C California Urban Water Agencies

Draft Report of
Review of Disinfection Technologies Project

California Urban
Water Agencies

MALCOLM PIRNIE

WQTS

3054004

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1. INTRODUCTION

1.1   Background

The drinking water providers throughout California navigate in an ever-changing landscape. Continued pressures on limited resources compound already challenging treatment and distribution issues. Water purveyors must not only be concerned with providing safe and reliable drinking water to their customers, but must also balance that concern with water management issues and Bay-Delta ecosystem integrity. California Urban Water Agencies (CUWA) has been addressing these issues through research and public education since 1990.

Most recently, water utilities throughout the country have been facing new drinking water regulations. Some of the most challenging are those that provide minimum levels of disinfection that must be provided, while at the same time limiting the by-products from the disinfection process. CUWA member agencies are now complying with the first stage of these Disinfection/Disinfection By-Product (D/DBP) rules, with a second stage pending proposal in late summer 2003. The Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR) may require additional inactivation or removal of *Cryptosporidium,* a chlorine-resistant pathogen. The Stage 2 Disinfection/ Disinfection By-Products (D/DBP) Rule provides maximum limits for chlorinated organic by-products (i.e., total trihalomethanes (TTHMs) and haloacetic acids (HAA5s), measured as a running annual average at each location in the distribution system (rather than averaging all points in the distribution system). Limits for ozone by-products (i.e., bromate) and chlorine dioxide (i.e. chlorite) are also maintained from the first stage of rulemaking.

Systems that would like to control TTHM and HAA5 formation, and at the same time provide additional *Cryptosporidium* inactivation or removal, may need to consider reducing the use of free chlorine by switching to alternative disinfectants such as ozone, or ultraviolet (UV) light for primary disinfection and/or chloramines for residual disinfection. The system may also consider using a physical removal process for chlorine-resistant pathogens (e.g., *Giardia* and *Cryptosporidium*) such as microfiltration (MF) or ultrafiltration (UF). Or a system may consider continuing their use of chorine by removing DBP precursors using treatment technologies such

as enhanced coagulation, granular activated carbon (GAC), powdered activated carbon (PAC), or other advanced process (e.g., magnetized ion exchange [MIEX]).

Water systems may have the opportunity to balance both source water quality and treatment efficiency to meet these challenges. In 1996, CUWA convened an Expert Panel to provide advice on source water quality characteristics that would allow their agencies to meet future drinking water quality goals based upon available technologies. The Record of Decision (ROD) under the CALFED Bay-Delta agreement introduced the concept of "equivalent public health protection" to the source water quality requirements forwarded by the expert panel. Under this concept, the CALFED agencies would achieve their goal of providing safe, reliable and affordable drinking water by either (i) achieving average bromide and TOC concentrations of 50 μg/L and 3.0 mg/L, respectively at Clifton Court Forebay and other central and southern drinking water intakes, or (ii) meeting an equivalent level of public health (ELPH) in terms of treated water quality using a cost-effective combination of alternative source waters, source control and treatment technologies (CALFED-Bay Delta Program, 2000).

CUWA initiated this project to:

- obtain up-to-date information on recent developments in disinfection and treatment technologies, which are specifically applicable to its member agencies for their unique constraints
- perform a preliminary analysis of the technologies that might be suitable for implementation by its member agencies.

Malcolm Pirnie, Inc. and Water Quality & Treatment Solutions, Inc. were retained as consultants to conduct this study.

1.2   Objectives

The main objectives of this study is to review and summarize the recent developments in disinfection and related treatment technologies, focusing on their applicability to source waters specific to CUWA member agencies. It is intended to include the following in this study:

- Determine the role of disinfection and treatment technologies in achieving the ELPH water quality goals, and

- Conduct a quantitative and professional assessment of individual CUWA member needs in terms of water supply options and multi-objective treatment opportunities.

The two key questions that this study will answer are:

- How well can the current treatment plants meet the "equivalent level of public health protection" with existing source water quality in terms of total organic carbon (TOC) and bromide concentrations?

- What additional treatment and disinfection technologies will be required to allow the water treatment plants to meet an "equivalent level of public health protection" with deteriorating raw water quality?

1.3   Defining Water Quality Goals

As a part of this effort, it was necessary to provide some working definition of "equivalent level of public health protection" in lieu of the raw water quality specifically identified in the ROD (i.e., 3 mg/L of TOC and 50 µg/L of bromide). The project team and members of the Project Advisory Committee (representatives of each CUWA member agency) agreed upon conducting the analysis in this effort based on finished water quality goals that are commensurate with the basis of ROD water quality goals. In developing the ROD concept of "equivalent level of public health protection" the previously convened Expert Panel based their recommendations upon the use of :

1) finished water quality goals of 40 µg/L for TTHM, 30 µg/L HAA5 and 5 µg/L for bromate. Various microbial inactivation goals were addressed including 1 and 2 log inactivation of *Giardia*, and 1 log inactivation of *Cryptosporidium*.

2) enhanced coagulation, ozone, GAC or membrane treatment in addition to conventional treatment.

Consequently, finished water quality goals were set at 40 µg/L TTHM, 30 µg/L HAA5, 5 µg/L Bromate and 1 log additional *Cryptosporidium* inactivation were used to conduct the evaluation of ELPH criteria.

1.4   Approach

The overall technical approach included:

- conducting an extensive literature search on technological advances for advanced drinking water treatment processes. This literature review includes information on alternative disinfection practices, DBP precursor removal processes, and *Cryptosporidium* inactivation/removal processes.
- compiling background information for designated treatment plants for each of the participating CUWA members. Twelve months of water quality and operational data were obtained for each of the participating water treatment plants. The Information Collection Rule (ICR) database was also reviewed to augment the data history. The member agencies were also interviewed to understand on-going efforts to upgrade and enhance treatment processes for meeting future regulations.
- assessing the applicability of available technologies for each member agency plant using the Water Treatment Plant (WTP) simulation model. The model was calibrated for each of the selected treatment plants using the historical plant operation and water quality data.

The technology assessments were performed under four scenarios:

Scenario 1 – Baseline: In this scenario, the model was run for the existing/ immediate future facility with existing average source water quality. Finished water quality was compared to the ELPH goals. Any inability to meet specific ELPH goal criteria were identified.

Scenario 2 – Process Improvements: WTP simulations were performed using the existing average source water quality and incorporating additional technologies to meet the ELPH

goals. Member agency representatives provided input regarding their preferred sequence of process improvement based upon their individual conditions. Conceptual costs were developed for the additional technologies once evaluated.

Scenario 3 – Raw Water Quality Boundary: The limiting source water quality that would allow plants to meet the ELPH goals with existing/immediate future facility was determined.

Scenario 4 – Modified Raw Water Quality Boundary: The limiting source water quality to meet ELPH goals, using the selected technology chosen (based on utility prioritization and costs) from those treatment options developed in Scenario 2, was determined.

1.5   Report Organization

This report is organized into the following six sections:

- Section 1 provides a brief description of the background and objectives of the study. The approach followed in conducting this project is also briefly described in this section.

- Section 2 provides a detailed review of the literature to cover new research reports on technologies related to disinfection and DBP precursor removal. A summary is provided at the end of the section that describes the relevance of these new research for the CUWA member agencies.

- Section 3 describes a brief history of the water treatment plant simulation model and provides a description of the method used to calibrated the central tendency model into the site specific versions of the model which are applicable for each of the member agency water plants evaluated during this study.

- Section 4 summarizes the technology assessments performed for the member agency water plants. While performing the technology assessments, the project team considered the priority of technology improvements suggested by the member agencies for their respective plants. Some planning level opinions of probably capital and operational costs for the future technology improvements are also provided in this section.

- Section 5 includes a summary and preliminary observations made during the efforts undertaken in this study.

■ Section 6 lists the references of various technical publications used in this study.

In addition to these sections, the report contains an appendix which captures the project team's activities in performing the technology assessments.

2.0   LITERATURE REVIEW

A literature review was conducted to provide an understanding of the state-of-the-art in drinking water treatment processes. Drinking water related, peer-reviewed articles published during the past five years in the following industry journals were reviewed:

1. *Journal of the American Water Works Association* [American Water Works Association]
2. *Water Research* [International Water Association]
3. *Environmental Science & Technology* [American Chemical Society]
4. *Journal of Environmental Engineering* [American Society of Civil Engineers]
5. *Ozone Science & Engineering* [International Ozone Association]

When there was insufficient material on a specific target technology in these peer-reviewed publications, conference proceedings were also reviewed.

The drinking water technologies were grouped according to the target short-list of treatment technologies identified by the Project Advisory Committee. The list of technologies includes:

- Chlorine/Chloramines
- Chlorine Dioxide
- Ozone
- UV
- MF/UF
- GAC/PAC
- MIEX$^{®}$
- Advances in Conventional Treatment Processes

This section summarizes the recent findings for each of the above categories of technologies. A more detailed literature review citing individual publications and specific findings is provided in Appendix A.

While a large number of drinking water-related articles have been published in the five selected journals over the past five years, the advances in water treatment technology, with the exception of UV irradiation, have been relatively minimal. In some instances, previous advances have

been corroborated and/or verified in additional water sources. Nevertheless, the literature includes many possible improvements to the existing technologies that are worth noting. The following sub-sections summarize these improvements for each of the technologies evaluated in this project.

2.1   Chlorine/Chloramine

The application of chlorine and/or chloramines to drinking water has been well characterized. The reviewed literature included some studies that focused on understanding the kinetics of chlorine decay and THM formation in drinking water. Some studies also evaluated chloramine by-products, including CNBr and CNCl. Of specific interest to CUWA agencies using ozone for primary disinfection and chloramines for secondary disinfection is the possibility of increased CNCl formation due to the reaction between chloramines and formaledyde. The latter is a common by-product of the reaction of ozone with natural organic matter. Although of interest, CNCl is not regulated in any current or pending future regulations.

2.2   Chlorine Dioxide

The results of an appreciable number of studies on chlorine dioxide (ClO_2) were reported in the literature during the past five years. Confirming earlier findings, new studies also determined that the use of chlorine dioxide for primary disinfection and free chlorine for secondary disinfection could cause odor problems in some homes. This problem is solved by either lowering the ClO_2 dose or by converting the distribution system from free chlorine to chloramines. A significant amount of work was reported on the formation and control of chlorite (ClO_2^-) from chlorine dioxide. It was noticed that the presence of chlorite and free chlorine in the distribution system could form some chlorine dioxide. In addition, trace levels of chlorite were reported to greatly assist in controlling nitrification in distribution systems because it acts as a biocide against ammonia-oxidizing bacteria.

The primary chlorite destruction technologies studied and reported were reduction by granular activated carbon (GAC) or reduction by ferrous (Fe^{2+}) iron before the water enters the

distribution system. GAC was shown to destroy about 60% of the chlorite formed. Interestingly, one study reported that chlorine dioxide actually increased the ability of GAC to remove NOM. It is not clear at this time how long the effectiveness of the GAC will last under field conditions.

The reduction of chlorite to chloride (Cl^-) with Fe^{2+} was also evaluated. The results showed that while Fe^{2+} can reduce chlorite, dissolved oxygen also imparts a demand on the Fe^{2+}, thus requiring a higher dose for effective chlorite control. It was also reported that the Fe^{2+} has to be added at or upstream of the rapid mix for effective filtration. Adding the Fe^{2+} anywhere between the rapid mix and the filters resulted in unacceptable filter performance.

Finally, some reported studies evaluated the inactivation of *Cryptosporidium* with chlorine dioxide. The results showed that up to 1-log *Cryptosporidium* inactivation could be achieved at CT values ranging from 40 to 250 mg/L-min. The potential to achieve such a high level of disinfection increases with increasing water temperatures. The combination of chlorine dioxide for primary disinfection and ferrous iron addition for chlorite reduction likely would be necessary in many situations, particularly in higher levels of disinfection were desired. In general, only under the more unique circumstances of temperature and high dosage (with ferrous iron to remove chlorite to maintain compliance) could chlorine dioxide achieve greater than 0.5 log inactivation of *Cryptosporidium*. Nevertheless, it can be used successfully to achieve at least 1 log *Giardia* inactivation given careful control of dosage and management of chlorite residuals with ferrous iron. In addition, the use of a dedicated chlorine dioxide contactor for disinfection is advantageous given the relatively rapid decay of chlorine dioxide and the desire to maximize t_{10}, thereby improving the efficiency of chlorine dioxide disinfection.

2.3   Ozone

Advances in the application and use of ozone in water treatment must be of paramount interest to CUWA members. Of the eight CUWA members participating in this study, only one (Sacramento) does not use or plan on using ozone. Research continues to focus on ozone disinfection capabilities, and on the formation and control of important ozonation by-products. Several studies evaluated the inactivation of *Cryptosporidium* oocysts with ozone. The proposed

LT2ESWTR, which will be published Summer 2003, will include a CT table for the inactivation of *Cryptosporidium* with ozone at various water temperatures. The CT table will utilize much of the reported literature during the past five years, but will also include an uncertainty factor that could greatly increase the CT values required for achieving *Cryptosporidium* inactivation. One issue the LT2ESWTR will not address is the impact of upstream ozonation on the inactivation of *Cryptosporidium* oocysts with chlorine or chloramines. This phenomenon has been coined the "synergistic effect" by many researchers. Typically, chlorine and chloramines are completely ineffective against *Cryptosporidium* under typical dose and contact time conditions at a water treatment plant. However, studies have shown that an additional 0.5 to 1-log inactivation of *Cryptosporidium* oocysts may be achieved if chlorine or chloramines disinfection is preceded by relatively low doses of ozone. For those CUWA members who use ozone and will be required to meet a specific log-inactivation of *Cryptosporidium,* this is an area that these members may want discuss with the Department of Health Services (DHS) after the State assumes primacy of the LT2ESWTR. Alternatively, on behalf of these members, CUWA may want to make an official comment on the proposed LT2ESWTR requesting that the Rule be modified to make it easier for utilities to demonstrate and obtain approval for this synergistic effect for meeting specific *Cryptosporidium* inactivation levels.

Research continues to be conducted and reported on the control of bromate; the only ozone by-product currently regulated. Due to the elevated bromide levels in Delta water, bromate formation is a serious problem for many CUWA members treating Delta water with ozonation. Bromate control strategies reported in the literature focused on two areas: The first is minimizing bromate formation, and the second is destroying bromate after it forms. The most reliable bromate minimization action continues to be pH depression. However, some studies have demonstrated that the combination of ammonia addition at low levels (up to 0.2 mg/L) with pH depression can significantly reduce bromate formation. If ozone is to be used for the meeting specific *Cryptosporidium* inactivation requirements, bromate minimization strategies may have to be implemented when the bromide concentration is at or above 50 µg/L, which is quite a low threshold by Delta water standards.

The other approach to meeting the bromate MCL is to remove some or all of the bromate after ozonation. Three technologies can achieve this goal. Two of these technologies involve chemical reduction of bromate to bromide, while the third technology utilizes bacteria that can use bromate as an electron acceptor and reduce it to bromide. This is similar to how denitrifying bacteria use nitrate as an electron acceptor and reduce it to nitrogen gas (N_2). The chemical reduction technologies are contact with GAC or with ferrous iron. This is similar to the reduction of chlorite by GAC and ferrous iron. Unfortunately, the capability of each technology to reduce bromate is severely limited under field conditions. Reduction of bromate with GAC is feasible in organic-free water. However, in the presence of any appreciable amount of NOM, the reduction reaction is greatly retarded making this process impractical. Similarly, while ferrous iron can reduce bromate, its effectiveness is greatly limited in the presence of dissolved oxygen because DO is preferentially reduced by ferrous compared to bromate. Finally, biological reduction of bromate in biological filtration downstream of ozone has been demonstrated in laboratory experiments. However, the biological reduction of bromate can only take place after the dissolved oxygen is reduced to below a maximum of 2 mg/L. Unfortunately, natural waters contain around 9 to 11 mg/L oxygen, which can also increase to >20 mg/L downstream of an oxygen-fed ozonation process. This renders this process virtually impractical for application at a water treatment plant.

Several studies are reported in the recent literature dealing with the use of ozone as a preoxidant in a water treatment plant. One study has shown that ozone can destroy two harmful algal toxins, microcystin LR and anatoxina-a, to below their detection limits. Other studies reported that ozone can achieve as much as 95% destruction of several pharmaceuticals that may be present in natural waters. On the other hand, the ozonation of water containing certain types of algae was found to increase THM precursor levels by as much as 100%. It is speculated that the destruction of the algal cells by the added ozone releases cellular material that produces high THM levels when contacted with chlorine.

Finally, several studies evaluated the formation and removal of biodegradable organic matter (BOM) in an ozone-biofiltration plant. The most useful information to CUWA members addresses the impact of filter operation on its performance as a biofilter. Studies have shown

that air-scour had no negative effect on the performance of the biofilters, which confirms the experience of some CUWA members who currently practice biofiltration. The impact of chlorine in the backwash water was also evaluated. In general, backwashing a biofilter with chlorinated water is not problematic except when all contributing factors are at their most unfavorable conditions. For example, backwashing with chlorinated water can impair the performance of a biofilter under cold-water conditions when treating water containing hard-to-remove BOM, and when using a high chlorine concentration in the backwash water.

2.4 UV

The emergence of UV light as an effective biocide against *Giardia* cysts and *Cryptosporidium* oocysts is the most significant event in the water industry during the past five years. It has certainly caught the attention of the USEPA and many water agencies. In the proposed LT2ESWTR to be published this summer (2003), the USEPA will include "IT" tables for the inactivation of *Giardia* and *Cryptosporidium* with UV.

Several papers on the application of UV treatment were found in the literature. Most of them dealt with measuring the inactivation of *Giardia* and *Cryptosporidium* with UV. While the results varied somewhat between researchers, it is reasonable to say that a UV dose between 10 and 40 mJ/cm^2 may achieve as much as 4-log inactivation of *Giardia* cysts or *Cryptosporidium* oocysts. These are quite low doses compared to those required for the inactivation of viruses. Similar doses were found to achieve significant inactivation of other organisms such as microsporidia. The cost of UV technology was also reported. Estimates show that it can be as much as 40% to 80% lower than that of ozone application to meet the same inactivation levels. However, it should be kept in mind that UV achieves none of the other benefits of ozone, such as destruction of T&O-causing chemicals.

Finally, several studies evaluated the impact of UV irradiation on the natural organic matter, and the possibility of forming any harmful by-products. The results were mixed. Some research showed that UV treatment might increase the BOM concentration in some waters, while others

showed no effect. No UV by-products have yet been identified at UV dosages appropriate for inactivation of pathogen cysts or viruses in drinking water.

2.5 MF/UF

The use of low pressure membranes – microfiltration (MF) or ultrafiltration (UF) – continues to increase in the US. ACWD will utilize UF at one of its plants (WTP #1). Research during the past five years has focused on trying to identify and understand the factors that cause membrane fouling. While it is widely believed that NOM is the primary cause of membrane fouling, significant research continues to identify the fouling mechanism with the hope of identifying means of reducing it. The results seem to indicate that high-molecular weight fractions of the NOM seem to be responsible for most of the fouling. The addition of PAC has been shown by many researchers to help reduce membrane fouling by adsorbing the NOM before it reaches the surface of the membrane. Nevertheless, due to differences in NOM characteristics between waters, it is very difficult to predict the extent of fouling of any membrane, and pilot testing remains an essential step before full-scale membrane implementation.

2.6 GAC/PAC

A significant amount of work on the use of GAC and/or PAC in water treatment was conducted during the past five years. One important finding is that the exposure of PAC to chlorine greatly reduces its ability to adsorb organics. In fact, studies have shown that chlorine contact with PAC that had already adsorbed DBP precursors causes the release of the precursors. Therefore, it is important to separate the PAC from the water before chlorine is added. This is important to understand because many utilities add chlorine and PAC simultaneously at the head of the plant with the intent to maximize control of T & O episodes. This research shows that this approach actually hinders the ability of the plant to achieve this goal. Other research showed that the same effect applies to GAC. Exposure of GAC to chlorine reduces its adsorption capacity for organics.

2.7   MIEX®

The Magnetic Ion-Exchange (MIEX®) process is a relatively new process to the water treatment industry. This process, which is proprietary, is very similar to the conventional ion-exchange process with the exception that the resin size is reduced to about 0.15 to 0.2 mm. The resin is also magnetized to promote its agglomeration and settling in a clarifier. The resin is added to the water as a slurry in a dedicated contactor and is then separated from the water in a dedicated clarifier, regenerated with salt, and reused. Data show that only 0.1% of the resin is lost on a continuous basis due to attrition. Because of its small size, this resin can rapidly adsorb significant amounts of NOM (between 60% and 90%). Since it is an anionic-exchange resin, it can also remove negatively charged ions such as bromide, sulfate, nitrate, and arsenic from the water.

Although there is only one peer-reviewed article on this technology, several papers were published in conference proceedings during the past several years. All the articles demonstrate the high NOM removal achieved by this process. They also show that the use of the MIEX® technology upstream of coagulation results in a significant decrease in the coagulant dose required.

The greatest hurdle to the use of this technology is the need to dispose of the spent regeneration brine which contains 10 to 12% sodium chloride (NaCl), as well as high concentrations of NOM, bromide, sulfate, and any other ions removed from the water. Unfortunately, none of the work done thus far addresses this issue. The 29 MGD plant in Australia discharges its brine to the ocean after a 1,000:1 dilution, and one plant in Virginia will be using an existing RO unit brine disposal system for the disposal of the MIEX® brine. Other designs being considered are evaluating discharge to the sanitary sewer. Clearly, the use of this technology at any large plant is greatly limited by the availability of an economical brine disposal solution.

The cost of this technology was reported in some articles, but without addressing the brine disposal cost. The cost of resin replacement has been reported at about $0.14/1000 gallons of

water treated. On the other hand, one paper reported the overall process cost (without brine disposal) at approximately $0.91/1000 gallons.

2.8   Conventional Treatment Processes

While conventional treatment is not a new technology, there were several papers that addressed modifications to improve its performance for removing contaminants. Some of the most notable are those dealing with the removal of algae. Tests showed that toxin-producing algae cells are well removed by conventional treatment (about 3-logs). However, the cells concentrated in the sludge broke down and released toxins in the sludge. This can have a significant impact on the approach adopted with regard to the recycling of washwater supernatant. As anticipated, some research showed that conventional coagulation, flocculation, sedimentation, and filtration were not effective in removing several pharmaceutical compounds.

Finally, several studies evaluated the use of high rate clarification processes such as dissolved air flotation (DAF) and ballasted flocculation (tradename ACTIFLO[®]). These processes are especially effective for treating waters with particles or organics that are difficult to flocculate and settle in conventional sedimentation basins.

3. WATER TREATMENT PLANT SIMULATION MODEL

3.1. WTP MODEL INTRODUCTION AND HISTORY

The original version of the USEPA Water Treatment Plant (WTP) model was developed by Malcolm Pirnie in 1992 to support the early stages of the development of the Disinfectant/Disinfectant By-Product (D/DBP) rules. The model was used extensively during the regulatory negotiation process of 1993-94 resulting in the 1994 DBP rule proposal. The model was later revised by Malcolm Pirnie under a contract with the University of Colorado-Boulder through USEPA funding. The model was significantly updated in 2001-02 as version 2.0. The WTP model version 2.0 was used in the Surface Water Analytical Tool (SWAT) for national compliance forecasting and regulatory negotiations for Stage 2 D/DBP Rule. In addition to the use of this model for regulatory development process, the tool was modified by Malcolm Pirnie for specific applications by Metropolitan Water District (MWD), the Philadelphia Water Department, Erie County Water Authority, Sacramento/San Joaquin River Delta, California Department of Water Resources and the City of Phoenix.

The WTP model is an empirical, central tendency model that can be used to predict DBP levels in potable water for different source water and treatment alternatives. The inputs for the model include source water quality parameters (e.g., alkalinity, bromide, hardness, pH, temperature, TOC, turbidity and UV254) and treatment process train design and operating criteria (e.g., detention times, chemical dosages).   The WTP model predicts finished water TOC, DBPs and other water quality parameters. The typical inputs and outputs from WTP model are shown in Figure 3.1.

The WTP model estimates: (i) NOM removal by the various treatment processes in a treatment train, (ii) disinfectant decay from demands exerted by NOM and other sources and (iii) DBP formation based upon disinfectant application and the associated water quality.   In this study, the WTP model was used to predict THMs, HAA5s, bromate, and *Cryptosporidium* inactivation.
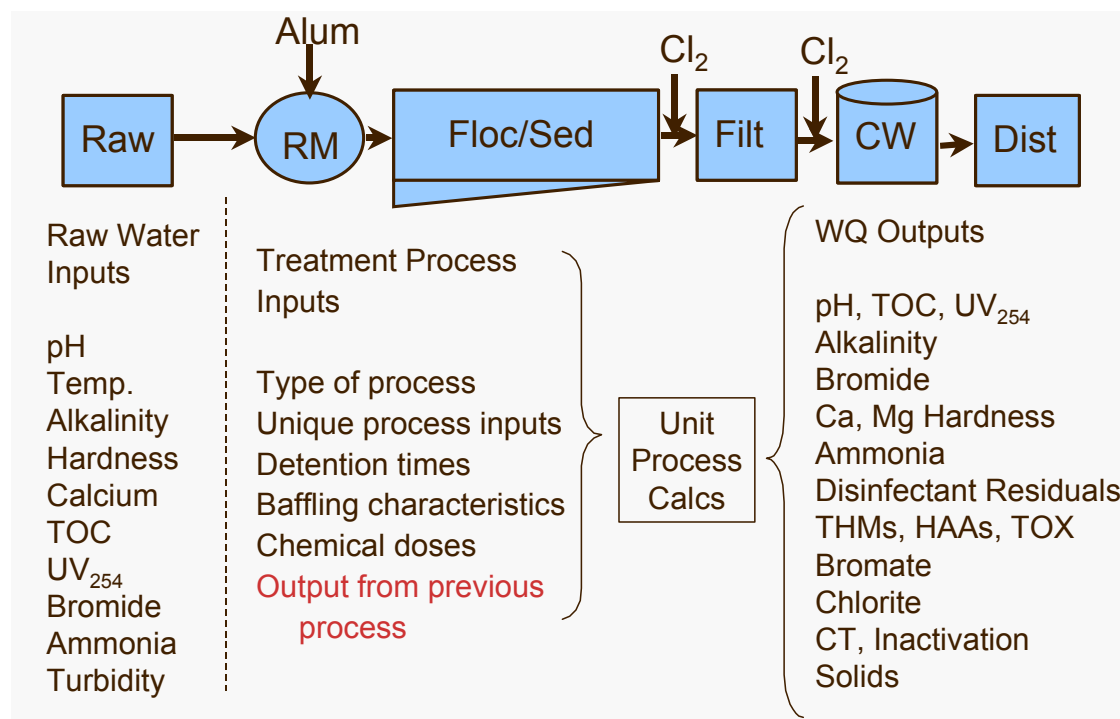
Figure 3.1. Illustration of Inputs and Outputs of WTP Model

3.2. SITE-SPECIFIC CALIBRATION

The WTP model predictions are expected to be relatively accurate on a central tendency basis. However, the results may not be as accurate for site-specific applications without calibration. For example, if 50 plants with various treatment configurations and chemical types were modeled, the average of all 50 plants' predictions would be reasonably accurate when compared to actual finished water data from these plants. Some of the plants are expected to be over-predicting and some are expected to be under-predicting. On average, however, they should be predicting with a reasonable degree of accuracy. For this reason, this type of central tendency model is considered useful for national average calculations in regulatory development. To use the model for site-specific applications, calibration is necessary. Once calibrated for a given water treatment plant, the model is expected to predict reasonably for the average water quality received at that plant.

To achieve site-specific calibration, the operations data are collected from each plant and these are used in modeling. The model predictions are then compared to the actual finished water data.

Typically, the parameters that are compared for accuracy are pH, TOC, TTHM and HAA5. From these comparisons, correction factors are determined that are written into the model code so that the calibrated model will predict more accurately. The more data available, the greater the confidence in the accuracy of the calibrated model.

3.3. CALIBRATION EXAMPLES

For each of the CUWA plants that were evaluated, the utilities provided a process schematic and 12 months of plant data. The plant data included influent and effluent water quality, and plant operation data such as chemical doses and basin size. Where these data were not available, data from the Information Collection Rule (ICR) database were used. A model file for each plant was created from the plant process schematics corresponding to each date that a complete data set was provided. The predicted effluent values for pH, TOC, TTHM and HAA5 were compared with the observed data. The ratio of predicted values to observed values resulted in a correction factor that was incorporated into the models equations, resulting in a calibrated model that predicts the finished water quality accurately for that plant.

The order in which the correction factors are applied is important. For example, if after running 12 months of data through the model, it is observed that pH was always over-predicted, a correction factor would then be applied to the pH equations so that the model would now predict pH correctly. The next step would be to re-run all the data through the model and compare predicted TOC with the measured TOC values. If there were a need to correct TOC, it would be made. Then, the next step would be to compare the predicted DBPs and measured DBPs and calibrate the model if necessary. This hierarchy of calibration is based on the model algorithm and is illustrated in Figure 3.2. The predicted pH is an input for TOC removal calculations, and subsequently the predicted pH and TOC become inputs for DBP formation calculations.

The following text illustrates four examples of how the calibration was applied to the 11 individual CUWA member agencies treatment plants evaluated in this study. The four examples are the Alameda County Water District's (ACWD) Water Treatment Plant No. 2 (WTP-2), Metropolitan Water District of Southern California's (MWDSC) Diemer Filtration Plant (FP),

San Francisco Water Department's (SFWD) Harry Tracy Water Treatment Plant, and Santa Clara Valley Water District's (SCVWD) Santa Teresa Water Treatment Plant.
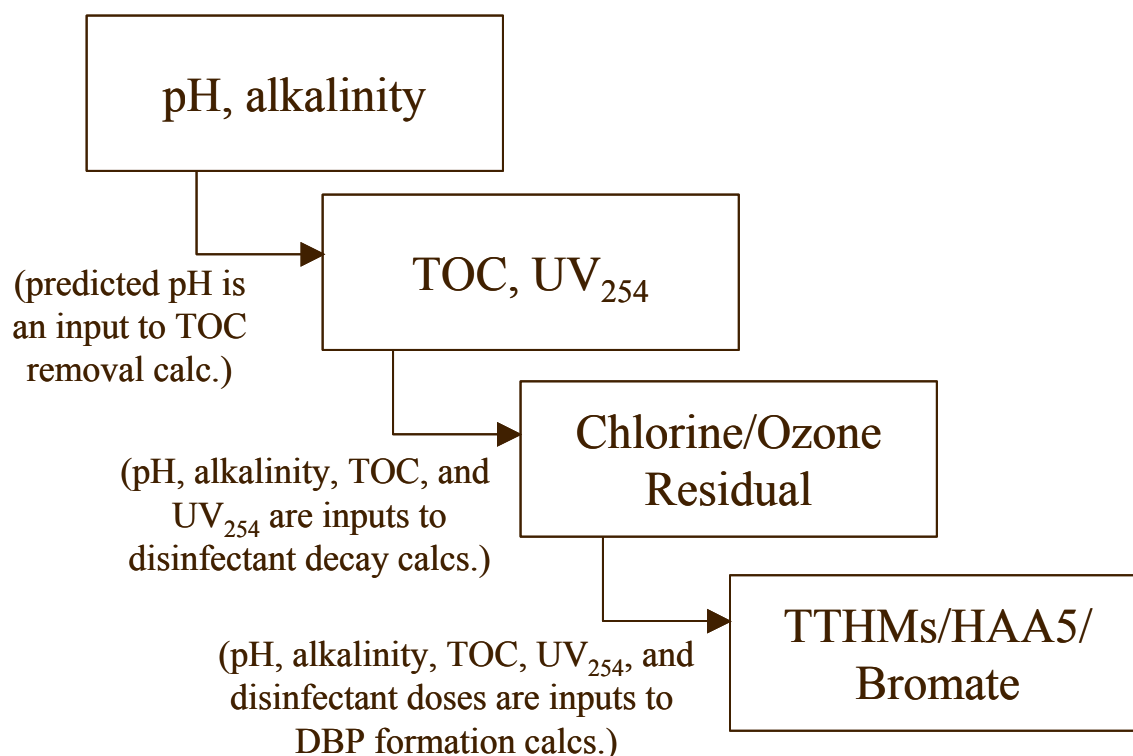
Figure 3.2. WTP Algorithm Calculation and Calibration Order

3.3.1.   Alameda County Water District – Water Treatment Plant 2

The WTP calibration for ACWD, TP-2 involved adjusting TOC and DBPs. Shown in Figure 3.3 is the box-and-whisker plot for measured and predicted pHs. The boxes outline the 25th and 75th percentile values, the whiskers show the 10th and 90th percentile values, and the solid line within the box indicates the median value.   The solid circles represent outliers, the points that are beyond the 10th and 90th percentiles. As shown in Figure 3.3, the predicted pH was slightly higher than the measured pH and therefore pH was not adjusted. The predicted pH was used by the model as an input for TOC calculations. The measured and predicted TOCs are shown in Figure 3.4. The TOC that was predicted before the model was calibrated (indicated as predicted-before) was significantly higher than the measured TOC. The median TOCs for measured and predicted-before calibration were 1.8 mg/L and 3.7 mg/L, respectively. The ratio of measured

TOC to predicted TOC was used as a correction factor to calibrate the model. The predicted TOC after calibration was much closer to the measured TOC (Figure 3.4).

Figure 3.3. Measured and WTP Simulation Model Predicted pHs for ACWD-WTP2

ACWD-WTP2
pH

The predicted pH and TOC were used as inputs to predict chlorine residual. The measured and predicted chlorine residuals are shown in Figure 3.5. Since the boxes for the measured and predicted chlorine residuals have significant overlap, no correction was performed for chlorine residuals. In particular, the median values were within 0.2 mg/L. The predicted pH, TOC, chlorine residuals, together with other plant information input, were used to compute DBPs. The measured and predicted TTHMs and HAAs are shown in Figures 3.6 and 3.7, respectively. Since the predicted TTHMs and HAAs were significantly higher than the observed values, the model was calibrated for both of these compounds. The post-calibration TTHMs and HAAs were similar to the measured values.

ACWD-WTP2
TOC

Figure 3.4. Measured and Predicted TOCs (Before and After Model Calibration) for ACWD-WTP2

ACWD-WTP2
Cl2 residual
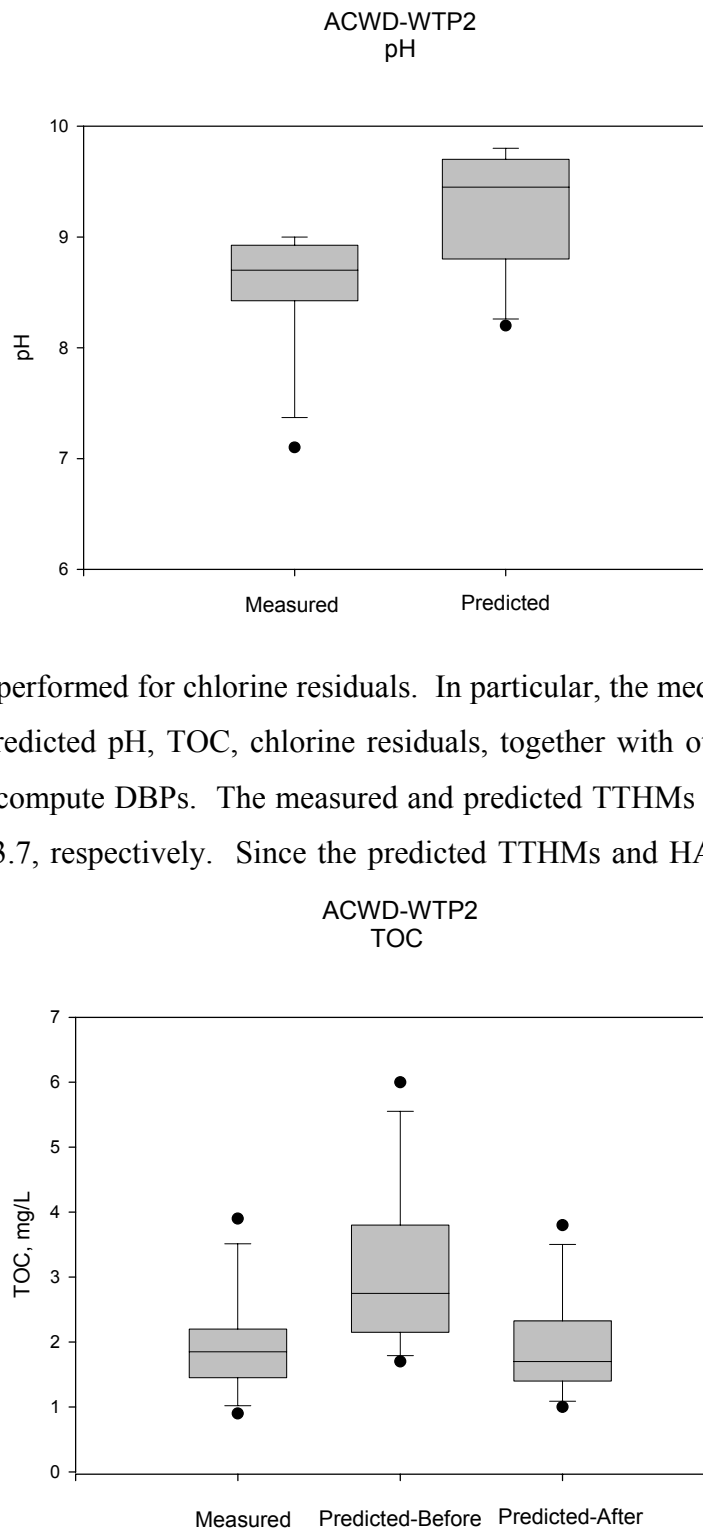
Figure 3.5. Measured and Predicted Chlorine Residuals for ACWD-WTP2
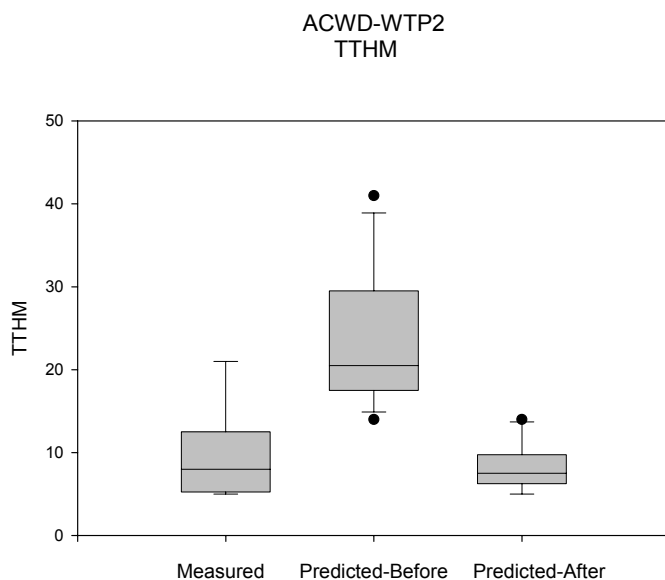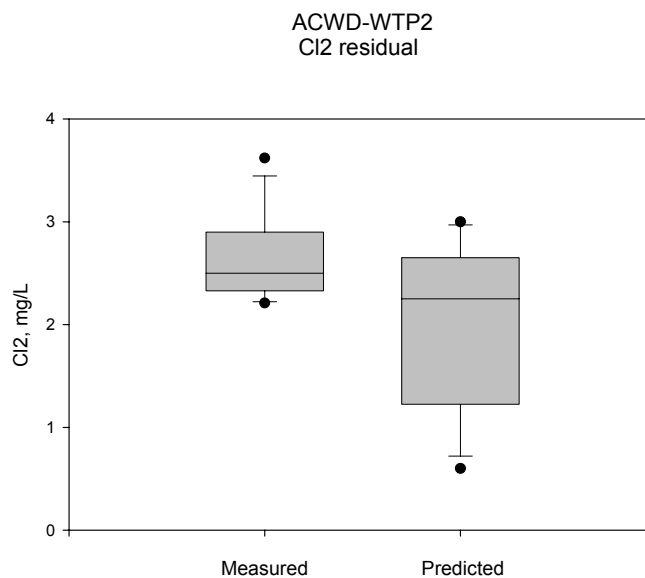
ACWD-WTP2
TTHM

Figure 3.6. Measured and Predicted THMs (Before and After Calibration) for ACWD-WTP2

Figure 3.7. Measured and Predicted HAAs (Before and After Calibration) for ACWD-WTP2

ACWD-WTP2
HAA5
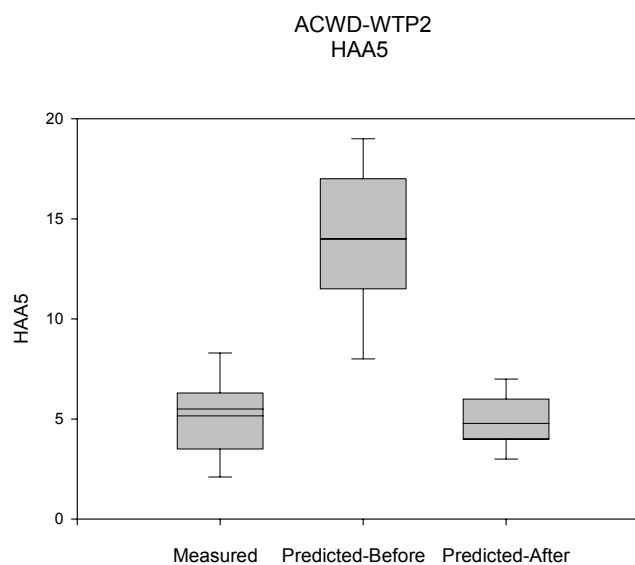
3.3.2. Metropolitan Water District of Southern California – Diemer Filtration Plant

For the MWDSC's Diemer FP, calibration was only required for TTHMs and HAAs. The predicted pH, TOC and chlorine residuals were in good agreement with the measured values (Figures 3.8, 3.9 and 3.10).   The predicted DBPs before calibration were higher than the measured DBPs (Figures 3.11 and 3.12).   After calibration, the TTHMs and HAAs were reasonably predicted.

