Blending Requirements for Water from Direct Potable Reuse Treatment Facilities
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EXECUTIVE SUMMARY

KEY FINDINGS

- The majority of the trace organic compounds sampled were below the detection limit or measured at levels below recommended limits in all blends.
- Advanced treated water generally had similar levels of biological corrosion as groundwater and surface water sources.
- Advanced treated water did not appear to produce more regrowth of opportunistic pathogens during simulated distribution than corresponding traditional potable water sources.
- All utilities and tested scenarios were highly effective at limiting regrowth of bacteria containing the antibiotic resistant genes of interest during 8 weeks of simulated distribution.
- Using a higher percentage of recycled water that had been treated by reverse osmosis and ultraviolet light advanced oxidation processes actually resulted in less regrowth of total and resistant heterotrophic plate counts than the use of traditional potable water sources.
- During the course of the 8-week pipe rig simulations, there was little evidence of nitrification in the pipe rigs.

BACKGROUND

Potable reuse – both indirect potable reuse (IPR) and direct potable reuse (DPR) – is moving forward nationally at a rapid pace. Research is underway to better understand critical potable reuse issues, including pathogen and chemical pollutant treatment requirements, monitoring requirements, risk assessment, engineered storage, public outreach, operator training, and finished water quality issues (e.g., blending, premise plumbing pathogens, and corrosion). These issues are addressed to different extents in regulations and/or guidelines for IPR and DPR that are under evaluation and/or development in Arizona, Florida, California, New Mexico, North Carolina, Oregon, and Texas.

Industry professionals understand that our water supplies are not pure and that most supplies contain some amount of treated wastewater. Existing national laws, including the Clean Water Act (CWA) and the Safe Drinking Water Act (SDWA), have succeeded in protecting public health and the environment; however, these laws were not developed based upon IPR and DPR realities. As we increasingly look to augment water supplies intentionally with recycled water, at percentages of 25 percent, 50 percent, or even higher, we must better understand the possible water quality and public health impacts from these practices. Central to this issue is defining what level of treatment and resulting water quality are required to make wastewater an acceptable raw water supply for conventional drinking water treatment.

This study aims to shed light on the impacts of blending different amounts and qualities of reclaimed water into conventional raw and potable water systems, examining the treatability at the water treatment plant (WTP) and the finished water quality both leaving the WTP and at the tap. Key issues include aesthetics, regulated and emerging contaminants, microbiology, and corrosion.
OBJECTIVES

Our goal for this project is to examine how reclaimed water quality, blend ratio, blend location, and blend water quality impact treatment, distribution systems, and water quality. The project examined the impact of the following variables:

- Different qualities of recycled water (ranging from tertiary effluent to purified water from various advanced treatment trains)
- Different proportions of recycled water to raw water (ranging from 0 to 50 percent)
- Different points of introduction/blending (prior to a water treatment plant and direct injection into the distribution system)

Bench-scale, pilot-scale, and full-scale testing were employed to examine these variables and related water quality. Importantly, premise plumbing rigs were used to simulate worst-case (stagnant conditions) of the blended waters in water distribution systems. A general review of reclaimed water treatment systems and blending water quality is shown in Tables ES.1 and ES.2.
Table ES.1
Tested reclaimed water treatment systems

<table>
<thead>
<tr>
<th>Utility</th>
<th>Process identification</th>
<th>Secondary treatment</th>
<th>Tertiary filtration</th>
<th>Tertiary chlorination†</th>
<th>O₃</th>
<th>BAF</th>
<th>Pasteurization</th>
<th>MBR</th>
<th>MF/UF</th>
<th>RO</th>
<th>UV AOP</th>
<th>Stabilization</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>O₃-BAC</td>
<td>BNR†</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1</td>
<td>AWP*</td>
<td>BNR†</td>
<td></td>
<td></td>
<td></td>
<td>X</td>
<td></td>
<td>X</td>
<td></td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>2</td>
<td>AWP*</td>
<td>BNR†</td>
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<td></td>
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<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2</td>
<td>Past- AWP*</td>
<td>BNR†</td>
<td></td>
<td></td>
<td></td>
<td>X</td>
<td></td>
<td>X</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>Tertiary</td>
<td>BNR† P‡</td>
<td></td>
<td>X</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>O₃-BAC</td>
<td>BNR† P‡</td>
<td></td>
<td>X</td>
<td></td>
<td></td>
<td></td>
<td>X</td>
<td></td>
<td></td>
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</tr>
<tr>
<td>4a</td>
<td>AWP*</td>
<td>BNR†</td>
<td></td>
<td></td>
<td></td>
<td>X</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>4b</td>
<td>Industrial AWP*</td>
<td>BNR†</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>X</td>
<td></td>
<td></td>
<td></td>
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</tr>
</tbody>
</table>

*AWP (advanced water purification) includes low pressure membrane filtration (MF or UF), reverse osmosis (RO), and ultraviolet light advanced oxidation process (UV AOP).
†BNR is a moderate to high SRT activated sludge process that nitrifies (removes ammonia) and for some facilities denitrifies (removes nitrate).
‡P connotes phosphorus removal in the biological treatment process.
§Tertiary chlorination consisted of a combination of total and free chlorine.
**Purified waters stabilized with zinc orthophosphate and calcium chloride in accordance with client preferences for other potable water reuse groundwater recharge projects.
**Table ES.2**

**Blending waters**

<table>
<thead>
<tr>
<th>Utility</th>
<th>Process identification</th>
<th>Blend water</th>
<th>Treatment post blending</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>O₃-BAC</td>
<td>Raw Surface Water</td>
<td>WTP*</td>
</tr>
<tr>
<td>1</td>
<td>AWP</td>
<td>Raw Surface Water</td>
<td>WTP*</td>
</tr>
<tr>
<td>2</td>
<td>AWP</td>
<td>Groundwater</td>
<td>-</td>
</tr>
<tr>
<td>2</td>
<td>Past- AWP</td>
<td>Groundwater</td>
<td>-</td>
</tr>
<tr>
<td>3</td>
<td>Tertiary</td>
<td>Raw Surface Water</td>
<td>WTP*</td>
</tr>
<tr>
<td>3</td>
<td>O₃-BAC</td>
<td>Raw Surface Water</td>
<td>WTP*</td>
</tr>
<tr>
<td>4a</td>
<td>AWP</td>
<td>Raw Surface Water and Groundwater</td>
<td>WTP¹ for blend with surface water only</td>
</tr>
<tr>
<td>4b</td>
<td>Industrial AWP</td>
<td>Groundwater</td>
<td>-</td>
</tr>
</tbody>
</table>

*WTP is a bench-scale water treatment plant, providing filtration and disinfection of the blended water.

**APPROACH**

**Workshop, Literature Review, and Survey**

The project began by conducting a pre-project workshop (July 2014) as part of a National Science Foundation (NSF) grant, *Workshop on Evaluating Approaches for Implementing Direct Potable Reuse*. This workshop was organized by the National Water Research Institute (NWRI), which assembled a team of academic and regulatory experts to address some of the remaining DPR challenges. In this two-day workshop, experts were presented with several real-world scenarios of blending purified water with potable water supplies. Issues of water quality and public health were discussed, and findings from this workshop helped direct the research objectives and technical approach for this project.

Once the workshop was conducted, a literature review was completed to expand on the workshop discussions. The purpose of this literature review was to provide background information on the range of regulations and treatment requirements that may affect DPR facilities, the water quality impacts of blending, and the impact of blending location on water quality.

**Water Quality Testing**

Informed by the pre-project workshop and literature review, water from utilities using different treatment technologies was selected for testing. In total, there were four utility partners, with some partners having two different tested treatment trains (Table ES.1). Utility 4 had waters tested from two different locations, one being a municipal wastewater treatment plant with purification technologies and the other being a high organic strength industrial wastewater treatment plant with purification technologies. The selected utilities and technologies represent a variety of realistic treatment and blending conditions. Water from each of the four utilities was blended at various ratios with raw water (untreated surface water) or groundwater (finished water), with raw water blends requiring further bench-top treatment simulating a water treatment plant. For each participating utility, blending waters were selected based upon their own specific
potential projects. For example, if the only potable water supply in a region is groundwater, then extracted groundwater was blended with a finished purified water (Table ES.2).

These treatment trains were not designed to provide equivalent levels of treatment, but were instead used for this project to provide insight into different treatment processes and the impact on final water quality after blending. All blends were tested for a number of water quality parameters. A subset of these blends was then used for premise plumbing tests (pipe rigs), which were meant to simulate household premise plumbing systems.

Water quality samples were collected at four different points for each treatment train and blending scenario (Figures ES.1 to ES.5): (1) pre-blending, (2) post-blending, (3) post simulated water treatment, (4) pre and post pipe rigs.

![Diagram of water treatment and blending process]

**Figure ES.1 Utility 1 detail of sample collection locations**
Figure ES.2 Utility 2 detail of sample collection locations

GW = Groundwater
AWP = Microfiltration or ultrafiltration, reverse osmosis, and ultraviolet light advanced oxidation process
Past-AWP = AWP plus pasteurization

Notes:
1. The blend refers to the percentage of purified reclaimed water.
2. Where two blend percentages are shown, two separate scenarios were conducted.
3. Water quality test 3 was conducted on the secondary effluent (prior to advanced treatment)
Figure ES.3 Utility 3 detail of sample collection locations
Figure ES.4 Utility 4a detail of sample collection locations

GW = Groundwater
SW = Surface Water
AWP = Microfiltration or ultrafiltration, reverse osmosis, and ultraviolet light advanced oxidation process

Legend

Utility 4a
1 Water Quality Test Set

Notes:
1. The blend refers to the percentage of purified reclaimed water.
2. Where two blend percentages are shown, two separate scenarios were conducted.
3. Water quality test 3 was conducted on the tertiary effluent (prior to advanced treatment).
Figure ES.5 Utility 4b detail of sample collection locations

At each of the four locations, a specific set of water quality tests was taken:

1) Pathogens, opportunistic pathogens (OPs), antibiotic resistant genes (ARGs)
2) Total trihalomethane formation potential (TTHM FP)
3) Bulk water quality, trace organic compounds (TOrCs), steroids, perfluorinated chemicals (PFCs), disinfection byproducts (DBPs), disinfectant residual stability, nutrients, biological dissolved oxygen demand, OPs, and ARGs
4) Bulk water quality, biological corrosion, trace metals, PFCs, DBPs, nutrients, OPs, and ARGs

RESULTS/CONCLUSIONS

For the different reclaimed water treatment trains, there are reasonable expectations of water quality that can be made based upon the selected treatment processes. For example, water that has been treated by reverse osmosis (RO) is expected to be low in total organic carbon (TOC) and aggressive in nature. Generally, the results can be correlated to three different types of reclaimed water:

- Municipal Advanced Water Purification (AWP) Applications: Very low organics, nutrients, salts, and microbiology. More aggressive water due to the lack of
constituents.

- Industrial AWP Applications: Similar to municipal AWP applications, but with higher nutrient and organic loads in the raw wastewater, resulting in higher nutrient and organic loads in the finished water.


The results are presented below in three categories (Treatment Systems, Distributions Systems, and Engineered Storage) with targeted subcategories (Recycled Water Quality, Blend Ratio, and Blend Location and Conventional Blend Water Quality).

**Treatment Systems**

Treatment of reclaimed water to potable standards can be engineered with or without RO, and can be equally protective of public health. A primary exception is salt removal, which requires RO or similar (e.g., nanofiltration) treatment if feed concentrations are high. For this project, the team examined RO-based purification trains that were intended to meet all regulated and health standards for potable water reuse, whereas the non-RO treatment trains represent “tertiary” or advanced tertiary (O₃-BAC) treatment. In parallel with this publication is the completion of extensive non-RO direct potable reuse demonstration efforts, which couple the O₃-BAC technologies studied in this project with ultrafiltration (UF), granular activated carbon (GAC), and ultraviolet (UV) (Altamonte Springs 2017). The testing of the tertiary and O₃-BAC treatment trains represent lower levels of purification compared to the O₃/BAC/UF/GAC/UV train referenced above.

**Recycled Water Quality**

Samples were analyzed to assess recycled water quality across the advanced treatment trains and blend ratios (post blending and treatment but pre-pipe rig simulation). In general, advanced water purification (AWP) waters were of higher quality than the O₃-BAC waters across blend ratios, primarily due to the use of RO. Both AWP and O₃-BAC blends had contaminant levels below regulated limits for most parameters considered, summarized below.

1) Nutrients

- Municipal AWP Applications: Total nitrogen, ammonia, nitrate, nitrite, and phosphorous were measured in this study, and samples in the blended waters were below regulated limits.

- Industrial AWP Applications: Similar to above, with the exception of high nitrate due to high nitrate levels in the feed water and incomplete removal by RO. Typically RO can remove more than 85 percent of nitrate (Metcalf & Eddy 2007), so high feed concentrations can result in RO permeate with measurable and impactful levels.

- Tertiary and O₃-BAC Applications: Similar to above, with sufficiently high nitrate levels in secondary effluent in some cases that can pass through the tested processes. For non-RO municipal projects, and depending upon the blend ratio, nitrate removal through biological processes may be required.
2) Bulk Water Quality

- Municipal AWP Applications: pH, turbidity, TOC, dissolved organic carbon (DOC), UV at 254 nm, conductivity, alkalinity, sulfate, and chloride were measured in this study. Blends with AWP generally had lower levels of these parameters than conventional drinking water sources. This is as expected with RO treatment.
- Industrial AWP Applications: Unlike municipal AWP applications, turbidity, chloride, conductivity, and pH were higher in industrial AWP blends than in conventional drinking water sources. DOC and alkalinity concentrations were similar. Despite these higher concentrations, none of these blends exceeded regulated limits.
- Tertiary and O₃-BAC Applications: As expected, unlike municipal AWP applications, pH, turbidity, TOC, DOC, UV at 254 nm, conductivity, alkalinity, sulfate, and chloride were generally higher than conventional drinking water sources. With the exception of sulfate and conductivity at specific blends, these higher levels in O₃-BAC blends did not exceed regulatory limits. For sulfate, a 50 percent surface water (SW)/50 percent O₃-BAC blend exceeded EPA's secondary standard. For conductivity, 100 percent O₃-BAC water exceeded the CA secondary maximum contaminant limit (900 µS/cm). For O₃-BAC type purification, salt removal may be required depending upon feed concentrations and finished water quality goals.

3) Disinfection Byproducts (DBPs)

- Municipal AWP Applications: N-nitrosodimethylamine (NDMA), total trihalomethanes (TTHM), the sum of five haloacetic acids (HAA5), and TTHM formation potential (FP) were measured in this study. For municipal AWP applications, TTHM FP tended to be lower than for conventional drinking water sources, which is consistent with findings from past drinking water studies and industry experience. With respect to TTHM and HAA5 formation, all blends were below the regulated values. Blending had an inconsistent impact; in some cases, the reclaimed water reduced the blended concentrations, and in others it did not. NDMA was partially removed by AWP trains; however, high levels of NDMA in the source water caused an exceedance above the 10 ng/L California notification level when 100 percent AWP water was used (CDPH 2013). NDMA feed concentrations vary based upon wastewater feed concentrations, formation through purification (e.g., chloramination, ozonation), and removal through purification (e.g., RO, UV, BAC). Both RO and non-RO treatment trains can be engineered to maintain NDMA at low levels. Reformation of NDMA has also been shown to occur after purification. RO allows NDMA precursors to pass through, most likely low molecular weight amines (e.g., dimethylamine or trimethylamine) that are able to reform NDMA in the presence of chloramines that were subsequently applied in the lab. Alternatively, ultraviolet advanced oxidation (UV/AOP) degraded NDMA into nitrite and dimethylamine, which then similarly allowed for NDMA to reform when chloramines were added (Plumlee and Reinhard 2007, Plumlee et al. 2008, Xu et al. 2009).
- Industrial AWP Applications: Similar to above. The 100 percent industrial AWP water also exceeded the 10 ng/L California notification level for NDMA due to high levels in the source water (CDPH 2013).
Tertiary and O₃-BAC Applications: Higher TTHM FP was observed in tertiary and O₃-BAC blends, which is consistent with findings from past drinking water studies and industry experience. O₃-BAC could reduce TTHM FP through the use of additional treatment, such as GAC (Altamonte Springs 2017). Like AWP applications, concentrations of TTHM and HAA5 were below regulated values. NDMA was observed in the tertiary effluent, and was partially removed by O₃-BAC; however, no NDMA concentrations above 10 ng/L were observed in tertiary and O₃-BAC applications.

Specific UV absorbance (SUVA) 254, was also calculated as part of this study to correlate SUVA₂₅₄ with TTHM FP. SUVA₂₅₄ is the ratio of UV₂₅₄ to DOC. In past work with drinking waters and surface waters, SUVA₂₅₄ was positively correlated to the TTHM FP, and this was attributed to the presence of reactive aromatic moieties (Reckhow et al. 1990, Liang and Singer 2003, Weishaar et al. 2003). However, this study found that caution must be applied when using SUVA₂₅₄ to predict the TTHM FP of blended waters with low DOC. For this work, TTHM formation under FP test conditions was significantly negatively correlated with SUVA₂₅₄ (R² = 0.79, p < 0.001). For these waters, low DOC values increased the SUVA₂₅₄, but also limited the formation of TTHM by the low concentration of total precursors present. This result is not surprising as organic matter character is important in DBP formation.

4) Pathogens: A direct measure of the pathogen levels in the advanced treated wastewater of interest is not sufficient to ensure the recycled water meets the EPA's allowable risk levels, as the detection levels for pathogens are not sufficiently low. The result is the use of a log removal approach to evaluate a water's safety. This log removal approach is based on the treatment train used and is intended to be conservative.

Municipal AWP Applications: Applying a strict log removal approach to the advanced treated wastewaters demonstrated that only the pasteurized-AWP water meets a 12/10/10 log removal goal set by California for IPR projects (CDPH 2015), given the treatment trains employed. Clearly, further log removal credits could be claimed by adding additional treatment, such as environmental buffers for IPR applications. For this study, pathogen levels in the advanced treatment source waters were also measured. Starting with this data and then applying assumed log reduction credits of the treatment trains considered, research found that both the pasteurized-AWP and AWP waters meet allowable risk levels discussed in the NWRI DPR Framework (Independent Advisory Panel 2015). Note that the pathogen sampling is limited and is not intended to represent a broad level of confidence in pathogen concentrations at each site.

Industrial AWP Applications: Similar to above, the industrial AWP application met EPA's allowable risk levels when evaluating the pathogen levels in the advanced treatment source waters; however, using a strict log removal approach, industrial AWP applications did not meet a 12/10/10 log removal goal set by California for IPR projects (CDPH 2015). Again, as stated above, further log removal credits could be claimed by adding additional treatment, such as the environmental buffers for IPR applications.

Tertiary and O₃-BAC Applications: Both approaches for measuring pathogen levels in the advanced treated wastewaters indicated that tertiary and O₃-BAC
applications do not meet the allowable risk levels, which was expected given the treatment methods employed. Other disinfection technologies could be added to different process trains to attain a similar level of disinfection, such as a downstream water treatment plant.

5) Organic Carbon: TOC reduction is an important treatment target to reduce DBP precursors and broad range organic pollutants in general. Additionally, TOC regulations are under consideration or have been adopted in certain states (e.g., California [0.5 mg/L] and Florida [3 mg/L]) for potable reuse applications. The basis for the difference between California and Florida is not justified in the literature, though recent extensive studies in Florida indicate that TOC levels in the range of 2.0 to 3.5 mg/L do not result in DBP exceedances (Altamonte Springs 2017).
   • Municipal AWP Applications: Reduction of TOC in secondary or filtered secondary effluent by recycled water processes is highly dependent upon the treatment processes employed, with RO-based purification expected to provide greater removal than other processes. In general, the RO process is a robust barrier to TOC, readily dropping TOC to <0.3 mg/L in RO permeate. In most cases in this study, concentrations of TOC, DOC, and biodegradable dissolved organic carbon (BDOC) were lower in municipal AWP blends than in conventional drinking water sources. BDOC concentrations were within the range of potable water in all blends.
   • Industrial AWP Applications: Industrial AWP applications did see reductions of TOC, DOC, and BDOC with the addition of industrial AWP water.
   • Tertiary and O₃-BAC Applications: Concentrations of TOC, DOC, and BDOC in tertiary and O₃-BAC applications were similar to, or slightly higher than, concentrations of these chemicals in conventional drinking water sources. This result is also expected, given the non-RO treatment train employed. Though non-RO processes are unable to attain such low TOC levels, work in Altamonte Springs (2017) reduced TOC to as low as 2 mg/L after O₃-BAC/GAC.

6) PFAs: Perfluoroalkyl substances (PFAs) represent an emerging concern for water supplies and can be challenging to remove with conventional treatment systems.
   • Municipal AWP Applications: This study saw little change in concentrations of PFAs for municipal AWP applications when compared to conventional drinking water sources. This indicates that concentrations of PFAs were low in the AWP source waters and/or RO sufficiently removed PFAs during treatment. For both municipal AWP applications and conventional drinking water sources, the concentrations of PFOA and PFOS were well below the EPA health advisory levels of 70 ng/L.
   • Industrial AWP Applications: Similar to above.
   • Tertiary and O₃-BAC Applications: PFAs were not effectively removed by O₃-BAC and tertiary treatment, resulting in higher concentrations of PFOA and PFOS in O₃-BAC and tertiary treated waters. No blend in this study, however, exceeded the EPA health advisory level. If treatment is needed, additional treatment barriers beyond O₃-BAC-based systems would be required. Recent work in Altamonte Springs (2017) indicated broad level PFAs removal by GAC in the purification train of O₃/BAC/UF/GAC/UV.
7) TOrCs: TOrCs include a broad range of hormones, pharmaceuticals, and personal care products. While typically well below any health threshold, TOrCs represent a public perception concern for potable water reuse projects.

- Municipal AWP Applications: The majority of the TOrCs were sampled in all blends (post-treatment, but pre-pipe rig conditioning). Of the advanced treatment trains considered, AWP blends had the fewest TOrCs above the detection limit. These chemicals included DEET and primidone.
- Industrial AWP Applications: Similar to above; however, industrial AWP applications also had detections of TCEP.
- Tertiary and O₃-BAC Applications: O₃-BAC blends had the most detections of TOrCs. The chemicals detected in O₃-BAC blends included DEET, primidone, TCEP, caffeine, fluoxetine, gemfibrozil, TCEP, sucralose, and meprobromate. None of the TOrC concentrations were above known health guidance values.

**Blend Ratio**

For this project, a blend number, such as 75 percent, refers to the percentage of other raw or finished water supply that is blended with recycled water.

For the chemicals found at higher concentrations in recycled water where a larger safety factor may be desired, either the addition of further purification or limiting the blend ratio may be effective strategies. In some cases, the advanced treated wastewater had lower levels of chemicals than the natural blend water, so lower blend ratios may not produce higher quality water. Many of these chemicals were found to act conservatively (e.g., concentration based directly upon the concentration in the two waters and the blend ratio) and thus can be easily predicted for different blend ratios. Below is a summary of these findings.

1) Nutrients: The primary nutrient of concern is nitrate.

- Municipal AWP Applications: Nitrate was below regulated limits for all blends. Thus, for the waters used in this study, limiting blend ratio was not necessary to control nutrient concentrations. Nitrate (and total nitrogen for that matter) in municipal secondary effluent is effectively treated by RO-based purification.
- Industrial AWP Applications: For industrial AWP applications, high nitrate levels in the industrial AWP source water caused concentrations above the EPA regulated value of 10 mg/L (EPA 1992), highlighting the need for blending with low nitrate source water, if nitrate removal through biological processes is not done. For the waters considered, a blend of 88 percent recycled water would be sufficient to limit high nitrate levels.
- Tertiary and O₃-BAC Applications: Like industrial AWP applications, the O₃-BAC applications measured in this study would need to be blended with low nitrate source water to meet EPA limits. At Utility 1, a blend of 50 percent recycled water would be sufficient, and at Utility 3, a blend of 60 percent recycled water would be sufficient to limit high nitrate levels. Alternatively, nitrate removal through biological processes may be required.

