes are heavily tuberculated, iron release problems following the transition to surface water may be more likely. If, however, the historic groundwater was relatively non-corrosive and the interior of the pipes contain calcium scale or other protective coatings and are only lightly corroded, red water problems may be less of a concern.
2. Continue the existing pipe replacement program. Water Utility Authority personnel have stated that a program is in place to replace old, corroded pipe, and that about \$6 million dollars per year is spent on this program (Robinson, personal communication). Continued investment in pipe replacement with a focus on areas of the community with the most heavily corroded pipe would be beneficial for minimizing water quality problems.
3. Minimize hydraulic changes to the distribution system. It has already been noted that the system has been designed to minimize hydraulic changes, but the operation of the system should be revisited specifically with regard to the potential for scouring of loosely-held iron scale in corroded pipes. If the only pipes subjected to flow reversal during the introduction of surface water are large-diameter mortar-lined pipes, and all distribution of surface water from reservoirs will follow historic flow patterns, hydraulic changes may be a non-issue.
4. Minimize rapid changes in water quality. A strategy of gradually blending surface water into the system over an extended period of time, perhaps a couple years, should help minimize iron release problems. The Water Utility Authority has indicated intent to operate the San Juan-Chama WTP at 25 percent of capacity during the first year and 50 percent of capacity

- during the second year. In addition, the surface water should mix with groundwater in the system reservoirs. This strategy may be of substantial benefit for minimizing iron release.
5. Consider the impact of alkalinity and dissolved inorganic carbon on iron release. The current operational strategy for the San Juan-Chama WTP is to lower the pH during coagulation, maintain the low pH during ozonation and filtration, and raise the pH with lime at the end of the treatment process. This strategy will reduce the concentration of dissolved inorganic carbon via stripping of carbon dioxide during the aeration process to reduce the dissolved oxygen concentration. Raising the pH prior to aeration would reduce the stripping of carbon dioxide and increase the alkalinity and buffering of the distributed water. Alternatively, pH adjustment with sodium carbonate instead of lime would increase the dissolved inorganic carbon concentration. It should be noted that modeling of changes in water quality during treatment is beyond the scope of this report, and the expected alkalinity and dissolved inorganic carbon from various treatment strategies has not been determined. Furthermore, any change in treatment that results in a change in pH or other water quality parameters would need to consider the impact on other treatment goals, such as TOC removal or DBP formation.
 6. Consider sampling in the distribution system to provide early detection of potential iron release problems. Sampling for iron concentration and other parameters in areas of the distribution system with old corroded pipe may reveal changes in water quality before they are detectable by customers. Previous studies have used turbidity and apparent color as useful surrogates for iron concentration.
 7. Review data and information from studies in which corroded pipe from distribution systems was exposed to water of varying quality. AWWARF, of which the Water Utility Authority is a member, has funded many studies of distribution system corrosion. Two recent studies that may be relevant to the current situation are:
 - *Development of Red Water Control Strategies*, by Jonathan Clement, et al. (AWWARF Project Number 368, Order No. 90883).
 - *Effects of Blending on Distribution System Water Quality*, by James S. Taylor, et al. (AWWARF Project #2702, Order No. 91065F).
 8. Consider obtaining guidance on water quality issues from an expert in corrosion and iron release in drinking water distribution systems.
 9. Depending on the information obtained in Numbers 7 and 8 above, consider conducting pipe loop studies using actual corroded pipe from the distribution system, similar to the studies in Boston, Champaign IL, Tampa area, and Tucson that were mentioned in this report. It should be noted, however, that these studies might be time consuming. Although they could not be conducted prior to the plant going on-line, the studies may be able to provide useful information before the plant is operating at full capacity, which is not scheduled to happen for a couple years.

As noted at the beginning of this section, distribution system problems are not limited to situations where a water utility puts a new water source into service, and corrosion is a common problem in water distribution systems. Prevention of iron release should be part of the maintenance program of any water utility to minimize customer complaints about perceived water quality problems. Thus, these recommendations could be considered prudent for any water utility.

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Dr. Kerry J. Howe, P.E., BCEE is a renowned expert in water quality and treatment with more than 20 years of experience in the profession. He has extensive experience in all aspects of water treatment engineering, including treatability studies, regulatory compliance evaluations, facility evaluations, master plans, pilot studies, predesign, detailed design, construction management, and plant startup. His areas of expertise include water chemistry, physical and chemical processes for water and waste treatment, membrane filtration, and reverse osmosis. He has both academic and professional engineering experience, having worked for environmental engineering consulting firms for 12 years before entering academia. While in consulting, he worked on the design of numerous water treatment plants, including a 100-million gallon per day surface water treatment plant (similar in capacity to the ABCWUA San Juan-Chama WTP) for the city of Tulsa, Oklahoma. Since entering academia, his areas of research expertise have been water chemistry, desalination, and membrane technologies for environmental treatment (including microfiltration and reverse osmosis). He has published in a number of environmental engineering journals, serves on national membrane technology committees for the American Water Works Association, and is a co-author of the textbook *Water Treatment: Principles and Design*.

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