Figure 3.8. Measured and Predicted pHs for MWDSC-Diemer FP

MWD-Diemer pH

MWD-Diemer TOC
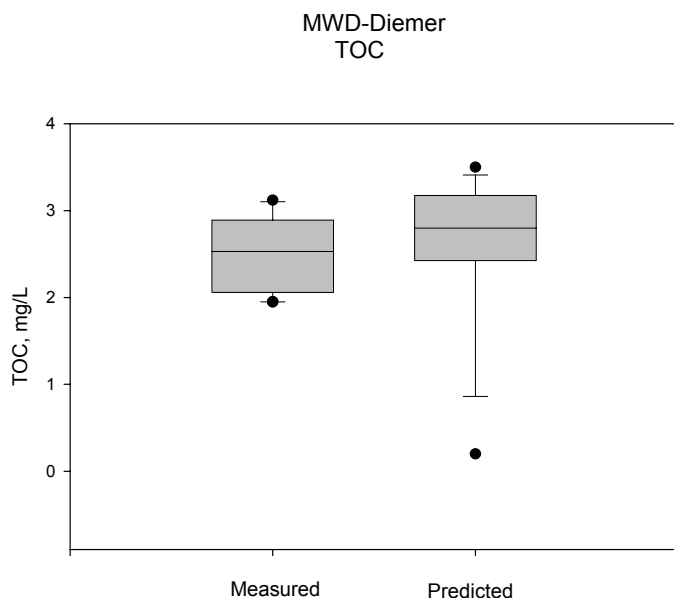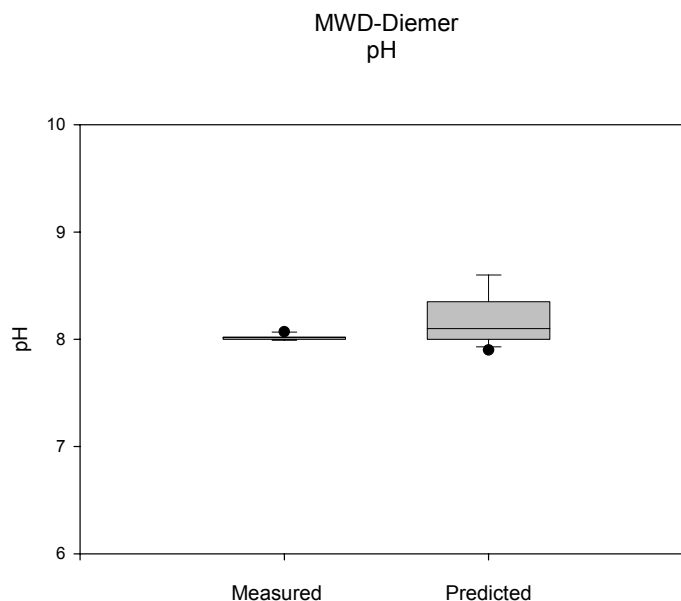
Figure 3.9.   Measured and Predicted TOCs for MWDSC-Diemer FP

MWD-Diemer
Cl2 Residual
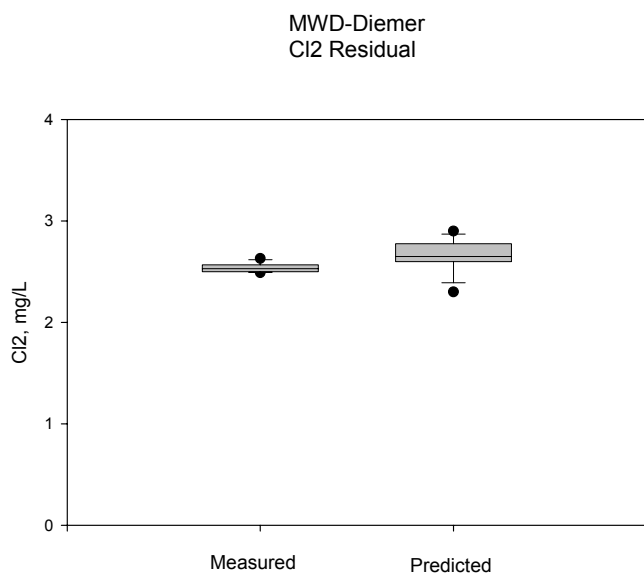
Figure 3.10. Measured and Predicted Chlorine Residual for MWDSC-Diemer FP

Figure 3.11. Measured and Predicted (Before and After Calibration) TTHMs for MWDSC-Diemer FP

MWD-Diemer
TTHM

MWD-Diemer
HAA5
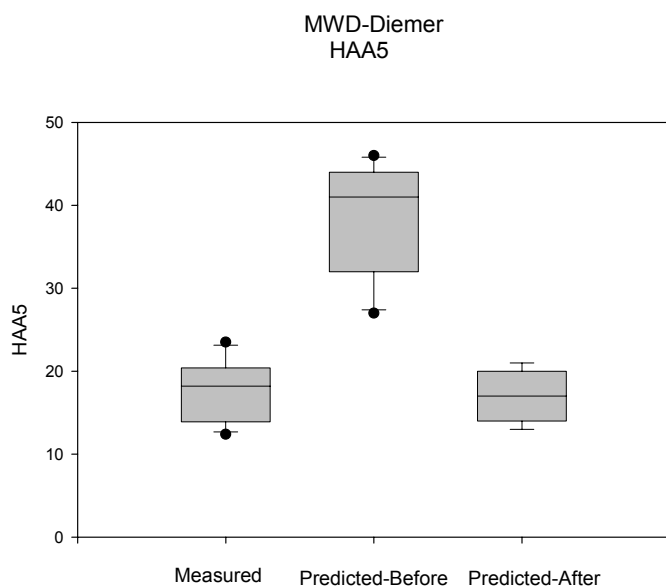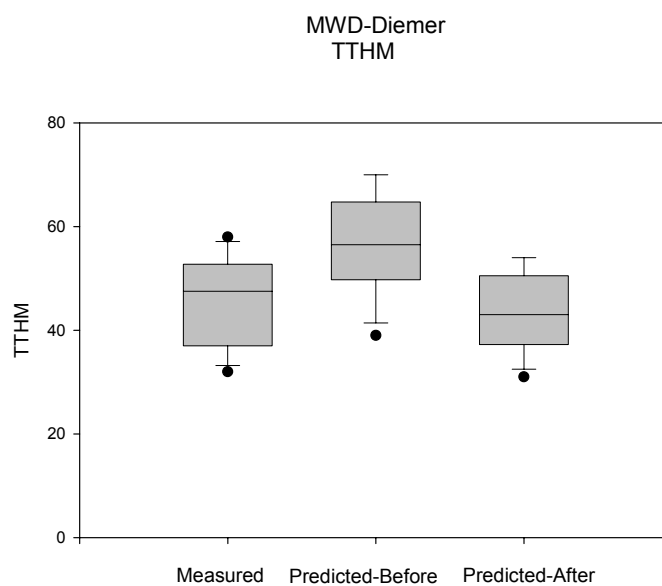
Figure 3.12. Measured and Predicted (Before and After Calibration) HAAs for MWDSC-Diemer FP

3.3.3. San Francisco Water Department – Harry Tracy Water Treatment Plant

For SFWD's Harry Tracy Plant, calibration was only required for the TTHMs and HAAs.   The rest of the parameters such as pH, TOC and chlorine residual were predicted reasonably (Figures 3.13, 3.14 and 3.15). The uncalibrated model over-predicted the TTHMs (Figure 3.16). After calibration the TTHMs were predicted accurately.

Figure 3.13.   Measured and Predicted pHs for SFWD's Harry Tracy Plant

SanFrancisco-Harry Tracy
pH
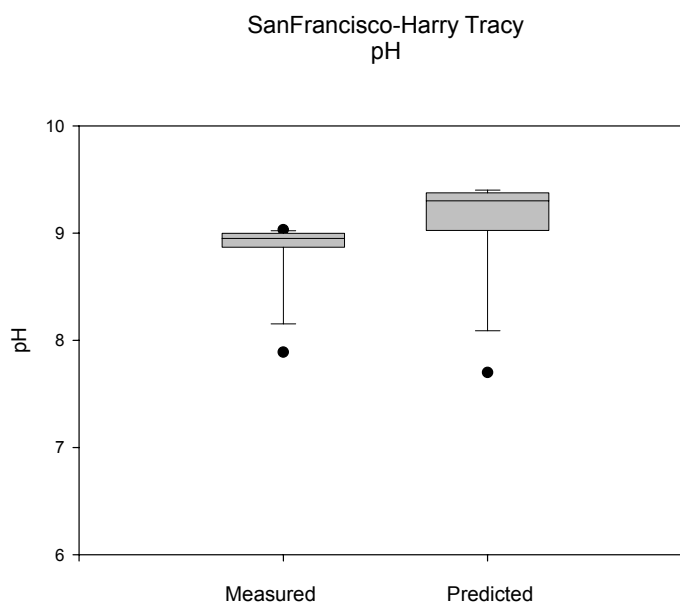
Figure 3.14.   Measured and Predicted TOCs for SFWD's Harry Tracy Plant

SanFrancisco-Harry Tracy
TOC

SanFrancisco-Harry Tracy
Cl2 Residual
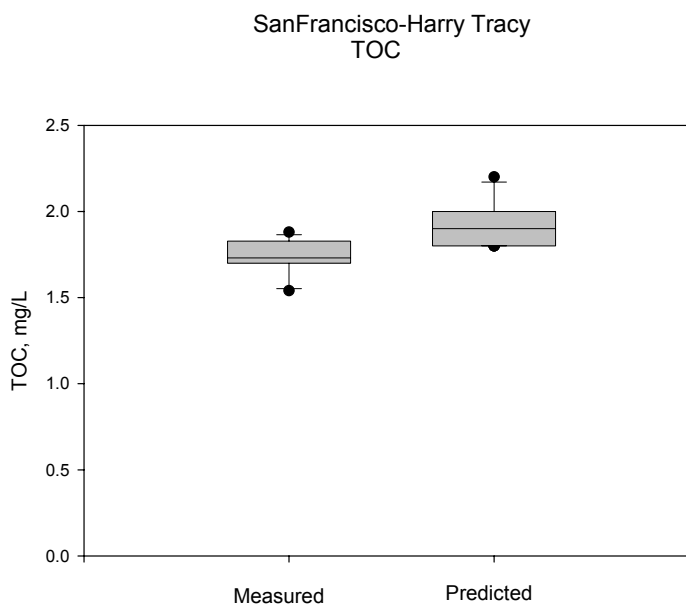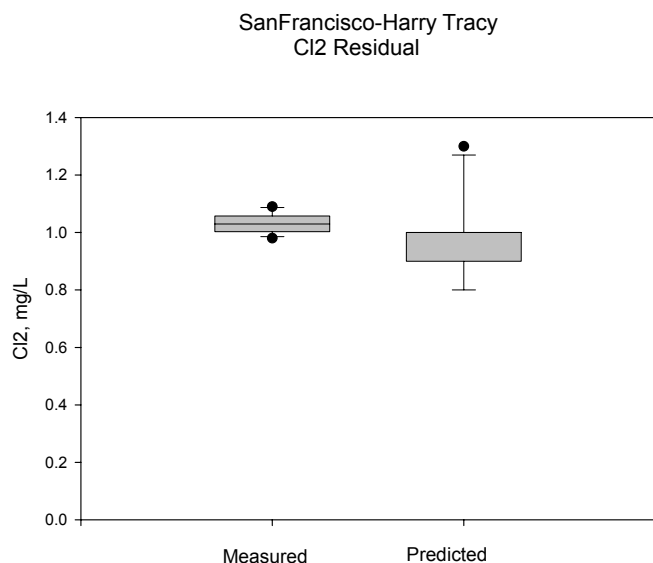
Figure 3.15. Measured and Predicted Chlorine Residual for SFWD's Harry Tracy Plant

SanFrancisco-Harry Tracy
TTHM

Figure 3.16. Measured and Predicted (Before and After Calibration) TTHMs for SFWD's Harry Tracy Plant
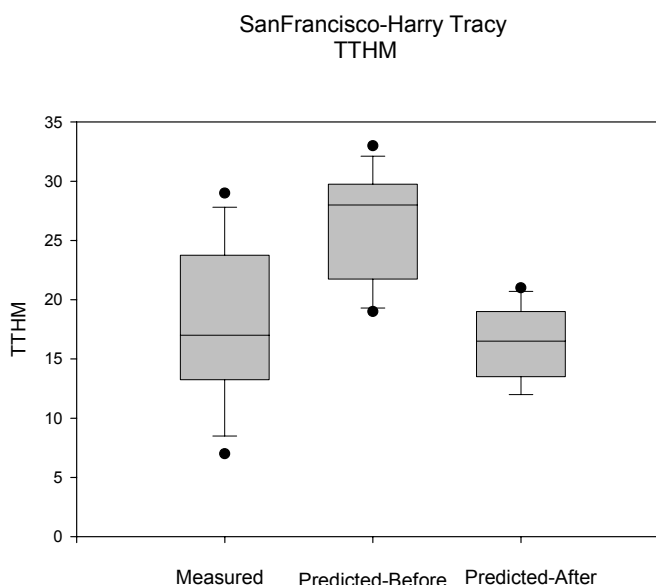
3.3.4. Santa Clara Valley Water District – Santa Teresa Water Treatment Plant

Similar to MWDSC's Diemer FP and SFWD's Harry Tracy Treatment Plant, the calibration for SCVWD's Santa Teresa Plant involved correcting for THMs and HAAs. Shown in Figures 3.17, 3.18 and 3.19 are the measured and predicted pHs, TOCs and chlorine residuals, respectively. Even though the predicted pHs were slightly lower than the measured pHs no correction was applied for pH prediction. The boxes for predicted TOCs overlap with measured TOCs and therefore no correction was applied. The predicted chlorine residuals had a wide range compared to the measured chlorine residuals.

Figure 3.17. Measured and Predicted pHs for SCVWD's Santa Teresa Plant

SCVWD-Santa Teresa
pH
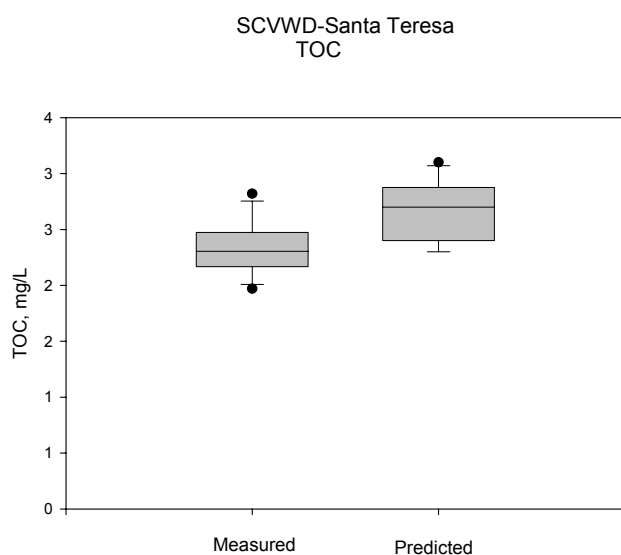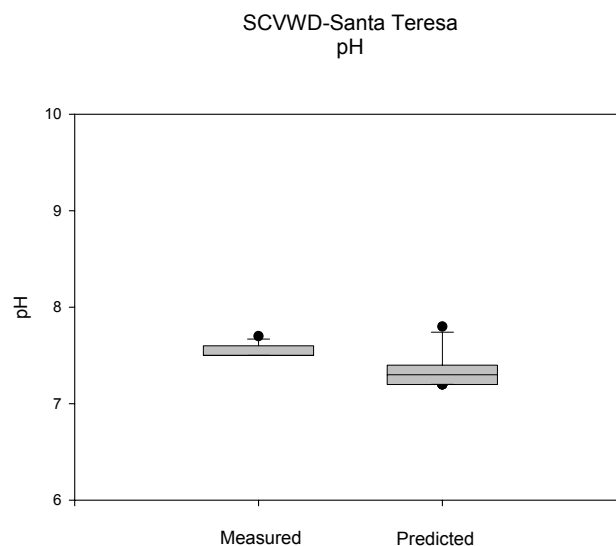
Figure 3.18. Measured and Predicted TOCs for SCVWD's Santa Teresa Plant

SCVWD-Santa Teresa
TOC

Figure 3.19. Measured and Predicted Chlorine Residuals for SCVWD's Santa Teresa Plant

SCVWD-Santa Teresa
Cl2 Residual
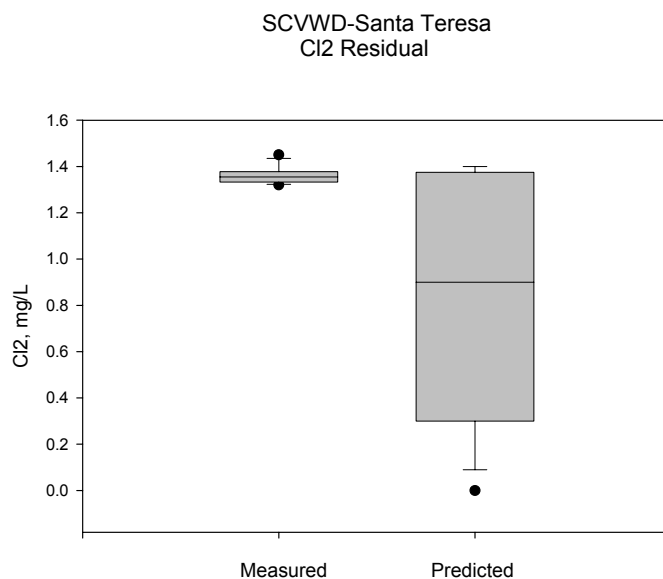
Shown in Figures 3.20 and 3.21 are the TTHMs and HAAs before and after calibration for Santa Teresa Plant. Calibration helped in predicting the TTHMs and HAAs more accurately.

Figure 3.20. Measured and Predicted TTHMs (Before and After Calibration) for SCVWD's Santa Teresa Plant

SCVWD-Santa Teresa
TTHM

SCVWD-Santa Teresa
HAA5
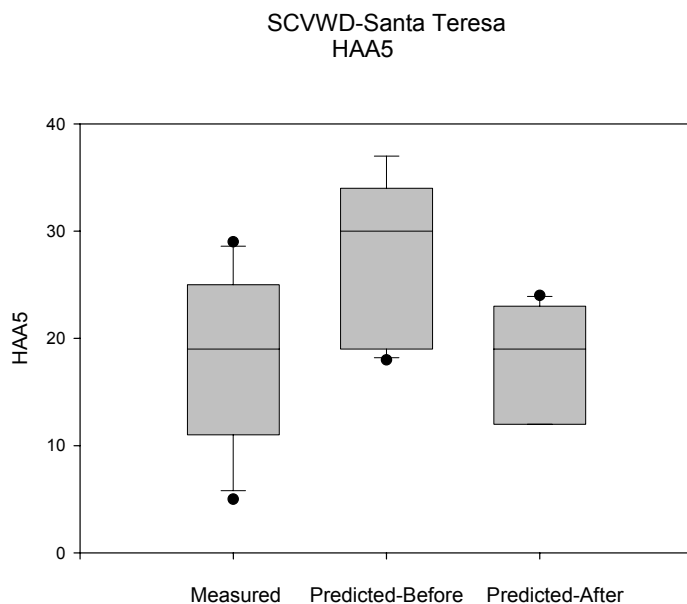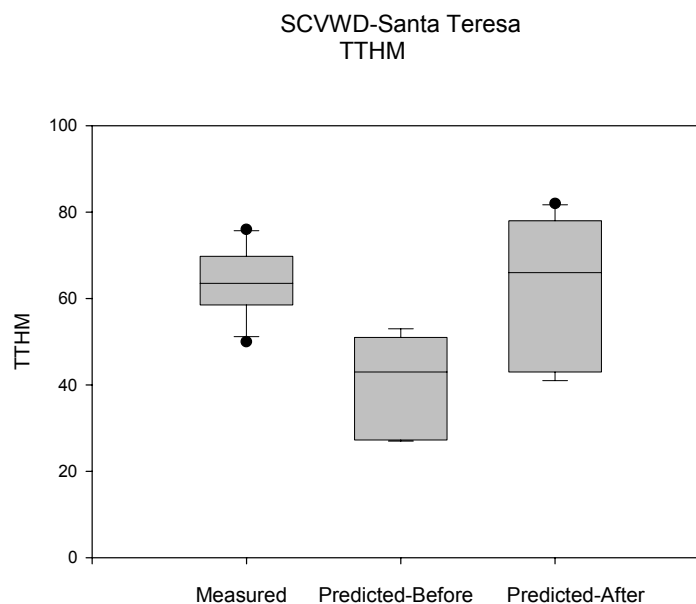
Figure 3.21. Measured and Predicted HAAs (Before and After Calibration) for SCVWD's Santa Teresa Plant

4. CASE STUDIES

4.1. BASIS OF ANALYSIS

4.1.1 Selection of Treatment Plants

The WTP simulation model was used to evaluate the performance of eleven water treatment plants operated by CUWA member agencies. Figure 4.1 shows the relative locations of these plants on a map of the State of California. Among these plants, six receive water from the San Francisco Bay Sacramento/San Joaquin River Delta. The other five plants receive waters that are not impacted by the operations of the Delta. The intakes for four of these plants are located upstream of the Delta region and therefore are not impacted by Delta operations. The fifth, the Los Angeles Aqueduct Filtration Plant (LAAFP) operated by the Los Angeles Department of Water and Power (LADWP) is not significantly affected by the operations of the Delta because the source water is obtained from Los Angeles Aqueduct, which emanates in the Owens Valley northeast of Los Angeles.

4.1.2 Data Collection

The project team collected information from the 11 water plants to define the water treatment process in terms of the physical facilities, water quality and operational parameters. The initial source of information in this data gathering effort was the Information Collection Rule (ICR) database as reported to the USEPA. In obtaining information from the ICR database, it was realized that several key pieces of information were missing because of the specific data validation requirements of the ICR. Consequently, the project team conducted on-site and telephone surveys for each of the facilities to obtain additional information about water quality and the treatment processes. For each of the plants, 12 months of plant operational and water quality data were compiled. The physical facility data included the size and type of the various unit processes. The water quality data included information on raw and treated water quality for a 12-month period. The operational data included flow and chemical dosages. While conducting the surveys, information was also obtained regarding plans for process upgrades and improvements. Improvements that are on-going and firmly planned for the immediate future were also included in defining the existing treatment processes for the purpose of this project.

Figure 4.1. Locations of All Plants Modeled

Plants in Bold Font with Bold Textbox
are Below Delta

City of Sacramento FWTP

City of Sacramento SRWTP

CCWD Bollman WTP

SFWD HTWTP

ACWD WTP2

SFWD Sunol WTP

SCVWD PWTP

SCVWD STWTP

LADWP LAAWTP

MWDSC Diemer WTP

MWDSC Mills WTP

4.1.3   Methodology for Plant Evaluation

Using the data collected from the 11 CUWA member agency plants, the project team constructed a calibrated water treatment plant simulation model for each of the eleven plants. As described in Section 3, the source code   for the original USEPA model was modified uniquely for each of the water plants to construct these calibrated versions of the model. Various input files were created for each of the water plants to reflect the existing processes (including the immediate future facilities) as well as various possible future modifications.

It is important to recognize that the calibrated models are estimation tools and not exact predictors of plant performance. They were constructed and enhanced over more than a decade using research and full-scale data. The models are believed to provide reasonable central-tendency predictions and with adjustments, are believed to provide reasonable information upon which to evaluate trends and technology application for each of the participating member agency treatment plants evaluated. But all models are based upon predictive equations that have boundary conditions for parameters used in development. It is possible that some conditions exceed those boundary conditions under unique circumstances, resulting in the model being used outside of its calibrated range. Therefore, the model results were used together with the best professional judgment of the project team and member agency representatives in an attempt to yield meaningful and realistic results.

The calibrated models along with the existing treatment processes (including immediate future facilities) were utilized to evaluate the eleven selected plants for compliance with the finished water quality goals that could approximate a working definition of "equivalent level of public health protection," as provided in the Record of Decision for the Bay-Delta Accord. As discussed in Section 1, the project team and project advisory committee members established finished water quality goals of 40 µg/L TTHM, 30 µg/L HAA5, 5 µg/L bromate and 1-log *Cryptosporidium* inactivation. These were the water quality goals used by the Expert Panel that resulted in the recommended source water goals of 3.0 mg/L for TOC and 50 ug/L for bromide in their 1998 report to CUWA. In the remainder of this report, the finished water quality as defined above will be termed "ELPH goals." It is recognized that this working definition was

developed by the consultant and member agency representatives for the sole purpose of this evaluation, and should not be confused as actual numerical criteria set forth in the ROD.

Technology assessments were performed under four scenarios:

1. ***Scenario 1 – Baseline*:** In this scenario, the model was run for the existing/ immediate future facility with existing average source water quality. Finished water quality was compared to the ELPH goals. Any inability to meet specific ELPH goal criteria were identified.

2. ***Scenario 2 – Process Improvements*:** WTP simulations were performed using the existing average source water quality and incorporating additional technologies to meet the ELPH goals. Member agency representatives provided input regarding their preferred sequence of process improvement based upon their individual conditions. Conceptual costs were developed for the additional technologies once evaluated.

3. ***Scenario 3 – Raw Water Quality Boundary:*** The limiting source water quality that would allow plants to meet the ELPH goals with existing/immediate future facility was determined.

4. ***Scenario 4 – Modified Raw Water Quality Boundary:*** The limiting source water quality to meet ELPH goals, using the selected technology chosen (based on utility prioritization and costs) from those treatment options developed in Scenario 2, was determined.

As stated above, treatment alternatives were selected with the counsel of each individual member agency representative. Available technologies included ozone (if not currently used), modifying ozone dosages (if ozone is currently used), adjusting ozonation or coagulation pH, increasing coagulant dosages, implementing chlorine dioxide, adding UV disinfection, adding MF/UF membranes, or potentially implementing emerging technologies such as MIEX®. The member agency representatives provided a priority for sequentially adding various technologies to attain ELPH compliance, based upon their unique conditions. This allowed the evaluation to be conducted in a manner that most fully considered the member agencies' existing treatment conditions.

It is important to recognize that any proposed modifications are based upon the constraints of the ELPH goals. They do not take into account other water quality needs, such as oxidation and/or taste and odor reduction. In some instances, the modeling outcome included a reduction in the applied ozone dose without regard to the impact of such action on the ability of the ozone process to control T&O occurrences. In those instances, the ozone dose was maintained above the minimum dose provided by the specific agency staff. In addition, annual average water quality, as obtained during the data collection effort, was used for these evaluations. These were used because the DBP and bromate criteria were based upon running annual average compliance. Nevertheless, it is recognized that significant variation in source water quality might result in different treatment selection or process criteria. Evaluations of this nature were beyond the scope of effort for this project.

4.1.4   Conceptual Costs

Conceptual costs for process improvements were developed using unit cost curves from *Technologies and Costs (T & C) for Control of Microbial Contaminants and Disinfection By-Products* (EPA, 2002). The unit costs for the different treatment technologies in the EPA report were prepared with certain design and operating criteria. Minor adjustments to these cost curves were made as they were utilized in this project to reflect differences in the design and operating assumptions for the CUWA member agencies. For example, the T & C document assumed certain acid and caustic feed rates for control of bromate during ozonation; however, the acid and caustic dosages were different for different plants among the CUWA members to produce the desired bromate control. Consequently, the project team adjusted the cost curves for bromate control for each of the specific applications during this project to develop a more realistic cost estimate.

4.2.   EVALUATION OF TREATMENT FACILITIES

4.2.1.   Summary

A summary of results for these evaluations is presented in Table 4.1. The average raw water TOC and bromide values are included. The source water TOC and bromide values are relatively lower in those plants unaffected by Delta operations compared to those at the plants downstream

of the Delta. This is primarily a result of the residual disinfectant used at the plants. The water plants unaffected by Delta operations use free chlorine for residual disinfection in the distribution system whereas the plants downstream of the Delta use chloramines for residual disinfection. As a result of the use of chloramines for residual disinfection, the water plants downstream of the Delta have better control on TTHM and HAA5 formation (formation of these DBPs practically ceases after the addition of ammonia to convert the free chlorine to chloramines) and thus can treat waters with higher levels TOC without violating the ELPH goals for TTHM and HAA5.

Of the eleven treatment plants that were evaluated only one plant (CCWD's Bollman WTP) was predicted to meet the ELPH goals with the existing or immediate future facilities, based upon the data provided. The finished water quality limitations for the other ten plants are also shown in Table 4.1. Eight of the eleven plants were predicted to not comply with the 1-log *Cryptosporidium* inactivation goal. Five plants were predicted to not comply with TTHM or HAA5 goals, and one plant did not comply with the bromate goal.

Four of the eleven evaluated plants currently do not have ozone in their treatment process or are not considering addition of ozone or UV disinfection in the near future. These plants could not meet the ELPH goal for 1-log *Cryptosporidium* inactivation under any raw water quality combinations since their current or immediate future facilities could not provide the necessary disinfection. Seven of the treatment facilities evaluated either currently treat with ozone or are in the process of implementing ozone in the near future. It is possible that these facilities could meet all the ELPH goals using certain combination of TOC and bromide in their source water.

Using the calibrated model, the project team estimated the limiting water quality conditions (shown in Table 4.1) under which these plants could meet the ELPH water quality goals. It should be noted that for the plants that include ozone in their immediate future facilities, the modeling activities in this project did not assume any changes in the current chlorination practice. Although some treatment plants may change their free chlorine disinfection practices after implementating ozone, the project team did not wish to arbitrarily reduce the chlorine contact time without consideration of other reasons for chlorination at the treatment facility.

Table 4.1. Assessment of Existing and Immediate Future Treatment Plants

Name of WTP	Design Flow (mgd)	Average Flow (mgd)	Average Raw Water TOC (mg/L)	Average Raw Water Bromide (mg/L)	Non-Compliance With Respect to ELPH Goals	Primary Disinfectant	Residual Disinfectant	Limiting Water Quality for Current or Immediate Future Facility
Upstream of Delta								
City of Sacramento – Fairbairn WTP	100	58.4	1.3	0.01[1]	Cryptosporidium inactivation < 1.0 log	Cl_2	Cl_2	Cryptosporidium inactivation cannot be achieved
City of Sacramento – Sacramento River WTP	135	50.5	1.5	0.01[1]	Cryptosporidium inactivation < 1.0 log	Cl_2	Cl_2	Cryptosporidium inactivation cannot be achieved
San Francisco WD – Sunol Valley WTP	160	80.0[2]	3.3	0.05	Cryptosporidium inactivation < 1.0 log	Cl_2	Cl_2	Cryptosporidium inactivation cannot be achieved
San Francisco WD – Harry Tracy WTP	180	54.9	1.9	0.05	Bromate > 5 µg/L	O_3	Cl_2	Br = 0.02 mg/L TOC = 2 mg/L
LADWP – LA Aqueduct FP	601	446	3.0	0.12	Cryptosporidium inactivation < 1.0 log TTHM > 40 µg/L	O_3	Cl_2	Br = 0.1 mg/L TOC = 2 mg/L
Downstream of Delta								
Contra Costa WD – Bollman WTP	75	37.3	3.3	0.16	None	O_3	NH_2Cl	Br = 0.3 mg/L TOC = 3 mg/L
Alameda County WD – Treatment Plant 2	20	17.7	3.5	0.22	Cryptosporidium inactivation < 1.0 log	O_3	NH_2Cl	Br = 0.06 mg/L TOC = 2 mg/L
Santa Clara Valley – Santa Teresa WTP	100	59.8	3.3	0.20	Cryptosporidium inactivation < 1.0 log TTHM > 40 µg/L	O_3	NH_2Cl	Br = 0.2 mg/L TOC = 2 mg/L
Santa Clara Valley – Penitencia WTP	40	25.4	3.4	0.18	Cryptosporidium inactivation < 1.0 log TTHM > 40 µg/L	O_3	NH_2Cl	Br = 0.2 mg/L TOC = 2 mg/L
MWDSC – Diemer FP	520	356	3.0	0.13	Cryptosporidium inactivation < 1.0 log TTHM > 40 µg/L	Cl_2	NH_2Cl	Cryptosporidium inactivation cannot be achieved
MWDSC – Mills FP	150	41.9	3.5	0.26	Bromate > 5 µg/L TTHM > 40 µg/L	O_3	NH_2Cl	Br = 0.1 mg/L TOC = 2 mg/L

Note: [1]Assumed value for WTP modeling purposes; [2]Assumed to be 50% of the design flow

Consequently, the TOC limits determined during this study may be lower for the plants that are in the process of implementing ozone in the near future, because the TOC limit is directly related to the free chlorine contact time for the production of chlorinated DBPs that are included among the ELPH goals.

Examples of developing calibrated models for member agency water plants are provided in Section 3. The following sections describes the use the calibrated models for the evaluation of the current status of compliance with the ELPH goals, determination of water quality constraints under the existing treatment process configurations, and an assessment of possible future technology improvements for meeting the disinfection and DBP goals defined for ELPH.

The following sections present the evaluations for the eleven member agency water treatment plants from north to south. Member agency plants upstream and unaffected by Delta operations are presented first, followed by those affected by Delta operations.

4.2.2   City of Sacramento – Fairbairn WTP

The process schematic, average chemical dosages, and average water quality for the Fairbairn WTP are provided in Appendix B, Slide B-2. This conventional plant uses raw water from the American River, upstream of its confluence with the Sacramento River, and significantly upstream of the Delta. This source is of a much higher quality than the Delta export as reflected by the low average raw water TOC (1.3 mg/L) and bromide (less than 0.01 mg/L). It uses pre- and post-chlorination, lime for pH control, and has eight GAC/sand filters. The current capacity of 90 mgd will be expanded to 200 mgd as part of a major construction project begun in 2002. In addition to new process basins, this project will include new chemical storage and handling facilities and a filter backwash water recycling system.

ICR data were used to calibrate the WTP model for this plant and evaluate its performance. Raw water bromide data were unavailable, since all were lower than the minimum reporting level in the ICR. Bromide levels were therefore assumed to be 0.01 mg/L for the purpose of this evaluation since the source water is upstream of the Delta. This plant currently relies on chlorine for primary disinfection and is not meeting the 1-log *Cryptosporidium* inactivation ELPH goal.

Since the immediate future facility also does not include treatment processes for *Cryptosporidium* removal and/or inactivation, no combination of source water quality is expected to provide ELPH compliance at this plant.

During consultation with this member agency, the utility indicated their preference for installing UV disinfection for compliance with 1-log *Cryptosporidium* inactivation if that becomes their water quality goal. Installation of UV disinfection will allow this WTP to provide *Cryptosporidium* inactivation. Therefore, the limiting source water quality will be dictated by the extent of coagulation at this conventional facility. By altering the alum dosages at this plant, the project team estimates that the plant will be able to comply with ELPH goals using raw water TOC concentrations of approximately 2 mg/L and raw water bromide levels of 0.05 mg/L with reasonable dosages of alum and lime (for pH adjustment). This TOC limitation assumes that the Fairbairn WTP will continue using free chlorine for secondary (residual) disinfection.

4.2.3   City of Sacramento – Sacramento River WTP

The process schematic, average chemical dosages, and average water quality for the Sacramento River WTP are provided in Appendix B, Slide B-7. This conventional surface water treatment plant takes water from the Sacramento River, which is of a lower quality than the American River. Its intake is downstream of the confluence of the two rivers. Its capacity will be expanded from 110 mgd to 160 mgd as part of the current construction project; eight new filters will be added to the existing 16, and new process basins will be added. This plant provides needed capacity to meet system demands when flows from the American River into the Fairbairn plant are constrained due to other legal commitments borne by the City.