2) Bulk Water Quality: Bulk water quality comparisons focused on parameters that may be predictive of corrosion, such as pH and alkalinity.

- Municipal AWP Applications: As described above, blends with higher percentages of municipal AWP water had lower turbidity, conductivity, and sulfate when
compared to existing potable water supplies, thus improving existing water quality. However, blends with higher percentages of AWP water also had lower alkalinity and pH when compared to existing potable water supplies, thus making the water more corrosive. Many of these parameters were found to act conservatively, so expected constituent concentrations can be determined for a utility’s unique water profile on a case-by-case basis when choosing an appropriate blend percentage. These changes to water quality with the introduction of AWP must be balanced by adjusting the blend ratio to find the optimal water quality for the potable water distribution system.

- **Industrial AWP Applications**: Industrial AWP waters saw higher turbidity, chloride, and conductivity than conventional drinking water sources, and lower sulfate concentrations than conventional drinking water sources. These parameters acted conservatively, so blended water quality can be balanced by adjusting blend ratio.

- **Tertiary and O₃-BAC Applications**: Blends with higher percentages of tertiary and O₃-BAC water had higher pH, turbidity, alkalinity, conductivity, and sulfate. The elevated levels of sulfate in O₃-BAC water exceed the EPA secondary standard at a 50 percent blend at Utility 3, and elevated conductivity in O₃-BAC exceeded the CA secondary maximum contaminant level at a 100 percent blend in Utility 1. It was also found that the tertiary blend used in this study would exceed the turbidity limit at a blend of 30 percent. Again, many of the bulk water quality parameters tested in this study were conservative, and utilities considering DPR can determine expected bulk water quality parameters for a particular blend by measuring parameter concentrations in the two waters considered for DPR.

3) **DBPs**: DBPs can be formed by some reclaimed water treatments (e.g., bromate by ozonation, NDMA by chloramination) and reduced by others (e.g., NDMA by BAC, NDMA by UV). Further, reclaimed water processes that result in less TOC are assumed to contain fewer DBP precursors.

- **Municipal AWP Applications**: Generally speaking, RO-based purification systems (e.g., AWP) have low DBP levels and low DBP formation potential. Thus, higher percentages of blend water are expected to improve water quality pertaining to DBPs. However, NDMA is only partially removed by AWP trains, so high levels of NDMA in the source water can cause high levels of NDMA in the blended water if high percentages of municipal AWP water are used.

- **Industrial AWP Applications**: Similar to municipal AWP applications.

- **Tertiary and O₃-BAC Applications**: The opposite is true for O₃-BAC water that tends to have a similar or higher TTHM FP to conventional raw water supplies (noting that further purification of O₃-BAC effluent with GAC substantially reduces both DBPs and DBP formation potential). In determining an appropriate blending ratio, the concentration and character of TOC in an AWP (low TOC concentration) versus an O₃-BAC treated water (typically higher concentration) needs to be considered.

4) **Organic Carbon**: Organic carbon concentrations are assumed to be linked to DBP formation.

- **Municipal AWP Applications**: Blends with higher concentrations of municipal AWP water generally had lower concentrations of TOC, DOC, and BDOC, due to...
RO treatment. Thus, blends with higher concentrations of this municipal AWP would lower concentrations of organic carbon in the blended water.

- Industrial AWP Applications: Similar to above.
- Tertiary and O$_3$-BAC Applications: The opposite is true for the tertiary and O$_3$-BAC applications where RO was not used. As concentrations of organic carbon are higher in these waters, limiting the blend ratio would be one way to control blended water organic carbon concentrations.

5) PFAs

- Municipal AWP Applications: PFAs act conservatively when blended. However, since concentrations of PFAs for municipal AWP applications were comparable to those in conventional drinking water sources, the impact of blending was minimal.
- Industrial AWP Applications: Similar to above.
- Tertiary and O$_3$-BAC Applications: PFAs are not removed well through O$_3$-BAC treatment; thus, their concentrations in the advanced water source water should be considered when picking an appropriate blend ratio. As PFAs act conservatively, their concentrations at a certain blend ratio can be estimated for a proposed DPR system by measuring the PFA concentrations in the two source waters.

6) TOrCs

- Municipal AWP Applications: The TOrCs measured all behaved conservatively; thus, their concentrations at a certain blend ratio can be estimated for a proposed DPR system. However, given their low levels in the waters considered in this study, and the general lack of evidence of health effects at ng/L levels, blending to control TOrCs was not necessary for these source waters.
- Industrial AWP Applications: Similar to above.
- Tertiary and O$_3$-BAC Applications: O$_3$-BAC blends had the most detections of TOrCs. However, like municipal and industrial AWP applications, the low levels of TOrCs measured in this study made blending to control TOrCs unnecessary.

**Blend Location and Conventional Blend Water Quality**

The scenarios tested also varied in both the point of blending and the source of the conventional drinking water used for blending. The different scenarios considered are summarized below.

- Utility 1: Surface water blended with advanced treated wastewater, then subjected to treatment via conventional water treatment techniques.
- Utility 3: Surface water blended with advanced treated wastewater pre-treatment, and then subjected to treatment via conventional water treatment techniques.
- Utility 4a: Two source waters: (1) groundwater blended with advanced treated wastewater post-treatment, and (2) surface water blended with advanced treated wastewater pre-treatment and then subjected to treatment via conventional water treatment techniques.
With these different scenarios, information on how blending with groundwater versus surface water and blending pre- versus post-treatment were considered. In general, groundwater versus surface water quality greatly impacted the blended water quality for some parameters. Additionally, pre- versus post- treatment affected pathogen and TOC concentrations.

1) Bulk Water Quality
   - Surface Water Blended Pre-Treatment: Surface water generally had low conductivity. However, at Utility 3, surface water had conductivity above the CA secondary maximum contaminant level, resulting in all blends exceeding the CA secondary maximum contaminant level.
   - Groundwater Blended Post-Treatment: Groundwater blends typically had higher alkalinity, sulfate, and chloride than the surface water blends considered in this study. For Utility 2 (blends with groundwater), all blends had concentrations of sulfate above the EPA's limit and conductivity above the CA secondary maximum contaminant level. In this case, conventional source water quality is important, as even a 50 percent blend with low conductivity, AWP water did not result in a conductivity level below the CA limit.

2) DBPs
   - Surface Water Blended Pre-Treatment: This study found that concentrations of TTHM and HAA5 in blends with surface water were similar or higher than concentrations of TTHM and HAA5 in blends with groundwater blended post-treatment. This is likely due to lower DOC concentrations in groundwater sources.
   - Groundwater Blended Post-Treatment: Same as above.

3) Pathogens
   - Surface Water Blended Pre-Treatment: The point of introduction for advanced treated wastewater is important for determining pathogen log removal credits. If blending occurs prior to water treatment, additional log removal credits can be obtained, with the level defined by the treatment techniques (e.g., 2/2.5/3 log removal credit for virus/Giardia/Cryptosporidium, respectively [Texas Commission on Environmental Quality 2016]). The additional log removal credits obtained for water treatment at Utility 2 were sufficient for the pasteurized AWP blend to meet the 12/10/10 log removal goal.
   - Groundwater Blended Post-Treatment: While the municipal AWP waters at Utility 1 and Utility 4a were blended post-treatment and thus did not receive log removal credits for water treatment, based on measured pathogen data in the source water, these blends meet the EPA's allowable risk level without the additional water treatment step. As mentioned above, however, the source water data collected in this study was limited for each source water considered, and is not sufficient for determining a recycled water's safety.

4) Organic Carbon
   - Surface Water Blended Pre-Treatment: For Utility 1 and Utility 3, where advanced treated wastewater was blended with surface water prior to treatment, the TOC, DBPs, and alkalinity found in the surface water can affect the required treatment technique in the final water treatment plant, as required by the Stage 1 Disinfectants and Disinfection Byproducts Rule to reduce DBP precursors. This rule was implemented to balance the risks between microbial pathogens and DBPs, as
disinfectants used to control pathogens can react with naturally occurring organic and inorganic matter to form DPBs. Systems using surface water and conventional filtration are required to remove specified percentages of organic materials (measured as TOC) unless they meet alternative compliance criteria. All the blends in Utility 1 and Utility 3 met alternative compliance criteria, and thus were not subject to specified TOC removal requirements. In most cases, the Utility 1 and Utility 3 blends were exempt because of their low TOC, TTHM, and HAA5 concentrations. However, in five cases where either the TOC or TTHM concentrations were higher, the blends' low SUVA254 values exempted these blends from this rule. Even though blending surface water with advanced treated wastewater can increase (or decrease) TOC, TTHM, and HAA5, the blend amounts of advanced treated wastewater considered in this study were not significant enough to trigger the Stage 1 Disinfectants and Disinfection Byproducts Rule.

- Groundwater Blended Post-Treatment: All of the groundwater blends had TOC levels that were below the Stage 1 Disinfectants and Disinfection Byproducts Rule threshold of 2 mg/L.

5) PFAs
- Surface Water Blended Pre-Treatment: PFAs pass through conventional water treatment, so the point of blending is not relevant for these parameters. No differences were seen in PFA concentrations for 100 percent groundwater and 100 percent surface water.
- Groundwater Blended Post-Treatment: Same as above.

6) TOrCs
- Surface Water Blended Pre-Treatment: Only sucralose, primidone, and meprobamate were measured at levels close to their respective MCLs in 100 percent surface water blends.
- Groundwater Blended Post-Treatment: Of the TOrCs measured in this study, none were present in 100 percent groundwater blends.

Distribution Systems

Recycled Water Quality

A primary goal of this study was to determine the effect on distribution system water quality of blending advanced treated wastewaters. Not only did this study look at the water quality immediately after blending and treatment, but it also quantified changes in water quality after spending time in a simulated premise plumbing system. Results are positive for both AWP and O₃-BAC blends. The key findings from this analysis are summarized below.

1) Nutrients
- Municipal AWP Applications: No significant changes were observed in nutrient levels pre- and post-pipe rig simulation for municipal AWP applications. Additionally, during the course of the pipe rig simulations for all utilities, results show the ammonia concentrations through the system did not change significantly, indicating that nitrification most likely did not occur. However, where chlorine was used for disinfection, ammonia concentrations were higher than where chloramines were used for disinfection.
- Industrial AWP Applications: Similar to above. No significant changes were observed in nutrient levels pre- and post-pipe rig simulation for industrial AWP applications.
- Tertiary and O₃-BAC Applications: Similar to above. No significant changes were observed in nutrient levels pre- and post-pipe rig simulation for tertiary and O₃-BAC applications.
- While these results are promising, in real distribution systems containing higher retention times, prolonged storage, and low flow conditions, elevated ammonia levels, if present, could still result in nitrification.

2) Bulk Water Quality
- Municipal AWP Applications: No significant changes in pH, alkalinity, chloride, conductivity, or sulfate were observed over the course of the pipe rig testing for municipal AWP applications. This is true for tests with and without stabilization (Utility 4a only).
- Industrial AWP Applications: Similar to above. No significant changes were observed in pre- and post-pipe rig simulation for industrial AWP applications. All industrial AWP applications included stabilization prior to pipe rig testing.
- Tertiary and O₃-BAC Applications: Similar to above. No significant changes were observed in pre- and post-pipe rig simulation for tertiary and O₃-BAC applications.

3) Disinfection Residual Stability: The initial disinfection dose of either chlorine or chloramines was added to inactivate pathogens in the blended and treated waters, and residuals were maintained. Of the four utilities, Utilities 1, 2, and 4 applied initial doses of chlorine followed by ammonia at their full-scale facilities to produce chloramines for disinfection with target doses of 1.8 to 2.2 mg/L, 3.8 mg/L, and 2.6 mg/L total chlorine, respectively. Utility 3 had a chlorine target dose of 1.5 mg/L free chlorine.
- Municipal AWP Applications: All municipal AWP applications used chloramines for disinfection. In general, almost all municipal AWP applications had limited or no increase in ATP concentrations post-pipe rig simulation. In most cases, these elevated ATP concentrations were still below 10 pg/mL, indicating a low level of activity and the efficacy of higher chloramine doses applied in these waters.
- Industrial AWP Applications: Similar to above. Limited increases in ATP concentration were observed in post-pipe rig simulation for industrial AWP applications. For all industrial AWP applications, chloramines were used for disinfection.
- Tertiary and O₃-BAC Applications: The two tertiary blends, which used chlorine instead of chloramines for disinfection, had much larger ATP increases and total chlorine decreases, most likely due to the higher relative demand as compared to other waters. It is also possible that the tertiary applications had larger ATP increases post-pipe rig simulation because the tertiary applications had lower amounts of disinfection prior to blending and higher organic loads (e.g., food sources) for microbiological activity.

4) DBPs
- Municipal AWP Applications: No significant changes in DBPs were observed over the course of the pipe rig testing for municipal AWP applications. This is expected, given that chloramination was used for disinfection.
- Industrial AWP Applications: Similar to above. No significant changes were observed in DBPs pre- and post-pipe rig simulation for industrial AWP applications. This is expected, given that chloramination was used for disinfection.
- Tertiary and O₃-BAC Applications: Utility 3, where chlorine was used for disinfection, had post-pipe rig exceedances for TTHM for the O₃-BAC blends. Utility 3 also saw a significant increase in TTHM for tertiary blends post-pipe rig simulations. Also, where chlorine was used, HAA5 tended to increase (significant for a few samples), but otherwise no significant changes occurred for HAA5. No HAA5 exceedances were observed.

5) Metals and Corrosion Potential
- Municipal AWP Applications: Given the use of RO for municipal AWP applications, it is expected that some corrosion and metals leaching would occur in these blends. Metals leaching was observed in this study. At Utility 1, blends with higher concentrations of AWP water tended to have higher concentrations of lead and zinc in the pipe rig effluent than conventional drinking water sources, but no significant change was observed in the concentration of copper. However, at Utility 2, there was not a significant change in lead concentration over the duration of premise plumping testing for AWP blends. Pasteurization-AWP blends at Utility 2 did have greater concentrations of copper, zinc, and lead than AWP blends at a 50 percent blend ratio.
- Industrial AWP Applications: While industrial AWP applications also used RO treatment, the industrial AWP applications stabilized their water. Thus, increases in lead, zinc, and copper leaching over conventional drinking water sources were not observed for industrial AWP applications.
- Tertiary and O₃-BAC Applications: Even though RO was not used for tertiary and O₃-BAC applications, some metals leaching was observed. O₃-BAC blends tended to have higher concentrations of copper and zinc in the pipe rig effluent than other advanced water treatment blends.
- To predict these increases in metal concentration, corrosion indices were calculated for all blends tested in this study. However, based on the results of this study, there was no universal corrosion index based on water quality parameters that accurately estimate the extent of lead corrosion that might occur after DPR implementation. There are individual instances when corrosion indices do work well for predicting lead, copper, and/or zinc corrosion. These instances appear to be case specific and indicate a need for a utility-specific study prior to DPR implementation.
- In some instances, the microbial gene density correlated well with lead corrosion from CDA360 brass although there was no correlation with copper or zinc corrosion. Considering Utilities 1, 2, and 3, the logarithm of 16S rRNA genes (universal to all bacteria and measured as a proxy for total bacteria count) in the biofilm correlated well with lead leaching in the pipe rigs (R² = 0.72). However, the experimental plan for this study was not designed to evaluate the mechanisms of this correlation. Therefore, this observation should be evaluated further in future studies. Utility 4 was excluded from this correlation because it was the only utility in this study that employed corrosion control.

6) Biological Corrosion: In general, advanced treated wastewater blends had similar levels of biological corrosion as groundwater and surface water sources. All water
blends saw moderate or aggressive sulfate reducing bacteria (SRB) and sulfate forming bacteria (SFB) biological corrosion.

- Municipal AWP Applications: Municipal AWP applications inconsistently saw acid-producing bacteria and denitrifying bacteria; however, these occurrences were typically moderate or non-aggressive occurrences.
- Industrial AWP Applications: No additional biological corrosion was observed.
- Tertiary and O₃-BAC Applications: Tertiary and O₃-BAC applications inconsistently saw acid-producing bacteria and denitrifying bacteria; however, these occurrences were typically moderate or non-aggressive occurrences.

7) Opportunistic Pathogens: Advanced treated wastewater did not appear to produce more regrowth of opportunistic pathogens (OPs) during simulated distribution than the corresponding traditional potable water source. This was true across municipal AWP, industrial AWP, tertiary, and O₃-BAC applications. As part of the study of OPs, 16S rRNA genes were also monitored. These genes are universal to all bacteria, and served as a proxy for total bacteria count. In all scenarios, regrowth of 16S rRNA genes consistently occurred.

8) Bacteria Containing Antibiotic Resistant Genes: All utilities and tested scenarios were highly effective at limiting regrowth of bacteria containing the antibiotic resistant genes (ARGs) of interest during 8 weeks of simulated distribution. An ARG was only detected in a single sample at above quantifiable levels: the vanA resistant gene in the biofilm of Utility 1’s 90 percent surface/10 percent AWP condition. While these results are very promising, indicating the DPR waters do not pose added risk in terms of proliferating ARGs in the distribution system, it is important to consider that these were short-term studies with young biofilms.

9) Total and Resistant Heterotrophic Plate Counts:

- Municipal AWP Applications: In this study, less regrowth of total and resistant heterotrophic plate counts (HPCs) was seen in AWP blends than in traditional potable water sources. However, some regrowth of HPCs in simulated distribution systems was consistently observed. HPCs capable of growth in the presence of several antibiotic classes, including ampicillin and oxacillin, were detected consistently across utilities.
- Industrial AWP Applications: Like municipal AWP applications, industrial AWP applications had less regrowth of total and resistant HPCs than in traditional potable water sources.
- Tertiary and O₃-BAC Applications: As expected, results indicate that tertiary applications have the potential to support regrowth of total and resistant HPCs in distribution systems.

Blend Ratio

Many of the parameters measured did not change during premise plumbing testing; thus, the effects of blend ratio on post-pipe rig water quality are no different than the effects outlined in the Treatment Systems section above. However, where changes were observed in post-pipe rig water quality, blend ratio was considered. The results of that analysis are shown below.

1) Total and Resistant Heterotrophic Plate Counts

- Municipal AWP Applications: As described above, less regrowth of total and
resistant HPCs was seen in AWP blends than in traditional potable water sources. Thus, it is possible that higher blends of municipal AWP applications could help reduce regrowth of HPCs. However, this should be studied further.

- Industrial AWP Applications: Similar to municipal AWP applications.
- Tertiary and O₃-BAC Applications: As described above, tertiary applications have the potential to support regrowth of HPCs in distribution systems. However, limiting the blend ratio to 5 percent was effective at reducing regrowth of total HPCs by approximately two orders of magnitude compared to a 10 percent blend ratio.

**Blend Locations and Conventional Blend Water Quality**

Many of the parameters measured did not change during premise plumbing testing; thus, the effects of blending location and conventional blend water quality on post-pipe rig water quality are as outlined in the Treatment Systems section above. Additionally, no significant and universal correlation between blend location or conventional blend water quality and metals leaching or corrosion was observed in this study. While conventional water treatment removes metals, the post-pipe rig lead, copper, and zinc concentrations were not significantly different when waters were blended pre- versus post-water treatment.

**Engineered Storage**

**Recycled Water Quality**

Engineered storage was not used for the treatment systems evaluated in this work. It is expected that engineered storage will be an important part of any DPR system to provide failure response time (FRT) and additional treatment for an advanced treated wastewater prior to blending. The time in storage is assumed to range from 30 minutes to as much as a day, though two hours of storage appears to be preferred by utility partners. As it is expected that engineered storage would be added to the system prior to blending with other water supplies, only the effect of recycled water quality on engineered storage was considered. It was assumed that, in many ways, an engineered storage buffer (ESB) would act similarly to a pipe rig. The results from this study, while critically important to understanding the implications of blending on downstream water quality for potable reuse projects, do not directly affect the sizing paradigm for ESBs. However, the results do affect the design of ESBs as described below.

1) Metals and Corrosion Potential: Corrosivity concerns are particularly acute for the ESB, as it will typically not have the benefit of blending with higher-alkalinity water prior to storage, so a compatible tank material or coating will be needed in combination with chemical stabilization of that water. Utilities 1, 2, and 3 did not chemically stabilize their blends, and saw mixed corrosion results in AWP blends. In some cases, higher blends of AWP water increased lead and zinc leaching, and no changes were observed in other cases. Pasteurized-AWP water did increase lead, zinc, and copper leaching; and O₃-BAC water increased copper and zinc leaching. Utility 4 was the only utility to stabilize its water blends. At this utility, metals leaching was generally low, and the presence of AWP water either did not affect or decreased the concentration of metals in the blends measured.
2) DBPs: It was found that for O₃-BAC waters where TTHM FP was higher than in conventional drinking water sources, disinfecting with free chlorine caused the concentrations of both TTHM and HAA5 to increase in the pipe rig. This trend was not observed in blends where chloramine disinfection was used. This finding, while not new, is important when considering using an ESB with free chlorine to achieve additional log reduction credits for pathogen removal.

3) Pathogens: Based on the limited data collected, log removal value requirements from secondary effluent to finished water were determined for *Giardia*, *Cryptosporidium*, and enteric virus with maximum values of 7.5-log, 4.6-log, and 12.6-log, respectively. This dataset is particularly interesting because it implies that less treatment is necessary for *Cryptosporidium* for every scenario studied than is required by current permitting processes; whereas, the maximum value for enteric virus implies a greater treatment need than California's 12-log standard for IPR. The results of this study lend some credence to the need for monitoring requirements in the water feeding a DPR system to better characterize the variability and concentrations of pathogens. With this pathogen source water characterization, a probabilistic quantitative microbial risk assessment (QMRA) can be performed to ensure the EPA risk levels are met (Independent Advisory Panel 2015). Such an analysis will document the value of free chlorination of virus, which could occur in the ESB should DBP formation not be a concern. If, through this risk assessment, it is determined that less log removal credit is needed to meet EPA risk levels than is currently required, the ESB size and detention time may decrease, thus lowering the cost and infrastructure impact of a DPR facility. A shorter detention time in the ESB would also limit DBP formation and corrosion concerns discussed above.

**Summary**

The first goal of this study was to examine the impact of reclaimed water quality on the downstream treatment and distribution systems. From this examination, it can be concluded that most of the reclaimed water blends met the majority of water quality regulations, indicating that reclaimed water quality did not detrimentally impact overall water quality. In particular, none of the TOrCs measured in any blend were above notification levels, guidance values, or regulations in the U.S., and, in fact, the majority of TOrCs were below the detection limit. This may be due to low levels of TOrCs in the advanced treated wastewaters' source water quality or the effectiveness of treatment and blending. Regarding OPs and ARGs, the reclaimed water blends did not produce more regrowth of OPs or ARGs during the simulated distribution than corresponding traditional potable water sources. Furthermore, this study found that reclaimed water blends that used RO and UV/AOP resulted in less regrowth of total and resistance HPCs in the simulated distribution system than the use of traditional potable water sources. While this is promising, more research is needed using older biofilms. Unstabilized reclaimed water did negatively affect corrosion in the simulated distribution systems, even after blending with conventional water supplies. Thus, stabilization of reclaimed water prior to blending is recommended.