ICR data were used to evaluate this plant. Similar to the Fairbairn plant, raw water bromide data were unavailable and the project team assumed 0.01 mg/L for the purpose of this evaluation since the source water is upstream of the Delta. This plant currently relies on chlorine for primary disinfection and is not meeting the 1-log *Cryptosporidium* inactivation ELPH goal. Since the immediate future facility also does not include treatment processes for *Cryptosporidium* removal and/or inactivation, no combination of source water quality is expected to provide ELPH compliance at this plant.

During consultation with this member agency, the agency staff indicated their preference for installing UV disinfection for compliance with 1-log *Cryptosporidium* inactivation if that becomes their water quality goal. Similar to the Fairbairn WTP, coagulant dosages could be altered in this conventional facility to accommodate higher TOC and bromide values. The project team estimated that this plant will also be able to comply with ELPH goals with raw water TOC of approximately 2 mg/L and raw water bromide of approximately 0.05 mg/L while maintaining reasonable coagulant dosages. Again, it was assumed that the Sacramento WTP will maintain its use of chlorine as a secondary (residual) disinfectant.

4.2.4   San Francisco Water Department – Sunol Valley WTP

The process schematic, average chemical dosages, and average water quality for the Sunol Valley WTP are provided in Appendix B, Slide B-12. This 160 mgd plant treats water mainly from the local (Bay Area) watershed and from the Calaveras and San Antonio Reservoirs, but it can also treat water from the Hetch-Hetchy pipeline as it passes through the area on its way to the South Bay and San Francisco. Free chlorine is the primary and residual disinfectant. Chlorine is added upstream of filtration at this plant. There are twelve sand/anthracite filters.

Twelve months of data were received from SFWD and the average influent water quality for this time period was evaluated. Raw water bromide data for this 12-month period were unavailable. Bromide levels were therefore assumed based on an average value for this plant from the ICR database. This plant currently relies on chlorine for primary disinfection and is not meeting the 1-log *Cryptosporidium* inactivation ELPH goal. Since the immediate future facility also does not include treatment processes for *Cryptosporidium* removal and/or inactivation, no combination of source water quality is expected to provide ELPH compliance at this plant.

During consultation with this member agency, the agency staff indicated their preference for installing UV disinfection for compliance with 1-log *Cryptosporidium* inactivation if that becomes their water quality goal. Consequently, the project team selected installation of UV disinfection for compliance with the ELPH goals. After installation of UV disinfection, the constraining raw water quality for compliance with the ELPH goals will be determined by the level of coagulation at this facility. The coagulant dosages currently used at this facility are

already approaching enhanced coagulation conditions. Nevertheless, the plant could possibly further optimize the coagulation process by slightly increasing the coagulant dosage. One of the most significant differences between the raw water qualities at this plant, compared to the Sacramento WTPs, is the higher level of alkalinity (greater than 100 mg/L as $CaCO_3$ for the Sunol Plant compared to approximately 20 mg/L as $CaCO_3$ for the Sacramento WTPs). Higher alkalinity inhibits pH depression by coagulants and therefore allows the application of higher coagulant dosages thereby resulting in greater TOC removal at reasonable coagulation pH values. Using the calibrated model, the project team estimated that the limiting raw water TOC and bromide values at this facility could be up to 3.5 mg/L and 0.05 mg/L, respectively.

4.2.5   San Francisco Water Department – Harry Tracy WTP

The process schematic, average chemical dosages, and average water quality for the Harry Tracy WTP are provided in Appendix B, Slide B-17. This plant's source water is primarily from the Hetch-Hetchy Reservoir, which flows into the San Andreas Reservoir located on the western peninsula of the San Francisco Bay. The source water is of a much higher quality than that originating in the Delta, with very low turbidity, TOC, and bromide levels. Raw water ozonation was added to this 144 mgd filtration plant in 1993. There are sedimentation basins but they are bypassed, making this plant operate as a direct filtration plant. The 10 sand/anthracite filters are biologically active. When the entire SFPUC system is converted from free chlorine to chloramines, this plant will begin adding ammonia. There are two clearwells that currently operate in parallel; these will be converted to series operation, and ammonia will be added in between the two.

Based on this evaluation using model results, the project team has estimated that the current operation at this facility is not meeting the ELPH goals due to the formation of bromate in excess of 5 µg/L. Using the calibrated WTP model for this plant, the project team estimated that the average raw water bromide at this facility should be less than 0.02 mg/L in order for the existing facilities to meet the ELPH bromate goal. The existing facilities can comply with the TTHM and HAA5 goals of the ELPH with raw water TOC of up to 2 mg/L using free chlorine for secondary (residual) disinfection. When the conversion to chloramines is completed, it is possible that this

plant will be able to treat raw water TOC values up to 4 mg/L while meeting the TTHM and HAA5 ELPH goals.

Under current average raw water quality at this plant, SFWD may consider pH reduction during ozonation or installation of UV for providing *Cryptosporidium* inactivation. Consultation with the water department staff indicated that the most likely method for compliance with the ELPH goals at this facility is to install bromate control by lowering the water pH. After the installation of bromate control through pH reduction during ozonation, this plant could treat raw water bromide values of up to 0.05 mg/L and raw water TOC values of approximately 3.5 mg/L while maintaining compliance with ELPH goals.

4.2.6   City of Los Angeles – Los Angeles Aqueduct Filtration Plant

The process schematic, average chemical dosages, and average water quality for the Los Angeles Aqueduct Filtration Plant are provided in Appendix B, Slide B-22. This direct filtration plant has a capacity of 600 mgd. Raw water ozonation is used ahead of ferric chloride addition, which is followed by high-rate filtration through deep-bed anthracite filters. The filters are biologically active; chlorine is applied after filtration, and free chorine is used as the residual disinfectant. The use of ferric chloride results in good arsenic removal; the sludge from this plant must be disposed of as a hazardous waste due to its noncompliance with one of the California WET test requirements. The reported THM data are quite variable due to the fact that during plant outages or water shortages, water in their distribution system comes from MWDSC's Jensen Filtration Plant, which they breakpoint chlorinate.

Twelve months of limited data were received from the utility and the average influent water quality for this time period was evaluated. Based on this evaluation, the project team has determined that the current operation at this facility is not meeting the ELPH goals due to the inability of the current process to provide 1-log *Cryptosporidium* inactivation and the production of TTHMs in the distribution system in excess of 40 μg/L. The current process at the plant includes the application of ozone at 1.1 mg/L without any pH reduction (raw water ozonation) and free chlorine for secondary disinfection. Based on the analysis performed during this project, it appears that this plant will be able to comply with the ELPH goals if the raw water

TOC is less than approximately 2 mg/L and the bromide concentration is less than 0.10 mg/L. The higher bromide limit for this water compared to the other non-Delta WTPs is due to the low ozone dosages applied at this plant. It should also be noted that after the conversion of the residual disinfection to chloramines, which is currently being evaluated by LADWP, this plant will probably be able to handle raw water TOC of 4 mg/L while still complying with the ELPH goals for TTHMs and HAA5.

For meeting the ELPH goals at this plant, the utility may consider additional technologies for TOC removal, reduction in pH during ozonation, or installation of UV for providing *Cryptosporidium* inactivation. Consultation with the utility indicated that the most likely method for compliance with the ELPH goals at this facility will be to install bromate control by lowering pH and increasing ozone dosage. By making these changes, it is anticipated that this plant will be able to handle raw water bromide levels of up to 0.2 mg/L. After the conversion of the distribution system to chloramines the raw water TOC limit could be higher (exceeding 4 mg/L).

4.2.7   Contra Costa – Bollman WTP

The process schematic, average chemical dosages, and average water quality for the Bollman WTP are provided in Appendix B, Slide B-27. This 75 mgd conventional surface water treatment plant receives raw water from the Delta after storage in the Mallard Reservoir. A small amount of chlorine is used as a pre-oxidant, and sulfuric acid is added to control the raw water pH that can be as high as 10 during algae blooms. After the sedimentation basins, ozone is used as the primary disinfectant. Due to its location in the process train, the pH of ozonation is typically around 6.5, which greatly helps reduce bromate formation. The plant has 6 GAC/sand biologically active filters; after filtration there is a few minutes of free chlorine contact, followed by ammonia addition to form chloramines.

Based on our evaluation, we have determined that the current operation at this facility is capable of meeting the ELPH goals using their existing ozonation system that achieves disinfection at the existing low pH conditions. Analysis performed during this project using the WTP simulation model indicates that this plant will be able to comply with the ELPH water quality goals with the existing process for raw water TOC levels up to 3 mg/L and raw water bromide levels up to 0.3

mg/L. The current treatment at this plant already includes enhanced coagulation and low ozonation pH. It appears that this plant has relatively little room to adjust their treatment processes to accommodate worse raw water quality conditions. As the analysis is indicating that this plant will be able to meet the ELPH goals without any changes, no treatment process improvements were determined to be necessary.

4.2.8   Alameda County Water District – Water Treatment Plant 2

The process schematic, average chemical dosages, and average water quality at the WTP2 are provided in Appendix B, Slide B-30. The source water for this 20 mgd plant is the South Bay Aqueduct, which originates in the Delta and can be highly variable both seasonally and daily. The plant uses pre-ozonation upstream of conventional treatment. It has six biologically active filters, three of which are anthracite/sand, and three are GAC/sand. After filtration, there is a short free chlorine contact period, followed by ammonia addition to form chloramines.

Based on our evaluation we have determined that the current operation at this facility is not meeting the ELPH goals due to the inability of the current process to provide 1-log *Cryptosporidium* inactivation. The current process at the plan includes the application of ozone at 2.3 mg/L while the pH during ozonation is lowered to 7.0 by feeding carbon dioxide. Using the calibrated WTP model for this plant, the project team has determined that the limiting source water quality for this plant are a concentration of TOC of 2 mg/L and a bromide concentration of 0.06 mg/L.

For meeting the ELPH goals at this plant under the current average raw water quality conditions, ACWD may consider additional technologies for TOC removal, further reduction in pH during ozonation, or installation of UV for providing *Cryptosporidium* inactivation. Consultation with ACWD staff indicated that the most likely method for compliance with the ELPH goals at this facility will be to install UV for additional disinfection. After installation of UV disinfection, the estimated raw water quality limits for compliance with the ELPH goals could be a TOC of 3.5 mg/L and a bromide concentration of 0.3 mg/L.

It should be noted that with relatively similar water qualities between CCWD's Bollman plant and Alameda County Water District's WTP2 the ELPH compliance with the existing plants are different. This is attributed to the fact that CCWD's Bollman Plant already practices bromate control by operating at an ozonation pH of less than 7.0. Furthermore, it is our understanding that CCWD utilizes the large raw water reservoir to dampen any excursions in Delta water bromide concentrations.

4.2.9   Santa Clara Valley Water District – Santa Teresa WTP

Process schematic, average chemical dosages, and average water quality for the Santa Teresa WTP can be found in Appendix B, Slide B-35. This is a 100 mgd surface water treatment plant, treating Delta water (primarily from San Luis Reservoir) as well as water from local reservoirs. Potassium permanganate is used as a pre-oxidant. District staff evaluated the use of chlorine dioxide, but decided to install ozone instead due to concerns about chlorite and chlorate formation. Settled water ozonation is currently under construction, and they expect the typical ozonation pH to be 6.8 – 7.0; a planned sulfuric acid feed system will be used when needed for additional bromate control. They will also have the capability to use hydrogen peroxide for advanced oxidation. They can add PAC for seasonal T&O control, but they have not found this to be completely effective. There are twelve sand/anthracite filters.

Twelve months of data were received from SCVWD staff and the average influent water quality for this time period was evaluated. The immediate future improvements for this plant include the addition of ozone at approximately 1 mg/L to achieve 0.5 log *Giardia* inactivation and the addition of GAC in the filters with no planned reactivation. Based on our evaluation, we have determined that the current operation at this facility is not meeting the ELPH goals due to the formation of TTHM in excess of 40 μg/L and the inability of the plant to provide 1-log inactivation of *Cryptosporidium*. Using the calibrated WTP model for this plant, we have determined that the limiting source water quality for this plant are TOC of up to 2 mg/L and bromide of up to 0.2 mg/L.

For meeting the ELPH goals at this plant under the average raw water quality conditions, SCVWD may consider additional technologies for TOC removal (enhanced coagulation),

reduction in pH during ozonation, or installation of UV for providing *Cryptosporidium* inactivation. Consultation with SCVWD staff indicated that the most likely method for compliance with the ELPH goals at this facility is to install bromate control by lowering pH and installation of UV disinfection. After installation of UV, this plant could treat raw water TOC levels of up to 3.5 mg/L and raw water bromide concentrations of up 0.3 mg/L while meeting the ELPH goals.

The raw water qualities and the treatment processes between this plant and the CCWD's Bollman plant are very similar, although the limiting raw water qualities are slightly different. This could be due to differences in the extent of coagulation (and hence the ozonation pH during settled water ozonation) and the extent of free chlorine contact time prior to the addition of ammonia for conversion of free chlorine to chloramines.

4.2.10 Santa Clara Valley Water District – Penitencia WTP

The process schematic, average chemical dosages, and average water quality for the Penitencia WTP are provided in Appendix B, Slide B-40. This 42 mgd conventional surface water treatment plant treats Delta water (primarily from the South Bay Aqueduct) as well as water from local reservoirs. The plant contains six filters. Settled water ozonation facilities are currently under construction at this plant with completion expected by 2004. After completion of the ozone project, the District plans to add sulfuric acid for controlling bromate, as well as an ozone residual quenching system.

Twelve months of data were received from SCVWD and the average influent water quality for this time period was evaluated. The immediate future improvements for this plant include the addition of ozone at approximately 1 mg/L to achieve 0.5 log *Giardia* inactivation and the addition of GAC in the filters with no planned reactivation. Based on our evaluation, we have determined that the current operation at this facility is not meeting the ELPH goals due to the formation of TTHM in excess of 40 μg/L and the inability of the plant to provide 1-log inactivation of *Cryptosporidium*. Using the calibrated WTP model for this plant, we have determined that the limiting source water quality for this plant are TOC of 2 mg/L and bromide of 0.2 mg/L

For meeting the ELPH goals under the current average raw water quality conditions, SCVWD may consider additional technologies for TOC removal, reduction in pH during ozonation, or installation of UV for providing *Cryptosporidium* inactivation. Consultation with SCVWD staff indicated that the most likely method for compliance with the ELPH goals at this facility is to install bromate control by lowering pH and installation of UV disinfection. After installation of UV disinfection, the model estimates that this plant can handle raw water TOC concentrations of up to 3.5 mg/L and raw water bromide concentrations of up to 0.3 mg/L.

4.2.11 MWDSC – Diemer Filtration Plant

The process schematic, average chemical dosages, and average water quality for the Diemer Filtration Plant are provided in Appendix B, Slide B-45. This plant treats a blend of State Project water and Colorado River water. The design flow is 520 mgd and the maximum flow is 710 mgd. It is a conventional surface water treatment plant with 48 anthracite/sand filters. Currently, pre-chlorination is practiced, but some other oxidant (either ozone, chlorine dioxide, or a combination of the two) will be added by no later than 2009, possibly sooner. Currently, they maintain a SUVA of 2.0 or less by adjusting the coagulant (ferric) dose and/or the raw water blend. If the plant treats less than about 25% state project water, then they meet the stage 1 DBP rule. However, if the fraction of SPW treated is higher, some alternative treatment technology is needed. For this plant, the cost of ozone vs. enhanced coagulation to meet the rule requirements is about the same, yet ozone has additional benefits such as improved disinfection and T&O control. They also have a finished water salinity target.

Twelve months of data were received from MWDSC staff and the average influent water quality for this time period was evaluated. This plant currently relies on chlorine for primary disinfection and is not meeting the 1-log *Cryptosporidium* inactivation ELPH goal. As discussed above, MWDSC has not yet determined which way to proceed with the immediate future facilities. For the purpose of this project we have assumed that the immediate future facility will not include treatment processes for *Cryptosporidium* removal and/or inactivation.   Consequently, no combination of source water quality is expected to provide ELPH compliance at this plant. Although MWDSC is considering installing ozone to achieve additional disinfection, the analysis

performed during this study indicates that the use of UV disinfection for achieving a 1-log *Cryptosporidium* inactivation may be less costly. Consequently, for the purpose of this analysis we have assumed that the plant will install UV disinfection for compliance with the ELPH goals. The only limiting source water quality after the installation of UV disinfection and reasonable increases in coagulant dosages at this plant would be a raw water TOC of 3.5 mg/L and a raw water bromide level of 0.3 mg/L. These limits are dictated by the ELPH goal of 40 μg/L for THMs.   Installation of ozone along with enhancement of the coagulation process would result in raw water quality limits of 3.5 mg/L of TOC and 0.2 mg/L of bromide.

4.2.12 MWDSC – Mills Filtration Plant

The process schematic, average chemical dosages, and average water quality for the Mills Filtration Plant are provided in Appendix B, Slide B-50. This conventional surface water treatment plant treats 100% State Project Water. Ferric is currently used as the primary coagulant, and raw water ozonation with a sulfuric acid feed system is currently being installed at this plant. There are 66 biologically active filters. For the purpose of this evaluation we have assumed that ozonation is a part of the existing facility.

Based on our evaluation, we have determined that the current operation at this facility is not meeting the ELPH goals due to the formation of bromate in excess of 5 μg/L and TTHM in excess of 40 μg/L. Using the calibrated WTP model for this plant, we have determined that the limiting source water quality for this plant include a TOC of 2 mg/L and a bromide of 0.1 mg/L. For meeting the ELPH goals under current average raw water quality conditions, MWDSC may consider additional technologies for TOC removal, reduction in pH during ozonation, or installation of UV for providing *Cryptosporidium* inactivation. Consultation with MWDSC staff indicated that the most likely method for compliance with the ELPH goals at this facility is to install bromate control by lowering pH and adjusting ozone dosages. Modeling results suggest that the lowering of ozonation pH may allow lowering of ozone dosage while still complying with the *Cryptosporidium* inactivation goal. This could allow the plant to increase the raw water bromide limit to 0.3 mg/L. The limiting raw water TOC concentrations could be up to 3.5 mg/L.

4.3   SUGGESTED PROCESS IMPROVEMENTS AND THEIR COSTS

4.3.1   Conceptual Cost Estimates

As discussed in Section 4.2, the project team utilized the calibrated Water Treatment Plant Simulation Model for evaluating the compliance with ELPH goals with the existing treatment processes and to determine the level of additional technologies needed, if any, for compliance with these goals. As discussed earlier, one of the eleven plants evaluated during this study is already in compliance with the ELPH goals and therefore does not require any additional treatment modifications. For the remaining plants evaluated in this project, specific technology improvement steps were determined and discussed in the Section 4.2. Table 4.2 summarizes the improvements identified for each plant and provides an opinion of conceptual cost estimates for these process improvements. A listing of raw water quality limits for these modified treatment processes are also included in this table. The process improvements shown in Table 4.2 were chosen by considering the prioritization obtained through interviews and a survey of the utilities, ability to meet ELPH goals, costs, and our professional judgment. For the most part, utility prioritizations were followed. Alternative technologies were considered only when the utility preferred technology was not able to meet all the ELPH goals or it was too expensive to implement the preferred technology.

Installation of UV treatment after filtration is the technology that is likely to be used by most systems that are not meeting the *Cryptosporidium* inactivation goal. Systems that are not meeting the DBP goals in addition to *Cryptosporidium* inactivation are likely to adopt pH adjustment and changes in their coagulation conditions to control DBP formation. Utilities that have ozonation (pre or intermediate) and have bromate formation issues are likely to resort to lowering the pH or decreasing ozone dosage (if it does not have any adverse impact on the plant performance).

Capital and operations and maintenance (O&M) costs were developed for all technology alternatives that can help in meeting the ELPH goals. These costs are presented in Appendix B. The costs for the most likely technology for each system are shown in Table 4.2. The costs

Table 4.2. Suggested Process Improvements for Each Treatment Plant

Name of WTP	Average Raw Water TOC (mg/L)	Average Raw Water Bromide (mg/L)	Process Improvements Needed to Meet ELPH Goals	Limiting Water Quality for Applied Process Improvements	Capital Cost	O&M Cost
Upstream of Delta						
City of Sacramento – Fairbairn WTP	1.3	0.01[1]	Install UV	Bromide = 0.05 mg/L TOC = 2 mg/L	$7.2 Million	$150,000 /year
City of Sacramento – Sacramento River WTP	1.5	0.01[1]	Install UV	Bromide = 0.05 mg/L TOC = 2 mg/L	$10.5 Million	$125,000 /year
San Francisco WD – Sunol Valley WTP	3.3	0.05	Install UV	Bromide = 0.05 mg/L TOC = 3.5 mg/L	$14.0 Million	$160,000 /year
San Francisco WD – Harry Tracy WTP	1.9	0.05	Install bromate control	Bromide = 0.05 mg/L TOC = 3.5 mg/L	$ 3.2 Million	$700,000 /year
LADWP – LA Aqueduct FP	3.0	0.12	Increase ozone dose	Bromide = 0.1 mg/L TOC = 2 mg/L	None	$1.0 Million /year
Downstream of Delta						
Contra Costa WD – Bollman WTP	3.3	0.16	None	Did not evaluate	None	None
Alameda County WD – Treatment Plant 2	3.5	0.22	Install UV	Bromide = 0.3 mg/L TOC = 3.5 mg/L	$2.0 Million	$ 65,000 /year
Santa Clara Valley – Santa Teresa WTP	3.3	0.20	Install bromate control	Bromide = 0.2 mg/L TOC = 3.5 mg/L	$1.8 Million	$5.2 Million /year
Santa Clara Valley – Penitencia WTP	3.4	0.18	Install bromate control	Bromide = 0.3 mg/L TOC = 3.5 mg/L	$770,000	$2.8 Million /year
MWDSC – Diemer FP	3.0	0.13	Install UV with pH adjustment	Bromide = 0.2 mg/L TOC = 3.5 mg/L	$47.5 Million	$6.3 Million /year
MWDSC – Mills FP	3.5	0.26	Install bromate control	Bromide = 0.3 mg/L TOC = 3.5 mg/L	$3.0 Million	$2.4 Million /year

Note: [1]Assumed value for WTP modeling purposes

shown in Table 4.2 include the costs for treatment or disinfection and additional pH adjustment steps that are associated with the technologies.

4.3.2   Benefits of Selected Treatment Technologies

Ozonation

Ozonation of the raw water prior to coagulation or ozonation of the settled water is currently practiced or is the process of implementation at majority of the water plants evaluated during this study. The primary reason for these plants to install ozone is to provide adequate primary disinfection against *Giardia*; however, the benefit of ozonation for the removal of taste and odor causing compounds (e.g., MIB and geosmin) is also a major factor for the decision of implementing ozone. In addition, ozone can assist utilities in implementing advanced oxidation for effective control of emerging contaminants such as MTBE, NDMA, and endocrine disruptors, although the dosages requirement for the oxidation of these contaminants are not clear at the present time. It is likely that the ozonation system in place now or in the process of being designed may need to be augmented with additional capacity in the future. It is believed, however, that the use of ozone as a disinfection and oxidation technology in the CUWA member agency water plants will continue to persist and may flourish even with the impending limitation of ozonation technology due to the formation of bromate in bromide containing waters. This is primarily driven by the frequent occurrences of T&O problems in Delta water, which are effectively eliminated by ozonation. None of the alternative technologies evaluated can meet this aesthetic water quality as can ozone.

UV Disinfection

Many of the member agencies of CUWA are leaning towards implementing UV disinfection to cope with a possible future 1-log inactivation requirement for *Cryptosporidium*. UV disinfection is a new technology. However, many installations are coming into existence primarily due to the attractive economics of this technology for *Cryptosporidium* and *Giardia* disinfection and the apparent lack of any disinfection by product formation. While UV technology can provide very effective disinfection against *Cryptosporidium*, this technology does not provide many of the other additional benefits that ozonation can provide as discussed above. It is conceivable,

however, that the use of UV technology may position water utilities favorably for advanced oxidation of certain emerging contaminants such as nitrosodimethyl amine (NDMA) or MTBE. The UV dosages needed for such advanced oxidation with UV may, however, be significantly greater compared to the dosages needed for *Cryptosporidium* disinfection.

Chlorine Dioxide

The use of chlorine dioxide is currently being evaluated by some of the CUWA member agencies for achieving inactivation of *Cryptosporidium*. To achieve adequate disinfection with chlorine dioxide the dosage necessary is expected to generate chlorite residuals in excess of the acceptable levels. For this reason, the use of chlorine dioxide will also need the implementation of ferrous sulfate addition for chlorite control. Application of ferrous sulfate is still in the evaluation stages and full-scale implementation has some process issues that need to be resolved. In addition to providing *Cryptosporidium* disinfection, chlorine dioxide is also capable of controlling certain types of taste and odor compounds, although it is not effective against MIB and geosmin, which are the primary causes of T&O problems in Delta water.

Low Pressure Membrane Filtration

Low pressure membrane filtration such as microfiltration (MF) or ultrafiltration (UF) is becoming a more common technology, even for larger water plants such as the ones operated by some of the CUWA member agencies. None of the plants evaluated during this study is actively considering the use of low-pressure membrane filtration for disinfection. However, the consideration for such technologies may be made in the near future. While these technologies provide a very effective physical barrier for particulate contaminants including *Cryptosporidium*, they are not very effective in removing DBP precursors unless associated with a coagulation process.

5.   FINDINGS, CONCLUSIONS AND RECOMMENDATIONS

5.1   OBJECTIVES AND APPROACH

Water purveyors throughout California are navigating an ever-changing landscape of source water availability, its associated variation in water quality, and evolving regulations that challenge existing water treatment processes. Ultimately, balancing both source water alternatives and treatment is desirable to meet increasingly stringent regulations. In 1998, an Expert Panel for CUWA recommended that source water quality including 3.0 mg/L of TOC and 50 µg/L of bromide would be required as Delta export in order to allow downstream agencies to meet potential future regulatory requirements for an array of incremental treatment processes including enhanced coagulation, ozone, GAC and membranes

We all recognize, however, that improvements in treatment technology and greater certainty in the regulatory future may alter specific source water quality targets. Accordingly, the Record of Decision for the Bay Delta Accord specified not only the numerical limits for TOC and bromide recommended by the Expert Panel, but also source water quality that allows "an equivalent level of public health protection."

This latter criterion raises a suite of questions, including:

1. What represents an "equivalent level of public health protection," also termed ELPH?

2. Can the existing treatment processes, including the near-term future improvements, implemented at the CUWA member agency plants meet the ELPH criteria? For the plants that can meet the ELPH criteria with the existing or immediate future facilities, what is the limiting source water quality (in terms of TOC and bromide) that continues to allow the existing (and near-term future) treatment processes to reliably meet the ELPH criteria?

3. What treatment process modifications are required at the member agency plants to comply with the ELPH criteria under current average raw water quality conditions? For the plants that need modifications for compliance with ELPH criteria, what are the

limiting source water qualities and how do they compare with the 3.0 mg/L for TOC and 50 µg/L bromide identified by the Expert Panel? Such an approach recognizes that incremental treatment represents a "step function" in treatment capability that may allow a more liberal definition of allowable source water quality.

4. What are the costs of the treatment options that allow CUWA member agencies to address ELPH goals?

This evaluation was initiated to address these questions. In addition to a literature review of the latest improvements and enhancements in water treatment technology, a water treatment plant simulation model – used in regulatory determination and other water quality evaluations over the past decade – was employed to identify source water quality associated with treatment options. This model was further improved through plant-specific calibration using full-scale data for individual treatment plants. The value-added component is that this model can continue to be used by CUWA member agencies to test treatment alternatives using a range of source water qualities and potential treated water quality endpoints.

No model, however, can completely replace expert judgment. Model development and use often provides baseline solutions that seed expert discussions to arrive at reasonable predictions based upon best professional judgment. The evaluation undertaken for this effort follows this precedence. Modeling was used to determine central tendency source and treated water quality predictions for various treatment options to better understand the relationship between ELPH treated water quality goals, alternative treatment processes, and limiting source water quality. Expert judgment was applied to modeling results to arrive at the conclusions and recommendations presented in this section.

5.2   DEFINING ELPH TREATED WATER QUALITY GOALS

When the Expert Panel convened between 1996 and 1998, they were requested to develop a plausible, long-term regulatory scenario that could guide CUWA member agencies in evaluating treatment process alternatives and accompanying source water quality requirements, as exported from the Delta. At that time, the M/DBP Rules were not finalized (the LT2ESWTR and Stage 2 D/DBP Rule had yet to be negotiated through a FACA process), although it was recognized that

specific finished water quality requirements associated with these two-stage regulations may not be sufficient in a long-term Delta solution that would be binding for decades to come. Accordingly, the Expert Panel evaluated short and long-term regulatory trends and arrived at the following treated water quality criteria in their analysis:

- 40 µg/L for TTHM and 30 µg/L for HAA5, measured as a running annual average.
- 5 µg/L for bromate, measured as a running annual average.
- 1 log of *Cryptosporidium* removal or inactivation, in addition to that expected in a well-operated conventional treatment plant using coagulation, flocculation, sedimentation and filtration.

The Expert Panel developed the 3.0 mg/L and 50 µg/L source water quality criteria for TOC and bromide, respectively, using this treated water quality scenario.

Given this context, the project team and representatives of the CUWA member agencies agreed that the above treated water quality criteria would be appropriate in defining ELPH goals. That is, regardless of the treatment processes employed, the ability to meet these finished water criteria would allow an assessment of limiting source water quality that would provide and "equivalent level of public health protection," consistent with the approach used by the Expert Panel in developing numerical source water constraints for specific treatment alternatives in 1998.

Since 1998, advances in treatment technology have occurred. Of particular interest, UV disinfection has evolved as a process that can provide high levels of *Cryptosporidium* and *Giardia* inactivation at cost-effective dosages. All of these developments, including the ability to implement technologies at a range of treatment plant capacities, were considered in this evaluation to describe the source water quality envelope that addresses ELPH goals for individual member agency treatment plants.

5.3   EXISTING AND IMMEDIATE FUTURE FACILITIES

As stated above, an evaluation was undertaken to determine whether existing and immediate future facilities at eleven CUWA member agency water treatment plants could meet the ELPH

water quality goals. For those that could, a range of source water quality was established to determine potential limiting conditions for these facilities. Table 4-1 summarized the assessment of compliance with ELPH goals and limiting source water quality (in terms of TOC and bromide) for the current and immediate future facilities.

Although the halogenated DBPs (TTHM, HAA5) and bromate present treatment challenges, the single factor limiting the ability of existing or immediate future facilities to meet ELPH goals is the requirement for 1-log additional inactivation/removal of *Cryptosporidium*. Of the eleven plants evaluated, only seven have current or immediate future facilities that can meet the *Cryptosporidium* inactivation criterion, all using ozone. Another plant, MWD's Diemer Filtration Plant, is currently evaluating advanced disinfection alternatives including ozonation and chlorine dioxide (chlorine dioxide is not being evaluated for *Cryptosporidium* inactivation).

The member agency plants that use ozone can meet all the ELPH goals with some combination of source water quality in terms of TOC and bromide. It is important to note that the LT2ESWTR may not require *Cryptosporidium* inactivation since the CUWA plants are most likely in the lower bin for compliance. Systems with average *Cryptosporidium* concentrations of <0.075/L fall into Bin 1 and are not required to provide any additional treatment. Nevertheless, the 1-log *Cryptosporidium* inactivation goal was consistent with the Expert Panel evaluation in 1998.

There are two plants unaffected by Delta operations that treat with ozone; SFWD's Harry Tracy Plant and LADWP's LAAFP. These plants can comply with the ELPH goals if the source water bromide and TOC levels are maintained below 0.02 mg/L and 2 mg/L, respectively. The bromide criterion is controlled by ozone treatment to meet *Cryptosporidium* inactivation. The TOC criterion is controlled by the halogenated DBP criteria (TTHM and HAA5) with minor improvements in coagulation and the continued use of free chlorine as a secondary (residual) disinfectant. It is recognized that SFWD is considering implementing chloramines at some time in the future, but this was not evaluated in this effort.

All plants affected by Delta operations, with the exception of MWDSC's Diemer Filtration Plant, use ozone in their treatment process and chloramines for residual disinfection. These plants

downstream of the Delta can meet the ELPH goals if the bromide concentrations are maintained below 0.06 to 0.3 mg/L. These levels vary from plant to plant depending upon ozone dosage, ozonation pH, and whether ozone is applied to the raw or settled water. TOC concentrations are predicted to be less than 2 to 3 mg/L, which also varies from plant to plant depending upon coagulant dosages, the extent of in-plant free chlorine contact time and coagulation pH. In this analysis, an attempt was not made to modify chlorine and ammonia addition points thereby changing the free chlorine contact time. The value of the model is that each member agency can continue to perform their own evaluations to determine the best combination of disinfectant contact times, coagulant dosages, and other operational changes to achieve the greatest flexibility in source water quality while meeting ELPH goals.

Five of the seven CUWA member agency plants using ozone may have to increase ozone dose or install UV treatment to meet the 1-log *Cryptosporidium* inactivation goal. One plant (CCWD Bollman WTP) currently meets the ELPH criteria based upon the data provided, and another (MWDSC Mills Filtration Plant) may have to limit source water quality in terms of TOC and bromide for their raw water ozonation process.

The four plants (three unaffected by Delta operations and one downstream of the Delta) that do not use ozone in their treatment process will not be able to meet the 1.0-log *Cryptosporidium* inactivation goal. According to the assumptions used in this analysis, these plants will have to implement additional process changes as described in the following section.

5.4   PROCESS IMPROVEMENTS AND COSTS FOR MEETING ELPH GOALS

All treatment plants with the exception of CCWD's Bollman WTP were evaluated to determine process improvements to meet ELPH goals. Each member agency was interviewed to determine their individual priority for process changes. Conceptual costs were estimated for the process changes selected. These results were summarized in Table 4-2.

Treatment plants with raw or settled water ozone will consider either 1) lowering ozone pH, or 2) reducing the ozone dose, if bromate compliance becomes an issue. Ozone dosages will only be lowered to the extent that other water quality objectives (e.g., T & O reduction) are met, based upon interviews with member agency representatives. Four of the seven treatment plants that

have ozone are projected to require bromate control, which can be achieved by lowering the pH during ozonation. The capital costs for bromate control ranged from $770,000 (SCVD's Penitencia WTP) to $3.2 Million (for SFWD's Harry Tracy WTP). The capital costs were mostly a function of the plant design capacity; larger plants require larger chemical feed systems to adjust pH. The O&M costs for bromate control ranged from $700,000 per year (for SFWD's Harry Tracy WTP) to $5.2 Million per year (SCVD's Santa Teresa WTP). The O&M costs are a function of the annual average flow and the acid/caustic dosage necessary to adjust pH. In addition to the lowering pH during ozonation, many of the member agency plants may also enhance their coagulation process by adding higher coagulant dose, decreasing coagulation pH through acid addition, or both. It should be noted that this process improvement also requires an increase in pH prior to distribution. One of the seven treatment plants that currently treats with ozone (ACWD's WTP2) prefers to install UV treatment for *Cryptosporidium* inactivation. This would allow a reduction in ozone dosage and thereby maintain bromate compliance with ELPH goals.

The utility survey and technology assessment results indicate that UV treatment will be preferred for the four plants without ozone in their current or immediate future facilities. Systems without ozone that are not meeting the DBP goals prefer pH adjustment during coagulation, in addition to UV treatment. One example of such a system is MWDSC's Diemer Filtration Plant. The capital costs for UV treatment ranged from $2.0 Million (ACWD's WTP2) to $47.5 Million (MWDSC's Diemer FP). The O&M costs for UV treatment range from $65,000 per year (ACWD'sWTP2) to $6.3 Million per year (MWDSC's Diemer FP).

Again, it should be recognized that the treatment processes envisioned to meet ELPH criteria, as defined for this study, may not be those currently envisioned for treatment plant upgrades. The 1-log *Cryptosporidium* inactivation/removal requirement may not be required in the near term for at least some of the CUWA member agencies. Consequently, these treatment plants may be considering other treatment options. For example, MWDSC is currently evaluating both chlorine dioxide and ozone for the Diemer plant. Chlorine dioxide is being evaluated to provide disinfection at levels that will not achieve 1-log *Cryptosporidium* inactivation credit. Further, both chlorine dioxide and ozone will address T & O reduction that will not be achieved through the addition of UV disinfection.

5.5   WATER QUALITY LIMITS FOR POTENTIAL FUTURE TREATMENT FACILITIES

Three of the five plants unaffected by Delta operations chose to install UV disinfection for compliance with the ELPH goals. In this analysis, it was assumed that these plants will continue with free chlorine as their residual disinfectant. Under these assumptions, the range of raw water quality that these plants can treat with their modified treatment process, while simultaneously meeting ELPH goals, includes raw water TOC range of 2 to 3.5 mg/L (depending on the extent of coagulation practiced at the member agency plants) and bromide levels of 0.05 to 0.10 mg/L (depending on the level of ozonation at the plant). These TOC values would likely increase if these utilities chose to implement chloramines as a residual disinfectant.

Two of the six plants downstream of the Delta chose UV disinfection to meet the *Cryptosporidium* inactivation target. The remainder of the treatment plants planned to modify ozone operation to improve bromate control. It was assumed that all of the plants downstream of the Delta would continue to use chloramines for residual disinfection and that chemical feed locations (and consequent detention times) for chlorine and ammonia would not be altered. Under these assumptions, up to 3.5 mg/L of raw water TOC can be treated. Corresponding raw water bromide levels were predicted to range from 0.2 to 0.3 mg/L. Again, the TOC values might be increased if free chlorine contact time were reduced while still achieving other water quality constraints.

Considering all the plants evaluated during this evaluation, it appears that the lowest level of raw water bromide that all the plants can treat while complying with the ELPH goals is 0.05 mg/L or 50 µg/L. This is consistent with the level estimated by the CUWA expert panel in 1998 and is constrained by bromate formation during ozone treatment. Based on the plant-by-plant analysis performed during this project, it appears that a portion of the plants could treat raw water bromide levels of up to 0.3 mg/L, depending upon their ozone location (settled versus raw water application), coagulant dosage, and ozonation pH. The lowest predicted TOC level of 2.0 mg/L was constrained by meeting halogenated DBP criteria when using free chlorine for residual disinfection in plants unaffected by Delta operations.   For the plants that are downstream of the Delta and use chloramines for residual disinfection, the raw water TOC limit was predicted to be

3.5 mg/L, a level similar to that suggested by the CUWA expert panel. Again, an effort was not undertaken in this study to modify the point of chlorine and ammonia application to potentially reduce chlorine contact and possibly increase the allowable source water TOC. The model developed in the study for each individual treatment plant could be used for this purpose.

6. REFERENCES

Acero, J. L., Haderlein, S. B., Schmidt, T. C., Suter, M. J. and von Gunten, U. (2001) MTBE Oxidation by Conventional Ozonation and the Combination Ozone/Hydrogen Peroxide: Efficiency of the Processes and Bromate Formation. *Environ. Sci. & Technol.* 35:21:4252 – 4259

Adams, C., Wang, Y., Loftin, K. and Meyer, M. (2002) Removal of Antibiotics from Surface and Distilled Water in Conventional Water Treatment Processes. *Jour. of Environ. Engin.* 128:3:253-260

Ahmad, R., Amirtharajah, A., Al-Shawwa, A. and Huck, P. M. (1998) Effects of Backwashing on Biological Filters *Jour. AWWA* 90:12:62 – 73

Asami, M., Aszawa, T., Morioka, T., Nishijima, W., Tabata, A. and Magara, Y. Bromate Removal During Transition from New Granular Activated Carbon (GAC) to Biological Activated Carbon (BAC), *Water Res.* 33:12:2797-2804

Baçaoui, A., Dahbi, A., Yaacoubi, A., Bennouna, C., Maldonado-Hódar, F. J., Rivera-Utrilla, J., Carrasco-Marín, F. and Carlos Moreno-Castilla, C. (2002) Experimental Design To Optimize Preparation of Activated Carbons for Use in Water Treatment. *Environ. Sci. & Technol.* 36:17:3844-3849

Bao, M. L., Griffini, O., Santianni, D., Barbieri, K., Burrini D. and Pantani, F. (1999) Removal of Bromate Ion from Water Using Granular Activated Carbon, *Water Res.* 33:13:2959-2970

Baribeau, H., Prévost, M., Desjardins, R., Lafrance, P. and Gates, D. J. (2002). Chlorite and Chlorate Ion Variability in Distribution Systems *Jour. AWWA* 94:7:

Bell-Ajy, K. et al (2000) Conventional and Optimized Coagulation for NOM Removal *Jour. AWWA* 92:10:44–58

Bin Alam, M. Z., Otaki, M., Furumai, H. and Ohgaki, S. (2001) Direct and Indirect Inactivation of Microcystis Aeruginosa by UV-radiation, *Water Res.* 35:4:1008-1014

Bolto, B., Dixon, D., Eldridge, R. and King, S. (2002) Removal of THM Precursors by Coagulation or Ion Exchange. *Water Res.* 36:5066-5073

Bolton, J.R. (2000) Calculation of Ultraviolet Fluence Rate Distributions in an Annular Reactor: Significance of Refraction and Reflection. *Water Res.* 34:13:3315-3324

Bolton, J.R. and Linden, K. G. (2003) Standardization of Methods for Fluence (UV Dose) Determination in Bench-Scale UV Experiments. *Jour. of Environ. Engin.* 129:3:209-215

Bukhari, Z., Hargy, T.M., Bolton, J.R., Dussert, B. and Clancy, J.L. (1999) Medium-pressure UV for Oocyst Inactivation *Jour. AWWA* 91:3:86 – 94

Bustamante, H. A., Shanker, S. R., Pashley, R. M. and Karaman, M. E. (2001) Interaction Between Cryptosporidium Oocysts and Water Treatment Coagulants, *Water Res.* 35:13:3179-3189

Bourke, M, Champlin, T., Perella, J, Topham, C. and Blattman, J. "Use of Magnetic Ion Exchange Resin to Improve DBP Precursor Removal and Reduce Coagulant Usage at Lee County's Olga WTP" presented at the Florida AWWA section conference, November 2002.

Cadee, K., O'Leary, B., Smith, P., Slunjski, M. and Bourke, M. "World's First Magnetic Ion Exchange (MIEX) Water Treatment Plant to be Installed in Western Australia" Proceedings of the AWWA Annual Conference and Exhibition, 2000.

CALFED-Bay Delta Program. Programmatic Record of Decision. August, 2000.

Campbell, A. T. and Wallis, P. (2002) The Effect of UV Irradiation on Human-Derived Giardia lamblia Cysts. *Water Res.* 35:963-969

Campos, C., Mariñas, B. J., Snoeyink, V. L., Baudin, I. and Lainé, J. M. (2000a) PAC-Membrane Filtration Process. I: Model Development *Jour. of Environ. Engin.* 126:2:97-103

Campos, C., Mariñas, B. J., Snoeyink, V. L., Baudin, I. and Lainé, J. M. (2000b) PAC-Membrane Filtration Process. II: Model Application *Jour. of Environ. Engin.* 126:2:104-111

Campos, C., Schimmoller, L., Mariñas, B. J., Snoeyink, V. L., Baudin, I. and Laîné, J. M. (2000c) Adding PAC to Remove DOC *Jour. AWWA* 92:8:69–83

Campos, C, Snoeyink, V. L., Mariñas, B., Baudin, I. and Lainé, J. M. (2000d) Atrazine Removal by Powdered Activated Carbon in Floc Blanket Reactors. *Water Res.* 34:16:4070-4080

Carlson, K. H. and Amy, G. L. (1998) BOM Removal During Biofiltration *Jour. AWWA* 90:12:42 – 52

Carlson, K. H. and Amy, G. L. (2001) Ozone and Biofiltration Optimization for Multiple Objectives *Jour. AWWA* 93:1:88–98

Carlson, K. H. and Gregory, D. (2000) Optimizing Water Treatment with Two-Stage Coagulation. *Jour. of Environ. Engin.* 126:6:556-561

Carlson, K. H., Via, S., Bellamy, W. and Carlson, M. (2000) Secondary Effects of Enhanced Coagulation and Softening *Jour. AWWA* 92:6:63–75

Carlson, M. and Hardy, D. (1998) Controlling DBPs With Monochloramine *Jour. AWWA* 90:2:95 – 106

Carroll,T., King, S., Gray, S. R., Bolto, B. A. and Booker, N. A. (2000) The Fouling of Microfiltration Membranes by NOM after Coagulation Treatment. *Water Res.*34:11:2861-2868

Chaiket, T., Singer, P. C., Miles, A., Moran, M. and Pallotta, C. (2002) Effectiveness of Coagulation, Ozonation, and Biofiltration in Controlling DBPs *Jour. AWWA* 94:12:81-95

Chakraborti, R. K., Atkinson, J. F. and Van Benschoten, J. E. (2000) Characterization of Alum Floc by Image Analysis. *Environ. Sci. & Technol.* 34:18:3969 - 3976

Chang, Y. J. and Benjamin, M. M. (2003) Modeling Formation of Natural Organic Matter Fouling Layers on Ultrafiltration Membranes. *Jour. of Environ. Engin.* 129:1:25-32

Chang, Y. J., Choo, K. H., Benjamin, M. M. and Reiber, S. (1998) Combined Adsorption–UF Process Increases TOC Removal *Jour. AWWA* 90:5:90 – 102

Chellam, S. and Jacangelo, J. G. (1998) Existence of Critical Recovery and Impacts of Operational Mode on Potable Water Microfiltration. *Jour. of Environ. Engin.* 124:12:1211-1219

Chen, W. J. and Wiesel, C. P. (1998) Halogenated DBP Concentrations in a Distribution System *Jour. AWWA* 90:4:151 – 163

Childress, A. E., Vrijenhoek, E. M., Elimelech, M., Tanaka, T. S. and Beuhler, M. D. (1999) Particulate and THM Precursor Removal with Ferric Chloride. *Jour. of Environ. Engin.* 125:11:1054-1061

Cho, J., Amy, G. and Pellegrino, J. (1999) Membrane Filtration of Natural Organic Matter: Initial Comparison of Rejection and Flux Decline Characteristics with Ultrafiltration and Nanofiltration membranes, *Water Res.* 33:11:2517-2526

Cho, M., Chung, H., and Yoon, J. (2002) Effect of pH and Importance of Ozone Initiated Radical Reactions in Inactivating Bacillus Subtilis Spore. *Ozone Sci. & Engin.* 24:145-150

Chow, C. W., House, J., Velzeboer, R. M., Drikas, M., Burch, M. D. and Steffensen, D. A. (1998) The Effect of Ferric Chloride Flocculation on Cyanobacterial Cells. *Water Res.* 32:3:808-814

Chow, C. W., Drikas, M., House, J., Burch, M. D. and Renate M. A. Velzeboer (1999) The Impact of Conventional Water Treatment Processes on Cells of the Cyanobacterium Microcystis Aeruginosa. *Water Res.* 33:15:3253-3262

Chung, Y., Choi, Y. C., Choi, Y. H. and Kang, H. S. (2000) Demonstration Scaling-up of the Dissolved Air Flotation. *Water Res.* 34:3:817-824

Clancy, J. L., Bukhari, Z., Hargy, T. M., Bolton, J. R., Dussert, B. W. and Marshall, M. M. (2000) Using UV to Inactivate Cryptosporidium *Jour. AWWA* 92:9:97–104

Clark, R. M. Thurnau, R. C., Sivaganesan, M. and Ringhand, P. (2001) Predicting the Formation of Chlorinated and Brominated By-Products. *Jour. of Environ. Engin.* 127:6:493-501

Clark, R. M., Sivagenesan, M., Rice, E. W. and Chen, J. (2002) Development of a CT Equation for the Inactivation of Cryptosporidium Oocysts with Ozone, *Water Res.* 36:3141-3149

Cook, D., Newcombe, G. and Sztajnbok, P. (2001) The Application of Powdered Activated Carbon for MIB and Geosmin Removal: Predicting PAC Doses in Four Raw Waters. *Water Res.* 35:5:1325-1333

Corona-Vasquez, B., Rennecker, J. L., Driedger, A. M. and Mariñas, B. J. (2002a) Sequential Inactivation of Cryptosporidium parvum Oocysts with Chlorine Dioxide Followed by Free Chlorine or Monochloramine, *Water Res.* 36:178-188

Corona-Vasquez, B., Samuelson, A., Rennecker, J. L. and Benito J. Mariñas, B. J. (2002b) Inactivation of Cryptosporidium Parvum Oocysts with Ozone and Free Chlorine, *Water Res.* 36:4053-4063

Cotton, C. A., Owen, D. M., Cline, G. C. and Brodeur, T. P. (2001) UV Disinfection Costs for Inactivating Cryptosporidium *Jour. AWWA* 93:6:82–94

Craik, S. A., Finch, G. R., Bolton, J. R. and Belosevic, M. (2000) Inactivation of Giardia muris Cysts Using Medium-pressure Ultraviolet Radiation in Filtered Drinking Water. *Water Res.* 34:18:4325-4332

Craik, S. A., Weldon, D., Finch, G. R., Bolton, J. R. and Belosevic, M. (2001) Inactivation of Cryptosporidium parvum Oocysts Using Medium- and Low-pressure Ultraviolet Radiation, *Water Res.* 35:6:1387-1398

Dąbrowska, A., Świetlik, J. and Nawrocki, J. (2003) Formation of Aldehydes Upon ClO_2 Disinfection. *Water Res.* 37:5:1161-1169

Delphos, P. J., Harr, K. and Bianco, T. L. "Selective DBP Precursor Removal with an Innovative Ion Exchange Process" Proceedings of the AWWA Water Quality Technology Conference, 2001.

Desjardins, C., Koudjonou, B. and Desjardins, R. (2002) Laboratory Study of Ballasted Flocculation. *Water Res.* 36:744-754

Diehl, A. C., Speitel, G. E., Jr., Symons, J. M., Krasner, S. W., Hwang, C. J. and Barrett, S. E. (2000) DBP Formation During Chloramination *Jour. AWWA* 92:6:76–90

Driedger, A. M., Rennecker, J. L. and Mariñas, B. J. (2001a) Inactivation of Cryptosporidium Parvum Oocysts with Ozone and Monochloramine at Low Temperature, *Water Res.* 35:1:41-48

Driedger, A. M., Rennecker, J. L. and Mariñas, B. J. (2000) Sequential Inactivation of Cryptosporidium parvum Oocysts with Ozone and Free Chlorine, *Water Res.* 34:14:3591-3597

Driedger, A. Staub, E., Pinkernell, U., Mariñas, B., Köster, W. and von Gunten, U. (2001b) Inactivation of *Bacillus Subtilis* Spores and Formation of Bromate During Ozonation. *Water Res.* 35:12:2950-2960

Drikas, M., Chow, C., House, J. and Burch, M. D. (2001) Using Coagulation, Flocculation, and Settling to Remove Toxic Cyanobacteria *Jour. AWWA* 93:2:100–111

Dugan, N. R., Fox, K. R., Owens, J. H. and Miltner, R. J. (2001) Controlling Cryptosporidium Oocysts Using Conventional Treatment *Jour. AWWA* 93:12:64–76

Dvorak, B. I. and Maher, M. K. (1999) GAC Contactor Design for NOM Removal: Implications of EBCT and Blending. *Jour. of Environ. Engin.* 125:2:161-165

Dyksen, J. E., Marshall, M. M., Gera, A. and Clancy, J. L. (1998) Cost of Advanced UV for Inactivating Crypto *Jour. AWWA* 90:9:103 – 111

Dykstra, T. S., Chauret, C. and Gagnon, G. A. (2002) Hydraulic Calibration and Fluence Determination of Model Ultraviolet Disinfection System. *Jour. of Environ. Engin.* 128:11:1046-1055

Ebie, K., Li, F., Azuma, Y., Yuasa, A. and Hagishita, T. (2001) Pore Distribution Effect of Activated Carbon in Adsorbing Organic Micropollutants from Natural Water. *Water Res.* 35:1:167-179

Edzwald, J. K., Tobiason, J. E., Parento, L. M., Kelley, M. B., Kaminiski, G. S., Dunn, H. J. and Galant, P. B. (2000) Giardia and Cryptosporidium Removals by Clarification and Filtration Under Challenge Conditions. *Jour. AWWA* 92:12:70–84

Edzwald, J. K., Tobiason, J. E., Amato, T. and Maggi, L. J. (1999) Integrating High-rate DAF Technology Into Plant Design *Jour. AWWA* 91:12:41–53

EPA, Technologies and Costs for Control of Microbial Contaminants and Disinfection By-Products. Draft Report. February 2002.

Escobar, I. C. and Randall, A. A. (2001a) Case Study: Ozonation and Distribution System Biostability *Jour. AWWA* 93:10:77–89

Escobar, I. C., Randall, A. A. and Taylor, J. S. (2001b) Bacterial Growth in Distribution Systems: Effect of Assimilable Organic Carbon and Biodegradable Dissolved Organic Carbon. *Environ. Sci. & Technol.* 35:17:3442 – 3447

Fan, L., Harris, J. L., Roddick, F. A. and Booker, N. A. (2002) Influence of the Characteristics of Natural Organic Matter on the Fouling of Microfiltration Membranes, *Water Res.* 35:18:4455-4463

Fonseca, A. C., Summers, R. S. and Hernandez, M. T. (2001) Comparative Measurements of Microbial Activity in Drinking Water Biofilters, *Water Res.* 35:16:3817-3824

Franceschi, M., Girou, A., Carro-Diaz, A. M., Maurette, M. T. and E. Puech-Costes, E. (2002) Optimisation of the Coagulation–Flocculation Process of Raw Water by Optimal Design Method. *Water Res.* 36:3561-3572

Frederick, H. T. and Cannon, F. S. (2001) Calcium–NOM Loading onto GAC *Jour. AWWA* 93:12:77–89

French, K., Guest, R. K., Finch, G. R. and Haas, C. N. (2000) Correlating Cryptosporidium Removal Using Dissolved Air Flotation in Water Treatment. *Water Res.* 34:16:4116-4119

Galapate, R. P, Baes, A. U. and Okada, M. (2001) Transformation of Dissolved Organic Matter During Ozonation: Effects on Trihalomethane Formation Potential, *Water Res.* 35:9:2201-2206

Galey, C. Mary-Dilé, V. Gatel, D. Amy, G. and Cavard, J. (2001) Controlling Bromate Formation *Jour. AWWA* 93:8:105–115

Gallard, H. and von Gunten, U. (2002) Chlorination of Natural Organic Matter: Kinetics of Chlorination and of THM Formation. *Water Res.* 36:65-74

Gillogly, T., Snoeyink, V. L., Elarde, J. R., Wilson, C. M. and Royal, E. P. (1998a) ^{14}C-MIB Adsorption on PAC in Natural Water. *Jour. AWWA* 90:1:98-108

Gillogly, T., Snoeyink, V. L., Holthouse, A. R., Wilson, C. M., and Royal, E. P. (1998b) Effect of Chlorine on PAC's Ability to Adsorb MIB *Jour. AWWA* 90:2:107 – 114

Gillogly, T., Snoeyink, V. L., Vogel, J. C., Wilson, C. M. and Royal, E. P. (1999) Determining GAC Bed Life *Jour. AWWA* 91:8:98–110

Gordon, G. (2001) Is All Chlorine Dioxide Created Equal? *Jour. AWWA* 93:4:163–174

Gordon, G., Gauw, R. D., Emmert, G. L., Walters, B. D., and Bubnis, B. (2002) Chemical Reduction Methods for Bromate Ion Removal. *Jour. AWWA* 94:2:91–98

Grens, B. K. and Werth, C. J. (2001) Durability of Wood-based Versus Coal-based GAC *Jour. AWWA* 93:4:175–181

Gyürék, L.L. and Gordon R. Finch, G. R. (1998) Modeling Water Treatment Chemical Disinfection Kinetics. *Jour. of Environ. Engin.* 124:9:783-793

Gyürék, L. L., Li, H., Belosevic, M. and Finch, G. R. (1999) Ozone Inactivation Kinetics of Cryptosporidium in Phosphate Buffer. *Jour. Of Environ. Engin.* 125:10:913-924

Hamm, E. and Bourke, M. "Application of Magnetized Anion Exchange Resin for Removal of DOC at the Coldiron Watkins Memorial Water Treatment Plant in Danville, KY" Proceedings of the AWWA Water Quality Technology Conference, 2001.

Heller-Grossman, L., Idin, A., Limoni-Relis, B. and Rebhun, M. (1999) Formation of Cyanogen Bromide and Other Volatile DBPs in the Disinfection of Bromide-Rich Lake Water. *Environ. Sci. & Technol.* 33:6:932 - 937

Henderson, R., Carlson, K. H., and Gregory, D. (2001) The Impact of Ferrous Ion Reduction of Chlorite Ion on Drinking Water Process Performance, *Water Res.* 35:18:4464-4473

Herner, H. A., Trosko, J. E. and Masten, S. J. The Epigenetic Toxicity of Pyrene and Related Ozonation By-products Containing an Aldehyde Functional Group. *Environ. Sci. & Technol.* 35:17:3576- 3583

Hijnen, W. A., Jong, R. and van der Kooij D. (1999) Bromate Removal in a Denitrifying Bioreactor Used in Water Treatment, *Water Res.* 33:4:1049-1053

Hoehn, R. C., Ellenberger, C. S., Gallagher, D. L., Wiseman E. V., Benninger, R. W., and Rosenblatt, A. (2003) ClO_2 and By-product Persistence in a Drinking Water System. *Jour. AWWA* 95:4:141-150

Howe, K. J. and Clark, M. M. (2002) Fouling of Microfiltration and Ultrafiltration Membranes by Natural Waters. *Environ. Sci. & Technol.* 36:16:3571 – 3576

Hozalski, R. M. and Bouwer, E. J. (2001a) Non-Steady State Simulation of BOM Removal in Drinking Water Biofilters: Applications and Full-Scale Validation, *Water Res.* 35:1:211-223

Hozalski, R. M., Zhang, L. and Arnold, W. A. (2001b) Reduction of Haloacetic Acids by Fe^0: Implications for Treatment and Fate. *Environ. Sci. & Technol.* 35:11:2258 - 2263

Huang, W. J. and Yeh, H. H. (1999) Reaction of Chlorine with NOM Adsorbed on Powdered Activated Carbon. *Water Res.* 33:1:65-72

Huber, M. M., Canonica, S., Park, G. Y. and von Gunten, U. Oxidation of Pharmaceuticals During Ozonation and Advanced Oxidation Processes. (2003) *Environ. Sci. & Technol.* 37:5:1016 – 1024

Huffman,D. E., Gennaccaro, A., Rose, J. B. and Dussert, B. W. (2002) Low- and Medium-Pressure UV Inactivation of Microsporidia Encephalitozoon Intestinalis, *Water Res.* 36:3161-3164

Joo, D. S., Choi, D. J. and Park, H. (2000) The Effects of Data Preprocessing in the Determination of Coagulant Dosing Rate. *Water Res*. 34:13:3295-3302

Judd, S. J. and Hillis, P. (2001) Optimisation of Combined Coagulation and Microfiltration for Water Treatment, *Water Res*.35:12:2895-2904

Karimi, A. A., Adham, S. and Tu, S. C. (2002) Evaluation of Membrane Filtration for Los Angeles' Open Distribution Reservoirs. *Jour. AWWA* 94:12:120-128

Karimi, A. A., Vickers, J. C. and Harasick, R. F. (1999) Microfiltration Goes Hollywood: The Los Angeles Experience *Jour. AWWA* 91:6:90–103

Katz, A. and Narkis, N. (2001) Removal of Chlorine Dioxide Disinfection By-products by Ferrous Salts, *Water Res*. 35:1:101-108

Kelley, W. A. and Olson, R. A. (1999) Selecting MF to Satisfy Regulations *Jour. AWWA* 91:6:52–63

Kirisits, M. J. and Snoeyink, V. L. (1999) Reduction of Bromate in a BAC Filter *Jour. AWWA* 91:8:74–84

Kirisits, M. J., Snoeyink, V. L. and Kruithof, J. C. (2000) The Reduction of Bromate by Granular Activated Carbon. *Water Res*. 34:17:4250-4260

Kirisits, M. J., Snoeyink, V. L., Inan, H., Chee-sanford, J. C., Raskin, L. and Brown, J. C. (2001) Water Quality Factors Affecting Bromate Reduction in Biologically Active Carbon Filters. *Water Res.* 35:4:891-900

Kirisits, M. J., Snoeyink, V. L., Chee-Sanford, J. C., Daugherty, B. J., Brown, J. C. and Raskin, L. (2002) Effect of Operating Conditions on Bromate Removal Efficiency in BAC Filters, *Jour. AWWA* 94:4:182–193

Korn, C., Andrews, R. C. and Escobar, M. D. (2002) Development of Chlorine Dioxide-related By-product Models for Drinking Water Treatment, *Water Res.* 36:330-342

Larson, M. A. and Benito J. Mariñas, B. J. (2003) Inactivation of Bacillus subtilis Spores with Ozone and Monochloramine, *Water Res*. 37:4:833-844

Lee, H., Amy, G. Cho, J., Yoon, Y., Moon, S. H. and Kim, I. S. (2001) Cleaning Strategies for Flux Recovery of an Ultrafiltration Membrane Fouled by Natural Organic Matter. *Water Res.* 35:14:3301-3308

Lee, N. H., Sinha, S., Amy, G. and Bourke, M. "Evaluation of Magentic Ion Exchange Resin for Preferential Removal of NOM Fractions" Proceedings of the AWWA Water Quality Technology Conference, 2002.

Lehtola, M. J., Miettinen, I. T., Vartiainen, T. Myllykangas T. and Martikainen, P. J. (2001) Microbially Available Organic Carbon, Phosphorus, and Microbial Growth in Ozonated Drinking Water, *Water Res*.35:7:1635-1640

Lehtola, M. J., Miettinen, I. T., Vartiainen, T. T., Rantakokko, P., Hirvonen, A. and Martikainen, P. J. (2003) Impact of UV Disinfection on Microbially Available Phosphorus, Organic Carbon, and Microbial Growth in Drinking Water. *Water Res*. 37:5:1064-1070

Li, H., Gyürék, L. L., Finch, G. R., Smith, D. W. and Belosevic, M. (2001a) Effect of Temperature on Ozone Inactivation of Cryptosporidium parvum in Oxidant Demand-Free Phosphate Buffer. *Jour. Environ. Engin*. 127:5:456-467

Li, H., Finch, G. R., Smith, D. W. and Belosevic, M. (2001b) Sequential Inactivation of Cryptosporidium Parvum Using Ozone and Chlorine. *Water Res*. 35:18:4339-4348

Li, H., Finch, G. R., Smith, D. W. and Belosevic, M. (2001c) Chlorine Dioxide Inactivation of Cryptosporidium Parvum in Oxidant Demand-Free Phosphate Buffer. *Jour. Of Environ. Engin*. 127:7:594-603

Liang, S. Palencia, L. S., Yates, R. S., Davis, M. K., Bruno, J. M. and Wolfe, R. L. (1999) Oxidation of MTBE by Ozone and Peroxone Processes *Jour. AWWA* 91:6:104–114

Lin, C. F., Huang, Y. J. and Hao, O. J. (1999) Ultrafiltration Processes for Removing Humic Substances: Effect of Molecular Weight Fractions and PAC treatment, *Water Res.* 33:5:1252-1264

Lin, C. F., Lin, T. Y. and Hao, O. J. (2000) Effects of Humic Substance Characteristics on UF Performance. *Water Res.* 34:4:1097-1106

Linden, K. G., Shin, G. A., Faubert, G., Cairns, W. and Sobsey, M. D. (2002) UV Disinfection of Giardia lamblia Cysts in Water *Environ. Sci. & Technol.* 36:11:2519 - 2522

Liu, X., Huck, P. M. and Slawson, R. M. (2001) Factors Affecting Drinking Water Biofiltration *Jour. AWWA* 93:12:90–101

Lundh, M., Jönsson, L. and Dahlquist, J. (2000) Experimental Studies of the Fluid Dynamics in the Separation Zone in Dissolved Air Flotation. *Water Res.* 34:1:21-30

Ma, J. and Liu, W. (2002) Effectiveness of Ferrate (VI) Preoxidation in Enhancing the Coagulation of Surface Waters. *Water Res.* 36:4959-4962

Madaeni, S. S. (1999) The Application of Membrane Technology for Water Disinfection. *Water Res.* 33:2:301-308

Magnuson, M. L., Kelty, C. A., Sharpless, C. M., Linden, K. G., Fromme, W., Metz, D. H. and Kashinkunti, R. (2002) Effect of UV Irradiation on Organic Matter Extracted from Treated Ohio River Water Studied through the Use of Electrospray Mass Spectrometry. *Environ. Sci. & Technol.* 36:23:5252 – 5260

Matsui, Y, Colas, F. and Yuasa, A. (2001) Removal of a Synthetic Organic Chemcial by PAC-UF Systems. II: Model Application. *Water Res.* 35:2:464-470

McCafferty, N. D., Callow, M. E., Hoggett, L., Holden, B. and Leadbeater, B. (2000) Application of a Method to Quantify Carbonate Precipitated on Granular Activated Carbon (GAC) Used in Potable Water Treatment. *Water Res.* 34:8:2199-2206

McGuire, M. J., Lieu, N. I. and Marie S. Pearthree, M. S. (1999) Using Chlorite Ion to Control Nitrification. *Jour. AWWA* 91:10:52–61

Mofidi, A. A., Meyer, E. A., Wallis, P. M., Chou, C. I., Meyer, B. P., Ramalingam, S. and Coffey, B. M. (2002) The effect of UV Light on the Inactivation of Giardia lamblia and Giardia muris Cysts as Determined by Animal Infectivity Assay (P-2951-01). *Water Res.* 36:2098-2108

Mofidi, A. A., Baribeau, H., Rochelle, P. A., de Leon, R., Coffey, B. M. and Green, J., F. (2001) Disinfection of Cryptosporidium Parvum with Polychromatic UV Light *Jour. AWWA* 93:6:95–109

Moll, D. M., R. Summers, R. S., Fonseca, A. C. and Matheis, W. (1999) Impact of Temperature on Drinking Water Biofilter Performance and Microbial Community Structure. *Environ. Sci. & Technol.* 33:14:2377 - 2382

Moore, B. C., Cannon, F. S., Metz, D. H., and DeMarco, J. (2003) GAC Pore Structure in Cincinnati During Full-scale Treatment/reactivation, *Jour. AWWA*, 95:2:

Najm, I., Tate, C. and Selby, D. (1998) Optimizing Enhanced Coagulation with PAC: A Case Study *Jour. AWWA* 90:10:88 – 95

Neumeister, S. M., Beatty, C. D. and Carlson, M. A. "The Quest to Reduce Disinfection By-products & Color Levels for Florida Groundwaters". Proceedings of the AWWA Water Quality Technology Conference, 2001.

Niquette, P., Prèvost, M., Servais, P., Beaudet, J. F., Coallier, J. and Lafrance, P. (1998a) Shutdown of BAC Filters: Effects on Water Quality *Jour. AWWA* 90:12:53 – 61

Niquette, P., Prevost, M., Maclean, R. G., Thibault, D., Coallier, J., Desjardins, R., and Lafrance, P. (1998b) Backwashing First-stage Sand BAC Filters *Jour. AWWA* 90:1:86 – 97

Nokes, C. J., Fenton, E. and Randall, C. J. (1999) Modelling the Formation of Brominated Trihalomethanes in Chlorinated Drinking Waters. *Water Res.* 33:17:3557-3568

Nowack, K. O., Cannon, F. S. and Arora, H. (1999) Ferric Chloride Plus GAC for Removing TOC *Jour. AWWA* 91:2:65 – 78

Parkinson, A., Barry, M. J., Roddick, F. A. and Hobday, M. D. (2001) Preliminary Toxicity Assessment of Water after Treatment with UV-irradiation and UVC/H2O2, *Water Res.* 35:15: 3656-3664, December 1999

Pedersen, E. J., Urbansky, E. T., Mariñas, B. J. and Margerum, D. W. (1999) Formation of Cyanogen Chloride from the Reaction of Monochloramine with Formaldehyde. *Environ. Sci. & Technol.* 33:23:4239 - 4249

Pelekani, C. and Snoeyink, V. L. (1999) Competitive Adsorption in Natural Water: Role of Activated Carbon Pore Size. *Water Res.* 33:51209-1219

Pinkernell, U. and von Gunten, U. (2001) Bromate Minimization during Ozonation: Mechanistic Considerations. *Environ. Sci. & Technol.* 35:12:2525 - 2531

Plummer, J. D. and Edzwald, J. K. Effect of Ozone on Algae as Precursors for Trihalomethane and Haloacetic Acid Production. *Environ. Sci. & Technol.* 35:18:3661 - 3668

Radziminski, C., Ballantyne, L., Hodson, J., Creason, R., Andrews, R. C. and Chauret, C. (2002) Disinfection of Bacillus subtilis Spores with Chlorine Dioxide: a Bench-scale and Pilot-scale Study, *Water Res.* 36:1629-1639

Rennecker, J. L., Mariñas, B. J., Owens, J. H. and Eugene W. Rice, E. W. (1999) Inactivation of Cryptosporidium Parvum Oocysts with Ozone, *Water Res.* 33:11:2481-2488

Rennecker, J. L., Driedger, A. M., Rubin, S. A. and Mariñas, B. J. (2000) Synergy in Sequential Inactivation of Cryptosporidium Parvum with Ozone/Free Chlorine and Ozone/Monochloramine, *Water Res.* 34:17: 4121-4130

Rennecker, J. L., Kim, J. H., Corona-Vasquez, B. and Mariñas, B. J. (2001) Role of Disinfectant Concentration and pH in the Inactivation Kinetics of Cryptosporidium parvum Oocysts with Ozone and Monochloramine *Environ. Sci. & Technol.* 35:13:2752 – 2757

Richardson, S. D., Thruston, A. D., Caughran, T. V., Chen, P. H., Collette, T. W., Floyd, T. L., Schenck, K. M., Lykins, B. W., Sun, G. and Majetich, G. (1999a) Identification of New Ozone Disinfection By-products in Drinking Water. *Environ. Sci. & Technol.* 33:19:3368 – 3377

Richardson, S. D., Thruston, A. D., Caughran, T. V., Chen, P. H., Collette, T. W., Floyd, T. L., Schenck, K. M., Lykins, B. W., Sun, G. and Majetich, G. (1999b) Identification of New Drinking Water Disinfection By-products Formed in the Presence of Bromide. *Environ. Sci. & Technol.* 33:19:3378 - 3383

Rodriguez, M. J. and Sérodes, J. B. (2001) Spatial and Temporal Evolution of Trihalomethanes in Three Water Distribution Systems. *Water Res.* 35:6:1572-1586

Rositano, J., Newcombe, G., Nicholson, B. and Sztajnbok, P. (2001) Ozonation of NOM and Algal Toxins in Four Treated Waters, *Water Res.* 35:1:23-32

Rossman, L. A., Brown, R. A., Singer, P. C. and Nuckols, J. R. (2001) DBP Formation Kinetics in a Simulated Distribution System. *Water Res.* 35:14:3483-3489
Ruffell, K. M., Rennecker, J. L. and Mariñas, B. J. (2000) Inactivation of Cryptosporidium parvum Oocysts with Chlorine Dioxide, *Water Res.* 34:3:868-876

Schäfer, A. I., Fane, A. G. and T. D. Waite (2001) Cost Factors and Chemical Pretreatment Effects in the Membrane Filtration of Waters Containing Natural Organic Matter. *Water Res.* 35:6:1509-1517

Schneider, O. D., and Tobiason, J. E. (2000) Preozonation Effects on Coagulation *Jour. AWWA* 92:10:74–87

Semmens, M. J, Burckhardt, M., Schuler, D, Davish, P, Slunjski, M., Bourke, M. and Nguyen, H. "An Evaluation of Magentic Ion Exchange (MIEX) for NOM Removal" Proceedings of the AWWA Annual Conference and Exhibition, 2000.

Sethi, S., Wiesner, M. R. and Dennis, J. E. (2001) Optimization of Hollow-Fiber Design and Low-Pressure Membrane System Operation. *Jour. of Environ. Engin.* 127:6:485-492

Sharpless, C. M. and Linden, K. G. (2001) UV Photolysis of Nitrate: Effects of Natural Organic Matter and Dissolved Inorganic Carbon and Implications for UV Water Disinfection. Environ. Sci. & Technol. 35:14:2949 - 2955

Shaw, J. P., Malley, J. P. and Willoughby, S. A. (2000) Effects of UV Irradiation on Organic Matter *Jour. AWWA* 92:4:157–167

Shim, Y., Lee, H. J., Lee, S., Moon, S. H. and Cho, J. (2002) Effects of Natural Organic Matter and Ionic Species on Membrane Surface Charge. Environ. *Sci. & Technol.* 36:17:3864 - 3871

Siddiqui, M., Amy, G., Ryan, J. and Odem, W. (2000) Membranes for the Control of Natural Organic Matter from Surface Waters. *Water Res.*34:13:3355-3370 Singer, P. C. and Bilyk, K. (2002) Enhanced Coagulation Using a Magnetic Ion Exchange Resin, *Water Res.* 36:4009-4022

Smith, P., Botica, C. and Long, B. "Design, Construction, Commissioning and Operation of the World's First Large Scale MIEX Water Treatment Plant." Proceedings of the AWWA Annual Conference and Exhibition, 2002.

Sotelo, J. L., Ovejero, G., Delgado, J. A. and Martínez, I. (2002) Comparison of Adsorption Equilibrium and Kinetics of Four Chlorinated Organics from Water onto GAC. *Water Res.* 36:599-608 Speitel, G. E., Symons, J. M., Mialaret, J. M. and Wanielista, M. E. (2000) AOP/Biofilm Processes for DOX Precursors *Jour. AWWA* 92:10:59–73

Speth, T. F. and Miltner, R. J. (1998) Adsorption Capacity of GAC for Synthetic Organics *Jour. AWWA* 90:4:171 – 174

States, S., Scheuring, M., Evans, R., Buzza, E., Movahed, B., Gigliotti, T. and Casson, L. (2000) Membrane Filtration as Post-treatment *Jour. AWWA* 92:8:59–68

States, S., Tomko, R., Scheuring, M. and Casson, L. (2002) Enhanced Coagulation and Removal of Cryptosporidium. *Jour. AWWA* 94:11:67-77

Świetlik, J., U. Raczyk-Stanisławiak, U., Biłozor, S., Ilecki, W. and Nawrocki, J. (2002) Adsorption of Natural Organic Matter Oxidized with ClO_2 on Granular Activated Carbon. *Water Res.* 36:2328-2336

Taniguchi, M., Kilduff, J. E. and Belfort, G. (2003) Modes of Natural Organic Matter Fouling during Ultrafiltration. *Environ. Sci. & Technol.* 37:8:1676 – 1683

Teodosiu, C., Pastravanu, O. and Macoveanu, M. (2000) Neural Network Models for Ultrafiltration and Backwashing, *Water Res.* 34:18:4371-4380 Ternes, T. A., Meisenheimer, M., McDowell, D., Sacher, F., Brauch, H. J., Haist-Gulde, B., Preuss, G., Wilme, U. and Zulei-Seibert, N. (2002) Removal of Pharmaceuticals During Drinking Water Treatment. *Environ. Sci. &. Technol.* 36:17:3855 - 3863

Tomaszewska, M. and Mozia, S. (2002) Removal of Organic Matter from Water by PAC/UF System. *Water Res.* 36:4137-4143 Tosa, K. and Hirata, K. (1999) Photoreactivation of enterohemorrhagic Escherichia coli following UV disinfection, *Water Res.* 33:2:361-366 Urfer, D., Huck, P. M., Gagnon, G. A., Mutti, D. and Smith, F. (1999) Modeling Enhanced Coagulation to Improve Ozone Disinfection *Jour. AWWA* 91:3:59 – 73

Vrijenhoek, E. M., Childress,, A. E., Elimelech, M., Tanaka, T. S. and Beuhler, M. D. (1998) Removing Particles and THM Precursors by Enhanced Coagulation *Jour. AWWA* 90:4:139 – 150

Volk, C. J. and LeChevallier, M. W. (2002) Effects of Conventional Treatment on AOC and BDOC Levels. *Jour. AWWA* 94:6:112-123

Volk, C, Bell, K., Ibrahim, E., Verges, D., Amy, G. and LeChevallier, M. (2000) Impact of Enhanced and Optimized Coagulation on Removal of Organic Matter and its Biodegradable Fraction in Drinking Water. *Water Res.* 34:12:3247-3257

von Gunten, U. (2003) Ozonation of Drinking Water: Part II. Disinfection and By-product Formation in Presence of Bromide, Iodide or Chlorine. *Water Res.* 37:7:1469-1487

von Gunten, U., Driedger, A., Gallard, H. and Salhi, E. (2001) By-products Formation During Drinking Water Disinfection: a Tool to Assess Disinfection Efficiency? *Water Res.* 35:8:2095-2099

Wang, G. S., Hsieh, S. T. and Hong, C. S. (2000) Destruction of Humic Acid in Water by UV Light–Catalyzed Oxidation with Hydrogen Peroxide, *Water Res.* 34:15:3882-3887 Westerhoff, P., Song, R., Amy, G. and Minear, R. (1998) NOM's Role in Bromine and Bromate Formation During Ozonation *Jour. AWWA* 90:2:82 – 94

Willemse, R. J. and Brekvoort, Y. (1999) Full-scale Recycling of Backwash Water from Sand Filters Using Dead-end Membrane Filtration. *Water Res.* 33:15:3379-3385 Xie, Y. F., and Zhou, H. (2002) Use of BAC for HAA Removal—Part 2, Column Study *Jour. AWWA* 94:5:126–134

Yuan, W. and Zydney, A. L. (2000) Humic Acid Fouling during Ultrafiltration. *Environ. Sci. & Technol.* 34:23:5043 – 5050

Yuksel, E., Sarikaya, H. Z., Ozturk, I. and Saatci, A. M., (2002) Ozonation and Direct Filtration. *Ozone Sci. & Engin.* 24:239-247

Zhang, M. and Song, L. (2000a) Mechanisms and Parameters Affecting Flux Decline in Cross-Flow Microfiltration and Ultrafiltration of Colloids. *Environ. Sci. & Technol.* 34:17:3767 - 3773

Zhang, M. and Song, L. (2000b) Pressure-Dependent Permeate Flux in Ultra- and Microfiltration *Jour. of Environ. Engin.* 126:7:667-674

Zhang, X. and Minear, R. A. (2002) Decomposition of Trihaloacetic acids and Formation of the Corresponding Trihalomethanes in Drinking Water. *Water Res.* 36:3665-3673 Zhang, M., Li, C., Benjamin, M. M. and Chang, Y. (2003) Fouling and Natural Organic Matter Removal in Adsorbent/Membrane Systems for Drinking Water Treatment. *Environ. Sci. & Technol.* 37:8:1663 - 1669

Zhou, H. and Xie, Y. F. (2002) Using BAC for HAA Removal—Part 1, Batch Study. *Jour. AWWA* 94:4:194–200

Appendix A

RECENT DEVELOPMENTS IN TREATMENT TECHNOLOGIES

A.1 OVERVIEW

A literature review was conducted to provide an understanding of the state-of-the-art in drinking water treatment processes. Drinking water related, peer-reviewed articles published during the past five years in the following industry journals were reviewed:

1. *Journal of the American Water Works Association* [American Water Works Association]
2. *Water Research* [International Water Association]
3. *Environmental Science & Technology* [American Chemical Society]
4. *Journal of Environmental Engineering* [American Society of Civil Engineers]
5. *Ozone Science & Engineering* [International Ozone Association]

When there was insufficient material on a specific target technology in these peer-reviewed publications, conference proceedings were reviewed.