The second goal of this study was to examine the impact of blend ratio on the downstream treatment and distribution systems. As expected, many measured chemicals behaved conservatively when blended. For these parameters, it should be sufficient to use the blend ratio to estimate expected blended concentration and determine the optimal blending percentage. If blending municipal AWP water, one should keep in mind that this water is generally very low in
organics, nutrients, salts, and microbiology. This water also tends to be more aggressive. If blending industrial AWP water, one should keep in mind that, while similar to municipal AWP, such water may be higher in nutrient and organic loads. If blending tertiary or O₃-BAC water, one should keep in mind that its water chemistry tends to be similar to raw surface waters with higher TOC, nutrients, and microbiology. Furthermore, particular attention should be paid to nitrate and NDMA, as no nitrate or NDMA removal was seen in the O₃-BAC process, and only partial removal was observed in the AWP trains. PFAs are also of concern, as limited to no removal of PFAs was observed during O₃-BAC. For these chemicals, limiting blend ratio or removing them with additional advanced treatment processes (e.g., GAC) may be necessary.

The third goal of this study was to examine the impact of blend location and conventional blend water quality on the downstream treatment and distribution systems. In general, blend location did not significantly impact downstream treatment or distribution systems, except for meeting pathogen reduction requirements. Conventional blend water quality was important and, in some cases, high levels of chemicals in the conventional blend water caused regulation exceedances. Groundwater had higher conductivity, alkalinity, sulfate, and chloride than surface water, but surface water had higher DOC concentrations than groundwater. This should be taken into consideration when determining an appropriate blend ratio.

APPLICATIONS/RECOMMENDATIONS

Several recommendations for future studies are presented below:

1) Better characterization of pathogen densities in source water (secondary effluent) is needed. This would help in developing uniform water recycling criteria for DPR that uses a probabilistic approach in applying log reduction credits. This approach is further discussed in the California State Water Resources Control Board's Expert Panel Report (Olivieri et al. 2016).

2) To calculate virus log removal credits, this study conservatively assumed that all gene copies were viable pathogens. However, gene copy numbers do not necessarily correlate to viable pathogens. More research is needed to determine the best approach for measuring accurate virus log removal credits given this limitation.

3) Further studies are needed to fully explore the role of biological corrosion in premise plumbing systems from all types of water blends.

4) Further study on the biofilm and lead leaching in premise plumbing systems is needed. One observation of this study was that 16S rRNA correlated fairly well with lead leaching. However further consideration should be devoted to this observation in future studies.

5) While this study is promising in its indication that these DPR blending scenarios do not present added risk in terms of enhancing re-growth of ARGs or antibiotic resistant bacteria as compared to conventional drinking water sources, it is important to consider that this was a short-term study. Longer term studies with mature biofilms would be of value.

6) This study limited the blend ratio to 50 percent. Future studies could look at higher blend ratios as it applies to corrosion. Those studies should include stabilization.
RELATED WRF RESEARCH

- Assessment of Techniques for Evaluating and Demonstrating Safety of Water from Direct Potable Reuse Treatment Facilities (project 4508)
- Critical Control Point Assessment to Quantify Robustness and Reliability of Multiple Treatment Barriers of a Direct Potable Reuse Scheme (project 4541)
- Model Public Communication Plan for Advancing Direct Potable Reuse Acceptance (project 4540)
- Potable Reuse Research Compilation: Synthesis of Findings (project 4645)
CHAPTER 1  
INTRODUCTION

BACKGROUND

Potable reuse is a critically important component of current and future water supplies throughout parts of the world. The vast majority of planned potable reuse is “indirect” potable reuse (IPR) with a much smaller percentage of “direct” potable reuse (DPR). The primary difference between IPR and DPR is the use of an environmental buffer before potable use of the water, such as a groundwater basin or a surface water reservoir (Figure 1.1). The environmental buffer provides a measure of treatment, blending, and response time in the event of a treatment process failure. DPR eliminates the environmental buffer, thus requiring a more detailed understanding of treatment process goals, performance, and monitoring. In contrast, *de facto* reuse refers to a drinking water supply that contains a significant fraction of wastewater effluent without the advanced treatment processes associated with planned potable reuse, typically from upstream wastewater discharges.

Water produced from a DPR facility can be inserted directly into the potable water distribution system (PWDS), added upstream of a potable water treatment plant (WTP), or added directly within a potable WTP between various processes. Little is known about the impact of blending in relation to treatability at the WTP and water quality in the finished potable water. For blending of purified recycled water with raw or finished potable water, there are key issues to be addressed:

- Regulated and Emerging Contaminants
- Microbiology
- Corrosion
- Aesthetics
- Location of Blending

This project addresses this data gap on the impacts of blending and will provide recommendations and guidance for the appropriate use of blending as part of a DPR project.

Tchobanoglous et al. (2011) concluded that blending advanced treated recycled water with other waters is not necessarily required to protect public health. However, unlike the International Space Station, real water systems will never form a 100 percent closed loop and so blending will be necessary for any municipal DPR project.
Source: Adapted from Trussell et al. 2013.

Figure 1.1 Summary of potable reuse paradigms
RESEARCH OBJECTIVES

Recent research on DPR has focused on pathogen barriers, treatment, risk, and engineered storage, providing a roadmap for DPR treatment from untreated wastewater to potable water, but not addressing what happens after final treatment at a WTP. Introduction of a purified water supply (as a blend with other raw or finished water supplies, or supplied directly to distribution) can change distribution system water quality, with potential implications (positive and/or negative) on DBP speciation and formation, pipe corrosion, disinfectant residuals, and taste and odor characteristics. Pipe corrosion can occur as a result of distribution system water quality changes from introduction of a new water supply in both transmission and service mains as well as in premise plumbing. Premise plumbing is a “worst case” location for microbial growth, corrosion, and generation of aesthetic (taste and odor) problems because of the large surface area to volume ratios, long stagnation times, reactive materials, higher temperatures, and other factors present in building plumbing (National Research Council 2006). Additionally, it is hard to address these problems because of the high capital cost and lack of utility jurisdiction associated with updating premise plumbing.

Our proposed goal for this project is to provide recommendations and guidance for the appropriate use of blending as part of a DPR project, including evaluations of treatment, impact of different water qualities, corrosion control issues, and impact on engineered storage, blending location, and blending percentages.

The research project objectives include examining the impact of three blending variables:

- Different proportions of recycled water to raw water (ranging from 0 percent to 50 percent)
- Different qualities of recycled water (secondary effluent and effluent from various advanced treatment trains)
- Different points of introduction/blending (prior to a water treatment plant and direct injection into the distribution system).

Bench-scale, pilot-scale, and full-scale testing will be employed to examine the three blending scenarios. Additionally, premise plumbing rigs will be used to simulate worst case (stagnant conditions) in water distribution systems.

Our intent for this work is to provide quality data that illustrates the potential issues for a wide range of DPR blending scenarios. Our participating utilities represent real world situations where such scenarios may become reality.

TECHNICAL APPROACH

The project began with a pre-project workshop conducted as part of a NSF grant, *Workshop on Evaluating Approaches for Implementing Direct Potable Reuse*. This workshop was organized by the NWRI, which assembled a team of academic and regulatory experts to address some of the remaining DPR challenges. In this 2-day workshop, these experts were presented with several real-world scenarios of blending wastewater with potable water supply. Issues of water quality and public health were discussed and findings from this workshop helped direct the research objectives and technical approach for this project.

Once this pre-project workshop was conducted, a literature review was completed to expand on the workshop discussions. The purpose of this literature review is to provide
background information on the range of regulations and treatment requirements that may impact DPR facilities, the water quality impacts of blending, and the impact of blending location on water quality. The results of this literature review can be found in Chapter 2.

During the experimental part of the project, water from four different utilities was selected, representing a variety of different conditions (e.g., groundwater, surface water, reservoir vs. river intake, etc.). Water from each of these 4 utilities was then blended at various ratios with water that had undergone different advanced treatments (e.g., microfiltration/reverse osmosis/ultraviolet advanced oxidation [MF/RO/UVAOP], ozone, and biologically active filters [O₃-BAF], etc.). Some of these blends then underwent simulated water treatment before all blends were tested for a number of water quality parameters. A subset of these blends was then used for premise plumbing tests (pipe rigs). These premise plumbing tests were meant to simulate household premise plumbing systems. Details of the experimental design can be found in Chapter 3 and a discussion of the results of both treated water quality and premise plumbing tests can be found in Chapters 4 and 5, respectively.

Conclusions and recommendations are presented in Chapter 6.
CHAPTER 2
LITERATURE REVIEW

INTRODUCTION

The quality of the purified recycled water used for blending depends strongly on the level of treatment provided to the wastewater and recycled water, which, in many cases, is controlled by treatment regulations. A summary of the regulatory framework for treatment of recycled water is presented in Appendix A. This literature review focuses on water quality impacts of blending and the impact of blending location on water quality.

WATER QUALITY IMPACTS OF BLENDING

Recent research on DPR has focused on the use of multiple barriers, a critical control point approach to process control, improved and more efficient treatment, and minimization or elimination of the acute risk. However very little research exists on the effects of DPR on distribution system water quality, and even less has been done regarding blending of potable water and purified recycled water of different qualities, or about the location where purified recycled water is introduced in a potable water system. This part of the literature review focuses on these aspects.

Perhaps the most immediate advantage of blending water of different qualities is to dilute an undesirable contaminant, e.g., total dissolved solids (TDS). In the case of IPR or DPR, blending purified water with other water sources provides dilution of the purified water. Dilution of purified water may be perceived as a loss of identity of the purified recycled water and may facilitate public acceptance. However, the effect of blending on water quality downstream of the point of blending or in a PWDS is difficult to predict, and presents many challenges on all aspects of water quality, which are described below:

- Microbial characteristics, including microbial regrowth in distribution systems.
- Chemical composition, including organic and inorganic chemicals, disinfectant stability, and DBP formation.
- Corrosiveness and aggressiveness, including metal release (e.g., lead, copper, iron).
- Aesthetics characteristics, including taste, odor, color, and mineral composition.

In a survey of 115 potable water providers that rely on multiple water sources (groundwater and surface water), Peet et al. (2001) observed that 63 percent of the respondents felt that some aspects of blending were a concern. Corrosion and loss of disinfectant residual were the main concerns reported (by 26 percent and 23 percent of the respondents, respectively), followed by taste (13 percent of the respondents), hardness (10 percent of the respondents), and general water quality issues (10 percent of the respondents).

Water quality may also degrade significantly in the PWDS once water leaves the treatment plant, with the largest problems occurring in premise plumbing. Premise plumbing is a “worst case” location for microbial growth, corrosion, and generation of aesthetic problems because of the large surface area to volume ratios, long stagnation times, reactive materials, higher temperatures, and other factors inherent to building plumbing (NRC 2006). Further, water providers have very little to no control over in-house plumbing.
Conservative vs. Non-Conservative Parameters

Physical, chemical, and microbial mechanisms are responsible for the formation, decay, degradation, and transformation of contaminants in water systems. The physical mechanisms include sedimentation of precipitates under low-flow conditions or sieving; the chemical mechanisms are precipitation, adsorption or co-precipitation, and generation; and the microbial mechanisms include sorption onto biofilms and bacterially-mediated reactions (Friedman et al. 2010, HDR 2006). These mechanisms explain why water quality parameters behave differently, being either conservative, or non-conservative. Conservative parameters do not change over time or due to blending water of different qualities, whereas non-conservative parameters exhibit changes over time, which may include changes in concentrations and/or speciation for gross or aggregate parameters (Table 2.1).

If mass balance calculations and the Law of Mixtures can be used to assess concentrations of conservative parameters upon blending, non-conservative parameters behave according to chemical reactions of various orders and estimating them typically requires bench- or pilot-scale testing. Examples of conservative parameters include TDS, conductivity, alkalinity, hardness, and individual species such as chloride and sulfate; whereas turbidity, disinfectant residual, DBPs, total organic carbon (TOC), microorganism density, dissolved inorganic carbon (DIC), nitrite and nitrate are non-conservative parameters. pH is an example of a complex conservative parameter because two-water blends can be predicted accurately using a mass balance approach, but blends of multiple waters requires a different approach where the resulting pH can be modeled using mass balances of alkalinity and acidity (Peet et al. 2001).

Peet et al. (2001) developed the MULTIBLEND model to predict conservative water quality parameters (total hardness, total alkalinity, conductivity, water temperature, and pH) when two or three waters blend together. The model was tested with a blend of up to five waters.

<table>
<thead>
<tr>
<th>Conservative parameters</th>
<th>Non-conservative parameters</th>
</tr>
</thead>
<tbody>
<tr>
<td>– No changes over time or when blending water of different qualities</td>
<td>– Change over time, including changes in concentration or speciation</td>
</tr>
<tr>
<td>– Modeled by mass balance calculations and the Law of Mixtures</td>
<td>– Evaluated using bench- or pilot-scale testing</td>
</tr>
<tr>
<td>– Examples: conductivity, alkalinity, hardness</td>
<td>– Examples: turbidity, disinfectant residual, DBPs, TOC, DIC, nitrite and nitrate</td>
</tr>
</tbody>
</table>

From a recycled water perspective, recognizing that certain parameters are conservative and others are not is important because dilution or blending is often considered as one option to decrease contaminants that may still be present in purified recycled water. However, the resulting water quality upon blending may be surprising, even for conservative parameters.
Chemical Characteristics

As noted above, despite the fact that chemical contaminants may not pose as critical of a risk as microbial contaminants, their wide variety, particularly in water supplies subject to anthropologic activities, make them an important group of contaminants. The occurrence of chemical contaminants in wastewater is a function of the proportion and composition of domestic and industrial wastewaters, and treatment strategies practiced at the wastewater and recycled WTPs.

Of critical importance are those pollutants that may present health risks, including some inorganic, radiologic, industrial, pesticide and pharmaceutical substances that are regulated by maximum containment levels (MCLs) (Tchobanoglous et al. 2011). As studied by Salveson et al. (2010) and Trussell et al. (2013), advanced wastewater treatment processes can remove the majority of both regulated and as-of-yet unregulated contaminants that may be present, but several trace level constituents may be found at nanogram per liter (ng/L) levels. A number of them have been proposed by an independent panel (NWRI 2013a) as indicators of treatment process effectiveness because they appear frequently in wastewater and at measurable concentrations. They include pharmaceuticals, drugs and antibiotics, sucralose, tris[2-chloroethyl]phosphate (TCEP), triclosan and N,N-diethyl-meta-toluamide (DEET). NWRI also proposes health-based threshold concentrations for these unregulated parameters, which are significantly higher than the concentrations typically found in secondary effluent, let alone advanced treated water (Trussell et al. 2013, Steinle-Darling et al. 2016, and others).

For this discussion, the chemical aspects of blending purified recycled water with other water supplies were separated into regulated inorganic constituents, regulated organic constituents, regulated disinfectants and DBPs, and regulated and unregulated microconstituents, recognizing that disinfectants, DBPs, and microconstituents can be either organic or inorganic.

Inorganic Chemicals and Minerals

Inorganic chemicals include a wide variety of compounds. This section discusses the main contaminants that are likely to be found at higher concentrations in wastewater than typically found in natural potable water sources, and that may present challenges when blending purified recycled and potable waters. These constituents include salinity (i.e., TDS, conductivity), bromide, iodide, metals, and nitrogen compounds (ammonia, nitrite, and nitrate).

From a blending perspective, many, but not all, inorganic chemicals are conservative parameters that do not change over time or when waters of different qualities are blended. As a result, if the inorganic contaminants discussed here tend to be present in higher concentrations in treated wastewater, blending with potable water sources may be beneficial to decrease the overall concentrations, particularly for TDS, bromide, iodide, and metals. However, studies have shown accumulation and potential release of inorganic chemicals in PWDS corrosion scales, precipitates, sediments, and biofilms, which represent reservoirs for these constituents that are sometimes also called “substrates” or “sinks” (Friedman et al. 2010, HDR 2006). Friedman and colleagues (2010; also documented in HDR 2006) showed how different inorganic chemicals and certain radioactive elements exhibit various tendencies to accumulate and be released from each contaminant reservoir. Any changes in hydraulic or physical conditions (e.g., flow rate and direction, pressure transients, construction activities), or in water quality and chemical conditions (i.e., water temperature, pH, oxidation-reduction potential (ORP) including disinfectant type and residual, competing ions, and concentrations of inhibitors, sequestrants, and chelating agents), may result
in contaminants accumulation or release in PWDSs. More specifically to the release, three chemical mechanisms were proposed by HDR (2006): chemical destabilization of existing scales resulting from a substantial change in water quality, progressive contaminant desorption due to prolonged changes in water quality, and frequent adsorption/desorption cycling due to minor changes in water quality.

**Salinity.** As stated by NWRI (2013b), “Increased salinity is one of the most under-recognized water quality threats in the Southwest,” and has been a concern since the 1980’s in potable supplies, irrigation water, recycled water, and groundwater sources. Because high salt concentrations prevent the use of the affected water for certain applications (e.g., crop and landscape irrigation, cooling towers), corrode distribution system structures, and may alter the water palatability, the EPA established a secondary MCL of 500 mg/L. However, being subject to only a voluntary standard, TDS removal is not as closely guided as that of chemicals with primary drinking water standards (TDS issues in purified recycled waters are well covered by Thompson et al. 2006).

TDS concentrations tend to be higher in recycled water than in other water sources due to human activities such as human excretion, chemicals added for self-regenerating water softeners and swimming pools, industrial and commercial discharges, and water and wastewater treatment (Thompson et al. 2006). TDS increases of 250 to 300 mg/L are considered typical (NWRI 2012). Studies have shown that residential self-regenerating water softeners used to treat potable water hardness may be partly responsible for the higher salt concentrations found in wastewater and treated recycled water (SCSC 2009).

Several treatment processes can remove particles and even organic contaminants, but only specific processes can remove TDS, namely RO or NF membrane filtration. As a result, TDS concentrations may remain high following secondary wastewater treatment, which may be a challenge in drinking water applications (Trussell et al. 2013).

**Bromide and Iodide.** Like TDS, bromide (and to some extent iodide) can be significantly higher in purified recycled water, as a result of the added TDS (NWRI 2012). Bromide and iodide influence DBP formation potential in subsequent potable water applications, as described later on in this literature review.

**Metals.** Metals can be present in potable water sources, as well as in wastewaters. Depending on the contribution of industrial wastewater, certain metals can be present at high concentrations. Fortunately, treatment strategies are well designed to remove them, and they are addressed by various regulations.

**Nitrogen Species.** Nitrite and nitrate are the only inorganic nitrogen species with public health implications. These compounds can be removed by wastewater treatment processes, namely nitrification and denitrification, though only some conventional wastewater treatment plants implement nitrification, and very few denitrify fully.

When blending purified recycled and potable waters, ammonia may present an additional challenge as it is responsible for nitrification. Nitrification is discussed in detail by AWWA (2013), and only summarized here. It is the biological conversion of ammonia to nitrite by ammonia oxidizing bacteria (AOB), then subsequently to nitrate by nitrite oxidizing bacteria (NOB). Ammonia may be present in the source water (both natural and recycled, depending on the treatment processes used), may result from poor ammonia dosage during monochloramine formation, or may be due to degradation of chloramine residual. Once initiated, nitrification is self-propagating and difficult to reverse: the nitrite produced by AOB reacts with chloramines to degrade disinfectant residuals, releasing ammonia, and thereby enhancing the nitrification process.
The resulting lack of a disinfectant residual may allow heterotrophic and coliform bacteria to grow. Nitrification may also be responsible for decreases in pH, dissolved oxygen, and alkalinity, which may enhance corrosion (AWWA 2013, Zhang et al. 2010a). The presence of excess ammonia in the chloraminated source waters may also contribute to the leaching of copper, as ammonia can complex copper and increase aqueous copper concentrations, although the relative importance of this complexation has not been demonstrated (Schock 1999, Trussell 1985).

**Phosphorus.** Although carbon is usually considered the limiting factor for microbial growth in North American waters, phosphorous is essential for bacterial growth and has been found as the limiting factor in certain types of water (Sathasivan and Ohgaki 1999). As such, blending with phosphate-containing waters (as may be the case in purified recycled water, depending on the treatment strategy used), could potentially enhance microbial regrowth.

Along the same lines, one may expect phosphorus-based corrosion inhibitors to have the potential to induce microbial regrowth in PWDSs. These inhibitors may contain either orthophosphate, polyphosphate, or a blend of the two. Polyphosphates cannot be directly metabolized by bacteria, but orthophosphates (e.g., $\text{PO}_4^{3-}$, $\text{HPO}_4^{2-}$, $\text{H}_2\text{PO}_4^-$, and $\text{H}_3\text{PO}_4$) are readily available for biological metabolism without prior degradation, thereby enhancing microbial growth. However, studies by Volk et al. (2000) and Appenzeller et al. (2001) showed that the benefits of corrosion inhibition far outweigh the potential for microbial regrowth, and less microbial regrowth (both suspended and fixed biomass) was observed in the presence of corrosion inhibitors.

**Organic Chemicals**

Organic chemicals constitute a wide variety of chemical constituents. This section discusses two of the main groups, i.e., organic carbon and organic nitrogen, and additional compounds are presented in the Microconstituents Section later in this literature review. Certain DBPs discussed later are also part of the organic chemical family.

**Organic Carbon.** Effluents from secondary and tertiary treatment processes may contain more organic carbon than many surface waters, which may result in microbial (re)growth, unstable disinfectant residuals and increased DBPs, impair the aesthetic quality such as taste, odor, and color, consume oxygen, and bind and transport trace level pollutants. Higher organic carbon content also affects treatment processes by requiring higher coagulant doses, which may result in increased sludge formation, or necessitating additional treatment processes such as ozone and biofiltration. On the other hand, purified recycled water, treated with RO or NF membranes for example, may contain less organic carbon than many surface waters, even treated surface waters, which may improve water quality after blending. Narashimhan et al. (2005) compare concentrations of DOC in potable and recycled waters.

As mentioned above, organic carbon in treated potable water is important because it is related to the growth of heterotrophic bacteria, which are the most common type of microorganisms in drinking water.

TOC can be separated into natural organic matter (NOM) and effluent organic matter, EfOM. If NOM is primarily composed of humic substances derived from the degradation of plants and animals in natural surface waters, EfOM can be composed of a variety of compounds, from degradable and soluble microbial products to refractory substances, and trace chemicals (Trussell et al. 2013).

In drinking water applications, UV absorbance at 254 nm ($\text{UV}_{254}$) and the specific UV absorbance (or SUVA, i.e., the ratio of $\text{UV}_{254}$ to DOC) are also used as indicators of organic
carbon. In conventional surface waters, SUVA has shown good correlations with the formation of the regulated DBPs (trihalomethanes, THMs, and haloacetic acids, HAAs). In wastewater applications, the ratio of UV absorbance measured at 280 nm (UV_{280}) to UV_{254} has been used to represent the relative protein content of water (Snyder et al. 2013). Other indicators have also been proposed, including fluorescence intensity, which is related to the excitation and subsequent emission of light (or relaxation of electrons during irradiation; Trussell et al. 2013). Results are represented on excitation-emission matrices (EEMs), where the ratio of humic- to protein-like peaks has been associated with the relative contribution of wastewater to natural water sources (Henderson et al. 2009).

From a DPR perspective, advanced water treatment processes are required to remove organic carbon. In California, a TOC limit of 0.5 mg/L of wastewater origin has been promulgated for indirect potable reuse projects using direct injection to groundwater (DDW 2017). In contrast, Florida has a 3 mg/L TOC limit for indirect potable reuse projects. A separate WE&RF project is currently underway to better define acceptable levels of TOC for DPR projects.