 The drinking water technologies were grouped according to the target short-list of treatment technologies identified by the Project Advisory Committee. The list of technologies includes:

- Chlorine/Chloramines
- Chlorine Dioxide
- Ozone
- UV
- MF/UF
- GAC/PAC
- MIEX®
- Advances in Conventional Treatment Processes

Where appropriate, each section is further divided into various sub-sections. For example, the ozone section is divided into sub-sections on ozone for disinfection purposes and ozone for oxidation purposes.

A.2 CHLORINE/CHLORAMINES

Gallard and von Gunten (2002) published a study on the kinetics of the formation of THMs when natural organic matter is chlorinated. They divided THM precursors into fast-acting and slow-acting fractions, and showed that the long-term chlorine demand and the formation of THMs could be described by second-order kinetics. The slow-acting THM precursors in this water consisted of phenolic compounds. The influence of various types of pre-treatment on THM formation and chlorine demand was studied. UV treatment did not alter THM formation, but did increase chlorine demand. Ozonation did not alter chlorine demand, but lowered THM formation. Chlorine dioxide reduced THM formation and also lowered chlorine demand. THM formation during chlorination, particularly the brominated species, was studied by Nokes et al. (1999). These researchers evaluated 17 different waters with varying bromide levels, and developed a model for individual THM species formation based on the ratio of bromide to chlorine dose. They found that this model matched the observed data well, in spite of a range of chlorination conditions used. It was not as good, however, in situations where the bromide was low compared to the NOM concentration. Clark et al. (2001) also developed a model to predict DBP formation when water is chlorinated. This model takes into account various factors such as initial chlorine concentration, chlorine consumption, bromide concentration, and pH. The model shows that higher levels of bromide in the raw water favor the formation of brominated compounds, and these form faster than the chlorinated ones.

Carlson and Hardy (1998) evaluated five utilities using free chlorine as a primary disinfectant followed by chloramine as a secondary disinfectant, looking at how common treatment variables such as chlorine dose, contact time, pH, temperature, etc. affect DBP formation. In all waters, increasing chlorine contact times led to increased THM and HAA levels, and the formation followed a two-phase model (rapid formation followed by slower formation). The ratio of THM:HAA varied considerably from one source to another. Low pH values led to slower THM

formation, as expected, although this effect was not as pronounced above a pH of about 7.5. The formation of HAAs as a function of pH was found to be "arbitrary and site-specific". Both THM and HAA formation increased linearly with temperature. Higher chlorine doses resulted in higher THM and HAA levels, particularly at a Cl_2:TOC below 1.0. This observed threshold Cl_2:TOC level is considerable less than the 6.0 reported in other studies, and the authors think this is due to the shorter contact times evaluated in this study. The authors conclude that the most important variable was free chlorine contact time.

Rodriguez and Sérodes (2001) measured THM levels throughout three distribution systems that used free chorine. They found significant spatial and temporal variations, especially at higher temperatures. They developed a model to describe the THM formation kinetics, and found that temperature was the most important factor in predicting THM concentrations. Rossman et al. (2001) created a simulated distribution system environment to determine how well THM and HAA levels generated in glass bottles matched those generated in distribution system pipes. Although chlorine decayed much more rapidly in the pipe system, there was not a corresponding decrease in HAA levels, and THM levels actually increased. Zhang and Minear (2002) documented the decomposition of HAAs and subsequent formation of THMs. Hozalski et al. (2001) showed that HAAs are reduced by contact with elemental iron, such as that found in unlined cast iron pipes.

DBPs formed during chloramination were studied by Diehl et al. (2000). In this study, three waters were tested at bench scale with pre-formed chloramine. The authors examined the effects of pH, Cl_2:N ratio, and bromide concentration on the formation of several different DBPs including THMs, HAAs, DOX, and cyanogen halides. Generally, low pH values, high Cl_2:N ratios, and high bromide levels led to the highest formation of disinfection by-products, although none of the reported THM and HAA levels were high (all were less than 40 and 30 µg/L, respectively). There were exceptions to the general trends, and the authors stated that this "necessitates testing of each water source". Pedersen et al. (1999) studied the chloramination by-product cyanogens chloride, in particular its formation when water containing formaldehyde is chloraminated. Formaldehyde is a common by-product of ozonation, and these researchers studied the formation pathway and kinetics of these reactions. They found that cyanogen

chloride could be formed at significant levels under drinking water treatment conditions. Chen and Weisel (1998) measured the concentrations of several DBPs in a chloraminated distribution system over a one-year period. They found that THM levels increased with increasing residence time, but the levels of haloacetonitriles, haloketones, chloropicrin, and HAAs all decreased with increasing residence time. They conclude that the maximum-residence-time location is not the appropriate sampling location for all DBPs in the distribution system.

The formation of cyanogen bromide (CNBr) in high-bromide water was studied by Heller-Grossman et al. (1999). These researchers evaluated DBP formation levels when a water source with very high bromide (1.9 mg/L) was disinfected with chlorine, chloramine, and chlorine dioxide. The highest concentration of cyanogen bromide (36 µg/L) was found at a chlorine dose of only 1.5 mg/L, a pH of 7.5, and a contact time of 10 hours. Cyanogen bromide yields were much lower when the water was treated with chloramine or chlorine dioxide. The CNBr concentration was found to be stable for ten days.

A.3 CHLORINE DIOXIDE

Chlorine dioxide is a strong disinfectant and an effective oxidant. It cannot be shipped and stored; it must be generated at the point of use. A variety of generation methods are used. In newer installations, generation is done by reacting chlorine gas with solid sodium chlorite, while older applications include the use of an aqueous sodium chlorite solution and sometimes an aqueous chlorine solution (hypochlorite). When added to natural waters, chlorine dioxide does not form appreciable amounts of THMs or HAAs, which is one of its major advantages when compared to chlorine. However, it can produce inorganic by-products, most notably chlorite and chlorate ions. Due to uncertainties in the quantities of these by-products formed and their health effects, the use of chlorine dioxide was limited in California prior to the Stage 1 DBP Rule effective date of January 1, 2002. At that time, EPA established an MCL for chlorite of 1.0 mg/L and a maximum residual disinfectant level (MRDL) for chlorine dioxide of 0.8 mg/L. In addition, California has had an action level for chlorate for some time (the current chlorate action level is 1.0 mg/L). As a consequence, the California Department of Health Services (DHS) has

become willing to more fully consider chlorine dioxide applications in California drinking water systems.

Several utilities have reported customer complaints of kerosene-like or cat-urine-like odors when chlorine dioxide is used. These odors have been linked to air-phase reactions between chlorine dioxide gas released from the water and organic compounds found in homes, most notably those used in new carpets. It has been reported that this odor problem can be solved by eliminating the chlorite at the treatment plant or by switching from free chorine to chloramine in the distribution system.

Gordon (2001) summarized various chlorine dioxide generation technologies and discussed the by-product types and levels that may be produced during generation and use of ClO_2. The chemistry is very complex, and not completely understood. Gordon (2001) thoroughly discusses issues of purity, yield, efficiency, by-products, etc. that are associated with the use of chlorine dioxide in drinking water treatment. Korn et al. (2002) developed a predictive model based on various water quality parameters to estimate chlorine dioxide consumption and chlorite production. They found that in the seven waters tested, over 90% of the predicted chlorite levels were within ± 0.1 mg/L of the measured levels.

Hoehn et al. (2003) performed a study at a full-scale system using chlorine dioxide and monitored the concentrations of chlorine dioxide and its by-products both within the treatment plant and throughout the distribution system. They also evaluated GAC for removing chlorite from the treated water. In this system, chlorine dioxide was added to the raw water. This was followed by conventional treatment, and then by GAC filtration. Free chlorine was used as the secondary disinfectant. Overall, chlorine dioxide doses were low (0.3 – 0.4 mg/L), and chlorine dioxide was consistently detected at low levels in the distribution system (0.02 – 0.2 mg/L) but not in the plant effluent. Chlorite in the distribution system was consistently less than the current MCL of 1.0 mg/L, but measurable at 0.6 – 0.8 mg/L. Consistent with other studies, the authors concluded that chlorine dioxide re-formed in the distribution system by the reaction of the chlorite formed with free chlorine. Customer complaints about objectionable odor were received from nine separate locations, and in all cases new carpets had been installed within the previous

six months. In two of these locations, chlorine dioxide was measured in the tap water at very low levels (0.03 and 0.07 mg/L). Turning off the chlorine dioxide feed or lowering the dose to around 0.25 mg/L resulted in no further odor complaints. The authors also found that the GAC removed about 60% of the chlorite, which was less than expected, and was not sufficient to prevent distribution system re-formation of chlorine dioxide. Baribeau et at (2002) also evaluated chlorine dioxide by-products in distribution systems. In this study, three different distribution systems were investigated; one in which chlorine dioxide was used as the residual disinfectant, and two in which free chlorine was used following chlorine dioxide use at the treatment plants. The authors attempted to track the fate of chorine dioxide and its by-products in the distribution systems as a function of residence time, pipe diameter and materials, temperature, and post disinfectants. Chlorine dioxide in the distribution systems was always below its detection limit (<0.2 mg/L). They found that chlorite decreased with time. Chlorate levels were stable and very low. In warm water, chlorate levels were higher, and decreased with increasing residence time.

Chlorine dioxide use as a primary disinfectant in conjunction with chloramine as a secondary disinfectant can be a viable option. McGuire et al. (1999) conducted a study of the ability of chlorite ion (a by-product of the use of chlorine dioxide) to inactivate ammonia-oxidizing bacteria. In bench-scale studies, they found high levels of inactivation of this type of bacteria. They evaluated five distribution systems in Texas and found that in those systems in which chlorite was present, there was less loss of chloramine and ammonia-nitrogen and thus less nitrification than in those system in which chlorite was not present. The authors suggested that this nitrification control strategy might be employed even by systems not using chlorine dioxide; that chlorite might be added in low concentrations to control nitrification.

Świetlik et al. (2002) investigated the effect of chlorine dioxide oxidation on the adsorption of natural organic matter by GAC. They found that even small doses of chlorine dioxide may significantly increase the adsorption of NOM on GAC. These authors also found very good removal of chlorite through the GAC filters. In a later study, these same researchers (Dąbrowska et al. 2003) found that aldehydes, similar to those formed by ozonation, can be formed by disinfection of water with chlorine dioxide.

Katz and Narkis (2001) investigated the removal of inorganic chlorine dioxide by-products by the use of ferrous (Fe^{2+}) salts as reducing agents, converting them to chloride ions. Studies were carried out both in the presence and absence of oxygen. Chlorate was virtually unaffected by the addition of the reducing agent, but chlorite was effectively removed. In the absence of oxygen, these researchers found that a lower dose of ferrous was needed; whereas when oxygen was present, it exerted a ferrous demand, and therefore a higher dose was needed for chlorite removal. For example, in water exposed to atmospheric oxygen, a ferrous dose of about 11 mg/L was able to reduce chlorite from a starting concentration of 3.5 mg/L to less than 0.2 mg/L.

As the chlorite is reduced to chloride, the ferrous ion is converted to ferric ion. Henderson et al. (2001) conducted a study to evaluation the effect of this ferric ion on downstream treatment processes. These researchers investigated ferrous addition at three different points in the process: pre-rapid mix, pre-settling, and pre-filtration. They found that chlorite was successfully reduced in all cases, but that only the pre-rapid mix injection location resulted in good performance of downstream processes. Adding the ferrous salt anywhere downstream of flocculation resulted in unacceptable filter performance.

Chlorine dioxide's ability to inactivate *Cryptosporidium parvum* has been studied by numerous researchers. Ruffell et al. (2000) examined the effects of pH and temperature on the inactivation of *Cryptosporidium* oocysts with chlorine dioxide. Generally, they found that pH was not very important, and that the kinetics were well-described by the Chick-Watson equation. They include a table of CT values (without a safety factor) that utilities using chlorine dioxide might consider for inactivation of *Cryptosporidium*. These values (approximately 40 to 250 mg/L-min for 1-log inactivation at typical Delta water temperatures) may be feasible in some drinking water treatment plants. Li et al. (2001c) also examined the effects of temperature and pH on chlorine dioxide's inactivation of *Cryptosporidium*, using animal infectivity method to determine oocysts viability. The CT values reported in Li et al's work are very close to those of the Ruffell study, and they also found that pH was not important. However, the temperature effects were quite different. Ruffell measured about a three-fold increase in CT for each 10ºC drop in

temperature, while Li measured about a two-fold increase. Li speculated that this may be due to the use of different methods to determine viability of the oocysts.

Corona-Vasquez et al. (2002a) looked specifically at the sequential use of chlorine dioxide followed by free chlorine or monochloramine and their impact on the inactivation of *Cryptosporidium*. Unlike the results of sequential ozone-chlorine or ozone-monochloramine, the sequential use of chlorine dioxide-chlorine and chlorine dioxide-monochloramine did not show any synergistic inactivation effects. Radziminski et al. (2002) studied the inactivation of *Bacillus subtilis* spores with chlorine dioxide, since these spores have been shown to be good surrogates for *Cryptosporidium* oocysts under certain conditions. In the bench-scale part of this study, the authors found good agreement between the inactivation of spores and published data on the inactivation of *Cryptosporidium*. At pilot scale, they found that they could achieve 2-log inactivation of the spores with chlorine dioxide while maintaining both chlorite and chlorine dioxide below regulated levels.

A.4 OZONE

The majority of CUWA members currently use ozone, or will be implementing ozone treatment at their water treatment plants. Ozone is used to meet multiple water quality and operational objectives. It is a powerful oxidant capable of destroying many chemicals that impart taste and odor in drinking water, as well as numerous micro-pollutants such as pesticides, methyl-tertiary-butyl-ether (MTBE), and others. The use of ozone has also been shown to help with coagulation and flocculation, reducing coagulant demand, and improving particle removal through filtration. Ozone is also a powerful disinfectant that is quite effective against *Cryptosporidium*. Contrary to chlorine, ozone does not form the regulated disinfection by-product groups of THMs and HAAs.

Nevertheless, ozone use generates its own unique by-products. Ozone reacts with the natural organic matter (NOM) present in all waters to produce several organic by-products such as aldehydes, ketoacids, and others. While some of these organic by-products are of health concern, none are currently regulated. Most of these organic compounds are highly

biodegradable, which can be a problem in a distribution system, since an increase in the amount of biodegradable organic carbon in the system increases the potential for bacterial regrowth and several associated problems such as loss of chlorine residual and positive coliform hits. Therefore, reducing their concentrations in the finished water is highly desirable. This is achieved by allowing for biological activity in the plant's media filters. Removal of these organic by-products with biological filtration varies greatly depending on the type of chemical being removed, water temperature, contact time through the filter, and filter media type.

The most difficult-to-remove by-product of the ozonation process is bromate, BrO_3^-, an inorganic ion formed when water containing bromide is ozonated. This by-product is not readily biodegradable and is regulated at an average finished water concentration of 10 µg/L.

A.4.1 Ozone as a Disinfectant

The use of ozone for the inactivation of virus and *Giardia* to meet the requirements of the Surface Water Treatment Rule (SWTR) has been well demonstrated for more than a decade. Much information has also been collected on the inactivation of *Cryptosporidium* with ozone. While chlorine and monochloramine are completely ineffective against *Cryptosporidium* under typical water treatment conditions, ozone has been proven to be a highly effective biocide against this pathogen, especially at warm temperatures.

Gyürék et al. (1998 and 1999) performed bench-scale studies over a pH range of 6 to 8 and temperatures between 1ºC and 22ºC using animal infectivity in an attempt to quantify the inactivation kinetics. These researchers found that pH did not affect the inactivation of the pathogen. They concluded with some disinfection design criteria for systems using ozone, and CT tables derived from their data. Cho et al. (2002) performed studies using *Bacillus subtilis* spores, which are generally thought to be good surrogates for *Cryptosporidium* oocysts. These researchers found that pH was important in ozone inactivation, and that higher levels of inactivation were measured at higher pH values. It is known that more hydroxyl radicals are formed at higher pH's, so the researchers added a hydroxyl radical scavenger, and repeated the experiments. They found that the inactivation levels were the same at all pH values. From this,

they concluded that hydroxyl radicals are important in ozone inactivation, and that the ozonation pH does affect inactivation rates.

As reported by Rennecker et al. (1999), results from different studies often do not produce consistent results. These authors used a modified *in vitro* excystation method to assess the viability of different strains of *Cryptosporidium* oocysts during inactivation studies. They found that ozone inactivation of *Cryptosporidium* was characterized by a "lag" phase, in which little-to-no inactivation occurred, followed by a pseudo-first-order loss of viability. They also explored the temperature dependence of the inactivation kinetics. They found that different strains produced different temperature-effect results. A paper by Li et al. (2001a) studied the ozone inactivation kinetics of *Cryptosporidium* over a range of pH values and temperatures, using animal infectivity. These authors concluded that pH had no significant impact, that temperature was very significant, and that the excystation method "is a less reliable viability assay than animal infectivity." In a paper by USEPA staff (Clark et al. 2002), an equation was derived to calculate the CT values required to achieve various log-inactivation of *Cryptosporidium* with ozone. Work is still ongoing to develop the CT table for the inactivation of *Cryptosporidium* with ozone in a water treatment plant. A CT table will be included in the upcoming proposed Long-Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR), which should be published this summer (2003).

It has been found that ozone in combination with other disinfectants can inactivate more *Cryptosporidium* than either disinfectant alone, suggesting a synergistic effect between the sequential disinfectants. Drieger et al. (2000) discuss a synergistic effect when exposure to ozone is followed by exposure to chlorine. *Cryptosporidium* oocysts that were pre-treated with ozone experienced a six-fold increase in the rate of inactivation compared to that with chlorine alone. This effect was quite pH-dependent, which is consistent with the concept that hypochlorous acid is the major species responsible for disinfection when free chlorine is used as the secondary disinfectant. Another study by the same research group (Renneker, 2000) confirmed the ozone/free chlorine synergy and found an even stronger synergy in the ozone/monochloramine combination. As in the earlier study, *Cryptosporidium* oocysts were pre-treated with ozone, and then exposed to chlorine or chloramine. Temperature effects were also

examined; the synergy between the ozone and the secondary disinfectant was more pronounced at lower temperatures. In experiments with monochloramine, for example, the inactivation rate was between 2.4 and 9.2 times faster with ozone pre-treatment than with monochloramine alone at 30ºC and 10ºC, respectively. The authors concluded that water utilities using ozone followed by chlorine or monochloramine can provide substantially greater protection against *Cryptosporidium* compared to that achieved with a single disinfectant. In another paper by this same group of researchers (Driedger et al. 2001a), the low end of the temperature range was further explored, with temperatures as low as 1ºC. They confirmed that the synergy between ozone and monochloramine was more pronounced at lower temperatures. The inactivation rates with monochloramine were five times faster at 20ºC and 22 times faster at 1ºC compared to monochloramine alone. They presented a CT table that utilities might consider for monochloramine preceded by ozone pre-treatment. Although no safety factor was included, these values are within the range achieved in many water treatment plants. An hour or two of contact time at reasonable monochloramine levels of around 2 – 3 mg/L might provide one or more log of inactivation of *Cryptosporidium*. Rennecker et al. (2001) did additional sequential inactivation studies exploring the effect of pH and disinfectant concentration. They concluded that ozonation pH was not significant within the range of 6 to 10, and the pH during subsequent chloramine exposure was not significant within the range of 8 and 10. Corona-Vasquez et al. (2002b) followed up on this work by exploring the kinetics of the sequential use of ozone and free chlorine. These researchers found that the kinetics varied with various oocysts lots, and with age of oocysts within a lot. They speculated that there were "weak" oocysts and "strong" oocysts, and mixtures of the two, and that the kinetics of their inactivation were difficult to quantify. They observed the same synergy as the earlier studies, but some inactivation curves exhibited a two-stage decay rate after the initial lag phase. Another paper by Li et al. (2001b) also explored the sequential inactivation of ozone and chlorine. Using animal infectivity, they found that ozone pre-treatment increased the effectiveness of free chlorine by about 4 to 6 times compared to that of chlorine alone. These authors concluded that "..the synergistic effect of ozone followed by free chlorine might compensate for the decrease in ozone efficacy at low temperatures." Larsen and Mariñas (2003) evaluated sequential disinfection (ozone and chloramine) of *B. subtilis* spores. The authors concluded that "..*B. subtilis* spores might not be conservative surrogates for *Cryptosporidium* oocysts for ozone disinfection at low tempertures"

due to their variability and potentially greater resistance to the disinfectants. Conversely, the spores may be overly conservative surrogates for *Cryptosporidium* inactivation by monochloramine at low temperatures.

A.4.2. Ozonation By-products – Bromate

The only ozone by-product that is currently regulated is bromate (BrO_3^-), which is formed from the reaction of the added ozone with the naturally occurring bromide ions (Br^-). This issue is of paramount concern to all Delta water users because of the elevated bromide concentrations in Delta water. A significant amount of research has been conducted on two bromate control approaches: One involves minimizing the formation of bromate, while the other involves the removal of bromate from water after it forms. The following subsections discuss the most recent published information on these two control approaches.

A.4.2.1. Removing Bromate

Removal of bromate has been the subject of much research. As reported by Bao et al. (1999), "it is almost impossible to both optimize disinfection and removal of micropollutants and to minimize bromate formation in a single treatment step." These researchers studied bromate removal by granular activated carbon (GAC). They found that under some conditions, GAC was capable of removing bromate. These researchers looked at the effect of GAC type, empty bed contact time, water characteristics, pre-loading with natural water, and thermal regeneration. In laboratory water, some GAC types worked well, others did not. When crushed and used as PAC, these better-performing carbons were not capable of removing bromate. When DOC and several anions were added to the water, bromate removal significantly decreased. They performed additional mass-balance studies and showed that the removal was due to reduction of the bromate to bromide, rather than adsorption onto the carbon. The authors concluded that source waters with moderate to high levels of DOC or anions such as bromide, nitrate and sulfate, can expect poor removal of bromate with GAC. This same general conclusion was reached by Kirisits et al. (2000) who investigated bromate reduction in laboratory and natural waters. Asami and co-workers (1999) also investigated bromate removal with GAC, specifically during the transition from new GAC to biologically-active carbon (BAC). Consistent with other researchers, these authors found that new GAC was capable of bromate reduction, but this

capability decreases dramatically as the GAC ages. After about 3 months, the GAC was not longer able to reduce bromate.

Another technique for bromate removal that has been investigated is the use of a fixed-film denitrifying bio-reactor (biological filtration). Hijnen et al. (1999) speculated that this process, also used for biological nitrate removal, might be effective for bromate removal. They found that the rate of bromate removal was much slower than that of nitrate removal, and that the process was not likely to be practical in a drinking water application. Kirisits and Snoeyink (1999) observed bromate removal during biological filtration. They investigated the effect of dissolved oxygen and nitrate concentrations (common electron receptors) and found that if these compounds were minimized, high levels of bromate reduction (around 80%) could be achieved. However, the dissolved oxygen concentration needed to be lowered significantly and they note that re-aeration might be necessary following filtration in a full-scale plant.   In another BAC/bromate removal study by this same research group, Kirisits et al. (2001 and 2002) the effects of filter operation on bromate removal were explored. They found that backwashing did not adversely affect bromate removal, and DO concentrations at or below 2 mg/L were best for bromate removal. DO was not present in the filter effluent under these conditions. Nitrite was observed periodically in the filter effluent, and perturbations of normal filter operation resulted in irreversible degradation of performance.

Gordon et al. (2002) explored bromate removal via chemical reduction to bromide. Their study focused on two reducing agents: sulfite ion (SO_3^{2-}) and ferrous iron (Fe^{2+}), and attempted to quantify their rates of bromate reduction under various conditions. They found that under drinking water conditions, Fe^{+2} is a good bromate reducing agent; while sulfite was not effective. At a pH of 8, 100 μg/L of BrO_3^- was reduced in 2 minutes. These experiments were done in the absence of oxygen, and the authors state that "Dissolved oxygen…is expected to react with SO_3^{2-} and Fe^{2+} under normal water treatment conditions. When applying SO_3^{2-} or Fe^{2+} as a removal strategy, water utilities must consider the competition between BrO_3^- and other species…and adjust concentrations accordingly."

A.4.2.2. Minimizing Bromate Formation

A pilot study conducted by Galey et al. (2001) confirmed what several other researchers have found: lowering the pH of ozonation can control the formation of bromate. These authors evaluated the effect of pre-ozonation and PAC addition on bromate formation in three plants outside of Paris, France. These plants need to remove atrazine and provide disinfection against *Cryptosporidium*, while minimizing bromate formation. They found that pre-ozonation (up to 1.2 mg/L ozone dose) and PAC addition did not have any impact on bromate formation during intermediate ozonation; while intermediate ozonation pH and ozone dose were quite significant. They concluded that an ozonation pH of 6.4 was necessary to achieve a 2-log *Cryptosporidium* inactivation, and that ozone residual quenching may be necessary to keep the bromate concentration less than 10 μg/L. It should be noted that the raw water bromide concentration in these waters was between 30 and 50 μg/L.

Pinkernell and von Gunten (2001) summarized the mechanisms of bromate minimization during ozonation. These authors developed a kinetic model for bromate formation resulting from various control options such as pH depression, ammonia addition, and addition of hydroxyl radical scavengers. For the water studied, ammonia addition was able to reduce bromate formation, but it was only effective up to a certain concentration. A further increase of ammonia beyond about 0.2 mg/L did not show an improvement in bromate control. This suggests that this method is not efficient in waters that already contain moderate to high levels of ammonia. With regard to pH, they found that about 2 μg/L of bromate formed regardless of pH, but additional bromate formation was successfully slowed by decreasing the water pH from 8 to 6. Von Gunten (2003) summarized many of the ozone disinfection by-product formation reactions and mechanisms, as well as some discussion of inactivation mechanisms. This author states that "...when bromide concentrations are above about 50 μg/L, it may be necessary to use control measures to lower bromate formation (lowering of pH, ammonia addition)."

Bromate formation and disinfection during ozonation were investigated by Driedger et al. (2001b). This study confirmed the pH depression and ammonia addition control strategies. Lowering the pH from 8 to 6 resulted in a 50% reduction in bromate formation while still achieving 2-logs of *B. subtillis* spore inactivation. Ammonia addition was somewhat effective.

The authors state that "...for bromide levels greater than approximately 50 μg/L, achieving 2-log inactivation of *Cryptosporidium* while limiting bromate formation to 10 μg/L may require the implementation of bromate control strategies."

A.4.3. Other Ozonation By-Products

In addition to bromate, other unique by-products are formed by the ozonation process. Ozone can degrade certain polycyclic aromatic hydrocarbons (PAHs). It is possible to completely oxidize these compounds to carbon dioxide, water, and straight-chain aliphatics. However, often there are by-products formed which can be as or more harmful than the parent compound. Herner et al. (2001) found that when pyrene was ozonated, the toxicity of the resulting mixture, as measured by an *in vitro* cell culture using rat liver cells, was often higher than that of pyrene itself. Richardson and co-workers conducted a study (1999a) that attempted to identify "as many compounds as possible" upon ozonation of low-bromide water, as well as in water that was chlorinated and chloraminated following ozonation. They found many organic by-products, many of which had not been identified previously. Most of the compounds they identified contained oxygen in their structures, with no halogenated DBPs found unless chlorine or chloramine was used after ozonation. They found that chloraminated waters generally produced the same by-products as chlorinated ones, but at lower levels. Most of the halogenated by-products were also formed at higher levels when the waters were treated with chlorine or chloramine alone. However, a few compounds were formed at higher levels when ozone-chlorine or ozone-chloramine treatment was used, compared to the secondary disinfectants alone. This indicates that ozone-chlorine or ozone-chloramine treatment is important in their formation. No information is yet available on the health effects of these compounds. This same research group conducted a similar study looking for new DBPs in ozonated waters containing high bromide levels (Richardson et al. 1999b). They found many identifiable and unidentifiable by-products.

A.4.4. Ozone for Oxidation

Work continues on studying the use of ozone, or ozone in combination with hydrogen peroxide (known as an Advanced Oxidation Process or AOP), to oxidize organic pollutants found in source waters. Liang and co-workers (2001) found that ozone alone and ozone in combination

with hydrogen peroxide, was capable of oxidizing (MTBE). However, unless the pH of ozonation was lowered, the resulting water contained bromate at levels higher than 10 μg/L. These authors concluded that further optimization of the process is necessary to minimize bromate formation while providing acceptable MTBE oxidation. Acero et al. (2001) also investigated oxidation of MTBE by ozone and by ozone/hydrogen peroxide. These authors developed rate constants for the reactions, and identified several degradation products of the reactions. In experiments with several natural waters, these authors found that with a bromide level of 50 μg/L, only 35 – 50% of the MTBE could be oxidized without exceeding the bromate standard of 10 μg/L. They stated that "...MTBE oxidation by conventional ozonation and the AOP O_3/H_2O_2 has to be carefully optimized with regard to bromate formation."

Plummer et al. (2001) investigated the effect of ozonation of cultured algae on THM and HAA formation. These authors found that the ozonation of certain types of algae produced significant amounts of THM and HAA precursor material. For example, *cyclotella*, a diatom, when ozonated at 1 mg/L increased chloroform formation by 5 – 26%, and by 39 – 109% when ozonated at 3 mg/L. This is significant since these diatoms are common in source waters. Relative to other sources of organic matter, these results suggest that algae blooms may contribute significantly to DBP precursor material. The authors state that most of the DBP precursors were attributable to the cellular material, and therefore removal of algae cells, especially the diatoms and blue-green algae, from a drinking water supply prior to chlorination will reduce THMs and HAAs. Galapate and co-workers (2001) also investigated the effect of ozonation on subsequent THM formation potential. These researchers found that reduction in measured dissolved organic carbon (DOC) concentrations during ozonation does not accurately predict the reduction in THM formation potential when the water is later chlorinated. They concluded that the dissolved organic matter is transformed by the ozonation process from a more reactive hydrophobic DOC to a less reactive hydrophilic DOC.

Not only do algae contribute to DBP precursor material, but they also can produce toxins in water, particularly the blue-green algae (cyanobacteria). Rositano et al. (2001) studied the ability of ozone to destroy some of these algal toxins. In their study, microcystin LR (an important hepatoxin) and anatoxin-a (a neuro-toxin) were ozonated in four different waters. The authors

found that complete destruction of these toxins could be achieved under conditions typically used in drinking water treatment. However, the saxitoxin class of compounds proved more resistant to ozone, and would require further treatment for removal.

There has been much concern lately regarding pharmaceutical compounds in drinking water. The ability of ozone to oxidize some of these compounds has been studied. Huber et al. (2003) investigated the ability of both ozone and ozone/hydrogen peroxide treatment to destroy nine selected pharmaceuticals. They developed rate constants in bench scale experiments, and concluded that "..ozonation and AOPs are promising processes for an efficient removal of pharmaceuticals in drinking water." Ternes et al. (2002) also evaluated the destruction and removal of pharmaceuticals, both in a laboratory and at pilot and full-scale plants. They found that coagulation and filtration was not effective at removing these compounds, but ozone was very effective in most cases. They also found GAC to be quite effective for the removal of most of the compounds studied. Adams et al. (2002) found that an ozone dose of 0.3 mg/L was able to achieve as much as 95% destruction of specific antibiotics.

A.4.5. Other Effects on Water Treatment

The use of ozone in a surface water treatment plant is not without effects on downstream processes. Urfer et al. (1999) developed a mathematical model that describes the effect of enhanced coagulation on the intermediate ozone system at a plant in Ontario, Canada. When the plant is operated in the enhanced coagulation mode, the pH of the settled water is lower and contains less organic carbon compared to the raw water. Both of these factors resulted in longer-lasting ozone residuals and thus higher CT values. On average, they found that operating in the enhanced coagulation mode increased their CT by about a factor of three.

Ozone has been used as a pre-treatment ahead of direct filtration. Yuksel et al. (2002) describe the effect of pre-ozonation on the effluent quality and headloss through a deep-bed filter operated in a direct filtration mode. In this study, the same water was treated through parallel filters, one with pre-ozone and the other with aeration. No coagulants were used in this study. The ozonated filter consistently showed lower turbidity values and lower particle counts than the non-ozonated filter. The authors speculated that the ozone reduced the surface charge of the

particles, leading to more effective charge neutralization. Schneider and Tobiason (2000) conducted a study in several different types of natural waters with different coagulants examining the effect of pre-ozonation on coagulation. They found that the effects were not consistent; in some cases, pre-ozonation improved performance, while in others it did not. Generally, when alum alone was used, performance with pre-ozonation was worse, and when cationic polymers were used, performance with pre-ozonation was improved.

Another issue with the use of ozone is the formation of biodegradable organic matter. It is believed that ozone converts large organic molecules into smaller, more biodegradable ones commonly quantifiable as assimilable organic carbon (AOC) or biodegradable organic carbon (BDOC). Elevated levels of AOC or BDOC may lead to increased bacterial growth in a distribution system. For this reason, ozone is often followed by biologically active filters which remove this biodegradable organic matter before it leaves the treatment plant. Escobar and Randall (2001a) evaluated data from a full-scale plant in Florida before and after the addition of ozone. This plant did not include a biofiltration step after ozonation. Samples from the distribution system were collected for one year before ozone went on line, and for another year after ozonation had started. Consistent with other researchers, they found a large (more than 200%) increase in AOC after the ozone system went on line. Hetrotrophic plate counts in the distribution system also increased significantly, although overall the numbers were low. In spite of the increased AOC levels, distribution system regrowth was repressed by the chlorine residual used for secondary disinfection. In another paper by these same researchers (Escobar et al. 2001b), data from this same Florida plant was compared to another plant using lime softening and nanofiltration. These authors found a good correlation between the measured AOC and HPC levels for both distribution systems. The study highlighted the importance of maintaining a disinfectant residual in the distribution system to control regrowth.

Fonseca et al. (2001) applied new techniques to assess the microbial communities in biofilters treating ozonated and non-ozonated waters. These authors attempted to measure the metabolic activity of the attached microorganisms in the filters using the reduction in a tetrazolium salt (INT). They found that this measurement showed a good correlation with DOC removal and that it was faster and less expensive than the phospholipid technique. Lehtola and co-workers (2001)

suggest that microbially available phosphorus (MAP) is also important to measure.   These researchers found that bacterial growth in distribution systems is sometimes phosphorus limited, rather than organic-carbon limited, and that the ozonation process can increase the amount of MAP in the distribution system, thus enhancing conditions for microbial growth.

Carlson and Amy (2001) discussed ozone and biofiltration optimization for multiple treatment objectives using case study examples. The ozone dose needed for disinfection, for example, is not always the same as that needed to optimize the performance of the biofilters. They break down BDOC into $BDOC_{rapid}$ and $BDOC_{slow}$; the former being removable by typical biofilters. They report that there is a unique ozone dose that will maximize the formation of $BDOC_{rapid}$, and increasing the ozone dose beyond this point will not result in any additional removal of BDOC in the biofilters. This optimum ozone dose was different for each water tested. They stated that when ozone is applied for optimal disinfection, the biological treatment process will be operating at close to its optimum. Additional optimization beyond that required for disinfection does not appear to be beneficial.

Removal of biodegradable organic matter in drinking water biofilters was simulated using a computer model called BIOFILT in a paper by Hozalski and Bouwer (2001). The authors found that BOM composition –relative amounts of readily degradable matter and more-slowly-degradable matter– was very important in predicting performance. A high concentration of readily degradable BOM was found to enhance the removal of the slowly degradable material primarily due to the ability to maintain a greater biomass on the filter. Consistent with full-scale results, BOM removal was not adversely affected by backwashing. The computer model was found to match full-scale plant data well. Chaiket et al. (2002) evaluated the performance of a pilot-scale biofiltration system for the removal of organic matter, particularly DBP precursors. These authors conducted their study on a high-alkalinity, low-SUVA water, one generally not amenable to enhanced coagulation.   They found that ozone-biofiltration was effective in reducing THM and HAA formation potentials, in spite of relatively low overall TOC removals. Ozone was the most effective process for reducing the concentration of THM and HAA precursors in this water because of its ability to alter the nature of the precursors, making them less reactive with chlorine. Another evaluation of post-ozone biofilter performance was done by

Carlson and Amy (1998) in which BOM levels before and after biofiltration were measured. These researchers also analyzed ten different ozone by-products to assess whether one might be used as a surrogate for BOM. None of the ten were found to serve this purpose. They found that the Empty Bed Contact Time (EBCT) through the filter was an important design parameter, but that the performance was fairly independent of hydraulic loading rates once the filters had reached steady state conditions. Xie and Zhou (2002) demonstrated that biofiltration is effective for removing the five regulated haloacetic acids (HAAs).

The effect of backwashing on biologically active filters is a concern since biofiltration is a relatively new process in the US. Ahmad et al. (1998) evaluated various backwashing strategies to control the amount of biomass attached to the media and also investigated the effect of sudden hydraulic transients. The techniques investigated included air scour, chlorinated backwash water, and different bed expansions. Air scour was not detrimental to biological filters, and adding chlorine to the backwash water was found to degrade the removal of AOC. While transient hydraulic loads resulted in significant immediate particle breakthrough from both biological and nonbiological filters, a sudden 30% increase in the hydraulic loading rate had a greater effect on the biological filter than on the conventional filter. The effect of taking a biological filter off-line was studied by Niquette et al. (1998b). In their study, dissolved oxygen dropped rapidly to below 2 mg/L within the first 2 hours of shutdown. Biomass densities declined quickly. When the filter that had been shutdown was restarted without backwashing, the effluent concentrations of DOC and ammonia were higher than before the shutdown. A backwash prior to returning the filter to service was found to eliminate these negative effects.

Lui et al. (2001) conducted a laboratory-scale study to evaluate the effect of various parameters on biofilter performance. They evaluated the effects of temperature, media type, backwashing with and without air scour, and chlorine and chloramine in the backwash water. When evaluated alone, none of the parameters had a large effect on performance, but the interaction between them was significant. For example, chlorine in the backwash water did not have a significant effect on performance except when anthracite filters were used to treat difficult-to-remove BOM at low temperatures using a high chlorine concentration in the backwash water. Substantial impairment in performance was only seen when all factors were at their most unfavorable

conditions. Air scour had no effect. Removal of acetate, formate, and formaldehyde, all of which are ozonation by-products, was found to be around 90%, while glyoxal removal was poorer. The effect of media type and backwashing on biofilter performance was also studied by Niquette et al. (1998a). These authors measured ammonia, BDOC, aldehyde and oxalate before and after backwashing, and found that backwashing did not adversely affect the filter's ability to remove these compounds, and activated carbon was better than sand. Moll et al. (1999) studied the effect of temperature on biofilter performance and found that NOM removal was poorer at lower temperatures.

A.5   ULTRAVIOLET (UV) IRRADIATION

Over the past few years, UV has emerged as a highly viable and economical disinfection technology. Bukhari et al. (1999) investigated the inactivation of *Cryptosporidium* using medium pressure UV light at bench scale, using a collimated beam apparatus, and at demonstration scale (215 gpm) using a UV reactor. These researchers used both an *in vitro* excystation method and *in vivo* (animal infectivity) method. Their data indicated that the *in vitro* assay grossly underestimated the inactivation of *Cryptosporidium* with UV. UV doses as low as 19 mJ cm^{-2} provided 3.9-log inactivation of *Cryptosporidium* oocysts.[1] They also concluded that the bench scale work agreed well with the demonstration scale studies.

In another study (Clancy et al. 2000), attempts were made to develop a dose-response curve for *Cryptosporidium* exposed to UV light and to quantify differences in lamp type (low and medium pressure). These authors also investigated inactivation of *Cryptosporidium* that had been spiked into recycled backwash water supernatant. Unlike prior work, this study included measurement of the actual irradiance distribution in the water using a radiometer and sensor across the beam. They found that quite low dosages of UV can be highly effective for inactivating oocysts, and that there were no significant differences between the low and medium pressure lamps. For example, they measured a 3.4-log inactivation at a dose of 3 mJ cm^{-2}. Also, they found that UV inactivation was effective in the presence of turbidity, although they caution that these data are quite preliminary. These authors discuss at length the inherent variability in the use of the

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[1] Since 1 J = 1 W•s, then 1 mJ cm^{-2} is the same as 1 mW•s cm^{-2}. Different studies use different units for UV dosage reporting.

animal infectivity method and caution that "for development of inactivation criteria for full-scale use, a conservative approach would be to use the lowest inactivation levels measured or to provide a significant safety factor."

Craik et al. (2001) developed a dose-response model for the inactivation of *Cryptosporidium* by UV. The resulting relationship is not linear, with a significant tailing effect at higher UV dosages. The authors measured a 2 log inactivation at a dose of 10 mJ/cm^2, and 3 log at a dose of 25 mJ/cm^2. They found that temperature, water type, and lamp type did not affect the results. Mofidi et al. (2001) also developed a dose-response curve for the inactivation of *Cryptosporidium* with UV light. Rather than animal infectivity, these authors used a human cell culture method which they acknowledge has uncertainties in its comparison with animal infectivity. Nevertheless, results from this study were similar to others in that a UV dose of only 6 mJ/cm^2 was needed for 2 log inactivation of *Cryptosporidium*. These researchers also found a good correlation between UV inactivation of *Cryptosporidium* and *E.coli*, and suggest that if confirmed, *E. coli* may be a good surrogate in future studies on the inactivation of *Cryptosporidium* with UV.

Giardia inactivation with UV has also been studied. Craik et al. (2000) exposed *Giardia muris* cysts to medium pressure UV light in filtered natural water. Animal infectivity, dye staining, and in vitro excystation methods were used to determine inactivation. The *in vitro* methods did not agree with the animal assays, consistent with the results of other researchers. These authors speculate that this is due to the germicidal action of the UV light. It is believed that UV causes damage to the cell's DNA, preventing successful reproduction, yet allowing excystation under laboratory conditions. They concluded that although the *in vitro* methods may be appropriate for other disinfectants such as ozone, they should not be used in UV disinfection studies. A UV dose as low as 5 mJ/cm^2 was shown to inactivate more than 2-logs of *Giardia* in this study.

Campbell and Wallis (2001) and Linden et al. (2002) also demonstrated inactivation of *Giardia* by UV. These two groups of researchers used *Giardia lamblia*, the species that is pathogenic to humans, and animal infectivity using Mongolian gerbils. Linden et al. found that inactivation was "very rapid and extensive" with UV, reaching the detection limit of >4 log within a dose of

10 mJ/cm^2. Campbell and Wallis reported inactivation levels of about 2 logs for this same dose. Campbell and Wallis also confirmed the lack of correlation between the *in vitro* methods and the *in vivo* (animal infectivity) method when UV is used to inactivate *Giardia*.

Mofidi et al. (2002) investigated the effect of low doses of UV on both *Giardia lamblia* and *Giardia muris*, using animal infectivity as the endpoint. They found that very low doses (3 mJ/cm^2) resulted in more than 2-log inactivation of both types of cysts. Like other researchers, these authors discussed the variability inherent in using live organisms, and cautioned that this variability needs to be taken into account when interpreting results. They also concluded that the responses of *G. muris* and *G. lamblia* were similar, so the easier-to-handle *G. muris* could be used for future studies.

The inactivation of other microorganisms with UV irradiation has also been studied. Tosa and Hirata (1999) conducted a study of the UV doses required to inactivate various strains of *E. coli*. Doses ranging from 5 to 12 mJ/cm^2 were reported to achieve 1 to 2 log inactivation of this bacterial strain. A higher dose was necessary after "photo-reactivation" or repair of cell damage caused by the UV exposure. This repair mechanism allows "inactivated" microorganisms to regain viability. The authors found that higher doses were necessary to inactivate these "re-activated" bacteria. Huffman et al. (2002) conducted a study of the inactivation of microsporidia (*Encephalitozoon intestinalis* spores) by UV. Both low and medium pressure lamps were used and the authors found that this emerging pathogen can be inactivated by UV at levels similar to those found effective for *Cryptosporidium*. A UV dose of 6 mJ/cm^2 was shown to inactivate more than 3.6 logs of microsporidia. Bin Alam et al. (2001) investigated inactivation of *microcystis aeruginosa*, a type of algae, by UV. Boats equipped with UV lamps have been used for controlling algae in some small eutrophic lakes in Japan; these researchers studied the effectiveness of UV in controlling algae. They found that UV can be effective in this application.

The cost of UV treatment for inactivation of *Cryptosporidium* has been examined. Dyksen et al. (1998) compared advanced UV system costs with those of other chemical disinfectant schemes such as ozone and chlorine dioxide. They concluded that UV "may be cost-effective…and could

be cost-competitive with conventional chemical treatment alternatives." Cotton et al. (2001) estimated capital, operations and maintenance, and total annualized costs for retrofitting existing plants with UV after filtration and before clearwell storage. Their costs were based on a UV dose of 40 mJ/cm^2, which they felt would ensure at least 2 log *Cryptosporidium* inactivation. They found that UV disinfection was relatively inexpensive; costs were lower than those of other technologies providing similar levels of *Cryptosporidium* removal/inactivation. For example, the UV costs to achieve > 2 log inactivation were 40 – 80% lower than ozone costs to achieve 1 – 2 log inactivation. This cost advantage with UV is gained without forming any known disinfection by-products.

When UV is used in a full-scale application, it is important that the UV dose be quantified and controlled. One method used is chemical actinometry using the photochemical decomposition of potassium ferrioxalate. Dykstra et al. (2002) studied the effects of hydraulic mixing on dose measurement. Under experimental conditions, chemical actinometry consistently over-estimated the dose being supplied with the reactor because of poor mixing and short-circuiting. They found that when they included dispersion and kinetics in their numerical model, their results agreed well with UV design curves. Bolton (2000) developed a mathematical model to calculate the dose distribution and the average dose within a UV reactor. This article describes in detail the optics of cylindrical UV reactors. Bolton and Linden (2003) propose a standardized protocol, along with construction specifications for a bench scale apparatus, that can be used for determining UV dose.

Aside from disinfection, UV's effects on other parameters have been studied. Shaw et al. (2000) looked at the effect of UV treatment on organic matter, specifically the fraction of the NOM that is biodegradable. Two surface waters and two groundwaters were examined. In each water source, the BDOC and AOC levels were measured before and after UV treatment. When all of the data were analyzed together, there was no statistically significant change in either parameter after UV treatment. However, in some individual waters, there were significant increases in biodegradable carbon concentrations. The authors concluded that utilities considering UV for waters with high or reactive organic matter concentrations in the absence of residual disinfectants should evaluate potential regrowth problems. Lehtola et al. (2003) investigated the

effect of UV on the formation of bio-available phosphorus and organic carbon. These researchers found that at levels used for drinking water disinfection, formation of these re-growth inducing parameters was not significant. Magnuson et al. (2002) also studied the effect of UV treatment on natural organic matter. These researchers analyzed the Extracted Organic Matter (EOM) from water that had been irradiated. When conventional spectroscopy and size exclusion chromatography were used, no significant changes were seen. However, electrospray ionization mass spectrometry (ESI-MS) showed significant changes in the organic matter resulting from UV treatment. The UV dose also appeared to affect the subsequent chlorine demand of the water. DBP formation tests did not reveal a significant increase in regulated DBPs following chlorination of the UV-treated water. The authors suggested that it might be important to consider the effects of UV treatment on organic matter in the water.

Sharpless and Linden (2001) investigated formation of nitrite when UV is used. There is concern that photolysis of nitrate to nitrite can occur during UV treatment, and these authors studied the effect of pH and NOM concentrations on this reaction. When low pressure mercury lamps was used (monochromatic UV light) nitrite formation via photolysis was not a problem. However, medium pressure lamps (broad range of wavelengths) may be a concern since the lower wavelengths are known to produce higher amounts of nitrite. The authors found that nitrite formation is "unlikely to pose a health concern" under conditions likely to be used in water treatment. At UV doses of up to $300 - 400$ mJ/cm^2, nitrite formation was less than 20% of the MCL of 1.0 mg/L. The potential toxicity of UV-treated waters was studied by Parkinson et al. (2001). These authors used three different toxicity tests (acute immobilization, cytotoxicity and Microtox) on two samples of water treated with UVA, UVB, UVC, and UV/H$_2$O$_2$. These tests indicated that the treated waters were non-toxic.

UV oxidation of natural organic matter, particularly in combination with hydrogen peroxide, has been studied. Wang et al. (2000) used a batch UV reactor to evaluate UV/H$_2$O$_2$ for the destruction of humic acids. Humic acid oxidation was found to be most rapid at very high (0.01% or 100 mg/L) peroxide concentrations. The presence of carbonate/bicarbonate species was found to have a negative affect, presumably due to their scavenging of the hydroxyl radicals. Speitel et al. (2000) also evaluated UV and peroxide for oxidation of DBP precursors, this time

in combination with subsequent biodegradation. They compared UV/H_2O_2 with ozone/H_2O_2 and ozone alone, all three followed by biofiltration. Their goal was not to completely oxidize the organic material with the AOP but to partially oxidize precursor material to enhance biodegradation. The UV/H_2O_2 biodegradation system performed comparably to the ozone biodegradation system, and both were better than the ozone/H_2O_2 biodegradation system for the waters tested in terms of DOXFP removal. The authors conclude that UV/H_2O_2 system is simpler than an ozone system, and might be a good choice for utilities treating this type of water (low SUVA).

A.6   MF/UF MEMBRANE FILTRATION

It is generally accepted that microfiltration (MF) and ultrafiltration (UF) membranes cannot remove a significant fraction of the natural organic matter (maximum of 10 to 15%), but organic matter is well removed with nanofiltration (NF) membranes. All membranes (MF through RO) are capable of *Giardia* and *Cryptosporidium* removal, but only UF and smaller membranes can remove viruses. Madaeni (1999) published a general review paper about membranes for water disinfection. This paper provides background material and references for the history of membrane use in drinking water treatment, rejection mechanisms, disinfection performance of various types, and fouling considerations. The author concluded with a listing of advantages and disadvantages of membranes for potable water disinfection. Interestingly, cost is not listed as one of the potential disadvantages.

In a study of low-turbidity surface waters, Siddiqui et al. (2000) evaluated several different membranes and found that the nanofiltration units were capable of removing most (86 – 97%) of the DBP precursor material. Microfiltration membranes demonstrated virtually no DBP precursor removal, and ultrafiltration membranes showed only modest (<30%) removal of these organics. In this study, the primary focus was on the nanofiltration membranes; the MF and UF units were intended as pre-treatment for the surface waters.

Schäfer et al. (2001) compared twelve different membranes of varying pore sizes from NF to MF, providing an estimate of cost as function of membrane performance. The pore diameters of

the membranes were calculated, and a relationship between DOC rejection and pore size was developed. This relationship clearly indicates that for a pore size of more than about 6 nm (corresponding to the difference between NF and UF) DOC rejection drops from >80% to about 10%. These authors present a Water Quality Parameter (WQP), which is a measure of the ability of the membrane to remove different contaminants. A total score is given, up to 300, which corresponds to the percent rejection of colloids of various sizes, DOC, and ions such as Ca^{2+} and Na^{+}. For example, an MF membrane was given a WQP score of 58, while an NF membrane had a score of 278. Then this WQP was correlated with overall costs, both for clean membranes and ones that had become fouled. They also investigate the effect of pre-treatment with a coagulant. Their cost calculations included both capital and operational factors. They conclude that membranes with tighter pore sizes (NF) are superior to more porous membranes with pre-treatment in terms of overall costs. However, if only turbidity removal is needed (no organics removal) then MF and UF have a clear cost advantage.

Judd and Hillis (2001) also investigated costs for membrane operation. In this study, microfiltration with ferric pre-treatment was analyzed. The authors analyzed the size of the particles formed as a function of time after coagulant addition, concluding that it took only a few minutes to form particles significantly larger than the membrane pore size, and therefore large flocculation basins were not necessary. When coagulant doses were too low, there was a deleterious effect on membrane performance, indicating that incomplete growth of the floc particles lead to irreversible clogging of the membrane. They plotted coagulant dose vs. cost, and found that there was a minimum. When the dose was too low, pumping and cleaning costs were high, and when dose was too high, chemical and solids handling costs were high.

Fan et al. (2001) studied the character of the organic matter responsible for fouling membranes, in this case MF membranes. They found that the high molecular weight fraction of NOM (>30 kDa) was responsible for most of the flux decline. The higher the aromaticity of the NOM in the water being treated (higher SUVA), the greater the flux decline. Lee et al. (2001) proposed cleaning strategies for UF membranes, suggesting that the cleaning regime be tailored to the nature of the organic foulant and the membrane material in order to best recover flux rates. They found that the hydrophobic NOM foulants were cleaned more effectively with acid and caustic

schemes, compared to the relatively hydrophilic NOM. An anionic surfactant was found to be ineffective on both types of foulants. The hydrophilic foulants were better removed by a high ionic strength (NaCl) cleaning.

Flux changes over time in membrane systems are difficult to predict. A paper by Teodosiu et al. (2000) presented the development and application of a mathematical model to predict flux losses over time. This neural network model was trained with a laboratory-scale system, and secondarily refined using pilot-scale membranes. The authors stated that it was able to accurately capture the non-linear dynamics of the flux decline, which should help predict operational changes.

The hybrid or combination process of adsorbents (mainly powered activated carbon) and ultrafiltration membranes has been investigated by several groups of researchers. Tomaszewska and Mozia (2002) studied this combination for the removal of organic matter, specifically humic acid and phenol. These researchers found that the addition of PAC to deionized water resulted in a flux decline, contrary to what other researchers had found. They speculate that this is due to the use of different types of PAC. When they examined the surface of the membranes, they found a relatively loose structure and high porosity layer of PAC on the membrane surface, regardless of PAC dose. This may be due to the fact that the outer layer of the cake is continuously removed by the flowing retentate. Generally, they found that PAC addition enhanced the removal of organics and reduced membrane fouling, since the organic material adsorbed onto the PAC (not the membrane), which is then easily removed in the backwashing process.

Zhang et al. (2003) investigated the effect of various adsorbents on fouling of membrane systems. They investigated the behavior of UF systems to which PAC, heated iron-oxide particles (HIOPs), or non-adsorbent SiO_2 particles were added. The PAC removed the most NOM from the water, the HIOPs removed less (40% vs. 60% removal), and SiO_2 removed essentially none. However, in the case of both PAC and SiO_2, increasing the dose of solids led to an increase in fouling, whereas the opposite trend applied when HIOPs were added. In the absence of NOM, none of the solids fouled the membranes. Thus, even though NOM causes

fouling, removing it from solution does not necessarily reduce fouling. The mechanism of removal can be just as important as the absolute removal if this removal occurs in a cake layer near the surface of the membrane. These authors suggest that the NOM binds PAC or SiO_2 particles to one another and to the membrane surface, so that the particles become part of the foulant. By contrast, the NOM appears to bind HIOPs to one another, but not to the membrane. This process leaves enough pore space in each cake layer for water to reach the membrane with minimal resistance.

In an article focusing exclusively on HIOPs as adsorbents in combination with UF membranes, Chang et al. (1998) evaluated sixteen different waters in bench scale tests. They compared performance of the hybrid process to that achieved by enhanced coagulation, determining whether the TOC removal targets in the D/DBP rule were met for each water. The HIOP/UF process showed as good or better removal of TOC in all waters tested compared to conventional coagulation. The authors conclude that the HIOP/UF system offers a "technically viable" alternative to enhanced coagulation for removing NOM from many waters. In this study, they captured, regenerated, and re-used the HIOPs. This was done by settling, centrifugation, caustic soaking, and subsequent pH adjustment back to near neutral. They stated that "...disposal options for the used regenerant solution and ways of separating natural colloidal matter from the HIOPs prior to regeneration have not yet been explored."

In three separate articles, Campos et al. (2000a, 2000b and 2000c) discuss the development, application, and verification of a mathematical model describing the PAC/membrane process. The objective was to develop and verify a model to predict the removal of organic compounds when PAC is applied to various membrane reactor configurations and operations. These authors discuss how the mass of organic material removed depends on the residence time of the carbon in the system. The dosing procedures (continuous or pulsed input, reactor configuration, wastage rate, etc.) determine the residence time. They modeled four different configurations, two with reactors (contact basins) upstream, and two in which the PAC was added directly to the membrane unit. They verified these four models with a bench scale flow-through apparatus, and found the model to accurately predict the removal of the target organic compound (4-nitrophenol in this case). These researchers went on to evaluate the impacts of the various modes of

operation on system performance.   They state that when there is no reactor upstream of the membrane, performance is better when the PAC is added all at once (i.e. a pulse input) rather than continuously. In this mode of operation, the residence time of all of the carbon is equal to the backwash frequency. Significantly lower carbon doses were needed to achieve performance equivalent to that achieved when the PAC was added continuously. Longer filtration times (lower backwash frequencies) and smaller PAC particles led to better organics removal. They state that when filtration times are short, an upstream reactor may be needed. Matsui et al. (2001) developed and tested a similar model for a PAC/UF system. As with the Campos et al. work, the model fit the experimental data well, except in the case of the pulse-input carbon dose. These researchers stated that "...there seems to be no clear benefit of pulse dose over continuous dose." They speculated that this might be due to the fouling effect of the NOM on the pore diffusion coefficient.

Chellam and Jacangelo (1998) conducted a pilot MF study investigating operational parameters. They found that operating at a constant pressure, rather than at a constant flux, improved performance; constant flux operation accelerated the rate of fouling. They believe this was due to membrane compaction and cake compression. They found a critical recovery (basically a function of the backwash interval) below which fouling rates were low and backwashing effectiveness was high. Changing the mode of operation from constant flux to constant pressure had no effect on filtrate water quality. The authors calculated that more membrane area (higher capital cost) is required, but less energy is required (lower operating cost) when constant pressure mode is used compared to constant flux.

There have been several fundamental studies done attempting to better understand membrane fouling. Taniguchi et al. (2003) investigated the modes of fouling of UF membranes by organic matter, particularly the role of aggregates. NOM size in the raw water source was characterized by parallel membrane fractionation and size exclusion chromatography.   Eight different membranes were tested, with MWCO's ranging from 10 to 1,000 kDa. They found that cake formation was the dominant mode of fouling in filtration of raw water containing aggregates. Pre-filtration to remove the aggregates and dilution to reduce their concentration reduced the fouling rate, but did not change the mode in the low MWCO membranes. In contrast, for the

high MWCO membranes pre-filtration prevented cake formation and shifted the mode to pore blockage. The authors state that an initial fouling layer of large aggregates can catalyze fouling by low molecular weight species. For the high MWCO membranes, the fouling layer could be removed by backwashing, but the lower MWCO ones exhibited some irreversible fouling, suggesting that the low molecular weight species were penetrating into the pore structure of the membranes. Shim et al. (2002) also conducted a study of membrane fouling mechanisms. These authors characterized the membranes and the NOM in terms of their charge properties, investigating the electrostatic interactions between the membrane surface and NOM. Chang and Benjamin (2003) developed three mathematical models to describe fouling of an ultrafiltration membrane by natural organic matter (NOM). These models attribute the fouling to: (1) an increase of the effective pore length; (2) formation of a uniform NOM gel layer on the membrane surface, or (3) narrowing of the membrane pores by sorption of a monolayer of NOM molecules. The parameters characterizing each model are identified and estimated based on data for flux and film growth gathered in the same system.

Carroll et al. (2000) also investigated membrane fouling. These researchers studied a particular water and compared pre-treatment (alum coagulation) with no pretreatment, and fractionated the organic material into four specific compounds based on hydrophobicity and charge. They found that when the raw water was filtered directly by the membrane with no pretreatment, fouling occurred quickly and colloids were mainly responsible for the fouling. When alum pretreatment was used, performance improved, although the rate of fouling was still appreciable. NOM was responsible for most of the fouling when no pretreatment was applied. They concluded that if the neutral hydrophilic substances could be targeted for removal upstream of this membrane, performance would be improved. Alternatively, improvements may result from selection of a membrane material specifically chosen for its resistance to fouling by these substances. Lin et al. (1999 and 2000) also fractionated humic material into hydrophobic and hydrophilic components and studied their relative impact on UF membrane fouling. The hydrophilic fraction was responsible for the worst fouling despite little rejection by the membrane. Those molecules with the largest molecular weights (6 – 23 kDa) exhibited the worst flux decline. These researchers found that adding PAC made fouling worse in this water. In another study that looked at fractionating the NOM and separating it based on other characteristics, Cho et al.

(1999) evaluated mechanisms of NOM rejection and fouling. When they separated the NOM into different molecular weight fractions, rejection of these fractions by particular membranes was not as expected, according to the manufacturer-provided MWCO of each unit. These authors provided a thorough summary of the very complex features of natural organic matter and state that "...when the degree of chemical complexity of NOM constituents is combined with the many physico-chemical aspects of the membranes and the filtration process, it will be very difficult to predict performance…In the future we are optimistic that correlations may be developed that will facilitate improved quantitative predictions for filtration of complex mixtures, such as NOM in drinking water."

Howe and Clark (2002) investigated UF membrane fouling mechanisms using five different source waters. These researchers sequentially filtered the waters, removing smaller particles with each step. In this way they could determine the fouling potential of the remaining components. Particulate matter (>0.45 μm) was relatively unimportant. Small colloids (3 – 20 nm) appeared to be very important foulants. When this colloidal material was removed, the remaining organic matter in the water caused very little fouling. The small colloids consisted of only about 10 – 15% of the total dissolved organic matter; the remainder of the DOM was smaller than 3 nm. The authors concluded that the majority of DOM, by itself, does not cause fouling; the actual foulant is a relatively small fraction of the bulk DOM. Yuan and Zydney (2000) conducted a similar study of the role of humic acid in membrane fouling. Two NOM isolates were tested on UF membranes ranging from 30 to 300 kDa. The purpose of the study was to better understand the relative contributions of humic acid adsorption and deposition onto the membranes (irreversible fouling), and concentration polarization (reversible fouling). They investigated the effects of MWCO and solution chemistry (pH, ionic strength, concentrations of various ions, etc.). This article also contains a good summary of work done to characterize fouling, pointing out apparently contrasting results among them.

Zhang and Song (2000a) investigated membrane fouling, looking at the mechanisms and various parameters that affect it. These authors state that the fouling rate is controlled by the dynamic process of cake formation and growth. Permeate flux declines as the cake layer grows, until it reaches a steady state. The time to reach this steady state (equilibrium thickness of the cake) in

cross-flow MF and UF membranes was modeled as a function of applied pressure, feed concentration, shear rate, and particle size. It was shown that the fouling due to large particles was more severe than that caused by smaller particles. These same authors (Zhang & Song, 2000b) developed a relationship between pressure and flux for UF and MF membranes, and found their model to adequately predict permeate flux. It should be noted that these experiments were carried out with a model water containing silica colloids and no organic matter.

Sethi et al. (2001) developed a numerical cost model for optimizing membrane systems. This model simulates a UF system operating under steady-state conditions, and optimizes it for parameters such as fiber radius, fiber length, crossflow velocity, transmembrane pressure, and system recovery. Cost minimization was accomplished using a sequential quadratic programming algorithm, with different raw water conditions and operating parameters. For a small to medium sized system with a water source that is inclined to fouling, the most cost-effective treatment was predicted to be narrow hollow fibers and high crossflow velocity. In contrast, water supplies characterized by low fouling potential were optimized using larger radii fibers and lower crossflow velocity.

The Los Angeles Department of Water and Power (Karimi et al. 1999 and 2002) investigated membranes for treating water in open finished water reservoirs. The 1999 paper details the results of testing of an MF membrane to determine design parameters such as the flux and chemical cleaning intervals. Water from the reservoir was dechlorinated with bisulfite ahead of the membrane unit. Various chemical dosing scenarios were investigated such as coagulant and PAC addition. Interestingly, the turbidity of the backwash water was low enough to allow direct discharge to the storm drain. The authors concluded that MF was a viable technology for this water, and that flux rates higher than those allowed at the time by the Department of Health Services for the membrane tested were justified. The 2002 paper describes the results of a study evaluating four membranes, two MF and two UF membranes. All four were shown to be capable of removing particulates from the water. Therefore, selection was based on flux rates, recoveries, and chemical cleaning requirements. One membrane had higher flux and recovery rates, and superior chlorine resistance. This membrane (an MF unit) was selected for full-scale design of LADWP's 6 to 10 mgd plant. The Pittsburgh Water and Sewer Authority also

investigated membranes to treat water from an uncovered finished water reservoir (States et al. 2000). Inactivated *Cryptosporidium* and *Giardia* were used to test the performance of five membrane systems. They also evaluated design parameters such as flux, backwash frequency, and chemical cleaning requirements. Based on their results, they selected membrane filtration as the treatment method most suitable for their 20 mgd plant.

Membrane filtration is used at a full-scale plant in the Netherlands (Willemse et al. 1999) for treating filter backwash water. An equalization/settling basin receives the spent filter backwash water, and the supernatant from this basin is treated with UF membranes. The permeate passes through a UV unit for disinfection, and then directly into the finished water clearwell. These authors report that the process works well, with very good product water quality and lower-than-expected power costs and high recovery (93%) in its first year of operation.

Kelly and Olson (1999) summarized the process the city of Marquette, Michigan went through to select the MF process, construct their 7 mgd plant, and operate it. The owners of this previously-unfiltered supply had to build a new treatment plant to meet the requirements of the surface water treatment rule. Originally, they considered direct filtration, but selected MF due to its ease of operation and small footprint. Another major consideration for the city was that they could get their entire 3-log *Giardia* reduction credit with MF, so their existing finished water reservoir would not have to be enlarged to meet CT requirements. The plant has been in operation since late 1997, and they report that it has been working well.

A.7   ACTIVATED CARBON (PAC AND GAC) ADSORPTION

Powdered Activated Carbon is often used seasonally to mitigate taste and odor causing compounds. Gillogly et al. (1998a) investigated ways to determine the necessary PAC dose to achieve MIB removal. Radio-labeled MIB was used in order to circumvent the analytical problems associated with MIB measurement at very low levels. These authors used the homogeneous surface diffusion model (HSDM) to analyze their data and develop predictions of percent removal of MIB at a particular carbon dose. They confirmed that the percent of MIB removed was independent of the initial MIB concentration in both laboratory and natural waters.

Gillogly et al. (1998b) also investigated the effect of chlorine on PAC's ability to remove MIB, again in both laboratory and natural waters. The presence of chlorine was found to significantly diminish the adsorption capacity, and the authors recommend minimizing or eliminating contact time between chlorine and PAC. Similar results were seen by Huang and Yeh (1999) in that chlorine adversely affected the performance of PAC. When water dosed with PAC was subsequently chlorinated, they found that organics, particularly DBP precursors, were desorbed and the resulting halogenated organic concentrations were sometimes higher than if the water had been simply chlorinated. Therefore, it is important that the organic-laden PAC be removed ahead of chlorination.

The necessary PAC dose for removing the taste and odor causing compounds MIB and geosmin from four raw waters was studied by Cook et al. (2001). PAC doses are difficult to predict since so many factors affect them (type of PAC, presence of competing natural organic matter, contact time, etc.). These authors used the homogeneous surface diffusion model (HSDM) to predict PAC dosages. In three of the four waters tested, predictions were very close to the experimental results. In the fourth water, the required dose was much higher than predicted by the model. The authors speculate that this lack of agreement was a result of the high turbidity of this particular source, which required a higher alum dose for optimum turbidity removal. The denser, heavier flocs in this water may have effectively reduced the contact between the water and the adsorbent.

Powdered activated carbon (PAC) can be used in an upflow floc blanket reactor configuration (also called a contact clarifier). In this type of reactor, the retention time of the carbon can be very long, which is expected to lead to high organic carbon removal rates, close to the maximum that would be predicted by isotherms. Campos et al. (2000d) explored various operating parameters that affect the performance of this type of system. Using atrazine as the model compound in the flow-through reactor, they found that the PAC was not used at its maximum capacity, as predicted by the isotherms. Carbon retention time decreased with increasing carbon dose and/or with increasing the hydraulic loading rate.

Najm et al. (1998) investigated the addition of PAC as part of an enhanced coagulation process for DBP precursor removal. In this study of Colorado River water, the authors compared the overall chemical costs (coagulants, pH adjustment chemicals, and PAC) and sludge production costs for the plant to meet the target DBP levels. They found that there is an optimum combination of ferric chloride and PAC doses which results in the lowest operating cost and least sludge produced. This combination is especially appropriate for low SUVA waters, which generally are not amenable to enhanced coagulation. They concluded that using PAC as part of enhanced coagulation is less expensive than using GAC alone for DBP removal. Enhanced coagulation combined with GAC was also studied by Nowack et al. (1999). In their study, pre-treatment with ferric chloride and settling ahead of GAC filtration removed a substantial fraction of the TOC, thus extending bed life. They also found that ferric increased the adsorbability of the remaining TOC, thereby improving performance.

Precipitation of calcium carbonate onto GAC is a problem that can have adverse effects during GAC regeneration and subsequent reuse. McCafferty et al. (2000) used a method involving acidifying a sample of GAC that has been in service for some time, causing CO_2 gas to evolve, which is then absorbed by sodium hydroxide and subsequently measured. They found this method to be reliable, and used it at a water treatment plant to quantify the amount of carbonate build up over time. They were not able to document an increase in the mass of carbonate on the GAC over the 15 month study period, but they did quantify carbonate removal by an acid washing of the GAC. They concluded that the method might be useful in avoiding over-regeneration of GAC (perhaps acid washing would be sufficient to recover capacity). Frederick and Cannon (2001) evaluated the loading of both calcium and organic matter onto GAC, and investigated methods for minimizing the calcium loading by adjusting the pH and adding coagulants. Generally, lower pH levels resulted in less calcium loading onto the GAC, and the addition of iron (added in the form of ferric chloride) also led to reduced calcium loading. The combination of pH adjustment and ferric addition was most effective, and they concluded that a utility operating in the enhanced coagulation mode should expect very little calcium loading onto their GAC.

When GAC is used seasonally to mitigate taste and odor problems, it is helpful to determine the remaining capacity of an existing GAC filter bed. Gillogly et al. (1999) conducted a study in which a small laboratory column was used to predict the performance of a pilot scale column. These laboratory columns were shown to simulate the pilot scale columns well. These authors also investigated the effects of influent MIB concentrations and chorine on the performance of the GAC. They conclude that GAC at moderate EBCT's (5-10 minutes) was not likely to be effective for controlling high levels of MIB, and that chlorine degrades performance.

In order to determine whether activated carbon can remove contaminants from water, isotherms of the contaminants in organic-free water are developed. These describe the relative strengths of adsorption for different compounds. Isotherms of regulatory interest (such as for disinfection by-products) have been developed and published (Speth and Miltner, 1998). These serve as a starting point for plant design, since many additional factors affect the adsorption capacity of the carbon in a real water system.

New sources of material for making GAC have been investigated. Baçaoui et al. (2002) investigated olive-waste cakes, a by-product of olive oil production, as a source for activated carbon. They found that the carbons created from this product had as good or better adsorption capacities as commercially-available carbons. They also investigated a one-step process, instead of the usual two-step process, for activating the carbon, which they found to be less expensive and produce good results. Grens and Werth (2001) evaluated the performance of wood-based GAC vs. coal-based GAC in a filter/adsorber application. Coal-based GACs generally exhibit higher abrasion resistance values than wood-based GACs, but the correlation of these values to long-term performance in filter-adsorbers is not known. Two pilot filters were operated simultaneously and subjected to 500 typical backwash-with-air-scour cycles. Measurements of bed height, pressure drop, filter effluent quality, and particle size characteristics were made. There were no significant differences in performance; the wood-based GAC appeared to be as durable as the coal-based GAC.

Sotelo et al. (2002) investigated the fundamentals of adsorption equilibrium and kinetics of organic compounds onto GAC. These authors assert that there is still much to learn about

adsorption mechanisms, as evidenced by the empirical nature of existing models and the prevalence of costly pilot plant studies. They studied and compared adsorption equilibrium and kinetics of various compounds onto GAC. A model assuming a bi-disperse structure (macro- and micropores) with each region having a different isotherm was used to describe the systems. The theoretical model used worked well for three of the four compounds evaluated, but not the fourth. They concluded that surface diffusion, which was neglected in the model, was not important for three of the compounds, but could not be neglected for the fourth. The role of pore size was investigated by Pelekani and Snoeyink (1999). This study focused on the impact of pore size on the competition mechanism between organic compounds. They used activated carbon fibers with narrow and broad pore size distributions and various types of organic matter. Their results support a direct size competition mechanism theory; larger molecules block access to the micropores, but do not penetrate into these micropores. Ebie et al. (2001) also evaluated the effects of pore size distribution on the competitive adsorption characteristics of NOM and micro-pollutants (agricultural chemicals). They performed single-solute adsorption experiments as well as simultaneous NOM and micro-pollutant adsorption tests. They found that the adsorption capacity of the chemicals was reduced significantly in the presence of NOM, and that this effect was more apparent in carbons with a larger portion of small pores. They tried modeling this behavior with ideal adsorbed solution theory (IAST) incorporated with the Freundlich isotherm expression, but were not successful until a pore-blockage effect was considered. By taking pore blockage into account, they were able to model the competitive adsorption behavior. They concluded that selecting an activated carbon with a broad pore size distribution, especially one with a significant fraction of its pores larger than 30 Å, could greatly alleviate the adverse affects of NOM on the carbon's ability to remove micro-pollutants.

GAC pore structure at a full-scale plant was studied by Moore et al. (2003) over several regeneration cycles. An on-site thermal regeneration facility processed this GAC through six cycles, roughly 3 – 9 months apart, and the pore structures were analyzed during each cycle. With each regeneration, total pore volume increased because of widening of the pores. By the sixth cycle, the GAC had become quite mesoporous (compared to the very microporous virgin GAC). However, this change did not affect the GAC's ability to remove TOC. Performance for all six cycles was very similar.

Practical GAC contactor design was studied by Dvorak and Maher (1999) with regard to the number of parallel contactors to install and the empty bed contact times (EBCTs) of each. Carbon usage rates can be minimized by blending the effluents of multiple parallel GAC contactors that have been placed in service at different times. In this way, the carbon in each column can be used for a longer period of time before it must be regenerated. These researchers compared GAC performance in two to ten parallel columns for removing TOC from Missouri River water at three different EBCTs. The most dramatic improvement in usage rates appeared between one and two parallel contactors. Significant improvement also occurred when going from two to three contactors. However, beyond three parallel contactors, the incremental improvement decreased significantly. The authors suggested that two to four parallel contactors, instead of eight or more, may be preferable in order to minimize the complexity of the system without compromising efficiency.

A.8   MAGNETIC ION EXCHANGE (MIEX®) PROCESS

In the peer-reviewed literature, only one article was found regarding the MIEX process. This study was done at the University of North Carolina with funding from USEPA, participating utilities, and the Orica corporation, the manufacturer of the MIEX resin. Singer et al. (2002) examined the process performance at bench scale. Nine waters were studied, one from each box in the 3 x 3 matrix of TOC and alkalinity ranges from the federal enhanced coagulation regulations. The researchers found that the MIEX resin reduced DBP precursors significantly. THM and HAA formation potential was reduced by at least 60% in all waters tested and nearly 90% in some of the waters. In the low alkalinity waters, they also observed bromide removal. In all cases, the MIEX resin was used as a pretreatment ahead of conventional coagulation with alum. Alum doses needed for subsequent turbidity removal after MIEX pre-treatment were dramatically lower in all cases. This report did not include any costs or ancillary requirements such as resin regeneration and brine disposal. It merely demonstrated that the MIEX process is promising for organic matter removal, particularly DBP precursors.