Beyond TOC, Trussell et al. (2013) have proposed a 90 percent EEM total fluorescence reduction through wastewater treatment trains as an indicator of EfOM removal, and performance- and health-based chemical indicators to characterize trace pollutant elimination. BDOC was also examined as an alternative to TOC to assess the removal of unregulated wastewater-derived organic compounds from purified recycled water, when used specifically for groundwater recharge via surface spreading (NWRI 2012). Studies have shown that BDOC removal correlates well with the removal of biodegradable CECs, and thus, could be used as an indicator of public health protection. However, highly treated purified recycled water, such as RO-treated waters, may need to monitor directly for trace pollutants as these waters may not contain sufficient organic material to sustain the biomass used during the BDOC assay (NWRI 2012).

**Organic Nitrogen.** In distribution systems, nitrogen may be present as organic or inorganic nitrogen, all of which may adversely affect water quality.

Studies suggest that EfOM may contain a greater proportion of nitrogenated organic compound than other water sources (Debroux 1998, as cited by NWRI 2012), because it is derived from algae and bacteria, as opposed to NOM that is derived from the degradation of plants and animals. However, carbon remains the dominant fraction of organic material. In a survey of 28 potable WTPs and a number of advanced purification facilities, Westerhoff et al. (2006) measured an average of 18 mg C per mg N, with variability across sampling sites and seasons.

In contrast to the ammonia-chlorine reaction that leads to the formation of inorganic chloramines (mono-, di-, and tri-chloramines), organic nitrogen can also react with chlorine to form organic chloramines. Unfortunately, these organochloramines are ineffective as disinfectants, and serve as intermediates for DBP formation (Westerhoff et al. 2006; Wolfe et al. 1985). Westerhoff and colleagues (2006) showed that organic nitrogen can be an important NDMA precursor. Also, the $N,N$-diethyl-$p$-phenylenediamine (DPD) colorimetric methods typically used by water providers to measure chlorine residual cannot distinguish between organic and inorganic chloramines (APHA 2005), which can lead to a high total chlorine residual measurement that has very little disinfection capability.
Disinfectants and Disinfection Byproducts

**Disinfectants.** Free chlorine reacts quickly with many substances including phenols, aromatic compounds, humic substances, iron, sulfides, and ammonia, some of which may be present in greater concentrations in recycled water than in drinking water, depending on treatment processes used.

With ammonia, chlorine reacts rapidly to form combined chlorine, also called chloramines. The chloramine speciation between mono-, di- and tri-chloramine, is a function of the chlorine-to-ammonia ratio and pH. Depending on the treatment processes used at the advanced treatment facility, sufficient ammonia may remain in the purified recycled water to convert a significant fraction of the chlorine added for disinfection into chloramine. The fact that free chlorine is a stronger disinfectant than chloramine to control microorganisms is already well documented (EPA 1991a).

Once formed, free and combined chlorine can react according to two general mechanisms:

- **Demand reactions,** where chlorine or chloramine reacts with water constituents such as organic materials, bromide, some metals (including iron scales and corrosion tubercles), nitrite, and others. Some products of these reactions are DBPs.
- **Decay reactions,** i.e., natural auto-decomposition of chlorine or chloramine, which is affected by the concentration of the disinfectant itself, chlorine-to-ammonia ratio in the case of chloramines, pH, water temperature, contact time, and residence time in the PWDS, bromide, and potentially alkalinity.

Detailed information about disinfectants that affect distribution system water quality, their associated DBPs, and control strategies is presented in Baribeau et al. (2017).

Depending on the characteristics of the blended waters and entry point of purified recycled water into a PWDS (either within the WTP or in the PWDS), disinfectant residuals and stability, DBP concentrations and speciation can change significantly.

Case studies conducted by Narashimhan et al. (2005) indicated that the most common problem encountered in recycled water distribution systems (RWDSs) was loss of disinfectant residual, coinciding with increasing bacterial concentrations. The case studies examined during the project conducted by Narashimhan et al. (2005) indicated significant consumption of disinfectant residual, and increases in bacterial counts and biofilm density. Microbial regrowth was significantly less when chlorine was present (this was also observed by Jjemba et al. 2010a and 2010b, and Storey and Kaucer 2009) and nitrogen species were limited. Some of the case studies also showed a decrease in organic carbon, and increased turbidity and metal release. In their study, Narashimhan et al. (2005) also include a literature review on the effects of disinfectant residuals on microorganisms in potable and recycled water distribution systems.

Pilot distribution systems made of different materials and receiving various blends of treated surface water, groundwater, and desalinated seawater showed that chlorine residuals were sensitive to reaction time, organic material as measured by UV254, and water temperature (Taylor et al. 2005). Results also showed lower free and total chlorine consumption rates in PVC and lined cast iron pipes, and higher decay in unlined cast iron and galvanized iron pipes, highlighting the importance of pipe materials. These results illustrate the numerous parameters that can influence disinfectant residuals upon blending, and thus, DBP concentrations and speciation, and how the
resulting water quality can be difficult to predict. Bench or pilot tests are needed to determine the resulting water quality.

**Disinfection Byproducts.** Mosher (2014) identified the removal of DBP precursors as one of the main risks of DPR, along with better documentation of pathogen and chemical removals, and a better understanding of microbial communities. The following DBPs and their respective target concentrations in potable water were proposed by NWRI (2013a) as indicators of wastewater treatment train performance evaluation, and are discussed in this section: TTHMs\(^1\) (80 \(\mu\)g/L), HAA5\(^2\) (60 \(\mu\)g/L), NDMA (10 ng/L), bromate (10 \(\mu\)g/L), and chlorate (800 \(\mu\)g/L). Background information about these DBPs is presented in Baribeau et al. (2017), NWRI (2013a), and Trussell et al. (2013). Baribeau et al. (2017) also presents detailed information about their fate in PWDSs.

DBP formation is influenced by many parameters such as disinfectant type and dose, contact time with the disinfectant, water temperature, and pH, as well as by the concentrations and composition of organic matter, and inorganic precursors such as bromide and iodide. For example, the aromatic fraction of the organic matter, which is captured by UV\(_{254}\), has been linked to TTHM formation potential, whereas the aliphatic fraction has been associated with HAA formation. Evidence suggests that the EfOM present in wastewater contains a greater proportion of aliphatic compounds, thereby increasing the proportion of HAAs formed when compared to other water sources (Debroux 1998, reported by NWRI 2012).

As described previously, purified recycled water may contain significantly greater concentrations of bromide and iodide than other water sources (NWRI 2012), thereby increasing TTHM and HAA formation potential upon chlorination (because disinfection of recycled water is required in many states), and shifting the speciation of these DBPs towards the more brominated and iodinated species as opposed to the chlorinated species. Weinberg et al. (2002) also showed that alternative disinfectants (i.e., chloramines, ozone, and chlorine dioxide) tend to increase the fraction of brominated and iodinated DBPs to total DBPs when compared with free chlorine. Because brominated and iodinated DBPs are heavier, on a molar weight basis, than their chlorinated counterparts, they tend to increase DBP concentrations. Also, recent studies suggest that brominated and iodinated DBPs may exhibit greater health risks than chlorinated DBPs (Kargalioglu et al. 2002, Plewa et al. 2002, Woo et al. 2002).

Nitrogenous DBPs (N-DBPs) have gained attention because of their potential health risks. Of particular interest are nitrosamines, a class of DBPs that represents a wide group of chemicals, with N-nitrosodimethylamine (NDMA) being the most common example within the group. Dickenson and colleagues (2013) determined that NDMA and N-nitrosomorpholine (NMOR) were the two most common nitrosamines detected in wastewater effluent. Several other nitrosamines may be formed during chloramination, but their precursors are not as evident in wastewater as those responsible for the formation of NDMA, which can be formed when chloramines react with nitrogenous precursors (e.g., dimethylamine) that are present in effluent (NWRI 2013a). Although not regulated at the national level, California and Massachusetts have adopted notification levels of 10 ng/L for N-nitrosodimethylamine (NDMA), the most commonly detected N-nitrosamines in drinking water systems (Russell et al. 2012, Woods and Dickenson 2015). NDMA precursors with high yields are compounds with quaternary or tertiary amines and include cationic polymers used in drinking water treatment (e.g. PolyDADMAC) (Park et al. 2009, Krasner et al. 2013 and

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1 TTHMs refer to the sum of the four regulated THMs, i.e., chloroform, bromodichloromethane, dibromochloromethane, and bromoform.

2 HAA5 refer to the sum of the five regulated HAAs, i.e., monochloroacetic acid, dichloroacetic acid, trichloroacetic acid, monobromoacetic acid, and dibromoacetic acid.
In addition to reaction with chloramines, NDMA can also be formed through ozonation of anthropogenic compounds with specific moieties (e.g., carbamates, hydrazones, hydrazines, sulfamides and thiosemicarbazides) (von Gunten et al. 2010, Marti et al. 2015). Therefore, in systems using chloramination or O₃, there is the risk of NDMA formation (Gerrity et al. 2015).

Being one of the strongest oxidants, ozone represents an attractive option to treat recycled water for potable applications. However, ozone reacts with bromide to form bromate, a regulated contaminant that presents health risks. Bromate can also be present in hypochlorite solutions. Dickenson and colleagues (2013) showed that ozone may also form nitrosamines and other byproducts such as hydrazones, semicarbazide, thiosemicarbazide, and carbamates.

Chlorate is one of the main byproducts of chlorine dioxide, and can be found in aged hypochlorite solutions (Bolyard et al. 1993). Ozone can also oxidize hypochlorite to chlorate. In addition, chlorate may be present in some industrial wastewaters, including paper mill effluents (NWRI 2013a).

From a blending perspective, the fate of disinfectant residual and DBPs that are already formed in recycled water and potable water is difficult to predict after blending. The final byproducts are highly dependent on the location of blending, additional treatment processes and disinfectants used, disinfecting conditions, and water quality of the blended waters (e.g., pH, TOC, bromide and iodide concentrations, target chlorine-to-ammonia ratio, etc.). Bench or pilot tests are needed to determine DBPs that would result upon blending.

**Microconstituents**

Microconstituents have been grouped into a number of well-known categories, such as pharmaceuticals and personal care products (PPCPs), biocides (pesticides, herbicides, and fungicides), perfluoroalkyl substances (PFAs), such as perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS), perchlorate, 1,4-dioxane, nanomaterials, and endocrine disrupting compounds (EDCs), although several of these groups have significant overlap. The term trace organic compounds or “TOrCs” provides an umbrella category for most of the above (with the exception of nanomaterials). TOrCs include organic compounds that are naturally occurring, synthetically produced, or formed during chemical reactions such as disinfection. They represent a growing issue both in drinking water sources and recycled water applications, and are discussed in detail in NWRI (2013a) and Trussell et al. (2013).

A number of publications have highlighted the wide variety and widespread occurrence of microconstituents that may be present in wastewater and their treated effluents, with their associated risks (e.g., Guo et al. 2010, Salveson et al. 2010, Sedlak et al. 2000). Their presence in purified recycled water is highly dependent on the nature of the sewage (including the proportion of domestic vs. industrial wastes), treatment processes used, season and weather, and others. Their occurrence has generated discussions about the need to remove them and the most appropriate techniques to do so. However, because they represent such a varied group of compounds, their removal may not be straightforward, and their presence cannot be directly linked to any specific health effects (Trussell et al. 2013). Recognizing that it is not possible to monitor for all of them, the identification of suitable indicators is also challenging.

Snyder et al. (2007) observed that a number of TOrCs occurred in both raw and treated drinking water: DEET and atrazine are examples. However, many more compounds are present in either water source, but not in both. Snyder et al. (2007) observed that carbamazepine, Dilantin, sulfamethoxazole, and meprobamate were the compounds that occurred in most raw drinking
waters, whereas meprobamate, Dilantin, ibuprofen and iopromide were dominant in treated drinking water. Moreover, Snyder et al. (2007) observed that conventional treatment processes such as coagulation, flocculation and sedimentation are inefficient at removing TORCs, and advanced processes are required (permeation membranes such as tight nanofiltration (NF) and RO, ozone and AOPs, activated carbon). It should be recognized that membranes and activated carbon are physical processes that remove the contaminants from the water, whereas ozonation and AOPs are chemical oxidants that can form reactive byproducts. This suggests that the highly treated (i.e., purified) recycled water would likely contain lower concentrations of TORCs than conventionally treated drinking water. From this perspective also, blending purified recycled and potable waters may improve drinking water quality.

Nanomaterials are compounds with at least one dimension less than 100 nm. They can be divided into two groups: nanoparticles (or discrete nanosized units), and nanostructured materials (or macroscale particles with nanoscale features, e.g., membranes). Like TORCs, they can be naturally occurring (e.g., silica, clay, iron oxyhydroxides), or synthesized by manipulation at the atomic or molecular levels. They are typically synthesized for specific applications, such as larger surface-area-to-mass ratio, increased reaction kinetics, improved disinfection capabilities, special surface coatings to increase sorptive capacity and reactivity, drug delivery, etc. (Trussell et al. 2013). The breadth of their properties is responsible for their widespread use, and as such, their increasing prevalence in water. Like TORCs, nanomaterials represent a wide group of compounds, which means their removal is not straightforward and their presence cannot be directly linked to any specific health effects.

Additional information about these microconstituents is presented in NWRI (2013a) and Trussell et al. (2013). The fate of these compounds upon blending is highly linked to their nature, composition, concentration, and interaction with other chemical constituents.

Microbiological Characteristics

A vital aspect of advancing water reuse for both potable and non-potable applications will be establishing a fundamental understanding of changes to water quality in distribution systems with respect to microbial regrowth and public health. Problems recently identified with microbial regrowth in PWDSs are illustrative, and similar problems may be discovered when recycled water is introduced to existing PWDSs, depending upon the water quality. The major concerns include aesthetics, microbially-induced corrosion (or microbially-induced corrosion, MIC) and opportunistic premise plumbing pathogens (OPPPs). Concerns about the potential spread of antibiotic resistant genes (ARG) in PWDSs are also worthy of investigation.

**General Microbial Issues in Distribution Systems**

PWDSs are now confronting a new generation of challenges, particularly with respect to microbial (re)growth and their colonization by opportunistic pathogens. These issues will be similar for potable reuse scenarios, and the blending process may impact the end results. Besides pathogens, various nuisance microbes such as nitrifying, sulfate-reducing, manganese-reducing and iron-oxidizing bacteria can lead to serious problems, including MIC, biofouling, loss of disinfectant residual, and poor aesthetic quality (CSIRO 2008, Narasimhan et al. 2005, Jjemba et al. 2010a, Jjemba et al. 2010b, AWWA 2011).

Heterotrophic bacteria are the most common type of microorganisms in PWDSs. These bacteria require carbon, nitrogen, and phosphorus nutrients in molar ratio of approximately
100:10:1. If carbon is typically, but not always, the limiting nutrient in natural waters, it is often the limiting nutrient in non-potable recycled water (Narashimhan et al. 2005), and is anticipated to be the limiting nutrient in purified recycled water.

Factors such as secondary disinfectant type and dose, hydraulics (sheer velocity, water age, stagnation), and water temperature are also known to be critical for controlling microbial re-growth and pathogens in PWDSs. Other factors such as pH, trace metals and pipe material can also sometimes have a profound effect, although their qualitative impacts are complex and inconsistent from system to system (Pruden et al. 2013b, van der Kooij and van der Wielen 2013, van der Kooij 2003).

Not all organic compounds are susceptible to microbial decomposition: of the TOC content of a water, the fractions that can be readily used to provide energy for growth can be quantified as BDOC, or assimilable organic carbon (AOC), whereas the fraction that is not amenable to biodegradation is referred to as refractory organic carbon. BDOC and AOC are complementary indicators: BDOC is the portion of the organic carbon that can be mineralized by heterotrophic microorganisms, whereas AOC is the portion of the biodegradable organic carbon that can be converted to cell mass (Huck 1990). These two fractions have been associated with biological regrowth in PWDSs, expressed as increases in coliform counts, HPCs, or biofilm density.

**Presence of Pathogenic, Opportunistic, and Non-Pathogenic Microorganisms**

Recent application of high-throughput DNA sequencing techniques has revealed that even disinfected oligotrophic PWDSs provide a rich habitat for a wide diversity of microbes (Wang et al. 2014, Pinto et al. 2012, Liu et al. 2012, Kwon et al. 2011, Hong et al. 2010). While many of the microbes inhabiting PWDSs may be harmless, opportunistic pathogens, such as *Legionella pneumophila*, *Mycobacterium avium*, *Pseudomonas aeruginosa*, *Acanthamoeba polyphaga*, and *Naegleria fowleri* can also establish as part of the distribution system microbial ecology. Preliminary efforts have demonstrated a similarly complex microbial ecology in non-potable recycled water distribution systems (RWDSs, also known as purple pipe systems) (Narasimhan et al. 2005, Jjemba et al. 2010a). Opportunistic pathogens inhabiting PWDSs are now thought to be the primary source of waterborne disease in developed countries (NRC 2006, Pruden et al. 2013b, Brunkard et al. 2011).

In PWDSs, investigations of the occurrence of *L. pneumophila*, *M. avium*, *P. aeruginosa*, and *Acanthamoeba* have implicated water chemistry, pipe material, and water age as key factors (Wang et al. 2012b, Wang et al. 2013). In particular, recent experiments using a simulated PWDS confirmed that a gradient of redox zones is established as water flows along the length of PVC pipes, beginning with a highly oxidizing zone with high levels of chlorine or chloramine and ending with reducing zones, which included denitrification and sulfate-reduction (Masters et al. 2012, Masters et al. 2015). These redox zones could play a key role in the amplification and/or attenuation of OPPPs or ARGs in water systems, with or without purified recycled water. Potential influencing factors, depending on the level of treatment, may include elevated nutrient and sulfate levels compared to conventional water sources.

While research related to RWDSs clearly represents a beyond worse-case scenario in the context of blending for potable reuse, examining the existing literature on RWDS is nonetheless illustrative. Initial studies have found no correlation between opportunistic pathogen regrowth and levels of AOC in simulated RWDSs, and even the relative impacts of chloramine versus chlorine on control of *Legionella spp.* were opposite to those established in PWDSs (Jjemba et al. 2010a, Jjemba et al. 2010b, Flannery et al. 2006). To our knowledge, there are only two published studies
(Fahrenfeld et al. 2013, Storey and Kaucer 2009) that track Legionella through RWDSs (WE&RF Project 12-05 titled Management of Legionella in Water Reclamation Systems, had not been published when this literature review was prepared), while other opportunistic pathogens have rarely been considered (Storey and Kaucer 2009). Transport of water through distribution systems can either attenuate or amplify risks for opportunistic pathogens via regrowth, predation, and/or disinfection as controlled by the interplay of microbial ecology and water chemistry.

Although these complex interdependencies have only been quantified in a few studies of RWDSs at bench and full scale, samples at different locations of a given RWDS on the same day commonly varied by more than 5-log for HPC, Legionella spp., and other pathogens due to regrowth (Storey and Kaucer 2009). In a short-term bench study of a simulated distribution system, Jembla (Jjemba et al. 2010b) also demonstrated 5-log increases in HPC, Legionella spp. and other pathogens after just 10 minutes of flow through PVC tubes with high residual chloramine (> 0.65-7 mg/L Cl₂).

**Antibiotic Resistant Microorganisms**

Antibiotic resistance is a growing public health concern and it has been argued that, at the current pace, antibiotics may soon cease to be functional for fighting and preventing deadly infections (Smith and Coast 2013, Carlet et al. 2011). There is growing attention on the potential for water reuse to contribute to this problem, including ARG persistence or even amplification through wastewater treatment (Rizzo et al. 2013, Zhang et al. 2009a, Schlüter et al. 2007). Few studies have directly examined the potential for recycled water to contribute to the spread of antibiotic resistance (Hong et al. 2013), and one recent study observed likely regrowth of several ARGs in two U.S. RWDSs (Fahrenfeld et al. 2013).

Results of other studies have traced ARG abundance as recycled water flows through other natural and engineered systems; such as aquifers (Böckelmann et al. 2009), soils (Knapp et al. 2011), constructed wetlands (Nõlvak et al. 2013, Anderson et al. 2013, Sidrach-Cardona and Bécares 2013), and aerobic and anaerobic digesters (Burch et al. 2013, Ma et al. 2011). The overall abundance of ARGs tends to decrease as water passes through such systems, but there have also been instances in which ARGs amplified by > 3-log (Knapp et al. 2011, Nõlvak et al. 2013, Burch et al. 2013, Ma et al. 2011).

In several studies ARG amplification has been linked to redox conditions, presence of specific ionic constituents, such as copper, availability of nutrients, and water temperature (Knapp et al. 2011, Nõlvak et al. 2013, Ma et al. 2011, Hoffman et al. 2010, Diehl and LaPara 2010). Proposed mechanisms of ARG attenuation under anoxic conditions include a change in which members of the microbial community are the dominant bacteria carrying ARGs (Ma et al. 2011, Diehl and LaPara 2010, Pei et al. 2007) and efforts of bacteria to conserve metabolism and thus rid themselves of fitness costs associated with carrying ARGs (Rysz et al. 2013).

Information on ARG removal through advanced treatment is limited, but processes that destroy or remove dissolved organics (e.g., advanced oxidation processes) would be expected to be effective in ARG control.
Corrosiveness and Aggressiveness

Corrosion is an electrochemical interaction between a metal surface (e.g., a pipe wall) and water. During this process, metal is oxidized and transferred to the water, or to another location on the metal surface. There are many forms of corrosion depending on the material, but the most important forms for drinking water are the following:

- Uniform corrosion, where the electrochemical interaction occurs along the inner pipe wall, resulting in a relatively uniform loss of metal across the surface.
- Non-uniform corrosion, where metal is lost from a localized point, causing pitting and mounding in some cases.
- Galvanic corrosion which comes from a coupling of dissimilar metals or internally in metallic alloys.

Biofilms and their extracellular polymeric substances (EPS) can create a discrepancy at the pipe surface, i.e., a disequilibrium that can trigger corrosion, a process referred to as microbiologically-induced corrosion (MIC). MIC contributes to about half of all water pipeline failures (Uhlig and Revie 1985, De Araujo-Jorge et al. 1992) which translates to $4 to $18 billion in annual costs for water systems (EPA 2002) and even higher costs are likely borne by building owners (NRC 2006, Edwards 2004). MIC is discussed in detail below.

While it is important to understand and control corrosion, metal release into the water is the process that drives drinking water regulations, and that may present the greatest risks to public health. Metal release is a function of the reactions that occur between the metal ions that are released, and the physical, chemical, and biological characteristics of the water and of the metal surface.

Pipe scales that build up on the metal surface are also important. They can include two types of compounds: (1) passivating films that form when pipe material and water react directly with each other; and (2) deposited scale material that forms when substances in the water (e.g., iron, manganese, aluminum, calcium) precipitate or sorb to, and then build up on the pipe surface. Scales can have several layers; they are influenced by treatment history, and they, too, can influence the effectiveness of subsequent corrosion control treatment process.

The characteristics of the scales and their structure dictate the amount of metal that can be released into the water during normal conditions and following physical disturbances (e.g., flushing, infrastructure work). If conditions favor the formation of insoluble, adherent scale to the inner pipe wall, then the rate of metal release will be low. However, if scales do not adhere well to the pipe wall or if they are very soluble, then metal release may be greater. Other compounds in the water (including aluminum, iron, manganese, and calcium) can significantly influence scale formation and properties.