There have been papers published in non-peer reviewed conference proceedings that describe the process in more detail. There is also some technical information on Orica's web site, the manufacturer of the resin (www.miexresin.com). Lee et al. (2002) conducted a bench scale study similar to Singer's in which the 3 x 3 matrix of waters with varying TOC and alkalinity levels was used. The purpose of this study was to characterize the natural organic matter removed by the resin, employing an XAD-8/4 resin fractionation process and size exclusion chromatography to investigate which fractions are effectively or preferentially removed by MIEX. They found that the MIEX resin is effective in targeting polar NOM in the form of transphilic and hydrophilic (low molecular weight) acids. This targeting is significant since coagulation and PAC adsorption are not effective at removing these NOM fractions.

There is one full-scale application at a 59 mgd groundwater treatment plant in Wanneroo, Australia (about 1/3 of the total flow is treated with the MIEX process and then blended with the conventional treatment plant flow). A 0.7 mgd surface water treatment plant is under construction in Mt. Pleasant, Australia. Another full-scale plant is under construction in the US in Franklin County, Alabama, which will serve a population of about 3,000. Several pilot studies have been done in the US; the ones conducted in Minnesota (Semmens et al. 2000), Florida (Neumeister et al. 2001), Virginia (Delphos et al. 2001) and Kentucky (Hamm and Bourke, 2001) are described in the various conference proceedings. Generally, all of the pilot studies demonstrated very good DOC and DBP precursor removal, consistent with the laboratory studies. All reported significant reductions in downstream chemical usage as well (coagulants, lime or caustic, and in some cases chlorine).

According to these papers, the MIEX resin (small beads 150 to 180 microns in size) is fed as a slurry into the raw water. This is done in a unit much like a rapid mix basin. The feed rates are reported at around 6 to 20 mL/L, which is a process variable that must be optimized. Orica suggests starting with 6 mL/L. MIEX addition is followed by a contact basin that acts like a CSTR, keeping the resin in suspension for a pre-determined period of time. Contact times in this basin are reported between 10 and 30 minutes, also selected during pilot testing. The contact basin is followed by a resin settling basin, which looks much like a conventional clarifier, but smaller. Due to the magnetic properties of the resin, the particles attract each other during

settling, forming large agglomerates with high settling velocities. Surface loading rates in these separation basins are reported to be around 3 to 4 gpm/ft^2 (Orica's web site says up to 6 gpm/ft^2). Incoming turbidity purportedly does not affect the ability of the resin to adsorb organic material, nor its ability to be separated out and re-used. A stream of concentrated, partially-spent resin is withdrawn continuously from the bottom of the separator. This is reported to be 20 – 30 % solids (v/v). Most of it (90-95%) is recycled back into the influent. A portion of it (5 to 10%) is diverted to the regeneration system. After regeneration, the resin is added back into the "fresh" resin feed line, along with some new make-up resin to compensate for any attrition. Nearly all (99.9%) of it is captured in the settling basins. The supernatant from the separation basins flows into the conventional surface water treatment plant. Generally, turbidity is unchanged; the process mainly removes dissolved organic carbon ahead of particulate removal processes.

At the Wanneroo plant (Smith et al. 2002), for a water flow rate of 29 mgd, the reported volume of resin pumped through the process is 0.6 mgd. Assuming this resin is in a 25% v/v slurry, about 2.4 mgd is continuously cycled through the basins. This must be done with low-shear pumps and large-radius pipes to avoid damage to the resin.

Regeneration of the resin is typically done as a batch process. Sodium chloride is used to make a 10 to 12% solution of regenerant brine. Sodium hydroxide and an acid (generally reported as hydrochloric) are also used in this process to aid in preparation of the regenerant solution. This regenerant must be adjusted each time it is used, which is usually done by adding more NaCl solution and caustic. It is reported that 5 to 9 adjustments can be done before the regenerant must be discarded. The waste brine, which contains concentrated organic material, other anions, and a very high concentration of salt, must be disposed of. No examples were found of plants in the US that had received a new NPDES permit for this waste stream. The Wanneroo plant employs an ocean outfall, and the waste is diluted by up to 1,000 times before it meets the ocean. The plant in Virginia, if it is modified, plans to use an existing RO unit brine disposal system. Semmens et al. (2000) successfully did a mass balance on TOC; they were able to predict the concentration of TOC in the brine.

In a presentation about the Wanneroo plant (Cadee et al. 2000), the authors reported that both the capital and operating costs of the MIEX process are about 60% of the cost of ozone/BAC. It should be noted that they assumed a 2 month replacement frequency for the GAC filters and an ozone generation power use of approximately 15 kWh/lb. They reported a cost of $0.24 per 1,000 gallons for the MIEX process, including regeneration but not including brine disposal. This number also included the calculated savings in alum costs, since they expect their alum dose to decrease from 75 to 25 mg/L, but did not include expected savings in lime costs.

It is not known how many times the resin can be regenerated before it needs to be replaced. During the pilot study in Australia, over 550 regeneration cycles were conducted with no measurable loss in performance. Generally, the only "new resin" costs are associated with the 0.1% attrition rate. Hamm and Bourke (2001) conducted a kinetic study of new vs. regenerated resins, and found no appreciable difference in rates of removal of UV_{254} absorbance. These authors also found that brine regeneration at ambient pH (8.1) and elevated pH (10.2) were equivalent. In the full-scale plant at Wanneroo, the process was added as a retrofit, and intermediate pumping was required to get the water through the clarifiers. The cost for this was not included in the paper. The settling basins needed to have an internal angle of 70º to minimize resin holdup. This meant that they had to excavate to a below-grade depth of 16 feet, necessitating special construction techniques.

The pilot study report from Florida (Bourke 2002) cites a cost of $0.15 per 1,000 gallons for the MIEX process, but no cost was given for the brine disposal. For this water, the expected ferric sulfate dose to achieve the water quality objectives was lowered by 180 mg/L with MIEX pretreatment. The authors calculated that the savings in ferric, pH correction chemicals and sludge disposal is about the same as the cost of the MIEX. The Kentucky study (Hamm and Bourke, 2001) showed a MIEX cost of $0.14 per 1,000 gallons, also not including brine (or waste regenerant) disposal. Their cost included resin make-up (11.4 cents) regenerant (NaCl, 1.5 cents) and power (1.1 cents). Delphos et al. (2001) used the same figure of $0.14 per 1,000 gallons for the MIEX costs. In this case, more than half of the expected overall savings resulted from not operating their existing RO plant. This same paper reported an overall capital cost of $3 million to add MIEX to their existing 7 mgd plant. Hamm and Bourke (2001) reported an

overall capital cost of $4.6 million for a 10 mgd plant. Neumeister et al. (2001) stated that "…the MIEX resin is typically regenerated for a one-year period before it is considered spent. At this point, the spent resin must be removed from the treatment process and replaced with virgin resin". While this is not consistent with the other papers, these authors calculate that the MIEX process will cost approximately $3.5 million per year for a 10.5 mgd flow rate (not counting brine disposal costs). They concluded that it is not the most suitable process for the water tested. Interestingly, they assumed the spent regenerant could be disposed of via the municipal sewer system.

A.9   IMPROVEMENTS IN CONVENTIONAL TREATMENT PROCESSES

Bustamante et al. (2001) investigated the relationship between coagulants (alum and ferric chloride), coagulant aids (poly-DADMACs), and *Cryptosporidium* oocysts. This fundamental investigation (conducted in laboratory water with no NOM) evaluated the zeta potential of the oocysts under various dosing schemes using micro-electrophoresis. Their results indicate that sweep flocculation may be more important when ferric is used, while charge neutralization is the more important removal mechanism when alum is used.

Numerous attempts have been made to optimize coagulant dosing (other than by jar testing and operator judgment). Franceschi et al. (2002) set out to develop a model to find the minimum coagulant dose resulting in the minimum turbidity based on various water quality parameters such as calcium and sulfate concentrations, pH, UV_{254}, turbidity, temperature, and clay concentration. They found that there were antagonistic influences of different factors, and that the process was quite complex. Along the same lines, Joo et al. (2000) used an artificial neural network model to predict optimum coagulant dosing based on operating data. However they found that the way in which the data were processed greatly influenced the outcome. They developed a pre-processing method that removes outliers, and subsequent predictions by the model were better. Chakraborti et al. (2000) developed an on-line analyzer that uses a photographic technique coupled with digital image processing software to quantify difference in fractal dimension of floc particles. As seen in practice, they found that there were measurable differences between sweep-floc coagulation and charge-neutralization coagulation. Another

potential method for optimizing coagulant dosing may be to use two-stage addition instead of a single stage (Carlson and Gregory, 2000). Adding alum in two stages, separated by about 60 seconds, was found to improve settled water turbidity and filter performance in a high-TOC water. This may be due, in part, to the absence of ideal mixing in the coagulant addition process.

Pre-oxidation ahead of alum coagulation by potassium ferrate (K_2FeO_4) was studied by Ma and Liu (2002). This chemical was studied as a possible alternative to chlorine, ozone, or permanganate as a pre-oxidant. In the three waters tested, use of this pre-oxidant significantly improved performance of the coagulation process.

Removal of THM precursors by coagulation was investigated by Bolto et al. (2002). Laboratory tests of waters composed of various types of NOM (based on its molecular weight, charge density, and polarity) were conducted using different coagulants. The authors tested the THMFP of each water, determined which fraction of the NOM was most responsible for the THMs, and targeted a coagulation scheme to maximize removal of that fraction. They compared precipitation of NOM (by coagulation) with adsorption (by ion exchange). The best treatment for each water depended on the characteristics of the organic matter. They concluded that "..the NOM being so spatially and temporally variable, it is not possible to state that a certain NOM fraction is the main THM precursor for all waters." Bell-Ajy et al. (2000) also investigated coagulation optimization for DBP precursor removal. In this study 16 waters were tested, and the authors confirmed that lower pH of coagulation resulted in improved TOC removal rates.

Volk and LeChevallier (2002) evaluated the effect of various surface water treatment processes on AOC and BDOC levels in 64 different plants. The objective was to identify treatment processes and operational practices that might control AOC levels in treated waters and to assess the effect of these processes on the formation or removal of biodegradable organic matter. Plants using GAC filter/adsorbers had lower AOC levels than systems using sand or mixed media, even though these GAC filters were operated with pre-chlorination and backwashed with chlorinated water. Microbial activity occurs in these filters even in the presence of a disinfectant residual. Systems using iron-based coagulants had lower AOC levels than those using alum or polymers. In many plants, AOC levels after treatment were significantly higher than in the raw water. Volk

et al. (2000) also documented improved BDOC removal with enhanced coagulation when compared with standard coagulation.

The impact of enhanced coagulation on the removal of *Cryptosporidium* was evaluated by States et al. (2002). These authors conducted a series of pilot plant trials in which large numbers of *Cryptosporidium* oocysts were spiked into water coagulated with alum, ferric chloride, or polyaluminum chloride. They found that TOC removal was indeed enhanced by lower pH values, and that *Cryptosporidium* removals were not compromised. They noted that turbidity and particle counts were not necessarily good indicators of oocyst removal. Childress et al. (1999) evaluated enhanced coagulation of State Project Water and Colorado River Water at pilot scale. They used ferric chloride and a cationic polymer at various doses, and measured turbidity removal as well as organic carbon removal (measured as TOC, SUVA, & THMFP). Their results were consistent with others in that the removal of particles and DBP precursors was better at higher ferric doses and lower pH values, and enhanced coagulation was not as effective for lower SUVA waters. A similar study evaluating simultaneous removal of particles and THM precursors was done by Vrijenhoek et al. (1998) in Colorado River Water and State Project Water, also at pilot scale. Alum and a cationic polymer were used in this study, and results indicated that the higher alum doses did improve TOC and DBP precursor removal, and did not degrade particle removal.

The secondary effects of plant operation in the enhanced coagulation mode have been discussed (Carlson et al. 2000). An expert workshop was held to identify and characterize these effects in an effort to assist utilities anticipate problems and mitigating them. Four process areas that may be affected by the change to enhanced coagulation were described including 1) inorganic contaminant control (primarily manganese and corrosion control), 2) primary disinfection, 3) particle removal, and 4) residuals handling and disposal. Potential mitigation strategies in each area were discussed.

The effectiveness of conventional treatment in removing *Cryptosporidium* oocysts was evaluated by Dugan et al. (2001). Oocysts were spiked into the influent of a pilot plant at high concentrations, and the settled water and plant effluent were monitored for oocysts as well as for

other surrogate parameters such as turbidity, particle counts, and endospores. They evaluated the effects of coagulant dose and type, filter media, and loading rates. Not surprisingly, sub-optimal coagulant dosing resulted in poor performance. Optimal coagulation conditions resulted in an average removal of oocysts of more that 5-log across the plant, and the surrogates correlated well with the oocysts, although they were conservative indicators of oocyst removal.

Algae removal, particularly toxin-producing cyanobacteria, was investigated by Drikas et al. (2001) and Chow et al. (1999). Conventional treatment (coagulation with alum, flocculation, sedimentation, and filtration) was used at bench and pilot scale, and cultured *Microcystis aeruginosa* were added to simulate an algae bloom entering a treatment plant. It is important that, whenever possible, these cells be removed intact in order to prevent the release of toxins and taste-and-odor causing compounds from their interiors. The authors found that 99.9% of the cyanobacterial cells were removed under normal operating conditions. The increase in the hepatoxin microcystin-LR was negligible. It was notable that the *M. aeruginosa* collected in the sludge broke down and released toxin immediately. This may have important implications for plants practicing recycle of spent filter backwash water or sludge dewatering supernatant. Chow et al. (1998) performed a study of the effect of ferric chloride on cultured algae. These authors found that the ferric at typical treatment plant doses did not cause the algae cells to break apart.

Antibiotics removal by conventional treatment was studied by Adams et al. (2002). Coagulation-flocculation-sedimentation with alum and iron salts was found to be ineffective at removing the seven antibiotics evaluated.

Dissolved air flotation has been studied. Edzwald et al. (1999) conducted a pilot-scale evaluation of this high-rate process (flocculation time of five minutes, DAF hydraulic loading rates of 12 – 16 gpm/sq. ft.) on two different water sources under cold water conditions, focusing on whether this technology could be integrated into an existing surface water treatment plant. Their results indicated that performance was good, that these short times were feasible, and that the process worked well upstream of standard dual-media filters and could be integrated into the plant design. Edzwald et al. (2000) compared the performance of a DAF system to a conventional sedimentation process in terms of their abilities to remove *Giardia* cysts and

Cryptosporidium oocysts. They found the DAF performance to be better, especially under challenging conditions, and suggested that this process be given at least as much removal credit in the Surface Water Treatment Rule as is given to conventional treatment with sedimentation. French et al. (2000) evaluated numerous published reports of *Cryptosporidium* removal with DAF in order to develop a model to predict performance. This model includes process factors such as recycle rate, feed water turbidity, and flocculation time. Lundh et al. (2000) performed studies in which they measured fluid velocities in the separation zone of DAF systems with an acoustic Doppler Velocimeter, analyzing density gradients and flow patterns in an effort to optimize the process. Chung et al. (2000) described the design and operation of a demonstration scale (2.4 mgd) DAF plant with an emphasis on scale-up issues from the pilot-scale plant (20 gpm). These authors found that information from the pilot plant translated well to full-scale, particularly with regard to the hydraulic loading rates and air dissolving tube pressures, but that in the flotation zone, the full-scale plant exhibited slightly poorer performance. They attributed this behavior to long retention times and turbulent mixing conditions during backwash, which resulted in breakage of the fragile flocs.

Ballasted flocculation, trade named ACTIFLO®, has been investigated by several research teams. In this modification to conventional treatment, the weight of floc particles is increased by the addition of high-density micro-sand particles, allowing higher surface loading rates and thus a smaller footprint and potentially lower cost for the sedimentation process. The micro-sand is separated from the sludge in a "hydrocylcone" (centrifugal separator) and reused. Desjardins et al. (2002) conducted a laboratory scale study of this process, attempting to validate a lab-scale method to predict performance rather than a pilot scale process. Their modified jar test procedure was able to predict the performance of the full scale plants with good accuracy.

Appendix B

Technology and Cost Assessment

INFLUENT — **American River (58.4 mgd)**

Cl2 = 1.73 mg/L

Grit Basin

ALUM = 1.3 mg/L

Rapid Mix

Flocculation (1.5 MG)

Sedimentation (6.1 MG)

Filtration (330,000 gal)

CL2 = .15 mg/L
CaOH = 6.8 mg/L

Reservoir (20 MG)

Distribution System

279 City of Sacramento
PWSID No. CA3410020
Sacramento, CA
Plant Name: E.A. Fairbairn Water Treatment Plant
ICR Plant ID No. 205
Treatment Type: conv
Design Flow: 100 mgd

12 Month Average

Influent WQ:
pH = 7.0
Temp = 15.4
TOC = 1.3
UV Abs = 0.034
Br = .01-above delta
Alk = 19.8
Cal hard = 16
Tot hard = 18.9
NH3 = .1
Turb = 2.8

Finished WQ:
pH = 8.6
Temp = 15.6
TOC = 0.8
UV Abs = 0.011
Br = ??
Alk = 22.7
Cal hard = 23.1
Tot hard = 27.1
Turb = 0.1

City of Sacramento - Fairbairn WTP

- ### Process improvement priority:
    1. Install UV inactivation for cryptosporidium inactivation
    2. Add chlorine dioxide with chlorite control for cryptosporidium inactivation
    3. Install ozonation with bromate control technologies (i.e., pH reduction during ozonation)
    4. Add GAC for TOC removal to allow continued free chlorine use for residual disinfection
    5. Convert residual disinfection to chloramines
    6. Install low pressure membrane filtration for physical disinfection of cryptosporidium
    7. Install MIEX technology for TOC and bromide removal

B-3

Technology Assessment Results for City of Sacramento - Fairbairn WTP

Current Condition

Deficiencies w.r.t. ELPH:
 - crypto inactivation < 1.0 log

1.0 Install UV after filters

Process Changes:
 - install UV

2.0 Add ClO2 with chlorite control

Process Changes:
-add 1.6 mg/L of ClO2 before RM
-add 12.0 mg/L of ferrous sulfate at RM
- reduce lime to 5.2 mg/L

3.0 Add O3 with bromate control

Process Changes:
-add 1 mg/L ozone
-add 4- 20,000 gal ozone chambers
-reduce lime to 5.2 mg/L

4.0 Install MF/UF after filters

Process Changes:
 -install membrane post filters

B-4

City of Sacramento - Fairbairn WTP

- Limiting Source Water Quality for Existing/Immediate Future Facility
  - Can't reach 1-log crypto credit
- Limiting Source Water Quality for Applied Technology Facility (UV)
  - Br = 0.01 mg/L
  - TOC = 1.3 mg/L

Technology Costs for City of Sacramento - Fairbairn WTP

Current Condition

Deficiencies w.r.t. ELPH:
- crypto inactivation < 1.0 log

1.0 Install UV after filters

Capital Cost: $7.2 Mi
O&M Cost: $150,000/Yr

2.0 Add CIO2 with chlorite control

ClO_2
Capital Cost: $815,000
O&M Cost: $200,000/Yr

Ferrous sulfate
Capital Cost: $800,000
O&M Cost: $845,000/Yr

3.0 Add O3 with bromate control

Capital Cost: $17 Mi
O&M Cost: $1.7 Mi/Yr

4.0 Install MF/UF after filters

Capital Cost: $84 Mi
O&M Cost: $5.1 Mi

Slide 1

```
                                                          279  City of Sacramento
  ┌─────────────┐     Sacramento River                    PWSID No.  CA3410020
  │  INFLUENT   │ ◄── (50.5 mgd)                                 Sacramento, CA
  └─────────────┘                              Plant Name:  Sacramento River Water
                                                             Treatment Plant
                                                          ICR Plant ID No. 204
     CL2= 2.2mg/L    CL2= 2.6 mg/L             Treatment Type: conv
                                               Design Flow:  135 mgd
```

Process flow diagram:

INFLUENT ← Sacramento River (50.5 mgd)

CL2= 2.2mg/L → Grit/Presed Basin
CL2= 2.6 mg/L → Aeration

ALUM = 3.4 mg/L

Grit/Presed Basin → Rapid Mix → Flocculation (755,355 gal) → Sedimentation → Filtration (52,263 gal)

Aeration → Flocculation (755,355 gal) → Clarification (1.8 MG) → Sedimentation → Filtration (52,263 gal)

CL2 = .32 mg/L
CAOH = 11.3 mg/L

Filtration (52,263 gal) → Clearwell (5 & 9.5 MG) → Distribution System

12 Month Average

Influent WQ:	Finished WQ:
pH = 7.3	pH = 8.1
Temp = 18.3	Temp = 17.5
TOC = 1.5	TOC = .9
UV Abs = 0.045	UV Abs = 0.014
Br = .01-above delta	Br = ???
Alk = 41	Alk = 42
Cal hard = 28	Cal hard = 39
Tot hard = 38	Tot hard = 51
NH3 = 0.1	NH3 = ???
Turb = 15.4	Turb = 0.1

B-7

Slide 2

City of Sacramento, Sacramento River WTP

- **Process improvement priority:**
  1. Install UV inactivation for cryptosporidium inactivation
  2. Add chlorine dioxide with chlorite control for cryptosporidium inactivation
  3. Install ozonation with bromate control technologies (i.e., pH reduction during ozonation)
  4. Add GAC for TOC removal to allow continued free chlorine use for residual disinfection
  5. Convert residual disinfection to chloramines
  6. Install low pressure membrane filtration for physical disinfection of cryptosporidium
  7. Install MIEX technology for TOC and bromide removal

B-8

Technology Assessment Results for
City of Sacramento, Sacramento River WTP

Current Condition

Deficiencies w.r.t. ELPH:
  - crypto inactivation < 1.0 log

1.0 Install UV after filters

Process Changes:
  - Install UV

2.0 Add ClO2 with chlorite control

Process Changes:
-add 1.2 mg/L of ClO2 before RM
-add 9.0 mg/L of Ferrous sulfate at RM
-reduce lime to 8.5 mg/L

3.0 Install O3 w/ bromate control

Process Changes:
-add 0.8 mg/L ozone
-install 4-.02 MG contactors
-reduce lime to 8.5 mg/L

4.0 Install MF/UF after filters

Process Changes:
-install membrane post filters
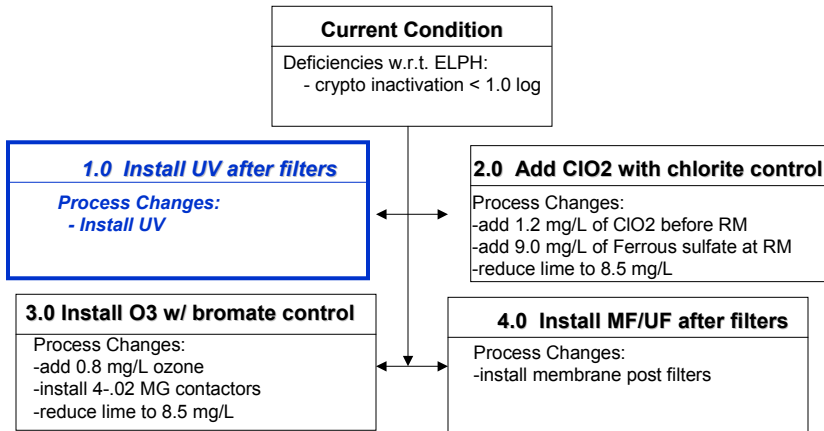
B-9

City of Sacramento, Sacramento River WTP

- Limiting Source Water Quality for Existing/Immediate Future Facility
  - Can't reach 1-log crypto credit
- Limiting Source Water Quality for Applied Technology Facility
  - Br = 0.01 mg/L
  - TOC = 1.8 mg/L

B-10

Technology Costs for
City of Sacramento, Sacramento River WTP

Current Condition

Deficiencies w.r.t. ELPH:
- crypto inactivation < 1.0 log

1.0 Install UV after filters

UV
Capital Cost: $ 10.5 Mi
O&M Cost: $ 125,000/yr

2.0 Add ClO_2 with chlorite control

ClO_2
Capital Cost: $ 580,000
O&M Cost: $ 115,000/yr
Ferrous sulfate
Capital Cost: $ 500,000
O&M Cost: $ 600,000/yr

3.0 Install O3 w/ bromate control

Ozone
Capital Cost: $ 16.4 Mi
O&M Cost: $ 120,000/yr

4.0 Install MF/UF after filters

MF/UF
Capital Cost: $ 104 Mi
O&M Cost: $ 4.5 Mi/yr

B-11

Hetch Hetchy Tunnel → **INFLUENT** ← San Antonio Reservoir
44% Nov 86% Dec (mgd)

Calaveras Reservoir
100% Jan-Oct 56%Nov
14% Dec

Alum = 32 mg/L
NaOCl = 3.1 mg/L
Cl2 = 1.0 mg/L

Rapid Mix

Flocculation (1.6 MG)

Sedimentation (6.7 MG)

Cl2 = .5 mg/L

Filtration (1.7 MG)

NAOH = 12.6 mg/L
Cl2 = 1.0 mg/L

Clearwell (324,000gal)

Reservoir (2.1 MG)

Distribution System

284 San Francisco Water Dept
PWSID No. CA3810001
Millbrae, CA
Plant Name: Sunol Valley WTP
ICR Plant ID No. 227
Treatment Type: conv
Design Flow: 160 mgd

12 Month Average

Influent WQ:	**Finished WQ:**
pH = 7.9	**pH = 8.6**
Temp = 15.9	**Temp = 15.4**
TOC = 3.3	**TOC = 2.4**
UV Abs = 0.090	**UV Abs = 0.040**
Br = .048 (from ICR data)	**Br = .02**
Alk = 119	**Alk = 118**
Cal hard = ???	**Cal hard = 98**
Tot hard = 124	**Tot hard = 124**
NH3 = ???	**Turb = 0.08**
Turb = 3.4	

B-12

SFWD - Sunol Valley WTP

- **<u>Process improvement priority:</u>**

  1. Install ozone with bromate control technologies (i.e., pH reduction during ozonation)

  2. Conversion to chloramines

  3. Install UV for cryptosporidium inactivation

  4. Add chlorine dioxide with chlorite control for cryptosporidium inactivation

  5. Install GAC for TOC control

  6. Install low pressure membrane filtration for physical disinfection of cryptosporidium

  7. Install MIEX technology for TOC and bromide removal

Technology Assessment Results for
SFWD - Sunol Valley WTP

Current Condition

Deficiencies w.r.t. ELPH:
- crypto inactivation < 1.0 log

1.0 Install O3 w/ bromate control

Process Changes:
-add 1.8 mg/L ozone
-add 4 – 20,000 gal ozone chambers

2.0 Convert to Chloramines

Process Changes:
-None-doesn't give crypto credits

3.0 Install UV after filters

Process Changes:
-Install UV

4.0 Add ClO2 with chlorite control

Process Changes:
-add 2.2 mg/L of ClO2 before RM
-add 16.5 mg/L of Ferrous sulfate at RM

5.0 Install GAC contactors

Process Changes:
-None-doesn't give crypto credit

6.0 Install MF/UF after filters

Process Changes:
-install membrane post filters

SFWD - Sunol Valley WTP

- Limiting Source Water Quality for Existing/Immediate Future Facility
  - Can't reach 1-log crypto credit
- Limiting Source Water Quality for Applied Technology Facility
  - $Br = 0.048$ mg/L
  - $TOC = 4.1$ mg/L

Technology Costs for
SFWD - Sunol Valley WTP

Current Condition

Deficiencies w.r.t. ELPH:
 - crypto inactivation < 1.0 log

1.0 Install O3 w/ bromate control

Ozone
Capital Cost: $ 22 Mi
O&M Cost: $ 0.5 mi/yr

3.0 Install UV after filters

UV
Capital Cost: $ 14 Mi
O&M Cost: $ 160,000/yr

4.0 Add ClO_2 with chlorite control

ClO_2
Capital Cost: $ 770,000
O&M Cost: $ 140,000/yr
Ferrous sulfate
Capital Cost: $ 550,000
O&M Cost: $ 1.9 Mi/yr

6.0 Install MF/UF after filters

MF/UF
Capital Cost: $ 123 Mi
O&M Cost: $ 6.5 Mi/yr

5. Did not cost Chloramines or GAC

INFLUENT ← San Andreas Reservoir (54.9 MGD)

Ozonation ← O3 dose = 4.9 mg/L

Rapid Mix ← Ferric = .47 mg/L as FeCl3*14H2O

Flocculation (716,000 gal)

Sedimentation (by-pass)

Filtration (1.4 MG)

Clearwell (81,000 gal)
← HFS = .98 mg/L as F
← Cl2 = 1.6 mg/L
← NAOH = 6.3 mg/L

8.0 MG   6.5MG

Distribution System

284 San Francisco Water Dept
PWSID No. CA3810001
Millbrae, CA
Plant Name: Harry W. Tracy WTP
ICR Plant ID No. 226
Treatment Type: df
Design Flow: 180 mgd

12 Month Average

Influent WQ:	Finished WQ:
pH = 7.9	pH = 8.9
Temp = 17.5	Temp = 61
TOC = 1.9	TOC = 1.7
UV Abs = 0.048	UV Abs = 0.018
Br = 0.037	Br = N/A
Alk = 50	Alk = 54
Tot hard = 55	Cal hard = 40
Turb = 0.63	Tot hard = 54
	Turb = 0.08
	Free Cl2 resid = 1.0

B-17

SFWD - Harry Tracy WTP

- **Process improvement priority:**

1. Install bromate control technologies (i.e., pH reduction during ozonation)

2. Conversion to chloramines

3. Install UV inactivation to reduce demand on ozonation for cryptosporidium inactivation

4. Add chlorine dioxide with chlorite control for augmenting cryptosporidium inactivation with ozone

5. Install GAC for TOC control

6. Install low pressure membrane filtration for physical disinfection of cryptosporidium

7. Install MIEX technology for TOC and bromide removal

B-18

Technology Assessment Results for
SFWD - Harry Tracy WTP

Current Condition*

Deficiencies w.r.t. ELPH:
- Bromate > 5 ug/L

*pH at ozonation – 7.9

** pH at ozonation – 7.5

1.0 Install bromate control**

Process Changes:
-lower ozone dose to 3.3 mg/L
-Or Add 2.3 mg/L H2SO4
-Increase NaOH to 7.9 mg/L

2.0 Convert to Chloramines

Process Changes:
-None-no effect on bromate formation

3.0 Install UV after filters

Process Changes:
-install UV
-lower ozone to <3.9 mg/L

4.0 Add ClO2 with chlorite control**

Process Changes:
-add .1 mg/L of ClO2 before RM
-add .8 mg/L of Ferrous sulfate at RM
-add 2.3 mg/L of H2SO4,7.9 mg/L of NaOH

5.0 Install GAC Contactors

Process Changes:
-None-no effect on bromate formation

6.0 Install MF/UF after filters

Process Changes:
- install membrane post filters
- lower ozone to <3.9 mg/L

B-19

SFWD - Harry Tracy WTP

- Limiting Source Water Quality for Existing/Immediate Future Facility
  - Br = 0.05 mg/L
  - TOC = 1.9 mg/L
- Limiting Source Water Quality for Applied Technology Facility
  - Br = ?? mg/L
  - TOC = ?? mg/L

B-20

Technology Costs for
SFWD - Harry Tracy WTP

Current Condition

Deficiencies w.r.t. ELPH:
- Bromate > 5 ug/L

1.0 Install bromate control

pH Adjustment
Capital Cost: $ 3.2 Mi (acid)
O&M Cost: $ 700,000 /yr

3.0 Install UV after filters

UV
Capital Cost: $ 14 Mi
O&M Cost: $ 140,000/yr

4.0 Add ClO$_2$ with chlorite control

ClO$_2$
Capital Cost: $ 870,000
O&M Cost: $ 125,000/yr
Ferrous sulfate
Capital Cost: $ 620,000
O&M Cost: $ 85,000/yr
pH Adjustment
Capital Cost: $ 1.6 Mi (acid)
O&M Cost: $ 0.2 Mi/yr

6.0 Install MF/UF after filters

MF/UF
Capital Cost: $ 138 Mi
O&M Cost: $ 4.8 Mi/yr

5. Did not cost Chloramines or GAC

B-21

INFLUENT ← L.A. Aqueduct (446 MG)

Ozonation ← O3 dose = 1.1 mg/L

Rapid Mix ← Ferric = 1.42 mg/L

Flocculation (3.4 MG)

Filtration (5.18 MG)

← CL2 = 2.6 mg/L
← HFS= .4 mg/L

Distribution System

262 City of Los Angeles
PWSID No. CA1910067
Los Angeles, CA
Plant Name: LA Aqueduct Filtration Plant
ICR Plant ID No. 163
Treatment Type: df
Design Flow: 601 mgd

12 Month Average

Influent WQ:
pH = 8.0
Temp = 18.9
TOC = 3.0
UV Abs = .081(assumed
based on similar upper delta
plants)
Br = 0.122
Alk = 102
Cal hard = 59
Tot hard = 89
NH3 = .027
Turb = 1.5

Finished WQ:
pH = 7.5
Temp = 17.0
TOC = 2.1
UV Abs = ???
Br = ND
Alk = 97
Cal hard = 62
Tot hard = 99
NH3 = .023
Turb = 0.06

B-22

City of Los Angeles – L.A. Aqueduct FP

- **<u>Process improvement priority:</u>**

  1. Increase ozone and use bromate control technologies (i.e., pH reduction during ozonation)
  2. Add chlorine dioxide with chlorite control for augmenting cryptosporidium inactivation with ozone
  3. Install UV inactivation to reduce demand on ozonation for cryptosporidium inactivation
  4. Install low pressure membrane filtration for physical disinfection of cryptosporidium
  5. Chloramines

B-23

Technology Assessment Results for
City of Los Angeles, L.A.Aqueduct FP

Current Condition

Deficiencies w.r.t. ELPH:
- Crypto inactivation < 1.0 log

1.0 Install bromate control

Process Changes:
-increase ozone to 1.5 mg/L

2.0 Add ClO2 with chlorite control

Process Changes:
-add 1.5 mg/L of ClO2 before RM
-add 11.3 mg/L of Ferrous sulfate at RM

3.0 Install UV after filters

Process Changes:
-install UV

4.0 Install MF/UF after filters

Process Changes:
-install membrane post filters

B-24

City of Los Angeles – L.A. Aqueduct FP

- Limiting Source Water Quality for Existing/Immediate Future Facility
  - Bromide = 0.04 mg/L
  - TOC = 3.5 mg/L
- Limiting Source Water Quality for Applied Technology Facility
  - Br = 0.12 mg/L
  - TOC = ?? mg/L

Technology Costs for
City of Los Angeles, L.A.Aqueduct FP

Current Condition

Deficiencies w.r.t. ELPH:
- Crypto inactivation < 1.0 log

1.0 Install bromate control

Increased Ozone Dosage
O&M Cost: $ 1.0 Mi/yr

2.0 Add ClO_2 with chlorite control

ClO_2
Capital Cost: $ 1.5 Mi
O&M Cost: $ 0.5 Mi/yr

Ferrous sulfate
Capital Cost: $ 1.1 Mi
O&M Cost: $ 3.0 Mi/yr

3.0 Install UV after filters

UV
Capital Cost: $ 50 Mi
O&M Cost: $ 0.6 mi/yr

4.0 Install MF/UF after filters

MF/UF
Capital Cost: $ 430 Mi
O&M Cost: $ 35 Mi/yr

INFLUENT ← **Mallard Reservoir (37.3 mgd)**

Flash Mix ← **NaOCl= 1.0 mg/L / Alum = 39.9 mg/L / H2SO4= 7.2 mg/L**

Aeration

Flocculation (1.6 MG)

Sedimentation (5.0 MG)

Ozonation ← **O3 = 1.2 mg/L**

Filtration GAC/Sand (330,000 gal)

← **CL2 = 3.6 mg/L / NaOH= 11.0mg/L / NH3= .5 mg/L**

Clearwell (10 MG)

Distribution System

12 Month Average

Influent WQ:	Finished WQ:
pH = 8.3	pH = 8.8
Temp = 17.2	Temp = 17.2
TOC = 3.3	TOC = 1.8
UV Abs = 0.090	UV Abs = 0.020
Br = 0.16	Br = .10
Alk = 63	Alk = 84
Cal hard = ??	Cal hard = ??
Tot hard = 99	Tot hard = 101
NH3 = .01	NH3=??
Turb = 2.6	Turb = 0.05

Contra Costa - Bollman WTP

- ### **Process improvement priority:**

  1. Add chlorine dioxide with chlorite control for augmenting cryptosporidium inactivation with ozone

  2. Install bromate control technologies (i.e., pH reduction during ozonation)

  3. Install UV inactivation to reduce demand on ozonation for cryptosporidium inactivation

  4. Install low pressure membrane filtration for physical disinfection of cryptosporidium

Contra Costa - Bollman WTP

- Limiting Source Water Quality for Existing/Immediate Future Facility
    - Br = 0.35 mg/L
    - TOC = 5.0 mg/L
- Limiting Source Water Quality for Applied Technology Facility
    - Immediate future plant meets ELPH goals, did not evaluate.