Erosion of pipe internal surfaces, linings and concrete reservoirs and tanks is a phenomenon that differs from metal corrosion. It derives from aggressive waters and mainly affects cement-mortar lined pipes, asbestos-cement pipes and cement-based storage reservoirs and tanks, which are composed of various calcium silicates and calcium aluminates that can dissolve in aggressive waters. This process can be responsible for the presence of sediments in the distribution system. It can often be prevented or limited by the formation of a thin layer of calcium carbonate (CaCO₃) on the inner pipe surface.
Factors Affecting Corrosion and Aggressiveness

Corrosion and aggressiveness are influenced by a myriad of water quality parameters, sometimes in conflicting ways. As such, they should not be examined individually but rather altogether. Also, the outcomes may vary from one material to another under similar water quality characteristics; i.e., water that may be passivating for one material may be corrosive for another. The parameters that influence corrosion and aggressiveness, and that are most relevant to purified recycled waters are listed here (Vik et al. 1996, Schock and Lytle 2011, Taylor et al. 2005):

- Water temperature influences all chemical and biological reactions. High temperature accelerates corrosion rates, but also increases the tendency of CaCO₃ to precipitate, thereby helping the formation of a protective layer on distribution system materials.
- pH: Low pH may increase corrosion rates of metals, whereas high pH tends to protect pipes and decrease corrosion rates. High pH can also be responsible for dezincification of brasses and increased copper release. pH also influences the effectiveness of corrosion control treatment.
- Alkalinity and dissolved inorganic carbon (DIC). While higher alkalinity waters tend to decrease corrosion rates, help form a protective film of CaCO₃, and control pH changes, it may increase corrosion of copper, lead, and zinc.
- Oxidants such as chlorine and chloramines. On one hand, oxidants, by impacting the ORP, may contribute to corrosion of metals such as copper, iron, and steel, and result in metal release. On the other hand, higher disinfectant concentrations may offer a better control of MIC. Because monochloramine is a less reactive oxidant than chlorine (i.e., it has a lower ORP), it does not affect materials and does not decrease alkalinity as much as free chlorine when used at similar concentrations. Wang et al. (2012a) showed how free chlorine and chloramine generated different corrosion scales and biofilm communities in a simulated recycled water distribution system. Potential synergistic effects of disinfectant combinations can also alter the corrosion process: Zhu et al. (2014) showed how UV and free chlorine can inhibit corrosion when compared with chlorine alone.
- Oxidation-reduction potential (ORP): Under certain conditions, ORP can have a significant impact on lead release. In the absence of a corrosion inhibitor or other interfering surface deposits, high-ORP waters could promote the formation of lead scales on pipe surfaces, thereby preventing lead from being released into the water. Conversely, a decrease in ORP may contribute to lead release.
- Dissolved oxygen (DO) often serves as electron acceptor (cathode) in corrosion reactions, thereby allowing the corrosion reactions to continue. High DO concentrations can also increase copper corrosion. In certain circumstances, however, high DO concentrations may be beneficial by facilitating the production of protective mineral oxide layers; this is the case for lead for example.
- Hardness and calcium concentration may favor the precipitation of calcium as CaCO₃, thereby providing protection inside distribution system structures. High calcium concentrations stabilize the calcium present in cement-mortar lined pipes, asbestos-cement pipes and concrete reservoirs and tanks, thereby preventing its release, and can reduce corrosion rates in unlined cast iron pipes. However, the link between calcium...
hardness and metal corrosion is not straightforward. One of the drawbacks of high calcium hardness is that it may cause excessive scaling or even turbidity.

- Total dissolved solids (TDS) and conductivity tend to increase corrosion rates by completing the electrochemical circuit responsible for corrosion reactions. The nature of the ions that compose TDS can also affect corrosion: for example, if bicarbonate and calcium ions are major TDS contributors, as opposed to chloride and sulfide ions.
- Chloride and sulfate may promote corrosion in metallic pipes by reacting with the metals in solution and causing them to stay in solution, or interfering with the formation of normal protective oxides and films. In this regard, chloride is typically three times more active than sulfate. Although the role of sulfate in iron corrosion should not be underestimated (Lytle et al. 2005), it tends to inhibit lead corrosion (Oliphant 1983). Pitting corrosion in copper piping has been associated with chloride and bicarbonate concentrations.
- Hydrogen sulfide (H₂S), may increase corrosion rates by reacting with metal ions to form non-protective insoluble sulfides.
- Ammonia forms soluble complexes with many metals, including copper and lead, thereby interfering with the formation of passivating films and increasing corrosion rates.
- Organic material may decrease corrosion rates by coating pipe surfaces, but some organics can complex metals (such as lead) and accelerate corrosion or metal release. Organics can also complex calcium ions, keeping them from forming a protective film of CaCO₃. Lastly, organic material can serve as food source for microorganisms, thereby potentially increasing biofilm density and microbiologically-induced corrosion (MIC).
- MIC is discussed later in this literature review.
- Corrosion and aggressiveness are highly dependent on hydraulic conditions, such as water velocity and water usage. High water velocity can reduce lead and copper release by transporting the corrosion inhibitor to the pipe surfaces at a higher rate. However, lead and copper corrosion can accelerate in some cases by increasing the rate at which the oxidants in water come into contact with the metal surface. High water velocity can mobilize loosely adherent scales and cause sporadic metal release. Conversely, low water velocity can decrease the effectiveness of the corrosion control inhibitor in forming a passivating scale. Increased water age can cause water quality changes that can increase corrosion and microbial growth. These phenomena can be exacerbated by flow reversals and hydraulic pressure transients, as well as physical disturbances such as infrastructure work (e.g., line repair or replacement, meter installation or replacement, valve shutoff, and pipe flushing).

Finished water quality goals may be set for each of the parameters described above to limit corrosion in the distribution system. When introducing a new source water or when blending waters, significant water quality changes in the distribution system should be avoided, and the quality of the “new” water should match that of the existing water as much as possible. Highly purified recycled water may exhibit low pH, TDS, hardness, and alkalinity compared to natural (untreated) water. Also, introducing RO/NF permeate may adversely impact existing treatment processes (i.e., coagulation, biofiltration). To limit corrosion, RO/NF permeates may require stabilization (i.e., augmentation of hardness and alkalinity with chemicals such as lime, CaCO₃ or
calcium hydroxide) prior to blending with raw water, unless it is blended with a water supply that is high in TDS, hardness and/or alkalinity, in which case blending of purified recycled and potable water may be beneficial (Duranceau et al. 2011 and Taylor et al. 2005). For surface waters with high chloride and/or sulfate concentrations, blending with RO treated purified recycled water may also decrease the corrosiveness of the finished water.

**Corrosion and Aggression Indices**

Several indices have been developed to quantify a water’s tendency to be corrosive or aggressive towards distribution system materials. Each index is calculated from a number of water quality parameters listed earlier, thereby providing a more thorough assessment than evaluations that are based on individual water quality parameters alone.

Many indices are based on corrosion control through CaCO₃ saturation, recognizing that a slight over-saturation of CaCO₃ may promote its precipitation resulting in deposition of a thin protective film inside distribution system pipes. The common indices that have been developed based on this principle include the Langelier Saturation Index (LSI), the Ryznar Saturation Index (RI), the Aggressiveness Index (AI), and the Calcium Carbonate Precipitation Potential (CCPP) (Langelier 1936, Schock and Lytle 2011, Snoeyink and Wagner 1996). Because CaCO₃ saturation is not directly linked to corrosion of metallic pipes and metal release, other indices were developed based on the role of anions (mainly chloride, sulfate) in promoting metal corrosion. These indices are better suited to provide indication of lead and copper release than the aggressiveness indices. They include the Larson Ratio (Larson and Skold 1957) and the Chloride to Sulfate Mass Ratio (CSMR; Carlson et al. 2000, Edwards et al. 1996, Edwards and Triantafyllidou 2007).

These indices, and particularly those that pertain to CaCO₃ precipitation, present several limitations that should not be ignored. The main limitations are listed here:

- The applicability and interpretation of an index is limited by the conditions in which it was developed. For example, the aggressiveness indicators (i.e., LSI, CCPP, RSI and AI) should not be used to evaluate lead or copper corrosion control. Also, the selection of the best corrosion index is typically site specific: there is no “one size fits all” index or parameter with regards to corrosion control. Lastly, the CSMR has been proposed fairly recently; thus, this index should be used cautiously considering the limited information that is available at this point.

- In poorly buffered (i.e., low-alkalinity and low-DIC) waters, the presence of localized pH changes may lead to significantly different corrosion and aggressiveness indices, thereby creating inconsistent corrosion reactions that can exacerbate the problem (Schock and Lytle 2011).

- Various models and calculators (often spreadsheets) are available to estimate the aggressiveness indices. They may provide different results depending on their level of accuracy.

- The significance of these indices in the presence of a corrosion inhibitor is questionable (Schock and Lytle 2011).

- The indices do not consider MIC, which may occur if the disinfectant residual is unstable and microbial (re)growth occurs.
Thus, corrosion and aggressiveness indices must be used carefully as there are many waters with non-corrosive or non-aggressive indices that are corrosive/aggressive, and vice versa (Singley 1981). Bench- or pilot-scale testing is important to confirm the trends suggested by the indices, and develop appropriate control strategies.

**Corrosion Control**

The selection of a corrosion control strategy needs to consider water quality, distribution system materials, and hydraulic conditions. As stated by Peet et al. (2001), the use of a corrosion control strategy may well override any impacts from blending, as long as care is taken to avoid diluting the control strategy by blending “treated” and “untreated” waters (“(un)treated” water here refers to water that has not received an optimized corrosion control treatment). If corrosion control is deemed necessary, the general approaches that can be used are the following: alkalinity and pH adjustment (which affects DIC concentration), and the use of corrosion inhibitors to form less soluble metal compounds (e.g., carbonates, silicates, or phosphates) that adhere to the inner pipe walls. Calcium hardness adjustment to increase CaCO₃ precipitation is often practiced to protect cement-based pipes, storage reservoirs and tanks, but it does not directly affect corrosion of lead, copper, and galvanized pipes.

**Alkalinity and pH Adjustment.** Corrosion inhibition by adjustment of alkalinity, pH, and DIC concentration can induce the formation of less solution compounds on the inner surface of metal pipes in a process called *passivation*. This can be performed by adding chemical compounds such as lime, soda ash, sodium bicarbonate, caustic soda, and carbon dioxide. Limestone contactors have also been used in small systems. When carbon dioxide concentrations are sufficient and pH is low, aeration has also shown sufficient reduction in DIC concentration to increase pH.

**Phosphate- and Silicate-Based Agents.** Corrosion inhibitors “*passify*” metals by preventing reactions at the anode (when anodic inhibitors are used) or the cathode (with cathodic inhibitors). The best corrosion inhibitors are those that provide both types of inhibition. Ortho- and poly-phosphates, zinc salts, and sodium silicates are the main chemicals used for corrosion inhibition by passivation.

Orthophosphates react with divalent metals (e.g., lead, copper, and iron), to form insoluble protective films, and are therefore recognized as effective agents to reduce metal release. Polyphosphates are not effective corrosion inhibitors. Their primary utilization is to reduce tuberculation of iron and steel, and sequester iron and manganese by binding the metals into their structures to prevent their precipitation. They can also sequester lead and copper, keeping them in the water, and thereby increasing concentrations measured at customer taps and the risk of exposure. Thus, polyphosphates are dissolved in the water rather than creating a film on pipe surfaces like orthophosphates. Zinc phosphates are bimetallic phosphates that generally combine 10 to 30 percent zinc with ortho- or poly-phosphates. Zinc is a cathodic inhibitor that prevents the contact between oxidizing agents (such as oxygen or chlorine) and the metal by precipitating at the cathode, thereby forming a non-conductive barrier. Zinc-based inhibitors are typically used to prevent leaching of calcium from cement mortar lined pipes, asbestos-cement pipes and cement-based reservoirs and tanks, and protect metal pipes by forming protective films. Phosphate blends are popular because they can offer multiple protections, e.g., corrosion protection, hardness stabilization, and red-water suppression. Silicate-based corrosion inhibitors are mixtures of soda ash and silicon dioxide. Although these compounds are not commonly used, they are efficient
anodic inhibitors that can sequester metal ions such as iron and manganese, and limit red-water occurrences by forming protective films.

**Calcium Hardness Adjustment.** Precipitation of a thin layer of CaCO₃ has been used to protect cement-based pipes, reservoirs, and tanks, but it does not affect corrosion of lead, copper, and galvanized pipes. If sufficient calcium is present in the water, increasing pH, alkalinity and/or DIC may be sufficient to favor CaCO₃ precipitation. Alternatively, a number of compounds can be used to increase calcium concentration, including slaked lime (i.e., calcium hydroxide, Ca(OH)₂), quicklime (i.e., calcium oxide, CaO), and CaCO₃. Hydraulic conditions that may allow or prevent the deposition of a protective layer of CaCO₃ are also important, and frequent changes in flow direction and velocities may limit the formation of a protective layer as films are constantly deposited and dissolved by the changing water flow and quality. An excessive calcium concentration is not desirable because it may limit a water system’s ability to raise pH (which limits metal corrosion) due to potential scaling problems.

**Microbiologically-Induced Corrosion**

As mentioned earlier, biofilms and their EPS can be responsible for microbiologically-induced corrosion (MIC). The process has been associated with rapid loss of disinfectant residual, contamination of potable water with high levels of copper, lead and iron (McNeill and Edwards 2001, Edwards et al. 2000, Zhang et al. 2009b), microbial corrosion failures (Videla 1996), and aesthetic problems (taste and odor; T&O) in potable water (NRC 2006). However, a number of studies have shown the opposite, i.e., decreases in corrosion resulting from interactions with microorganisms and biofilms (Teng et al. 2008, Wang et al. 2012a, Zhang et al. 2010b, Zhu et al. 2014), which highlights the complexity of the MIC phenomenon.

Copper pipes are particularly vulnerable to bacterial attacks, where bacteria, including heterotrophic bacteria commonly found in drinking water distribution systems, may enhance copper solvation (cuprosolvency). Increases in cuprosolvency have been correlated with water pH, LSI, chloride, alkalinity, TOC, and soluble phosphate concentrations, and are related to the microbial species that are present (Critchley et al. 2003).

Sutherland et al. (2009) observed the formation of zinc sulfide microcrystals from a galvanized pipeline used to transport raw brackish groundwater. The proposed pathway included the reduction of sulfate from the groundwater to sulfide, dissolution of zinc from the pipe wall, and the formation of microcrystals, all mediated by sulfate-reducing bacteria colonizing the pipe wall. After periods of stagnation (greater than 24 hours) in the pipe, turbidity spikes were measured as high as 85 NTU and appeared “milky” when flow was restarted. Approximately 80 percent of the particles counted were less than 5 µm in diameter.

**Aesthetic Characteristics**

This section focuses on aesthetic characteristics of purified recycled water and blends of purified recycled water with other water sources. The importance of potable water aesthetic characteristics (i.e., taste, odor, color, and mineral composition) cannot be understated because it is directly linked to customers’ acceptance of a water supply, and are the main cause of customer complaints. However, aesthetic characteristics are typically not related to public health, and thus, do not have primary regulatory standards, although some taste and odor causing compounds have secondary standards that may be enforced in certain states (WRF 2014). Nonetheless, many
consumers may relate the aesthetics of the water to its safety, making these characteristics even more important from a DPR perspective.

A national survey conducted by Mackey and colleagues (2003) showed that safety and healthiness were the main drivers for using an alternative to tap water (for 68 percent of the customers that filter their tap water, and 56 percent of the customers that prefer to use bottled water), along with taste (for 26 and 30 percent of the customers that filter their water or use bottled water, respectively). A complementary study conducted by the same group (Mackey et al. 2004) added that taste, including chlorinous flavors, was often associated with a perception of tap water unhealthiness and lack of safety. A subsequent study that analyzed survey results obtained in the USA, Canada, and France led to the same observations (Doria 2006).

Tastes and odors in water can come from numerous constituents, including both chemical and microbiological parameters. These parameters can come from the source water and pass through the treatment processes without being removed (e.g., MIB, geosmin), can be formed during treatment (e.g., disinfectants, DBPs), or can be generated during water distribution (e.g., alkyl benzenes resulting from inadequately cured coatings) (Khiari et al. 2002, WRF 2014). Khiari and colleagues (2002) highlighted the complexity of distribution generated taste and odor compounds and noted that “Every material that is in contact with water has some potential to affect the flavor of the water, given the right conditions.”

To optimize the acceptance of purified recycled water for drinking water applications, Trussell et al. (2013) have proposed two aesthetic goals:

1. “The recycled water must be free of wastewater properties that are obvious to uninformed customers.” To meet this goal, the targets stated in the secondary standard regulations were proposed, i.e., color (≤15 apparent color units, ACU), threshold odor number ≤3 and no off flavor, and mineral content (TDS and hardness must be similar to local water supplies).
2. “The recycled water must be free of wastewater properties that are evident to informed customers.” While not aesthetic goals, Trussell et al. (2013) proposed two indicators:
   a. The recycled water must be free of dissolved organic material (DOM) of wastewater origin. A TOC concentration of ≤0.5 mg/L C, or 90 percent EEM total fluorescence reduction through wastewater treatment trains, were proposed as indicators of EfOM removal.
   b. Trace level pollutants should be eliminated to acceptable levels, using performance- and health-based chemical indicators.

**Interactions between Taste- and Odor-Causing Compounds**

When blending water from multiple sources, the resulting blends can lead to various tastes and odors. Background water quality influences taste and odor threshold levels, with both synergistic and antagonistic effects. The literature often reports cases where chlorinous tastes may mask other tastes. Although results were not statistically significant, Mackey and colleagues (2004) observed that high TDS concentrations may decrease the sensitivity to chlorinous flavors of both trained and untrained panelists. The presence of earthy and musty tastes also showed this antagonistic effect (e.g., Khiari et al. 2002, Oestman et al. 2004).

Khiari and colleagues (2002) examined the effect of blending treated water of different origins and qualities. Among others, results show that differences in odors tend to appear before differences in flavors, and that the introduction of a small proportion of one water with certain
taste and odor causing compounds (e.g., medicinal flavors) may be enough to significantly change
the flavor profile of the resulting blend. Likewise, a high proportion of a non-flavorful water may
be necessary to mask certain tastes and odors such as chlorine.

The fact that certain taste and odor causing compounds can be formed during treatment
highlights the importance of a careful selection of treatment strategies for purified recycled water,
in light of the characteristics of the potable water with which it will be blended. Also, the masking
effect that certain taste and odor causing compounds have on others suggests that highly treated
recycled water (by RO or NF for example) may cause aesthetic challenges because of its low
hardness, alkalinity, mineral and salt content. This is where blending of highly treated recycled
water with potable water may offer an advantage, by averaging the mineral and salt content. It
should be noted that managing aesthetic characteristics is one reason that justifies blending, as
opposed to dilution of contaminants that present health concerns, which should not be used for
public health reasons (Tchobanoglous et al. 2011).

Another point in consideration is the fact that after its introduction into a PWDS, the
blended water may cause taste and odor challenges that would not occur with the individual water
sources. In addition, differences in tastes and odors are more noticeable at higher water
temperatures, which may be concerning from a DPR perspective because water suppliers that are
likely to consider DPR are those that are affected by drought conditions, and are therefore located
in the warmer areas of the USA. Ultimately, the aesthetic quality of the distributed water must be
maintained to ensure public acceptance of the blended DPR water.

Lastly, Khiari and colleagues (2002) reinforced the need for bench- or pilot-scale testing
to assess the true outcome of blending water from different sources or qualities on tastes and odors,
considering that most of the taste and odor causing compounds are non-conservative parameters
that cannot be evaluated by the Law of Mixtures.

**Distribution System Characteristics**

In addition to chemical, microbial, and aesthetic characteristics, other factors also influence
distribution system water quality regardless of its type (PWDS or RWDS). A literature review
conducted by Narashimhan et al. (2005) summarized these factors as follows:

- Although most RWDSs are relatively new, the age of the distribution system influences
  bacterial counts and regrowth.
- Pipe and distribution system materials affect microbial counts and biofilm density, as
  well as disinfectant stability and DBP formation. Certain materials are also more prone
to corrosion, and accumulation and release of inorganic chemicals as discussed earlier.
- Hydraulic conditions, including flow velocity and system flushing, affect water age,
  and thereby the fate of the non-conservative parameters.

**BLENDING LOCATIONS**

Water quality resulting from blending is largely based on the quality of the different water
sources. The location of blending is also important and influences both water quality (as detailed
below), and which laws and regulations apply (Tchobanoglous et al. 2011).

Because blending waters of different qualities may locally disrupt the natural “ecology” of
the receiving environment, blending in a storage reservoir or within a potable WTP is generally
more predictable. It may also eliminate the negative effects that can be observed when blending
occurs in the PWDS, which may lead to biofilm detachment, or disruption of the corrosion scales if the intruded water is more aggressive or corrosive. The negative effects of blending in PWDS pipes can be exacerbated by variations in hydraulic conditions and water demands, which may change the exact blending location in the PWDS. This was confirmed by Peet et al. (2001) who observed that contact time between the water and the pipe was more important than changes in water chemistry when evaluating the effect of blending on the corrosiveness/aggressiveness of the resulting water in distribution system pipes.

Although no regulations are in place, it can be anticipated that the “point of compliance” for DPR water quality criteria will vary by jurisdiction. For example, California will likely require DPR water quality criteria to be met before the point of blending with raw water. Considering that the point of introduction of purified recycled water into a potable water system (whether upstream of a potable WTP, in the WTP, or after treatment) influences the purified recycled water treatment strategies, meeting these criteria may require excessive treatment of the wastewater if blending is performed upstream of the potable WTP. Other states may consider the entire treatment envelope, including conventional WTPs, as part of the treatment process for DPR, as Texas does now, which alters the parameters influencing the selection of an optimal blending location. Regardless of regulatory paradigm, aesthetic criteria must be met immediately prior to the first water customer.

Perhaps another point to consider is simply the location of the advanced purification facility with respect to the potable WTP. Based on the treatment processes used at the advanced WTP, a number of challenges may be encountered in the RWDS, including rapid decay of disinfectant residual, microbial growth, odors, pipe corrosion, etc. Finally, the blending location will most likely be dictated by geographic factors, such as the locations of existing treatment facilities and the availability of space to construct advanced treatment facilities; with water quality considerations, including blending considerations, addressed to accommodate these constraints.

**Blending Upstream of a Drinking Water Treatment Plant**

The water quality effects of blending purified recycled water with untreated water are summarized in Table 2.2. The wastewater treatment strategies that are examined as part of this project were used to illustrate the complexity of the resulting blends and potential effects on water quality. Typical surface waters and groundwaters as observed in the USA were assumed, and a 50/50 blend of purified recycled and raw water was considered. The scenario where secondary effluents would be blended with untreated surface or groundwater was not examined considering that this situation would most likely be unacceptable because it would increase the concentrations and composition of most parameters examined here.

Another point to consider when blending purified recycled water upstream of a potable WTP is the fact that the resulting water quality may have a positive or a negative impact on the existing treatment processes. For example, blending purified recycled water pretreated with RO upstream of an existing conventional surface WTP may adversely affect the existing coagulation process by 1) reducing the alkalinity to a point where additional alkalinity must be supplemented to promote coagulation, or 2) decreasing the turbidity, which minimizes particle removal, thereby reducing the pathogen removal “credit” often granted to conventional treatment.