B-29

INFLUENT ← South Bay Aqueduct (17.7 mgd)

CO2 = 20 mg/L

254 Alameda County Water Dist
PWSID No. CA0110001
Fremont, CA
Plant Name: WTP 2
ICR Plant ID No. 136
Treatment Type: conv
Design Flow: 20 mgd

O3 dose = 2.3 mg/L

Ozonation

Flash Mix ← ALUM = 17.3 mg/L

Flocculation (513,900 gal)

12 Month Average

Sedimentation (1.1 MG)

Filtration (543,600 gal)

CL2 = 3.7 mg/L

Clearwell (73,000 gal) ← NAOH = 40 mg/L

NH3 = .6 mg/L

Distribution System

Influent WQ:	Finished WQ:
pH = 7.9	pH = 8.5
Temp = 17.6	Temp = ??
TOC = 3.5	TOC = 1.9
UV Abs = 0.129	UV Abs = 0.033
Br = 0.219	Br = .2
Alk = 95	Alk = 101
Cal hard = ??	Cal hard = 24
Tot hard = 125	Tot hard = ??
NH3 = <.1	Turb = 0.06
Turb = 9.3	Total Cl2 resid = 2.6

B-30

Alameda County Water District - TP2

- **<u>Process improvement priority:</u>**
    1. Install UV inactivation to reduce demand on ozonation for cryptosporidium inactivation
    2. Add chlorine dioxide with chlorite control for augmenting cryptosporidium inactivation with ozone
    3. Install low pressure membrane filtration for physical disinfection of cryptosporidium
    4. Install MIEX technology for TOC and bromide removal

Technology Assessment Results for ACWD, TP2

Current Condition

Deficiencies w.r.t. ELPH:
- Crypto inactivation < 1-log

1.0 Install UV after filters

Process Changes:
- install UV

2.0 Add ClO2 with chlorite control

Process Changes:
-add 1.9 mg/L of ClO2 before RM
-add 14.3 mg/L of Ferrous sulfate at RM

3.0 Install MF/UF after filters

Process Changes:
- install membranes after filters
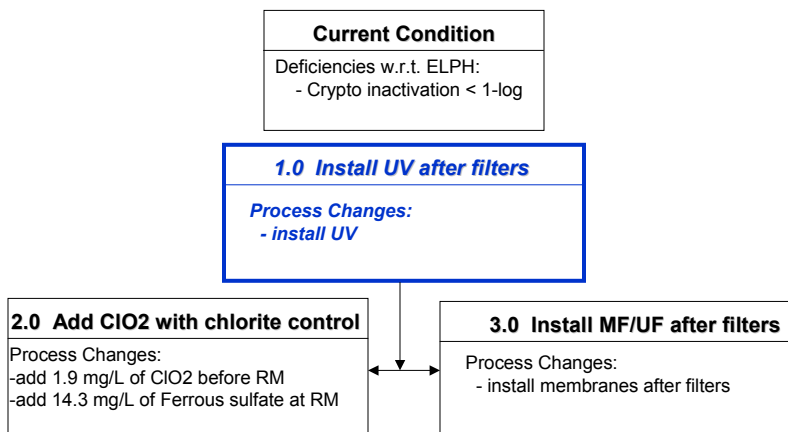
Alameda County Water District - TP2

- Limiting Source Water Quality for Existing/Immediate Future Facility
  - Br = 0.19 mg/L
  - TOC = 1.7 mg/L
- Limiting Source Water Quality for Applied Technology Facility (UV)
  - Br = 0.3 mg/L
  -TOC = 5.0 mg/L

Technology Costs for ACWD, TP2

Current Condition

Deficiencies w.r.t. ELPH:
- Crypto inactivation < 1-log

1.0 Install UV after filters

UV
Capital Cost: $2.0 Mi
O&M Cost: $65,000/Yr

2.0 Add ClO$_2$ with chlorine control

ClO$_2$
Capital Cost: $300,000
O&M Cost: $70,000/Yr
Ferrous sulfate
Capital Cost: $250,000
O&M Cost: $225,000/Yr

3.0 Install MF/UF after filters

MF/UF
Capital Cost: $19 Mi
O&M Cost: $1.7Mi/Yr

Slide B-35

```
INFLUENT  ←  Source Water
              (59.8 MGD)
    ↓
Rapid Mix  ←  Cl2 = 2.1 mg/L
           ←  ALUM = 24.1 mg/L
    ↓
Flocculation
(1.6 MG)
    ↓
Sedimentation
(5.3 MG)
    ↓
Ozone (future)  ←  O3 = 1.0 mg/L
    ↓  ←  Cl2 = 1.0 mg/L
Filtration
(987,000 gal)
    ↓  ←  NH3 = .43 mg/L
       ←  NAOH = 6.8 mg/L
Clearwell
(10 MG)
    ↓
Wholesale
Customer
```

286 Santa Clara Valley Water District
PWSID No. CA4310027
San Jose, CA
Plant Name: Santa Teresa WTP
ICR Plant ID No. 236
Treatment Type: conv
Design Flow: 100 mgd

12 Month Average

Influent WQ:	Finished WQ:
pH = 7.8	pH = 7.5
Temp = 17.0	Temp = 17.8
TOC = 3.3	TOC = 2.3
UV Abs = 0.1	UV Abs = 0.050
Br = 0.2	Br = .1
Alk = 84	Alk = 80
Cal hard = 56	Cal hard = 56
Tot hard = 185	Tot hard = 110
Turb = 1.8	Turb = .1

B-35

Santa Clara Valley - Santa Teresa WTP

- **Process improvement priority:**
  1. Install bromate control technologies (i.e., pH reduction during ozonation)
  2. Install UV inactivation to reduce demand on ozonation for cryptosporidium inactivation
  3. Add chlorine dioxide with chlorite control for augmenting cryptosporidium inactivation with ozone
  4. Install low pressure membrane filtration for physical disinfection of cryptosporidium
  5. Install MIEX technology for TOC and bromide removal

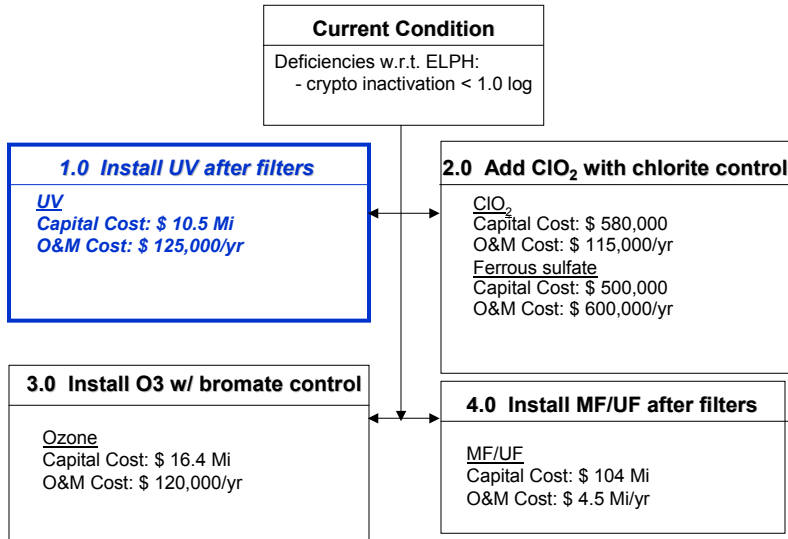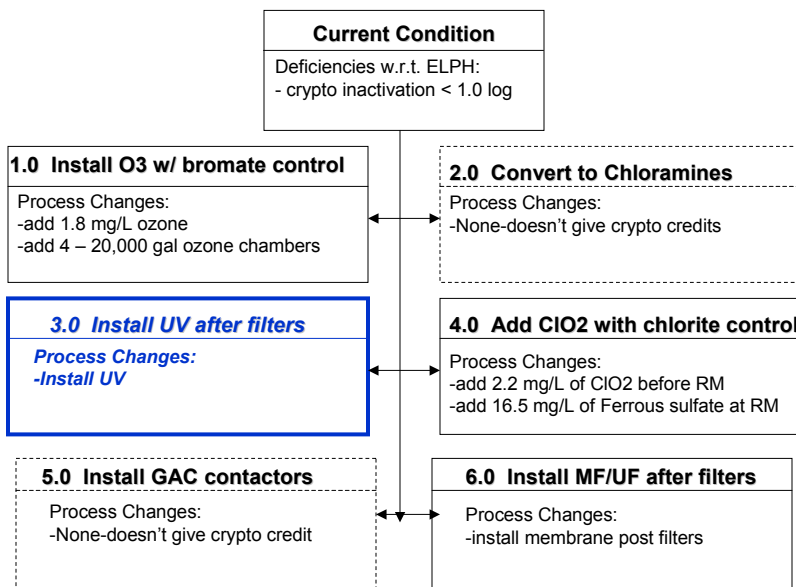B-36

Technology Assessment Results for
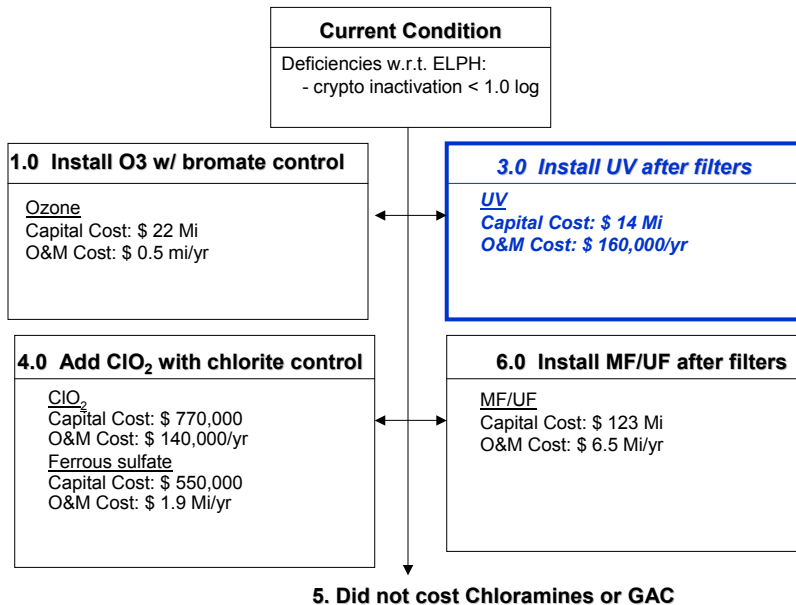## Santa Clara Valley - Santa Teresa WTP

Current Condition*

Deficiencies w.r.t. ELPH:
- crypto inactivation < 1.0 log
- TTHMs > 40 ug/L

*pH at ozonation – 7.0

** pH at ozonation – 5.4

***pH at ozonation – 7.1

****pH at ozonation –5.6

1.0 Install bromate control**

Process Changes:
- add 61 mg/L of H2SO4
- add 56 mg/L of NaOH

2.0 Install UV after filters**

Process Changes:
- install UV
- add 61 mg/L of H2SO4
- increase NaOH to 56 mg/L

3.0 Add ClO2 with chlorite control***

Process Changes:
-add 2.1 mg/L of ClO2 before RM
-add 15.8 mg/L of Ferrous sulfate at RM
-lower NaOH to 5.4 mg/L

4.0 Install MF/UF after filters****

Process Changes:
- install membrane post filters
- add 58 mg/L of H2SO4
- add 54 mg/L of NaOH

Note: H2SO4 doses >50 mg/L are not NSF
approved, therefore alternative pH reduction
methods may be combined with the H2SO4
addition to achieve the desired results

Santa Clara Valley - Santa Teresa WTP

- Limiting Source Water Quality for Existing/Immediate Future Facility
  - Bromide = 0.25 mg/L
  - TOC = 1.8 mg/L
- Limiting Source Water Quality for Applied Technology Facility
  - Br = 0.2 mg/L
  - TOC = 3.3 mg/L

Technology Costs for
Santa Clara Valley - Santa Teresa WTP
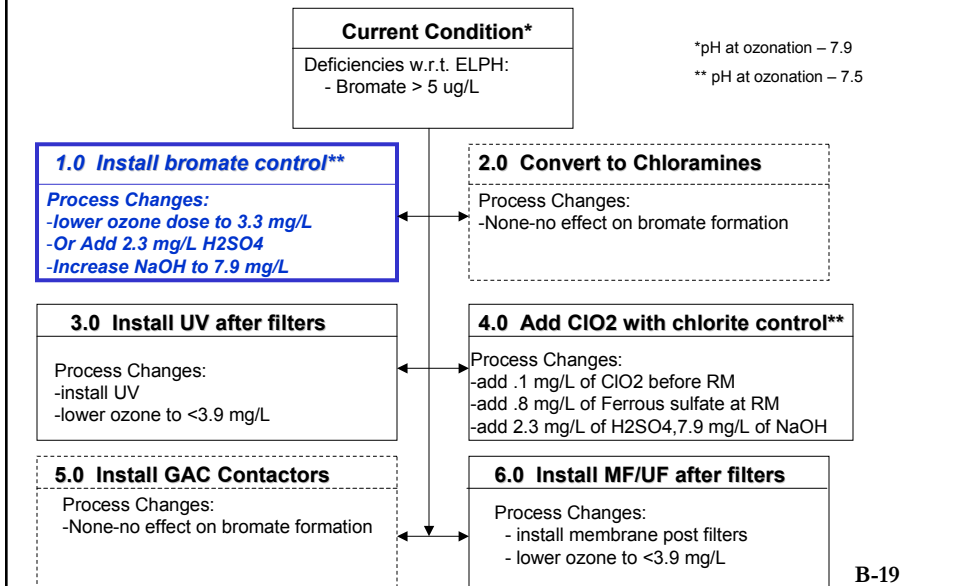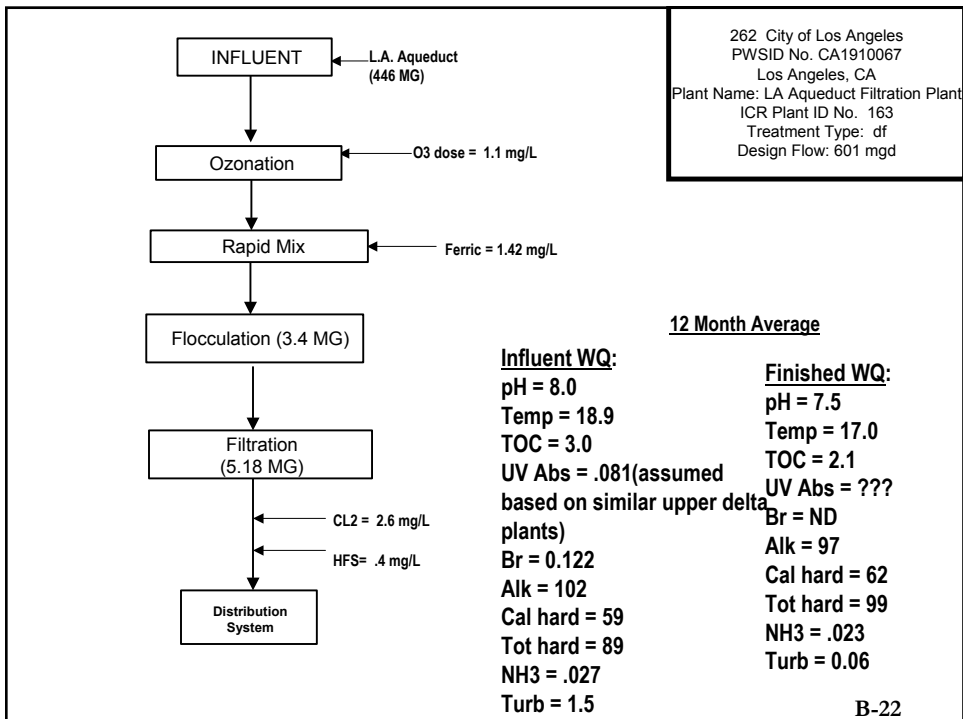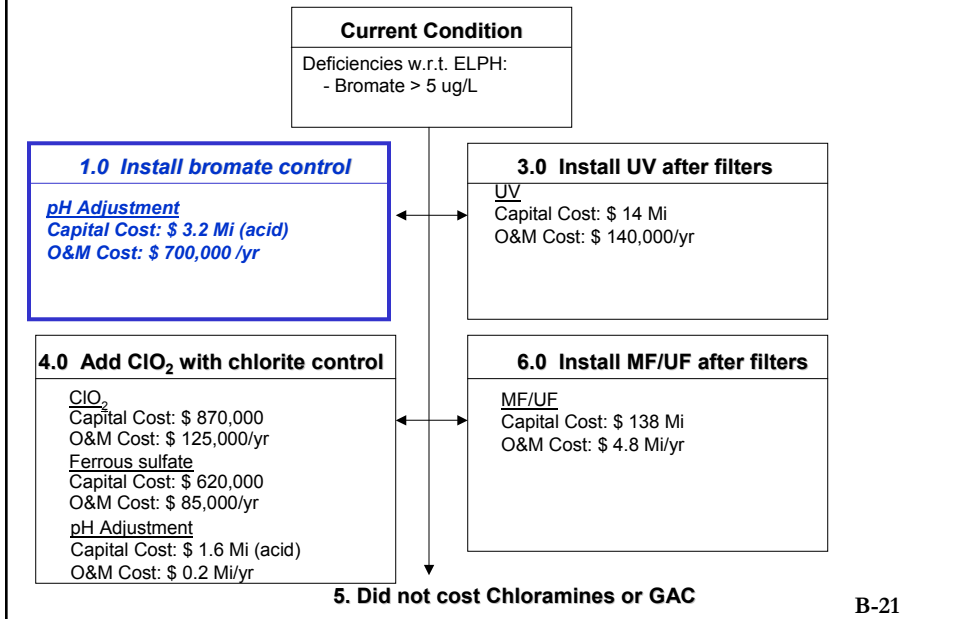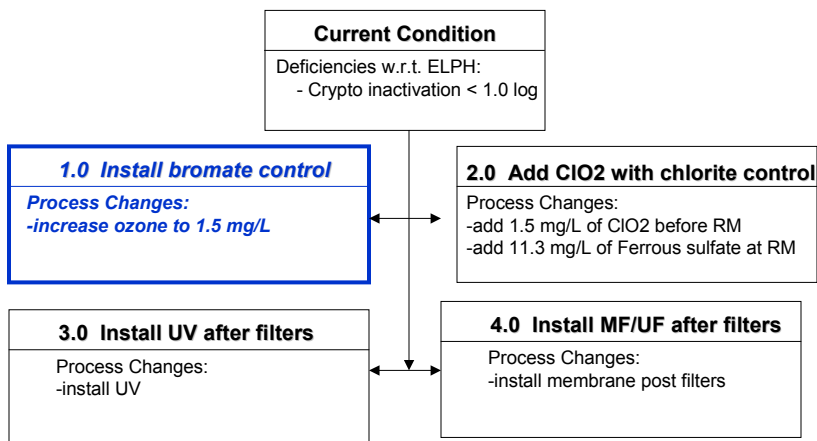
Current Condition

Deficiencies w.r.t. ELPH:
- crypto inactivation < 1.0 log
- TTHMs > 40 ug/L

1.0 Install bromate control

pH Adjustment
Capital Cost: $ 1.8 Mi (acid)
O&M Cost: $ 5.2 Mi/yr

2.0 Install UV after filters

UV
Capital Cost: $ 7.2 Mi
O&M Cost: $ 155,000/yr

pH Adjustment
Capital Cost: $ 1.8 Mi (acid)
O&M Cost: $ 4.7 Mi/yr

3.0 Add ClO_2 with chlorite control

ClO_2
Capital Cost: $ 815,000
O&M Cost: $ 360,000/yr
Ferrous sulfate
Capital Cost: $ 800,000
O&M Cost: $ 1.6 Mi/yr
pH Adjustment
O&M Cost: $ minimal

4.0 Install MF/UF after filters

MF/UF
Capital Cost: $ 84 Mi
O&M Cost: $ 5.2 Mi/yr

pH Adjustment
Capital Cost: $ 1.8 Mi (acid)
O&M Cost: $ 4.4 Mi/yr

B-39

INFLUENT ← **South Bay Aqueduct (25.4 MGD)**

NaOCl = 2.0 mg/L
Alum = 29 mg/L
→ Rapid Mix

Flocculation
(278,000 gal)

Sedimentation
(2.0 MG)

Ozone
(future) ← **O3 = 1.0 mg/L**

NaOCl = 1.5 mg/L
→ Filtration
(400,860 gal)

NH3 = .3 mg/L
NAOH = 3.8 mg/L
→ Clearwell
(3.2 MG)

Wholesale Customer

286 Santa Clara Valley Water District
PWSID No. CA4310027
San Jose, CA
Plant Name: Penitencia WTP
ICR Plant ID No. 238
Treatment Type: conv
Design Flow: 40 mgd

12 Month Average

Influent WQ:
pH = 7.9
Temp = 18.9
TOC = 3.4
UV Abs = 0.154
Br = 0.18
Alk = 67
Cal hard = 58
Tot hard = 85
Turb = 9.0

Finished WQ:
pH = 7.6
Temp = 20.5
TOC = 2.1
UV Abs = 0.047
Br = .13
Alk = 61
Cal hard = 59
Tot hard = 86
Turb = 0.6

B-40

Santa Clara Valley - Penitencia WTP

- **<u>Process improvement priority:</u>**

    1. Install bromate control technologies (i.e., pH reduction during ozonation)

    2. Install UV inactivation to reduce demand on ozonation for cryptosporidium inactivation

    3. Add chlorine dioxide with chlorite control for augmenting cryptosporidium inactivation with ozone

    4. Install low pressure membrane filtration for physical disinfection of cryptosporidium

    5. Install MIEX technology for TOC and bromide removal

Technology Assessment Results for
Santa Clara Valley - Penitencia WTP

Current Condition*

Deficiencies w.r.t. ELPH:
 - crypto inactivation < 1.0 log
 - TTHMs > 40 ug/L

*pH at ozonation – 7.0

** pH at ozonation – 5.6

***pH at ozonation – 5.5

****pH at ozonation –6.9

1.0 Install bromate control**

Process Changes:
-add 45 mg/L of H2SO4
-add 41 mg/L of NaOH
-Increase ozone to 2.6 mg/l

2.0 Install UV after filters**

Process Changes:
 - install UV
 - add 47 mg/L of H2SO4
 - add 43 mg/L of NaOH

3.0 Add ClO2 with chlorite control***

Process Changes:
-add 2.1 mg/L of ClO2 before RM
-add 15.8 mg/L of Ferrous sulfate at RM
-increase NaOH to 4.7 mg/L

4.0 Install MF/UF after filters**

Process Changes:
 - install membrane post filters
 - add 46 mg/L of H2SO4
 - add 42 mg/L of NaOH
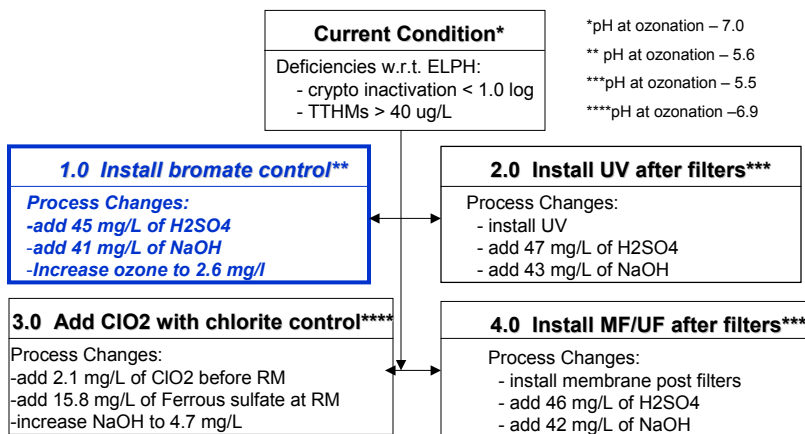
Santa Clara Valley - Penitencia WTP
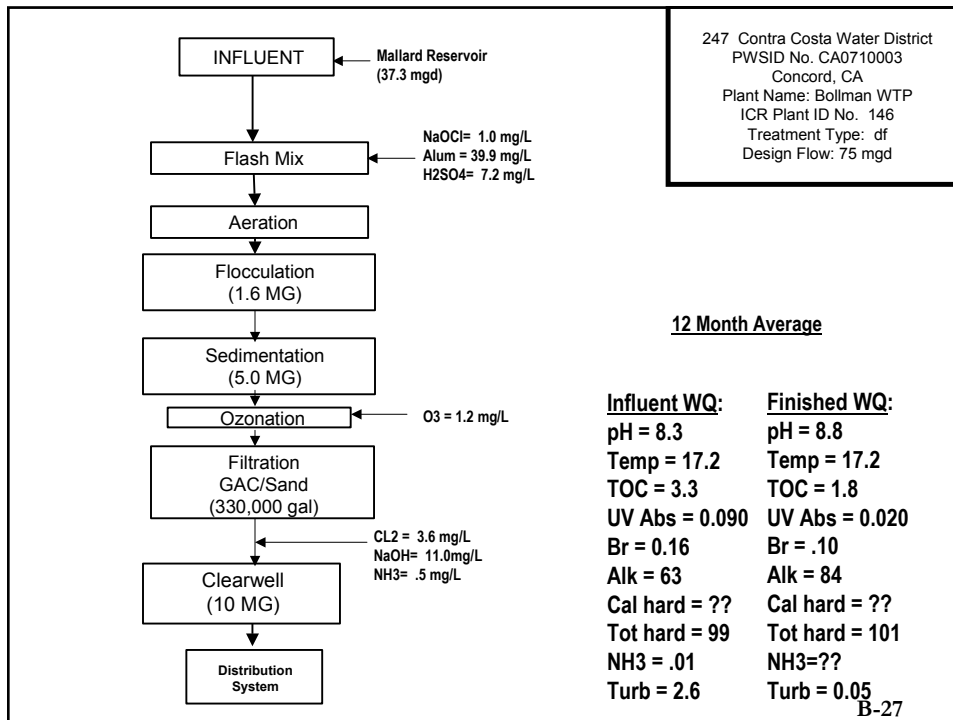
- Limiting Source Water Quality for Existing/Immediate Future Facility
  - Bromide = 0.32 mg/L
  - TOC = 1.8 mg/L
- Limiting Source Water Quality for Applied Technology Facility
  - Br = 0.22 mg/L
  - TOC = 3.4 mg/L

Technology Costs for
Santa Clara Valley - Penitencia WTP

Current Condition

Deficiencies w.r.t. ELPH:
  - crypto inactivation < 1.0 log
  - TTHMs > 40 ug/L

1.0 Install bromate control

pH Adjustment
Capital Cost: $ 770,000 (acid)
O&M Cost: $ 2.8 Mi/yr

2.0 Install UV after filters

UV
Capital Cost: $ 4.0 Mi
O&M Cost: $ 95,000/yr

pH Adjustment
Capital Cost: $ 770,000 (acid)
O&M Cost: $ 2.8 Mi/yr

3.0 Add ClO_2 with chlorite control

ClO_2
Capital Cost: $ 600,000
O&M Cost: $175,000/Yr
Ferrous sulfate
Capital Cost: $ 400,000
O&M Cost: $1.2 Mi/Yr
pH Adjustment
O&M Cost: $ minimal

4.0 Install MF/UF after filters

MF/UF
Capital Cost: $ 38 Mi
O&M Cost: $ 2.4 Mi/yr

pH Adjustment
Capital Cost: $ 770,000 (acid)
O&M Cost: $ 2.4 Mi/yr

261 Metro Water Dist of So Calif
PWSID No. CA1910087
Los Angeles, CA
Plant Name: R. B. Diemer FP
ICR Plant ID No. 169
Treatment Type: conv
Design Flow: 520 mgd

INFLUENT ← Colorado River State Project Water (356 MGD)

Rapid Mix ← Cl2 = 1.6 mg/L ← Ferric = 8.0 mg/L

Flocculation (7MG)

Sedimentation (33.6 MG)

Filtration (4.0 MG) ← Cl2 = 2..5 mg/L

Clearwell (24.6 MG) ← NH3 = .56 mg/L ← NAOH = 5.3 mg/L

Distribution System

12 Month Average

Influent WQ:	Finished WQ:
pH = 8.3	pH = 8.0
Temp = 18.6	Temp = 18.6
TOC = 3.0	TOC = 2.5
UV Abs = 0.055	Alk = 112
Br = 0.131	Cal hard = 136
Alk = 115	Tot hard = 233
Tot hard = 232	Turb = 0.05
Turb = 1.5	

B-45

MWDSC - Diemer FP

- **Process improvement priority**

1. Install O3 with bromate control technologies (i.e., pH reduction during ozonation)

2. Install UV for cryptosporidium inactivation

3. Install GAC contactors for TOC control

4. Install low pressure membrane filtration for physical disinfection of cryptosporidium

5. Add chlorine dioxide with chlorite control for cryptosporidium inactivation

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Technology Assessment Results for
MWDSC - Diemer FP

*coagulation pH– 7.0
** pH at ozonation – 6.6
***coagulation pH – 7.3
****coagulation pH –5.6
***** coagulation pH – 7.4

Current Condition*

Deficiencies w.r.t. ELPH:
  - crypto inactivation < 1.0 log
  - TTHM > 40 ug/L

1.0 Install O3 w/ bromate control**

Process Changes:
-add 45 mg/L of H2SO4
-add 42 mg/L of NaOH
-add 4.1mg/L O3 & 4 - .20 MG chambers

*2.0 Install UV after filters****

Process Changes:
- install UV
-add 5 mg/L of H2SO4
- add 9.5 mg/L of NaOH

3.0 install GAC contactors

Process Changes:
-None-can't reach crypto inactivation

4.0 Install MF/UF after filters***

Process Changes:
- install membrane post filters
-add 4.6 mg/L of H2SO4
- add 9.0 mg/L of NaOH

5.0 Add ClO2 with chlorite control*****

Process Changes:
-add 2.1 mg/L of ClO2 before RM
-add 15.8 mg/L of Ferrous sulfate at RM
-NaOH decrease to 3.9 mg/L

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MWDSC - Diemer FP

- Limiting Source Water Quality for Existing/Immediate Future Facility
  - Can't reach 1-log crypto credit
- Limiting Source Water Quality for Applied Technology Facility
  - Br = 0.13 mg/L
  - TOC = 3.0 mg/L

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Technology Costs for
MWDSC - Diemer FP

Current Condition

Deficiencies w.r.t. ELPH:
- crypto inactivation < 1.0 log
- TTHM > 40 ug/L

ELPH Compliance

1.0 Install O3 w/ bromate control

Ozone
Capital Cost: $ 54 Mi
O&M Cost: $ 4.2 Mi/yr

pH Adjustment
Capital Cost: $ 9.1 Mi (acid)
O&M Cost: $ 34 Mi/yr

2.0 Install UV after filters

UV
Capital Cost: $ 43 Mi
O&M Cost: $ 0.5 Mi/yr

pH Adjustment
Capital Cost: $ 4.5 Mi (acid)
O&M Cost: $ 5.8 Mi/yr

4.0 Install MF/UF after filters

MF/UF
Capital Cost: $ 370 Mi
O&M Cost: $ 28 Mi/yr

pH Adjustment
Capital Cost: $ 4.5 Mi
O&M Cost: $5.8 Mi/yr

5.0 Add ClO$_2$ with chlorite control

ClO$_2$
Capital Cost: $ 1.3 Mi
O&M Cost: $ 0.5 Mi/yr

Ferrous sulfate
Capital Cost: $ 0.9 Mi
O&M Cost: $ 3.0 Mi/yr

3.0 Did not cost GAC

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INFLUENT ← **State Project Water (41.9 mgd)**

261 Metro Water Dist of So Calif
PWSID No. CA1910087
Los Angeles, CA
Plant Name: H. J. Mills FP
ICR Plant ID No. 172
Treatment Type: conv
Design Flow: 150 mgd

Ozonation (future) ← O3 = 2.0 mg/L

Rapid Mix ← Ferric = 16.7 mg/L

Flocculation (1.6 MG)

Sedimentation (8.2 MG)

CL2 = 4.4 mg/L

Filtration (508,800 gal)

NAOH = 10 mg/L
Cl2 = .4 mg/L
NH3 = .6 mg/L

Clearwell (50 MG)

Wholesale Customers

12 Month Average

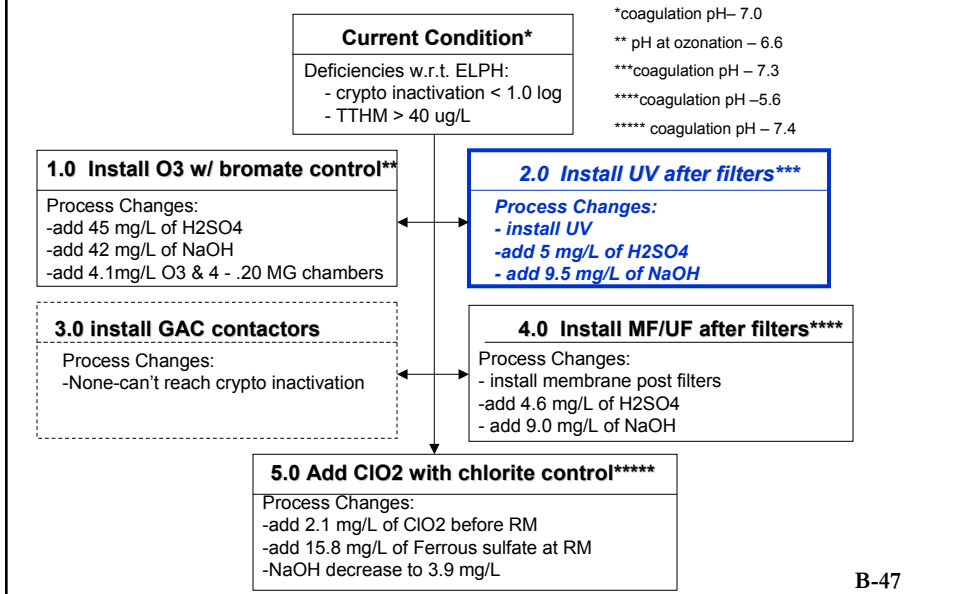| Influent WQ:	Finished WQ:
pH = 8.1	pH = 8.3
Temp = 16.1	Temp = 17.6
TOC = 3.5	TOC = 2.3
UV Abs = 0.096	UV Abs = ???
Br = 0.261	Br = ???
Alk = 80	Alk = 75
Cal hard = ??	Cal hard = 53
Tot hard = 113	Tot hard = 113
NH3 = *ND*	Turb = 0.07
Turb = 1.4	

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MWDSC - Mills FP

- **<u>Process improvement priority:</u>**

  1. Install bromate control technologies (i.e., pH reduction during ozonation)

  2. Install UV inactivation to reduce demand on ozonation for cryptosporidium inactivation

  3. Install low pressure membrane filtration for physical disinfection of cryptosporidium

  4. Add chlorine dioxide with chlorite control for augmenting cryptosporidium inactivation with ozone

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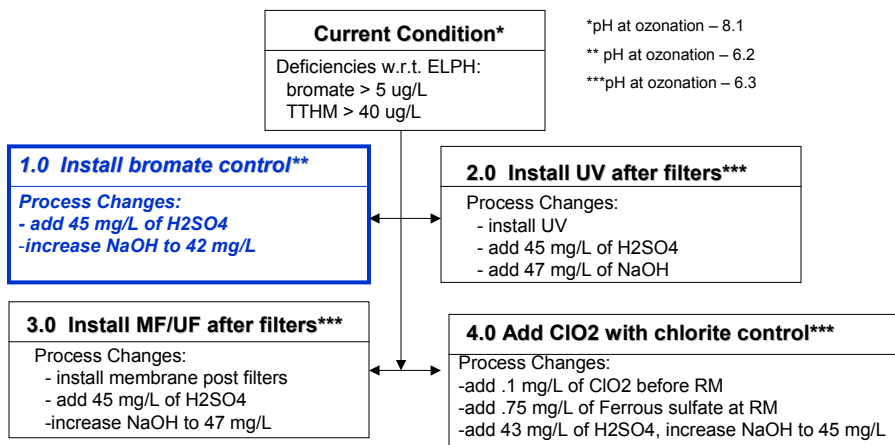# Technology Assessment Results for MWDSC - Mills FP

Current Condition*

Deficiencies w.r.t. ELPH:
  bromate > 5 ug/L
  TTHM > 40 ug/L

*pH at ozonation – 8.1
** pH at ozonation – 6.2
***pH at ozonation – 6.3

1.0 Install bromate control**

Process Changes:
- add 45 mg/L of H2SO4
-increase NaOH to 42 mg/L

2.0 Install UV after filters***

Process Changes:
  - install UV
  - add 45 mg/L of H2SO4
  - add 47 mg/L of NaOH

3.0 Install MF/UF after filters***

Process Changes:
  - install membrane post filters
  - add 45 mg/L of H2SO4
  -increase NaOH to 47 mg/L

4.0 Add ClO2 with chlorite control***

Process Changes:
-add .1 mg/L of ClO2 before RM
-add .75 mg/L of Ferrous sulfate at RM
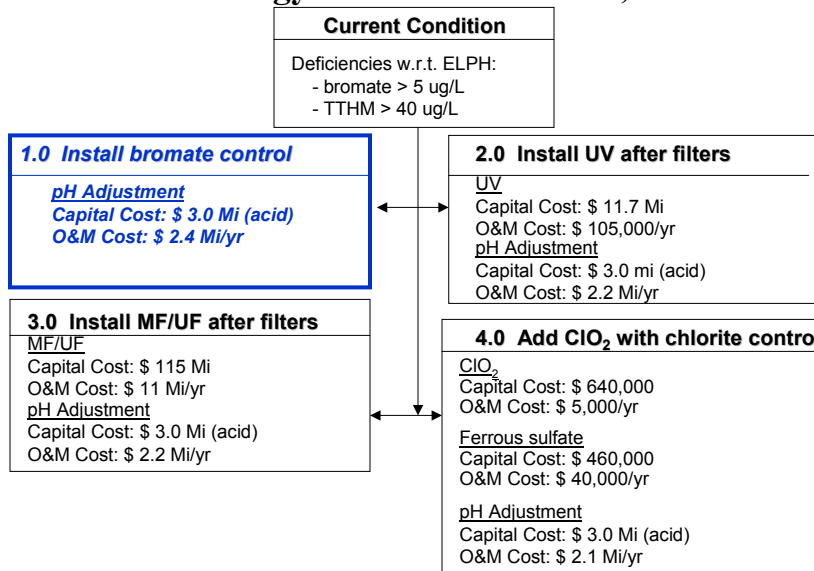-add 43 mg/L of H2SO4, increase NaOH to 45 mg/L

MWDSC - Mills FP

- Limiting Source Water Quality for Existing/Immediate Future Facility
  - Br = 0.07 mg/L
  - TOC = 3.5 mg/L
- Limiting Source Water Quality for Applied Technology Facility
  - Br = 0.26 mg/L
  - TOC = 3.5 mg/L

Technology Costs for MWDSC, Mills FP

Current Condition

Deficiencies w.r.t. ELPH:
- bromate > 5 ug/L
- TTHM > 40 ug/L

1.0 Install bromate control

pH Adjustment
Capital Cost: $ 3.0 Mi (acid)
O&M Cost: $ 2.4 Mi/yr

2.0 Install UV after filters

UV
Capital Cost: $ 11.7 Mi
O&M Cost: $ 105,000/yr
pH Adjustment
Capital Cost: $ 3.0 mi (acid)
O&M Cost: $ 2.2 Mi/yr

3.0 Install MF/UF after filters

MF/UF
Capital Cost: $ 115 Mi
O&M Cost: $ 11 Mi/yr
pH Adjustment
Capital Cost: $ 3.0 Mi (acid)
O&M Cost: $ 2.2 Mi/yr

4.0 Add ClO$_2$ with chlorite control

ClO$_2$
Capital Cost: $ 640,000
O&M Cost: $ 5,000/yr

Ferrous sulfate
Capital Cost: $ 460,000
O&M Cost: $ 40,000/yr

pH Adjustment
Capital Cost: $ 3.0 Mi (acid)
O&M Cost: $ 2.1 Mi/yr