Table 2.2 illustrates the complexity of blending different water qualities and the various factors that need to be considered, which reinforces the importance of conducting pilot-scale testing before introducing blended water in a full-scale distribution system.
<table>
<thead>
<tr>
<th>Issue</th>
<th>UF/RO/UV AOP blended with surface water</th>
<th>UF/RO/UV AOP blended with groundwater</th>
<th>O₃/BAF blended with surface water</th>
<th>O₃/BAF blended with groundwater</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pathogens</td>
<td>Purified recycled water reduces overall pathogen load</td>
<td>Both water supplies low in pathogen concentrations</td>
<td>To be determined, as raw water may have more or less pathogens then the O₃/BAF.</td>
<td>Groundwater expected to have less pathogens then the O₃/BAF.</td>
</tr>
<tr>
<td>Organic material and nutrients</td>
<td>The contribution of the purified recycled water is likely to decrease the organic content of the resulting blend</td>
<td>The resulting blend is likely to contain very little organic content</td>
<td>Depending on the efficiency of the wastewater treatment process, the purified recycled water may increase the organic content of the resulting blend</td>
<td>Depending on the efficiency of the wastewater treatment process, the purified recycled water is likely to increase the organic content of the resulting blend</td>
</tr>
<tr>
<td>Trace level pollutants</td>
<td>The purified recycled water will reduce the concentration and composition of trace level pollutants in the resulting blend</td>
<td>Both water supplies low in pollutant concentrations, though legacy groundwater pollutants may be problematic</td>
<td>The purified recycled water may increase the concentration and composition of trace pollutants in the resulting blend</td>
<td>The purified recycled water may increase the concentration and composition of trace pollutants in the resulting blend</td>
</tr>
</tbody>
</table>

(continued)
### Table 2.2 Continued

<table>
<thead>
<tr>
<th>Issue</th>
<th>UF/RO/UV AOP blended with surface water</th>
<th>UF/RO/UV AOP blended with groundwater</th>
<th>O₃/BAF blended with surface water</th>
<th>O₃/BAF blended with groundwater</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Disinfectant stability and DBPs</strong></td>
<td>The purified recycled water is likely to stabilize the disinfectant residual and decrease DBP formation, but a greater proportion of brominated and iodinated DBPs may be found in the resulting blend</td>
<td>The resulting blend is likely to have a very stable disinfectant residual and lower DBP concentrations, but a greater proportion of brominated and iodinated DBPs may be found</td>
<td>Depending on the efficiency of the advanced treatment process, disinfectants may be unstable and DBPs may form in greater concentrations and different composition; the concentrations of brominated, iodinated, and nitrogenous DBPs is likely to increase in the resulting blend</td>
<td>Depending on the efficiency of the advanced treatment process, disinfectants are likely to be less stable and DBPs are likely to be formed in greater concentrations and different composition; the concentrations of brominated, iodinated, and nitrogenous DBPs is likely to increase in the resulting blend</td>
</tr>
<tr>
<td><strong>Corrosion and chemical stability</strong></td>
<td>The purified recycled water is likely to decrease the corrosiveness and aggressiveness of the resulting blend by averaging pH, TDS, hardness, alkalinity, chloride, and sulfate; but it may also affect the existing corrosion scales, which could lead to increases or decreases in corrosion rates, and metal releases and/or precipitation</td>
<td>The corrosiveness and aggressiveness of the resulting blend is highly dependent on the groundwater quality; it is possible that the resulting blend will be more corrosive/aggressive</td>
<td>Blending with purified recycled water is likely to increase the corrosiveness and aggressiveness of the SW, due to greater TDS concentrations</td>
<td>Blending with purified recycled water is likely to increase the corrosiveness and aggressiveness of the groundwater (GW), due to greater TDS concentrations</td>
</tr>
</tbody>
</table>

(continued)
<table>
<thead>
<tr>
<th>Issue</th>
<th>UF/RO/UV AOP blended with surface water</th>
<th>UF/RO/UV AOP blended with groundwater</th>
<th>O₃/BAF blended with surface water</th>
<th>O₃/BAF blended with groundwater</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aesthetics</td>
<td>Blending is likely to improve the aesthetic characteristics and mineral content of the purified recycled water</td>
<td>Blending is likely to improve the aesthetic characteristics and mineral content of the purified recycled water</td>
<td>The aesthetic characteristics of the resulting blend are largely dependent on the relative qualities of the water sources.</td>
<td>The aesthetic characteristics of the resulting blend are largely dependent on the relative qualities of the water sources.</td>
</tr>
</tbody>
</table>

*When assessing the water quality resulting from blending, mass balance calculations and the Law of Mixture may apply for some of the parameters responsible for corrosion and chemical stability. However, the complexity of the corrosion phenomenon warrants that each water blend be examined individually.*
Blending within a Drinking Water Treatment Plant

The level of advanced treatment provided to the wastewater effluent is very likely to determine at what point it can be introduced in a potable WTP. Starting from Table 2.2, wastewater treated by UF/RO/UV AOP could potentially be introduced towards the end of the potable water treatment process, before final disinfection and corrosion control. The corrosion control would need to be adapted to the new water blend.

It is likely that wastewater treated by O3/BAF would need to be introduced at the beginning of the potable WTP to remove any remaining organic materials, improve disinfectant stability, and/or decrease DBP formation potential. As mentioned earlier, the potential presence of remaining trace pollutants deserves further attention, and the corrosion control strategy should be adapted to the new water blend.

Even for conservative parameters, the resulting water quality upon blending may be surprising, as found by Sutherland et al. (2009), who observed the formation of zinc and sulfide nanocrystals on membranes used to treat brackish groundwater. A similar situation may occur if purified recycled water were to be blended in a potable WTP, and RO or NF membranes were to be used.

Blending into the Potable Water Distribution System

The effects of blending purified recycled water with treated drinking water directly in the PWDS are summarized in Table 2.3, using only the RO-based processes. In this example, it was assumed that a conventional treatment scheme is used to treat surface water, i.e., coagulation, flocculation, sedimentation, granular media filtration, and disinfection.

Blending of waters with different qualities has been studied in potable water systems, particularly with regards to corrosion and aggressiveness. Peet et al. (2001) showed how a careful selection of the blend ratio at the bench scale can decrease the aggressiveness of a water and render it non-corrosive, although pipe loop experiments showed little change in corrosion (particularly when older pipes were used), even with significant changes in water quality resulting from blending. In a full-scale PWDS, routine blending will force the onset of a new equilibrium, whereas temporary switches may bring undesirable disturbances in water quality, which could take various forms. As such, Peet et al. (2001) concluded that contact time between the water and the pipe was more important than changes in water chemistry when evaluating the effect of blending in distribution system pipes, and recommended that desktop, bench- and/or pilot-scale studies be undertaken to evaluate the potential changes that may be encountered.
### Table 2.3
Expected water quality resulting from blending purified recycled water with treated drinking water directly in a potable water distribution system

<table>
<thead>
<tr>
<th></th>
<th>UF/RO/UV AOP blended with surface water</th>
<th>UF/RO/UV AOP blended with groundwater</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Pathogens</strong></td>
<td>The contribution of the purified recycled water <em>must</em> decrease the microbial concentration and composition of the resulting blend</td>
<td>The contribution of the purified recycled water <em>must</em> decrease the microbial concentration and composition of the resulting blend</td>
</tr>
<tr>
<td><strong>Organic material and nutrients</strong></td>
<td>The purified recycled water will decrease the organic content of the resulting blend</td>
<td>The resulting blend is likely to contain very little organic content</td>
</tr>
<tr>
<td><strong>Microconstituents</strong></td>
<td>The purified recycled water will reduce the concentration and composition of microconstituents in the resulting blend</td>
<td>The resulting blend is likely to contain very little trace level pollutants, unless there are legacy pollutants in the groundwater</td>
</tr>
<tr>
<td><strong>Disinfectant stability and DBPs</strong></td>
<td>The purified recycled water is likely to stabilize the disinfectant residual and decrease DBP formation, but a greater proportion of brominated and iodinated DBPs is likely to be found in the resulting blend</td>
<td>The resulting blend is likely to have a very stable disinfectant residual and lower DBPs concentrations, but a greater proportion of brominated and iodinated DBPs may be found</td>
</tr>
<tr>
<td><strong>Corrosion and chemical stability</strong></td>
<td>The purified recycled water is likely to decrease the corrosiveness and aggressiveness of the resulting blend by averaging pH, TDS, hardness, alkalinity, chloride, and sulfate; but it may affect the existing corrosion scales, which could lead to increases or decreases in corrosion rates, and metal releases and/or precipitation</td>
<td>The corrosiveness and aggressiveness of the resulting blend is highly dependent on the GW quality; the existing corrosion scales could also be affected, leading to increases or decreases in corrosion rates, and metal releases and/or precipitation</td>
</tr>
<tr>
<td><strong>Aesthetics</strong></td>
<td>Blending is likely to improve the aesthetic characteristics and mineral content of the purified recycled water</td>
<td>Blending will improve the aesthetic characteristics and mineral content of the purified recycled water</td>
</tr>
</tbody>
</table>

*When assessing the water quality resulting from blending, mass balance calculations and the Law of Mixture may apply for some of the parameters responsible for corrosion and chemical stability. However, the complexity of the corrosion phenomenon warrants that each water blend be examined individually.*
A pilot distribution system study involving various blends of treated surface water, groundwater and desalinated seawater showed that iron and copper corrosion and metal release from aged pipes varied with blended water qualities and pipe materials (Tang et al. 2006a, Taylor et al. 2005). Iron concentrations increased when water with a different quality was introduced in the simulated distribution system. pH and lower alkalinity appeared to be the main factors responsible for increased iron release, along with increases in chloride, sulfate, sodium, dissolved oxygen, and water temperature. Pipe surface characteristics also affected iron release, with unlined cast iron leading to the highest iron concentrations, followed by galvanized steel, then lined ductile iron, and finally PVC. Regarding copper release, high alkalinity and low pH were the water quality parameters responsible for increased releases, along with increases in sulfate and temperature, and decreases in silica concentration. Lead release was also driven by decreases in pH and increases in alkalinity, with sulfate (decreases) and chloride (increases) concentrations playing a minor role. Calcium, dissolved oxygen, and chlorine residual did not correlate with copper or lead release (Taylor et al. 2005). Results showed higher releases of iron, copper, and lead in the chloraminated pilot distribution systems than in the free chlorinated systems for most blends examined.

Modeling studies that have investigated the effects of blending multiple water sources on distribution system water quality are limited. Taylor et al. (2005) used field plant evaluation and modeling of water quality changes in pilot distribution systems receiving desalinated seawater, treated surface water, groundwater, or their blends (water was treated using advanced and conventional treatment processes that simulated full-scale WTPs), to evaluate the impacts on distribution system water quality. Using these pilot distribution systems, Taylor et al. (2005) developed empirical models to predict color (as a surrogate for iron release), copper and lead releases. Different models were required for different pipe materials and hydraulic conditions. Models were also developed to explain free and total chlorine decay, as well as suspended and fixed biomass growth (expressed as HPCs). However, Taylor et al. (2005) specified that these models were applicable to these pilot systems only, and would not be appropriate without consideration of site-specific features such as pipe material, pipe geometry (length and diameter), hydraulics (flow rate or velocity), and water quality.

**Storage Buffer**

A storage buffer is a reservoir that can be either natural, also referred to as “environmental” (e.g., groundwater aquifer), or engineered (e.g., storage tank) in which treated effluent remains for a specific period of time before being withdrawn for potable water applications. The main benefit of such buffers is to allow time to monitor and react should treatment be inadequate due to process variability, unreliability, or unforeseen events. Another advantage of these reservoirs is to provide storage of the purified recycled water when excess is available. This section discusses both types of reservoirs, i.e., environmental and engineered.

Environmental buffers can also provide additional treatment through a range of potential processes, including photo-transformation, sorption, and biodegradation. For example, Salveson et al. (2010) observed adsorption of trace level pollutants during their transport in surface canals. While environmental buffers can add value, these buffers may also expose the purified recycled water to potential contaminants. Mobilization of metals (e.g., arsenic) has been observed in aquifer storage and recovery (ASR) projects in Florida (Trommer 2014). Natural surface waters may contain higher concentrations of pathogens and undesirable chemicals than purified recycled water. An additional drawback of environmental buffers is the need for a suitable groundwater
basin or surface water reservoirs, which may not be available everywhere. The presence of a natural or environmental buffer is the primary difference between IPR and DPR.

An engineered storage buffer (ESB) is designed to retain treated wastewater for a sufficient period of time to allow the measurement and reporting of specific constituents (referred to as “failure response time,” or FRT, Salveson et al. 2015), and ensure that water quality meets applicable standards before being used for potable water applications. The concept of FRT was used by Salveson et al. (2015) to propose a framework to adequately size ESBs. Further, the ESB will typically store purified water with very low DBP formation potential, allowing free chlorination of the ESB to provide significant pathogen removal - and the associated credits for it. Engineered storage systems are thus a recommended concept for DPR, considering that they offer opportunities for advanced monitoring and treatment.
CHAPTER 3
MATERIALS AND METHODS

WATER SOURCES, BLENDING, AND SIMULATED TREATMENT

The research project objectives include examining the impact of three blending variables:

- Different proportions of recycled water to raw water (ranging from 0 percent to 50 percent)
- Different qualities of purified water (different treatment process trains)
- Different points of introduction/blending (blending with the source water of a water treatment plant or direct introduction into the distribution system)

Four utilities (Utilities 1, 2, 3, 4a, and 4b) considering different blending scenarios were included in this study. The advanced water treatment process trains, the waters used for blending, location of blending, and water treatment plant processes varied across the utilities.

A key difference between the scenarios is the location of blending. To assess the impacts of blending prior to conventional drinking water treatment, samples were collected from untreated surface water and the purified recycled water. These waters were blended at specified blend ratios and the blend was subject to bench-scale simulated drinking water treatment designed to mimic the full-scale water treatment plant processes at the utilities. Finished drinking waters from full-scale treatment plants were compared to the treatment of raw surface waters at the bench-scale to determine if the bench-scale treatment used for the blended scenarios could effectively mimic the water quality seen at full-scale. Post-processing with SigmaPlot applied the Tukey’s honest significant difference (HSD) test with an alpha of 0.05. Results with p values less than 0.05 were considered to be significant.

To assess the impacts of blending after conventional water treatment, samples were collected from finished surface or groundwater and the purified recycled water. These waters were blended and tested without any additional simulated bench scale treatment.

Another key difference between utilities was the differences in the advanced purification process trains. These treatments included the following:

- Ozone and biological aerated filters (O₃-BAC)
- Microfiltration or ultrafiltration / reverse osmosis / ultraviolet advanced oxidation (AWP)
- Pasteurization / Microfiltration (MF) or ultrafiltration (UF) / reverse osmosis / ultraviolet (UV) / RO / UV advanced oxidation (past.-AWP)
- Tertiary Treatment, a Modified Johannesburg process for biological nitrogen and phosphorus removal with dual media filtration (Tertiary)

These treatment trains were not designed to provide equivalent levels of treatment, but were instead designed to provide insight into different treatment processes. The AWP process was operated to meet California groundwater recharge regulations while the O₃-BAC process was operated as further treatment to tertiary treated water. One of the key differences between the AWP and O₃-BAC was that AWP is an RO based process while O₃-BAC is not. The tertiary treatment
was included for comparison. Advanced treatment processes included either O₃ and biological activated carbon (O₃-BAC) or MF-RO-UV/AOP (AWP) at full- or pilot-scale. The feed water to the advanced treatment process trains was either secondary or tertiary effluent from full-scale WWTPs.

The following sections describe the blending scenarios for the four utilities. Table 3.1 provides a summary of the blending scenarios, and includes details on the purified recycled waters, conventional source waters, conventional water treatment processes, and abbreviations used throughout this report. Figures 3.2 through 3.6 show the blending scenarios schematically.

**Utility 1**

The Utility 1 blending scenarios are described as follows:

- Surface water (source water for their drinking water treatment plant) was blended with purified recycled water: AWP (MF-RO-UV/AOP) at a ratio of 10 percent AWP and 90 percent surface water, and 50 percent AWP and 50 percent surface water.
- Surface water was blended with purified recycled water: O₃-BAC, at a ratio of 10 percent O₃-BAC and 90 percent surface water.
- The blended waters and a 100 percent surface water were subjected to bench-scale simulated drinking water treatment consisting of the following: O₃ at a dose of 2 mg/L, coagulation with alum (38 mg/L), cationic polymer (1.5 mg/L), and non-ionic polymer (0.18 mg/L), flocculation, sedimentation, filtration (1.5 μm glass fiber filter), and chlorine followed by the addition of aqua ammonia to form chloramines (target of 1.8 to 2.2 mg/L total chlorine after 11 minutes).
- For comparison with the bench-scale simulation, a finished drinking water sample was also collected; this water had previously undergone the same treatment processes described above, but at the full-scale WTP.
- For the purposes of quantifying pathogen reduction, a sample of the secondary effluent that feeds the purification process trains, was collected. This secondary treatment process included some nitrification and denitrification.

**Utility 2**

The Utility 2 blending scenarios are described as follows:

- Treated groundwater (TDS range of approximately 1,000 to 1,600 mg/L; treated for iron and manganese removal) was collected and blended with purified recycled water: AWP (MF-RO-UV/AOP) at a ratio of 10 percent AWP and 50 percent treated groundwater.
- Treated groundwater was collected and blended with purified recycled water that included a pre-treatment step of pasteurization followed by UF-RO-UV/AOP (past.-AWP). The blending ratios applied were 10 percent past.-AWP and 90 percent groundwater, and 50 percent past.-AWP and 50 percent groundwater.
- The blended waters were subjected to secondary disinfection with chlorine followed by the addition of aqua ammonia to form chloramines at a target residual of 3.8 mg/L
and no additional treatment. In this case, the blending location is the distribution system. pH was not controlled in this treatment process.

- For the purposes of quantifying pathogen reduction, a sample of the secondary effluent that feeds the purification process trains was collected. This secondary treatment process included nitrification and denitrification.

**Utility 3**

The Utility 3 blending scenarios are described as follows:

- Surface water (source water for their drinking water treatment plant) was blended with purified recycled water: O3-BAC, at ratios of 10 percent O3-BAC to 90 percent surface water, and 50 percent O3-BAC to 50 percent surface water.
- Surface water was blended with tertiary effluent at a ratios of 5 percent effluent to 95 percent surface water, and 10 percent effluent to 90 percent surface water.
- The blended waters and a 100 percent surface water were treated via bench-scale simulated water treatment processes including: coagulation with ferric chloride (0.6 mg/L), flocculation, sedimentation, filtration (0.7 μm glass fiber filter), and chlorination (target dose of 1.5 mg/L after 2 hours). This was the only utility to employ chlorine instead of chloramines as a secondary disinfectant.
- For comparison with the bench-scale simulation, a finished drinking water sample was also collected; this water had previously undergone the same treatment processes described above, but at the full-scale WTP.
- For the purposes of quantifying pathogen reduction, a sample of the tertiary effluent that feeds the purification process trains was collected. The tertiary treatment process was a Modified Johannesburg process for biological nitrogen and phosphorus removal with dual media (anthracite/sand) filtration.

**Utility 4**

The Utility 4 blending scenarios are divided into 4a and 4b, where 4b included purified recycled water from an industrial process. The Utility 4a blending scenarios are described as follows:

- Treated groundwater (TDS range of approximately 300 to 700 mg/L; treated using GAC to remove VOCs) was blended with purified recycled water: AWP (MF-RO-UV/AOP) at a ratio of 10 percent AWP and 90 percent groundwater.
- Raw surface water was blended with purified recycled water: AWP (MF-RO-UV/AOP) at a ratio of 10 percent AWP and 90 percent groundwater.
- In both of the above-mentioned blends, the AWP was stabilized prior to blending to reduce its corrosivity with a goal LSI from -0.5 to 0.5 or close to zero (to see a further explanation of this parameter please refer to Chapter 5). To do this, the pH was raised to approximately 8 and then calcium chloride was added.
- The stabilized AWP blend was treated at the bench-scale by O3 (0.5 mg/L), coagulation with ferric chloride (1.5 mg/L) and cationic polymer (1.2 mg/L), flocculation, sedimentation, filtration (1.5 μm glass fiber filter), and pre-formed monochloramines
for chloramination (target dose of 2.6 mg/L total chlorine after 16.5 minutes), and finally the additional of zinc orthophosphate.

- The treated groundwater and AWP blend was subject to secondary disinfection and final addition of zinc orthophosphate.
- For the purposes of quantifying pathogen reduction, a sample of the tertiary effluent that feeds the purification process train was collected.

The Utility 4b blending scenarios are described as follows:

- Finished surface water was blended with purified recycled water from an industrial source: AWP (MF-RO-UV/AOP) at a ratio of 10 percent AWP and 90 percent groundwater. The industrial-AWP was a pilot-scale system treating the effluent from industrial food and beverage production with different water quality than a typical municipal wastewater.
- Prior to blending, the corrosivity of industrial-AWP was reduced by raising the pH to approximately 8 and adding calcium chloride with the end goal of a LSI value of 1.
- Following blending, the water was chloraminated with monochloramines at a target dose of 2.6 mg/L total chlorine. This disinfection was the only additional treatment added to the waters to determine how blending post-conventional treatment impacted this water.
- For comparison with the bench-scale simulation of surface water treatment in Utility 4a, a finished surface water sample was also collected; this water had previously undergone treatment at the full-scale level by O₃ (0.5 mg/L), coagulation with ferric chloride (1.5 mg/L) and cationic polymer (1.2 mg/L), flocculation, sedimentation, filtration (1.5 μm glass fiber filter), and chlorine followed by the addition of aqua ammonia for chloramination (target dose of 2.6 mg/L total chlorine after 16.5 minutes), and finally the additional of zinc orthophosphate.
- For the purposes of quantifying pathogen reduction, a sample of the secondary effluent that feeds the purification process trains was collected. This is secondary effluent from an MBR process.

All samples collected during this project were shipped overnight to SNWA without any additional preservative but on ice to keep them at approximately 4 °C. Upon receipt, samples were either immediately tested for various water quality parameters, used in blending and simulated treatment, or stored at 4 °C until the start of the experiments. For all utilities, blending was conducted in clean (rinsed with water and soap, rinsed with tap water, and rinsed with DI water) 40-gallon stainless steel containers.
<table>
<thead>
<tr>
<th>Utility, source water, and blend ratio</th>
<th>Pipe rig conditioning?</th>
<th>Scale</th>
<th>Simulated bench-, pilot-, or full-scale treatment</th>
<th>Sample date: water treatment</th>
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<td>Utility 1 FSW</td>
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</tr>
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<td></td>
<td>O₃, coagulation, filtration, chloramination</td>
<td>5/12/15</td>
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<td>O₃, coagulation, filtration, chloramination</td>
<td>5/12/15</td>
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<tr>
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<td>Bench</td>
<td>O₃, coagulation, filtration, chloramination</td>
<td>7/13/15</td>
</tr>
<tr>
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<td>7/13/15</td>
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<td>Pilot</td>
<td>O₃, BAC</td>
<td>5/12/15</td>
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<tr>
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<td></td>
<td>NA</td>
<td>5/12/15 and 7/13/15</td>
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<td></td>
<td>Fe/Mn removal and chlorination</td>
<td>2/29/16</td>
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<td>Chloramination</td>
<td>12/7/15</td>
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<td>Chloramination</td>
<td>12/7/15</td>
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<tr>
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<td>Chloramination</td>
<td>2/29/16</td>
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<td>50% GW/50% Post-AWP</td>
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<td>Chloramination</td>
<td>2/29/16</td>
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<tr>
<td>SE</td>
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<td></td>
<td>NA</td>
<td>12/7/15 and 2/29/16</td>
</tr>
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<td></td>
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<td>6/7/16</td>
</tr>
<tr>
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<td></td>
<td></td>
<td>Coagulation, filtration, chlorination</td>
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<tr>
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<td>90% SW/10% Tertiary</td>
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<td>Coagulation, filtration, chlorination</td>
<td>6/7/16</td>
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<tr>
<td>90% SW/10% O₃-BAC</td>
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<td>Bench</td>
<td>Coagulation, filtration, chlorination</td>
<td>10/13/15</td>
</tr>
<tr>
<td>50% SW/50% O₃-BAC</td>
<td>Yes</td>
<td>Bench</td>
<td>Coagulation, filtration, chlorination</td>
<td>10/13/15</td>
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<td></td>
<td></td>
<td>NA</td>
<td>10/13/15, 6/7/16</td>
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(continued)
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<tr>
<th>Utility, source water, and blend ratio</th>
<th>Pipe rig conditioning?</th>
<th>Scale</th>
<th>Simulated bench-, pilot-, or full-scale treatment</th>
<th>Sample date: water treatment</th>
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<td>GW</td>
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<td>Full</td>
<td>Chloramination</td>
<td>8/16/16</td>
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<tr>
<td>90% GW/10% AWP</td>
<td>Yes</td>
<td>Bench</td>
<td>Chloramination</td>
<td>8/16/16</td>
</tr>
<tr>
<td>90% SW/10% AWP</td>
<td>Yes</td>
<td>Bench</td>
<td>O₃, coagulation, filtration, chloramination, stabilization</td>
<td>8/16/16</td>
</tr>
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<td>AWP</td>
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<td>Pilot</td>
<td>MF, RO, UV-AOP</td>
<td>8/16/16</td>
</tr>
<tr>
<td>TE</td>
<td>No</td>
<td>Full</td>
<td>NA</td>
<td>8/16/16</td>
</tr>
</tbody>
</table>

| **Utility 4b**                       |                        |       |                                                  |                             |
| FSW                                  | Yes                    | Full  | O₃, coagulation, filtration, chloramination, stabilization | 8/29/16                     |
| 90% FSW/10% Industrial AWP           | Yes                    | Bench | Chloramination                                    | 8/29/16                     |
| Industrial AWP                       | No                     | Pilot | MF, RO, UV/AOP                                   | 8/29/16                     |
| SE                                   | No                     | Full  | NA                                                | 8/29/16                     |

AWP = advanced water purification, FSW = finished surface water, GW = groundwater, SW = surface water, O₃-BAC = ozone and biological aerated filters
SIMULATED DISTRIBUTION SYSTEMS: PIPE RIGS

Following blending and simulated treatment, waters were stored in amber, glass gallon bottles at 4 °C. Stored waters were subsequently added to pipe rigs to replicate the drinking water distribution system (Figure 3.1). Water was stored in the pipe rigs for a total of 8 weeks and changed out every 2 to 3 days depending on the weekly schedule. To generate enough water to replace the pipe rigs, the blended and simulated treated water was produced twice during the total 8-week test in weeks 1 and 4. After the production of enough water to last for 4 weeks, half of it was pasteurized for use during weeks 3 and 4. Pasteurization was done by setting the amber, gallon jugs in a water bath for 3-4 hours to raise the temperature within the test water to 72 °C. Once samples reached 72 °C, samples were pasteurized for 30 minutes. Following pasteurizing, the chlorine or chloramine dose was boosted to its original target to maintain the designated residual (Escobar and Randall 1999).

![Image of pipe rigs](image.png)

**Figure 3.1** Pipe rigs used to simulate drinking water distribution system (left) and a close-up of the brass rods inside the pipe rigs (right)

Of the waters that were collected for this project only a subset was conditioned in the pipe rigs, which are shown in Table 3.1 and Figures 3.2 through 3.6. The pipe rigs were designed and fabricated by Virginia Tech researchers to simulate household premise plumbing system and were made up of PVC pipes with two CDA 360 brass rods inside. The brass rods were 36 cm long with a ½ inch diameter and contained 60 percent copper, 35 percent zinc, and 3 percent lead. Duplicate pipe rigs were created for each water type. The duplicate pipe rigs were stored at room temperature, which was approximately 20 to 22 °C.

Eight weeks was chosen for the pipe rig tests as a balance between providing sufficient time for a biofilm to form and the practical limitations of shipping and storing a sufficient amount of water.
SAMPLING PLAN

Water quality (WQ) samples were collected at five different points during the sample collection, blending, simulated water treatment, and pipe rigs, with four different sets of water quality parameters collected at these five points. These points are highlighted in Figures 3.2 through 3.6, and the water quality parameters tested at each sample point are summarized in Table 3.2.

Figure 3.2 Utility 1 detail of sample collection locations

Notes:
1. The blend refers to the percentage of purified reclaimed water.
2. Where two blend percentages are shown, two separate scenarios were conducted.
3. Water quality test 3 was conducted on the secondary effluent (prior to advanced treatment)
Figure 3.3 Utility 2 detail of sample collection locations

Notes:
1. The blend refers to the percentage of purified reclaimed water.
2. Where two blend percentages are shown, two separate scenarios were conducted.
3. Water quality test 3 was conducted on the secondary effluent (prior to advanced treatment)
Figure 3.4 Utility 3 detail of sample collection locations

Notes:
1. The blend refers to the percentage of purified reclaimed water.
2. Where two blend percentages are shown, two separate scenarios were conducted.
3. Water quality test 3 was conducted on the secondary effluent (prior to advanced treatment)
Figure 3.5 Utility 4a detail of sample collection locations

Notes:
1. The blend refers to the percentage of purified reclaimed water.
2. Where two blend percentages are shown, two separate scenarios were conducted.
3. Water quality test 3 was conducted on the tertiary effluent (prior to advanced treatment)
These samples were collected in order to understand the pathogen load entering the advanced water treatment facilities tested during Utility 1-Utility 4 and to understand how these data compare to the raw surface water samples. One blended water sample was also analyzed from Utility 1 (50 percent SW/50 percent AWP). The pathogens included:

- *E. coli*
- fecal coliform,
- total coliform,
- *Giardia sp. *
- *Cryptosporidium sp. *
- Total Culturable Virus,
- Enterovirus,
- norovirus G1A, G1B, and G1I.

To detect *E. coli*, fecal coliform, total coliform, *Giardia sp.*, and *Cryptosporidium sp.* Standard Methods were used at SNWA. The four viruses were determined by BioVir. Virus samples were collected on site (at the water treatment facility) instead of being shipped to SNWA to prevent long old times of these sensitive samples.
**WQ Test 2**

In this test, the TTHM formation potential (FP) was determined for the samples listed in Table 3.2. Samples for WQ Test 2 were collected post-blending and simulated treatment, but prior to disinfection. This procedure is Standard Method 5710B and determines the concentration of maximum TTHM precursors present. Briefly, the procedure involves spiking samples with 10mM of phosphate buffer to insure a pH of 7 and dosing samples with sodium hypochlorite to ensure a residual of 3 to 5 mg/L of free chlorine at the end of a 7-day holding period. Samples were stored in the dark at room temperature. At the end of the 7-day exposure, samples were quenched with sodium thiosulfate and four THM species were quantified with EPA Method 524.3. They included bromoform, bromodichloromethane, chloroform, and chlorodibromomethane.

**WQ Test 3**

These analyses were chosen to understand the effect of bench-scale treatment as compared to full-scale treatment and compare bench-scale evaluation of various blending scenarios. WQ Test 3 included the following classes of TOrCs:

- PPCPs,
- steroids,
- PFAs, and
- DBPs.

The individual compounds and their method detection limits (MRL) can be found in Table 3.2 and were analyzed according to previously published procedures (Appleman et al. 2013, Appleman et al. 2014, Benotti et al. 2009, Holady et al. 2012, Vanderford and Snyder 2006). In addition to these TOrCs, several basic water quality parameters were measured including alkalinity, ammonia, nitrate, nitrite, TOC, DOC, total phosphate (as P), UV absorbance at 254 nm, UV absorbance at 280 nm, pH, turbidity, free chlorine, total chlorine, and adenosine triphosphate (ATP). Researchers at Virginia Tech also sampled for:

- BDOC,
- opportunistic pathogen (OP) gene markers,
- ARG, and
- antibiotic resistant HPC.

The sampling methods used for these analyses are described in Appendix B.

**WQ Test 4**

Samples for WQ Test 4 were collected in duplicate at influent and effluent to the pipe rigs. While water was changed out of the pipe rigs during the 8-week experiment, the WQ Test 4 samples were gathered twice in week 6 and week 8 of the influent and effluent of the pipe rigs. The specific parameters in WQ Test 4 are also shown in Table 3.2 and results were generated with the purpose of understanding the impact the distribution system has on potential changes in water quality.
Table 3.2
WQ test parameters, methods, and method reporting limits

<table>
<thead>
<tr>
<th>Bulk parameter</th>
<th>WQ Test</th>
<th>Analyte</th>
<th>Method</th>
<th>Reporting limit</th>
<th>Lab</th>
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<tbody>
<tr>
<td>Bulk water quality</td>
<td>Test 3 and Test 4 influent</td>
<td>Turbidity</td>
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*See Appendix B - Research Methods.
CHAPTER 4
WATER TREATMENT

BACKGROUND

This section compares the water quality of each blend at each of the four utilities as measured directly after blending or after simulated water treatment, as discussed in the sampling plan in Chapter 3. Water quality at this location should provide insight into the expected water quality entering the distribution system for all of the blends considered.

Comparing blends after treatment provides information on how the different source waters affect the water treatment process. For example, comparing the finished water quality of 100 percent surface water to 90 percent surface water / 10 percent AWP water provides insight into how adding AWP water to a water treatment plant will affect plant performance. Additionally, looking across all water sources and blends, inferences can be made regarding the most effective blend ratios.

RESULTS

Nutrients

Parameters collected to understand nutrient values included total nitrogen as N, ammonia as N, nitrate as N, nitrite as N, and phosphorous as P. Nutrients are important not only because they can be directly harmful to human health, but also because they can create the conditions that allow for harmful bacterial growth within the distribution system (EPA 1992, Zhang et al. 2009b). Nitrate and nitrite (both as N) are regulated at the entry to the distribution system by the EPA at levels of 10 mg/L and 1 mg/L, respectively (EPA 1992), and both can cause serious illness to infants including shortness of breath and blue-baby syndrome (EPA 1992). Phosphorous is not regulated by the EPA, but some states have taken on standards for lakes/reservoirs and rivers/streams that act as drinking water sources. Beyond protecting the receiving environment from adverse effects of eutrophication, an additional rationale behind these standards is to prevent the potential growth of unwanted microorganisms in drinking water systems potentially causing taste and odor issues or other water quality problems mentioned above with respect to nitrogen.

Although ammonia is not directly regulated, the presence of ammonia can lead to nitrification and potentially be disruptive to the disinfection process (Routt and Skadsen 2013). Water treatment plants typically target a ratio of Cl2:NH3-N 5:1 to generate chloramines, specifically monochloramine. If this ratio is not met and the ammonia level is lower than expected, free chlorine could be present in the system and potentially produce disinfection byproducts or if the ammonia is higher than expected, nitrification could occur. Nitrification occurs when nitrifying bacteria proliferate and oxidize ammonia to nitrate and then nitrite. These bacteria can also result in the loss of disinfectant residual, decreases in the pH and dissolved oxygen concentrations, and potential increase in corrosion of premise plumbing systems (Zhang et al. 2009b, Zhang et al. 2010a, Routt and Skadsen 2013). The complex set of interactions involving nutrients illustrates the importance of understanding their presence in finished drinking water. Within the context of this project, the effect of blending and simulated water treatment on nutrients was evaluated.

Nutrient concentrations varied considerably across the different utilities based on source water type and wastewater treatment processes. Finished drinking water samples for Utility 1 and 3
(utilities with bench- and full-scale comparison samples) showed mixed results in their ability to mimic the concentrations generated by the simulated water treatment at bench-scale as shown in Figure 4.1. The bench-scale samples at Utility 3 were generally able to mimic the full-scale comparison samples. However, in Utility 1, the nitrate and phosphorus data did not match with the data collected at the full-scale. The difference in nitrate and phosphorus observed may be attributed to the variability in water samples collected from the raw surface water used in simulation tests and the full-scale finished water which were collected on different dates.

![Figure 4.1 Concentrations of nutrients in Utility 1 (bottom) and Utility 3 (top) samples comparing the full-scale (FSW) and bench-scale (SW) results; unfilled bars are below the MRL](image)

Ammonia concentrations in the blended and simulated treated waters were low for waters from Utility 1 and 3 as seen in Figure 4.2. In Utility 1, the concentration of ammonia was decreased with the addition of O₃-BAC, which would improve the water quality. The concentrations found in Utility 2 samples show that there is a potential risk for AWPAWP based advanced treated wastewaters to drive the concentration of ammonia up when blended with low-ammonia surface waters or groundwaters. RO is effective in treating ammonia (> 85 percent), but the final effluent water quality is still dependent on influent ammonia concentrations (Voorthuizen et al. 2005, Metcalf & Eddy 2007). In Utility 2, levels of ammonia in the blends were <1.5 mg/L, but this concentration could potentially affect the amount of chlorine necessary to form chloramines or inadvertently allow for excess ammonia entering the drinking water distribution network. As mentioned above, the latter can be harmful due to the negative consequences of potential nitrification. The concentrations of ammonia in Utility 3 were all below the MRL. In Utility 4, changes in the concentration of ammonia with respect to added recycled water were limited. Although advanced treatment provided additional removal of ammonia, low influent concentrations are attributed to upstream wastewater treatment plant processes.
Nitrate in advanced treated wastewater from Utilities 1, 3, and 4 (Figure 4.3) resulted in concentrations above the EPA regulated value of 10 mg/L (EPA 1992), highlighting the need for these systems to be blended with a source water containing low levels of nitrate. The test blends for Utilities 1, 3, and 4 would have exceeded the regulation for a 50 percent, 60 percent, and 88 percent blends, respectively. These exceedances are a result of the nitrate present in both AWPAWP and O3-BAC advanced treated wastewaters. Although RO removed more than 85 percent of nitrate, the resulting concentration may still be relatively high based on the influent concentration (Metcalf & Eddy 2007). O3-BAC was not expected to provide removal of nitrate. These three utilities demonstrate the potential risk of blended with either a biological filtration-based treatment or membrane-based treatments. In Utility 4a, GW/AWP, the opposite trend was observed where the groundwater contained higher concentrations of nitrate as compared to the advanced treated wastewater and the addition of advanced treated wastewater improved the water quality.

Figure 4.2 Concentration of ammonia relative to the percentage of recycled water added for the four utilities
Nitrite was not detected in any of the blended and simulated treated waters with the method reporting limit at 0.02 mg/L.

Key conclusions on nutrients are shown in Figure 4.4.
Bulk Water Quality

During this project, pH, turbidity, TOC, DOC, UV at 254 nm, conductivity, alkalinity, sulfate, and chloride were collected as a part of the bulk water quality analyses. In general, the pH of RO permeate was low due to the removal of the dissolved HCO₃⁻ and CO₃²⁻ and subsequent equilibrium shift in the presence of atmospheric CO₂. This trend was observed in Utilities 1, 2, and 4a (Figure 4.5) in which the addition of higher percentages of recycled water decreased the pH. In Utility 3, O₃-BAC and tertiary effluent samples had increases in the pH as compared to the SW; however, in the 100 percent O₃-BAC sample, the pH decreased significantly (p < 0.001). The pH in Utility 4b increased with the addition of recycled water, where base and calcium chloride were added during the simulated water treatment to change the LSI of the water to minimize metal leaching. This adjustment resulted in an increase in pH from approximately 7 to 8.

Figure 4.5 pH relative to the percentage of recycled water added for the four utilities

The turbidity, TOC and DOC followed similar trends in all four utilities. Turbidity is associated with the presence of harmful pathogens and thus has been used as a surrogate for the filtration treatment efficacy. Turbidity levels are regulated by the EPA depending on the type of treatment applied with direct filtration/conventional systems required to produce water with a MCL < 1 Nephelometric Turbidity Unit (NTU) at all times and 95 percent of monthly samples < 0.3 NTU (EPA 1989). Utilities with filtration systems other than conventional or direct filtration, must follow state limits and are required to limit turbidity to below 5 NTUs always.

All the samples tested were below the 0.3 NTU regulation as shown in Figure 4.6. In Utilities 1, 3, and 4, the addition of AWP water lowered the turbidity and improved the water quality. In systems with advanced treatment using a tertiary effluent or BAC, the opposite trend was observed. The tertiary effluent system was the only one that appeared as though it could potentially exceed this level when blended at 30 percent (not tested, calculated based on presumed effect of dilution). Note, this tertiary effluent was tested as an extreme worst-case scenario and is not currently under consideration.
Although both TOC and DOC were tested, the difference between the two was limited to < 0.5 mg/L for all utilities with the majority being the dissolved fraction. Instances where the DOC did not match with the TOC were limited to O3-BAC waters and the tertiary effluent where colloids and particulate organic matter have not been physically removed or degraded during treatment. Comparing simulated treatment and full-scale treatment, the DOC concentrations for Utility 1 were within the standard deviation and the simulated treatment DOC concentrations for Utility 3 were only slightly higher than the full-scale treatment DOC concentrations as shown in Figure 4.7. This was another strong indicator that the simulated treatment tests were effective in re-creating the water quality seen at the full-scale. While no comparison data was generated in Utility 2 and 4, the simulated treatment data is considered to be an accurate representation of the water quality that would be observed at full-scale based on the DOC and nutrient data (phosphorous, nitrate, nitrite, and ammonia) generated in Utilities 1 and 3. Additionally, the bench scale systems followed the full-scale operations as closely as possible. While the bench scale systems may not perfectly simulate the full-scale system, the finished water at the bench scale is used as a relative comparison for the blending scenarios.

Figure 4.6 Turbidity changes relative to the percentage of recycled water added for the four utilities; dotted red line indicates monthly EPA limit for direct filtration/conventional treatment systems
TOC has an EPA required treatment technique for drinking water. The percent removal of TOC is water specific, based on the raw water TOC and alkalinity concentrations. Additionally, TOC regulations are under consideration or have been adopted in certain states (e.g., California and Florida) for potable reuse applications. The current regulation from the California Division of Drinking Water for groundwater recharge requires that water systems reduce their effluent TOC to below 0.5 mg/L of wastewater origin (CDPH 2015). Florida requires a recycled water TOC of less than 3 mg/L of wastewater origin for indirect potable reuse. The Water Environment and Reuse Foundation is currently funding research investigating appropriate TOC guidelines for potable reuse, which may account for type of potable reuse application and geographical location. Figure 4.8 includes the 0.5 mg/L (wastewater origin) TOC limits as a conservative regulatory threshold. However, this limit only applies to results from analysis of 100 percent recycled water, namely the industrial AWP and AWP in Utility 4, and the O₃-BAC in Utility 1. Both the AWP waters met the 0.5 mg/L threshold, but the O₃-BAC well exceeded the threshold.

In Utility 1, the difference between O₃-BAC and AWP AWP was highlighted by the net effect on the DOC concentration (Figure 4.8). With the addition of O₃-BAC at a DOC of 5.6 mg/L, the DOC of 10 percent blended water increased to 3.23 mg/L, up from 2.68 mg/L in the bench-scale simulated treated surface water. In contrast, the addition of a higher percentage of AWP the DOC decreased from 4.47 mg/L with 10 percent to 3.07 mg/L with 50 percent. Utility 2 blends were groundwater-based and therefore significantly lower (p < 0.001) than the surface water-based blends in Utility 1. With the similar levels of DOC in Utility 2 groundwater and the advanced treated wastewater (with pasteurization or without), the blending ratios only caused slight changes in the concentration. In Utility 3, the O₃-BAC and tertiary effluent contained 4.4 mg/L and 6.4 mg/L, respectively. Both these additions increased the DOC of their blended waters. The two scenarios in Utility 4 both had waters treated by AWP AWP that reduced the concentration of DOC in the blended waters. Overall, DOC concentrations illustrated the importance of source water and the type of advanced treatment and in general, the surface water and O₃-BAC concentrations were higher than those observed in groundwater and AWP AWP.
Figure 4.8 Concentration of DOC in all four utilities under various blending scenarios; red line is representative of the CA regulation applied to unblended recycled waters in CA; blue, dotted line differentiates between scenarios a and b for Utility 4.

SUVA is a measure of the light absorbance at a given wavelength normalized to the concentration of DOC. When calculated at 254 nm, SUVA is a measure of the aromaticity of the organic matter present in the sample (Weishaar et al. 2003). SUVA\textsubscript{254} is also used as an indicator of the formation of disinfection byproducts, specifically trihalomethanes. The highest SUVA\textsubscript{254} values were observed in waters from groundwater or treated with AWPAWP as seen in Figure 4.9. SUVA\textsubscript{254} and its relationship to disinfection byproducts is further discussed in the disinfection byproduct discussion below.
Alkalinity, sulfate, and chloride are important corrosion potential indicators. Sulfate and chloride are listed as secondary standards by the EPA due to aesthetic effects and are limited to concentrations below 250 mg/L to prevent the water from being detected as “salty” by customers (EPA 1979, EPA 1986, EPA 1991b). Understanding if these parameters act conservatively in the blended waters is important for predicting future changes under blending scenarios not specifically examined in this report.

Alkalinity concentrations in groundwater, O3-BAC, and tertiary effluent samples were significantly higher than those found in AWP or surface waters (p < 0.001) as seen in Figure 4.10. When these waters were blended, the alkalinity behaved as an additive parameter with dilution dictating the final concentrations. Sulfate concentrations were similarly high in the groundwater, O3-BAC, and tertiary effluent samples. All of the waters in Utility 2 were higher than the secondary standard of 250 mg/L and the 50 percent SW/50 percent O3-BAC blend of Utility 3 exceeded this value at 280 mg/L. The O3-BAC in Utilities 1 and 3 and tertiary effluent in Utility 3 did not provide removal and instead increased the sulfate concentrations in any blended water. In contrast,
RO-UV/AOP is effective in removing > 90 percent of the sulfate present (Metcalf & Eddy 2007), which means that treatment with AWP reduced the concentration of sulfate in the blended water.

Figure 4.10 Chloride, sulfate, and alkalinity concentrations in all four utilities under various blending scenarios; blue, dotted line differentiates between scenarios a and b for Utility 4

Chloride concentrations were relatively similar across almost all of the waters and all were below the 250 mg/L Secondary standard. In Utility 1, the O3-BAC contained 203 mg/L of chloride, which could have been a problem for the blended sample had the surface water not been significantly lower at 80 mg/L (p < 0.001). Utilities 2 and 3 advanced treated wastewaters and source waters had similar levels of chloride and upon blending the concentrations did not change. The AWP of Utility 4 had higher chloride levels than the groundwater or surface samples and increased the concentration when blended with recycled water. Alkalinity, sulfate, and chloride all behaved as conservative water quality parameters with concentrations depending on the source water and the type of advanced treatment, which showed that future water quality predictions could be generated for the water quality of the two blended waters.
Conductivity is not regulated by the EPA, but is included in the CA Secondary Maximum Contaminant Levels. There are three levels listed in the Secondary Water Standards including a recommended value of 900 µS/cm, an upper limit of 1,600 µS/cm, and a short-term limit of 2,200 µS/cm (DDW 2017). The recommended level was exceeded by the 100 percent O₃-BAC of Utility 1 and all of the scenarios for Utilities 2 and 3 as shown in Figure 4.11. The upper limit was exceeded by three samples in Utility 2 including: 100 percent GW, 90 percent GW/10 percent AWP, and 90 percent GW/10 percent past.-AWP. The high levels of conductivity in Utility 3 were driven by concentrations found in the 100 percent SW, tertiary effluent, and O₃-BAC blended waters. In Utilities 2 and 4, the highest concentrations were found in the GW. Conductivity is considered to be a surrogate for total dissolved solids.

Key conclusions on bulk water quality are shown in Figure 4.12.

![Figure 4.11 Conductivity concentrations in all four utilities under various blending scenarios; blue, dotted line differentiates between scenarios a and b for Utility 4](image-url)
Disinfectant Residuals, Disinfection Byproduct Formation, and ATP

The initial disinfection dose of either chlorine or chloramines was added to inactivate pathogens in the blended, treated waters and residuals were maintained. Of the four utilities, Utilities 1, 2, and 4 applied an initial dose of chlorine followed by ammonia at their full-scale facilities to produce chloramines for disinfection with target doses of 1.8-2.2 mg/L, 3.8 mg/L, and 2.6 mg/L total chlorine, respectively. Utility 3 had a chlorine target dose of 1.5 mg/L free chlorine.

For Utilities 1, 2, 3, and 4, the majority of samples were able to meet the target dose after and during storage and were subsequently spiked with additional chloramines or chlorine to retain this residual as seen in Figure 4.11. The 100 percent O₃-BAC from Utility 1 did not meet its target and only had a residual of 0.4 mg/L (not shown on Figure 4.11). For the other utilities, the dominant species of either chloramines or free chlorine were as expected.
Figure 4.13 Total chlorine and ATP concentrations in all four utilities under various blending scenarios; blue, dotted line differentiates between scenarios a and b for Utility 4

The impact of these disinfectants on the ATP is shown in Figure 4.13. In general, treated drinking waters contain 1-10 pg/mL of ATP (Vital et al. 2012, Nescerecka et al. 2016). The comparison samples from Utility 3 (FSW and SW) were similar in terms of the resulting ATP post-treatment, but the ones collected from Utility 1 were not. As was discussed for the nutrients, this difference is most likely due to the collection time for the raw surface water vs. the finished drinking water. The low concentration of total and free chlorine in the 100 percent O₃-BAC from Utility 1 resulted in an ATP value of 78.4 pg/mL (not shown in Figure 4.13), which is significantly higher than the average drinking water effluent ATP (p < 0.001). Before disinfection, a 100 percent O₃-BAC (from Utility 3) sample contained 504 pg/mL of ATP, meaning that the Utility 1 sample was much higher than other drinking waters, but was significantly reduced through disinfection.

DBPs can form from the reaction of chlorine or chloramines and NOM, EfOM, and anthropogenic compounds. While over 600 DBPs have been reported, only a subset is regulated by the EPA (Richardson and Ternes 2011, EPA 2010) including four THMs and five HAAs as a result of their links to negative human health effects including increased risk of cancer and liver, kidney, or central nervous system issues (EPA 2010). The sum of four THMs (bromoform,
chloroform, bromodichloromethane, and dibromochloromethane) are limited to 80 µg/L. HAAs include dichloroacetic acid, trichloroacetic acid, monochloroacetic acid, bromoacetic acid, and dibromoacetic acid. These five compounds are regulated as a sum at 60 µg/L. Brominated THMs and HAAs are produced by a series of complex reactions of organic matter, ambient bromide ions, and the disinfectant (chlorine or chloramines). These brominated DBPs are of concern because they tend to be more toxic than their chlorinated analogs (Richardson and Ternes 2011, Krasner et al. 2006). To meet these EPA regulations, many utilities have switched their disinfectant from chlorine to chloramines, which have lower yields for THMs and HAAs (Seidel et al. 2005).

DBPs can be mitigated by removing the precursors or the compounds themselves. For THMs and HAAs, precursor mitigation can be determined by tracking the bulk organic carbon, specifically DOC with high percentage of aromatic moieties (i.e., high SUVA254 values) (Reckhow et al. 1990, Liang and Singer 2003). DOC and bromide are both found at significantly higher concentrations in wastewater as compared to traditional source waters such as surface water and groundwater. Additionally, AWPAWP treatments will dramatically reduce the concentration of DOC and bromide. O3-BAC treatments will alter the ratio of DOC and bromide and lead to changes in the relative proportion of brominated to chlorinated DBPs that are formed because bromide is not removed.

Across all utilities, ambient TTHM and HAA formation was under the regulated values of 60 and 80 µg/L. Considering the dose of chlorine or chloramines added, the relative formation of TTHMs and HAAs were highest in Utility 3 (free chlorine only). The highest dose of chloramines added in Utility 2 did not result in a higher concentration of TTHMs or HAAs. Under the same disinfection conditions, the groundwater in Utility 4 formed lower TTHMs and HAAs than the FSW, which can be attributed to the lower DOC in the groundwater (Figure 4.30).

In Utility 1, the surface water used in blending was collected at two different time points. In the first round of samples, the SW and 90 percent SW/10 percent O3-BAC were tested and resulted in similar concentrations of TTHMs and HAAs (Figure 4.14 and 4.15). However, for the second round of surface water sampling, the concentrations of TTHMs and HAAs were significantly higher. This is possibly due to the change in DOC concentration from 3 mg/L to ~4 mg/L³ of the surface water collected for the first round vs the second round. However, surprisingly an analogous increase in TTHM FP was not observed. Thus, it is more likely that during the second round of water sampling there was an unexpected variability in the disinfection protocol and source water. For Utility 2, low concentrations of TOC and TTHMs were seen and the 90 percent GW/10 percent AWP and 50 percent GW/50 percent AWP had improved water quality as compared to the GW source water. For the pasteurized samples, the 90 percent GW/10 percent AWP-past, had lower concentrations of TTHMs than the 50 percent blend, similar to the GW. This was most likely due to the lower concentration of total chlorine. However, these trends were not observed for the HAAs of Utility 2, where the blend did not significantly impact their formation. For Utility 3, the blended waters resulted in increases in the TTHM and HAA concentrations as compared to the treated surface water. This clearly demonstrated the impact of using an O3-BAC treatment train as compared to AWPAWP. The lowest concentrations of TTHMs and HAAs were observed in Utility 4 where the TOC was low, chloramines were applied, and waters were blended with AWPAWP treated waters. A lower concentration of TTHMs formed in scenario 4a compared to scenario 4b, possibly indicating that the fraction of organic matter in the surface water blend was less reactive to form TTHMs even though it had TOC values similar to

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³ A concentration of ~4 mg/L was estimated using the data collected and the assumption that DOC is a conservative parameter.
the samples in scenario 4b. The 90 percent SW/10 percent AWP in Utility 4a had a higher concentration of HAAs as compared to the 90 percent GW/10 percent AWP and GW sample.

Figure 4.14 TTHM concentrations in all four utilities under various blending scenarios; blue, dotted line differentiates between scenarios a and b for Utility 4
In past work, the ratio of chloride to bromide and the ratio of bromide to DOC have been calculated to compare the relative importance of brominated DBPs. During this study, bromide ion concentrations were not measured; therefore, to compare the relative contribution of brominated to chlorinated DBPs, the bromide substitution factor (BSF) was calculated. The BSF was introduced to compare the integration of bromine across different classes of DBPs (Gould et al. 1983, Hua et al. 2006, Hua and Reckhow 2012) by determining the molar contribution of brominated species to the total (e.g., TTHM and HAA5 in Eq. 4.1 and Eq. 4.2). The HAA5-BSF was calculated because only five HAA species were included in this study and therefore the dihaloacetic acid and trihaloacetic acid BSF values used in previous work were not calculated (Hua and Reckhow 2007). The Fourth Unregulated Contaminant Monitoring Rule (UCMR4) includes the sum of nine HAA species known as HAA9 (EPA 2016c). It has four species the HAA5 did not include: bromodichloroacetic acid, chlorodibromoacetic acid, tribromoacetic acid, and bromochloroacetic acid. If HAA9 were used, then the DHAA- and THAA-BSF values could be
determined. Importantly, the additional species all contain bromine and have been shown to co-occur, which would result in increases for the DHAA- and THAA-BSF values. The UCMR4 also included a contaminant known as HAA6Br, which is the sum of bromochloroacetic acid, bromodichloroacetic acid, dibromoacetic acid, chlorodibromoacetic acid, monobromoacetic acid, and tribromoacetic acid. The HAA6Br species each include bromine and therefore could not be used to calculate a BSF.

\[
TTHM - BSF = \frac{CHBrCl_2 + 2 \times CHBr_2Cl + 3 \times CHBr_3}{3 \times (CHCl_3 + CHBrCl_2 + CHBr_2Cl + CHBr_3)}
\]  \hspace{1cm} (4.1)

\[
HAA5 - BSF = \frac{2 \times DBAA + 1 \times MBAA}{3 \times TCAA + 2 \times (DCAA + DBAA) + (MCAA + MBAA)}
\]  \hspace{1cm} (4.2)

The BSF for TTHM and HAA5 are presented in Figure 4.16. In the utilities with chloramines (Utility 1, 2, and 4), the BSF values were generally higher than in the chlorinated Utility 33. However, this could also be due to the higher residuals applied in these three utilities. In Utility 1, the percentage of bromine integration for both TTHM and HAA5 was higher for the O3-BAC and surface water samples potentially owing to a higher ratio of bromide to DOC. In Utility 2, the 90 percent GW/10 percent AWP and 50 percent GW/50 percent AWP waters were higher than those observed for the pasteurized AWP, most likely based on different bromide concentrations found at these two pilot plants. BSF values for TTHMs in Utility 3 were highest in the surface water samples which is surprising given that the tertiary treated wastewater and O3-BAC waters would be expected to add bromide to the four blends. However, it is possible that the lower pH value in the surface water as compared to the blends resulted in a higher BSF (Hua and Reckhow 2012). In Utility 4, both the 100 percent AWPAWP treated waters contained lowest BSF ratios, which were expected based on the removal of bromide by RO. With the GW, 90 percent GW/10 percent AWP, and 90 percent SW/10 percent AWP in scenario a, the percentages were consistent. Similarly, the FSW and the 90 percent FSW/10 percent industrial-AWP in scenario b were consistent.
Figure 4.16 BSF for TTHM and HAA5 in all four utilities under various blending scenarios; blue, dotted line differentiates between scenarios a and b for Utility 4

TTHM FP tests were conducted on waters prior to disinfection to assess the level of precursors present. The results, as seen in Figure 4.17, show a different picture than the water disinfected per each utility’s specification. Utility 1 contained the highest concentration of precursors and showed the difference between blending a surface water with O₃-BAC and the AWP. The SW and 90 percent SW/10 percent AWP of Utility 1 contained higher concentration of precursors than the 50 percent SW/50 percent AWP sample indicating that the blending of these waters improved the water quality. In Utility 2, the groundwater, acting as the source water, contained lower concentrations of precursors than the surface water in Utility 1, which allowed the overall TTHM FP to be lower. The lower concentrations of precursors were composed of a majority of brominated species (bromoform and chlorodibromomethane) which would indicate that the groundwater contained a higher ratio of bromide to DOC than found in Utilities 1, 2, or 3. Surprisingly, Utility 3 contained lower concentration of precursors than found in Utility 1 despite using tertiary effluent as a blending water. This is attributed to the relatively lower precursors in the surface water of Utility 3 as compared to the surface water in Utility 1. For Utility 3, the blends of surface water with O₃-BAC and tertiary effluent increased the TTHM FP slightly, as compared to the unblended surface water. Lastly in Utility 4, the AWP waters both reduced the concentration of precursors when blended with either surface water or groundwater improving the overall water quality of the blended waters. Despite the influent to the industrial AWP, the effluent water quality was very similar to that observed in the municipally-fed treatment plant. The TTHM FP results indicate that blending with AWP can reduce the TTHM FP while
blending with O₃-BAC results in similar or increased TTHM FP. Additionally, the precursors in the surface water or groundwater blended with advanced treated wastewater also need to be considered when evaluating the results of a TTHM FP test.

In past work with drinking waters and surface waters, SUVA₂₅₄ has been positively correlated to the TTHM FP and this was attributed to the presence of reactive aromatic moieties (Reckhow et al. 1990; Liang and Singer 2003, Weishaar et al. 2003). However, in Utilities 1 through 4, TTHM formation under FP test conditions was significantly negatively correlated with SUVA₂₅₄ ($R^2 = 0.79$, $p < 0.001$). For these waters, low DOC values drove up the SUVA₂₅₄, but also limited the formation of TTHMs by the low concentration of total precursors present. This was especially apparent in the samples from Utilities 2 and 4 in which GW was blended with AWP treated water and all the waters had DOC concentrations < 1 mg/L (highlighted by a black box in Figure 4.18). UV₂₅₄ ($R^2 = 0.71$, $p < 0.001$) and DOC ($R^2 = 0.89$, $p < 0.001$) were significantly positively correlated with the TTHM FP. This indicates that both are important factors in the formation of TTHMs under FP conditions, but that caution must be applied when using SUVA₂₅₄ to predict the TTHM FP of blended waters. SUVA₂₅₄, DOC, and UV₂₅₄ were not significantly correlated with ambient THMs or HAAs formed during simulated disinfection.

**Figure 4.17** TTHM formation potential in all four utilities under various blending scenarios; blue, dotted line differentiates between scenarios a and b for Utility 4.
(presented above) due to the combined effects of the different doses applied and disinfectant (chlorine vs chloramine) type.

![Figure 4.18 TTHM FP concentrations as compared to DOC and SUVA254 in all four utilities under various blending scenarios; samples within the black box have DOC values < 1 mg/L](image)

NDMA can also be formed during AWP and O3-BAC processes through chloramination (used in AWP processes) and ozonation reactions. NDMA was present at concentrations below the CA notification level of 10 ng/L in Utilities 1 and 3 (Figure 4.19) and below the MRL of 2.5 ng/L in Utility 2 (CDPH 2013). Higher concentrations were observed in the tertiary effluent (Utility 3) and waters blended with O3-BAC (Utilities 1 and 3) because of the presence of NDMA that was not completely biodegraded during BAC and present in the system from the reaction with O3. It is presumed that NDMA levels are due to ambient levels either in the treated wastewater or formed during upstream chloramination at the advanced facility or due to the application of chloramines as a secondary disinfectant (for Utilities 1 and 4).

Utility 4 had two detections of NDMA above the notification level both in 100 percent AWP treatment trains. Removal efficiency of RO and UV-AOP at a similar facility has been approximately 70 percent (combined) with 25 to 50 percent attributed to RO and 40 to 65 percent attributed to UV-AOP (Plumlee et al. 2008). There are a couple of potential explanations for this high level of NDMA: there was a breakthrough of NDMA that was in the treated wastewater and/or formed upstream during the application of chloramines to prevent biofouling of membranes, the RO allowed precursors to pass through (most likely low molecular weight amines: dimethylamine or trimethylamine) and reform NDMA in the presence of chloramines, or UV/AOP degraded NDMA into nitrite and dimethylamine that similarly allowed NDMA to reform when chloramines were added (Plumlee and Reinhard 2007, Plumlee et al. 2008, Xu et al. 2009).

Key conclusions on DBPs are shown in Figure 4.20.
Figure 4.19 NDMA concentrations in all four utilities under various blending scenarios; blue, dotted line differentiates between scenarios a and b for Utility 4; red line is at the 10 ng/L notification level (CDPH 2013)
Figure 4.20 DBPs key conclusions

Pathogens

One of the main goals of U.S. drinking water regulations is to ensure that treated waters are microbiologically safe. For the EPA, this means that the water must be treated to prevent the outbreak of illnesses, but also to minimize any effect of drinking water on endemic level of disease (Macler and Regli 1993). To achieve this, a risk-based approach is used to limit the annual risk of infection to 1 in 10,000. This corresponds to a *Giardia sp.* concentration of $< 6.8 \times 10^{-6}$ cysts/L, a *Cryptosporidium sp.* concentration of $< 3.0 \times 10^{-5}$ oocysts/L, and a virus concentration of $< 2.2 \times 10^{-7}$ MPN/L. To meet this goal, pathogens must be four to seven orders of magnitude below the detection limit. Currently there are no federal regulations specifically for DPR; however, California and Texas have adopted this 1 in 10,000 risk goal as a means of regulating potable reuse. Additionally, there are often further criteria to ensure removal and control of otherwise unregulated trace level pollutants (CDPH 2014, Crook et al. 2013, Trussell et al. 2013). For this report, *Giardia sp.*, *Cryptosporidium sp.*, Total Culturable Virus, Enterovirus, Norovirus GIA, Norovirus GIB, and Norovirus GII were measured.

Two metrics were used to evaluate the quality of the advanced treated wastewater considered in this study with respect to pathogens for each of the four utilities. The first metric is a direct comparison to the 1 in 10,000 risk level concentrations listed above. This approach is in alignment with the regulatory approach to DPR in Texas, which requires extensive characterization of an effluent source water and bases log removal requirements on the difference between pathogen levels in the effluent and the drinking water goals listed above.

The second metric considered in this study is a comparison to the standardized log reduction goals for virus, *Giardia sp.*, and *Cryptosporidium sp.* established by the State of California. While the state has not yet adopted regulations for DPR, it has adopted groundwater
recharge IPR regulations for pathogens. These regulations are a useful starting point when considering possible future DPR regulations. The existing IPR regulations for direct injection of recycled water into a drinking water aquifer specify a 12-log virus reduction, 10-log *Giardia sp.* reduction, and 10-log *Cryptosporidium sp.* log reduction. These standardized values were calculated based on the difference between a conservative estimate of pathogen concentrations in raw wastewater and the 1 in 10,000 risk level concentrations listed above.

In order to better quantify the low levels of pathogens expected in the advanced treated wastewater and evaluate the water blends by the metrics described above, untreated feed water for the advanced treatment trains was sampled, and then product water pathogen levels were estimated using assumed log removal credits for the treatment processes used. A schematic of the water treatment and blending process is presented in Figure 4.21 along with the location of each water sampled and calculated for pathogens.

The focus of this analysis is on the water quality of the advanced treated wastewaters that are blended with each utility's current water supply. Although raw water from each utility's current water supply (the water blended with the advanced treated wastewaters) was sampled, it was assumed that once treated, this water would meet all applicable water quality pathogen standards. This is a reasonable assumption because this raw water is currently the water supply for each utility. In this study, the only new variable that could introduce pathogens into the system is the advanced treated wastewater.

The log removal credits assumed for each treatment process are shown in Table 4.1. Log removal credits are established for *Giardia sp.*, *Cryptosporidium sp.*, and virus. In this study, virus data was collected for total culturable virus, Enterovirus, Norovirus GIA, Norovirus GIB, and Norovirus GII. An important difference between the total culturable virus test and the other tests is the use of a culture to measure viable organisms in the former, while the measurement of gene copies measured via quantitative polymerase chain reaction (qPCR) analysis is used in the latter. Gene copy numbers do not necessarily correlate to viable pathogens and this is a current topic of research within our industry. A conservative approach would be to assume all gene copies to be viable pathogens. In this report we follow this conservative approach, using the enterovirus qPCR counts for virus counts. In the data collected, the ratios of enterovirus to total culturable virus for the advanced treated wastewaters ranged from 17 to 800,000 indicating that our approach is indeed conservative.
Figure 4.21 Pathogen sampling scheme
Table 4.1
Log removal credits

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</tr>
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<tr>
<td>Pasteurization</td>
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*Log removal credits based on the Texas Commission on Environmental Quality 2016 report.
†Log removal credits based on EPA requirements for surface water treatment.
‡Log removal credits based on a 2017 study by Carollo Engineers, Inc. and a 2015 study by Branch, A. and LE-Clech, P.
§Log removal credits based on the Framework for Direct Potable Reuse (Independent Advisory Panel 2015). Note that these credits were estimated in general alignment with the permitting approach taken in California and may not accurately represent the credits that might be assigned to treatment processes in other states.

The measured pathogen data for each of the four utilities is shown in Figure 4.22 for Giardia sp., Figure 4.23 for Cryptosporidium sp., Figure 4.24 for Total Culturable Virus, and Figure 4.25 for Enterovirus. Both the untreated secondary effluent used as feed water for the advanced treatment trains as well as the untreated surface or groundwater blended with the advanced treated product waters are shown. As presented in the figures, all pathogens measured in the untreated surface and groundwater were below the detection limit which ranged from 0.005 to 2/Liter.

Using the measured data presented in the figures and assumed log reduction credits for the treatment trains considered, a risk analysis was conducted for each water source and is summarized in Table 4.2 and Figures 4.26 through 4.28. In the figures, the overall length of the bar shows the total log removal credit assumed for each advanced treated wastewater considered in this study. The lower edge of each bar shows the measured starting concentration for each water blend. As shown in the table and figures, only the AWP treated waters are below the required drinking water risk level for all pathogens shown. For both the tertiary treated water and the O₃-BAF waters that undergo simulated water treatment, two of the three pathogen drinking water risk levels are exceeded. The O₃-BAF water that does not undergo further treatment in Utility 1 exceeds all three drinking water risk levels.

A comparison of the assumed log reduction through the treatment trains considered to the 12/10/10 (virus/Giardia sp. / Cryptosporidium sp.) log reduction requirements was also calculated and is shown in Table 4.3. As shown in the table, only the pasteurized advanced treatment train in Utility 2 meets all the log reduction requirements currently in place for groundwater recharge for
IPR in California. Furthermore, if an ESB with free chlorine were used with any of the AWP waters, an additional 6-log reduction virus credit and 3-log reduction *Giardia* sp. credit would be achieved. Using an ESB with free chlorine along with the AWP waters in all utilities would achieve the log reduction requirement currently in place for groundwater recharge for IPR in California.

Key conclusions on pathogens are shown in Figure 4.29.

![Graph showing measured Giardia in different utilities](image-url)

**Figure 4.22 Measured *Giardia***
Figure 4.23 Measured Cryptosporidium
Figure 4.24 Measured total culturable virus
Figure 4.25 Measured enterovirus
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<th>Post simulated water treatment assumed value</th>
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<td>1.61 x 10$^{-5}$</td>
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Table 4.2 Continued

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<th>Utility</th>
<th>Water</th>
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<th>Post advanced treatment assumed value (AWP or O₃-BAC)</th>
<th>Post simulated water treatment assumed value</th>
<th>Risk level</th>
<th>Meets drinking water risk level?</th>
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(continued)
## Table 4.2 Continued

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<tr>
<th>Utility</th>
<th>Water</th>
<th>Measured value</th>
<th>Post advanced treatment assumed value (AWP or $O_3$-BAC)</th>
<th>Post simulated water treatment assumed value</th>
<th>Risk level</th>
<th>Meets drinking water risk level?</th>
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</table>

*Existing water supply meets drinking water risk goals.
†Represents all AWP waters that were blended with surface water or groundwater (depending on the utility) and underwent simulated water treatment.
‡Represents all $O_3$-BAF waters that were blended with surface water or groundwater (depending on the utility) and underwent simulated water treatment.
§Represents the $O_3$-BAF water that was not blended or further treated.
**Represents all past.-AWP waters that were blended with surface water or groundwater (depending on the utility) and underwent simulated water treatment.
††Represents all tertiary waters that were blended with surface water or groundwater (depending on the utility) and underwent simulated water treatment.
Figure 4.26 Calculated concentrations of enterovirus with assumed log removal

---

Simulated Water Treatment was not included in the 100% AWPPP-B water, but was included in the 90% raw / 10% AWPPP-B water.
Figure 4.27 Calculated concentrations of *Giardia* with assumed log removal.
Figure 4.28 Calculated concentrations of *Cryptosporidium* with assumed log removal

--- Simulated Water Treatment was not included in the 100% AWPPP-B water, but was included in the 90% raw / 10% AWPPP-B water.

---

**Figure 4.28 Calculated concentrations of *Cryptosporidium* with assumed log removal**
Table 4.3
Calculated log reduction for each blend

<table>
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<th>Utility</th>
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<th>Cryptosporidium log reduction</th>
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<td>O₃-BAC</td>
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</tr>
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<td>O₃-BAC</td>
<td>5</td>
<td>3</td>
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<td>Utility 2</td>
<td>AWP</td>
<td>10.5</td>
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<td>Past.-AWP</td>
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<td>18.5</td>
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<td>O₃-BAC</td>
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<td>6</td>
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</table>

Pathogens: Key Conclusions

- Pathogen levels in the advanced treatment source waters were measured. Starting with this data and applying assumed log reduction credits, both the pasteurized-AWP and AWP water meets allowable risk levels discussed in the NWRI DPR Framework (Independent Advisory Panel 2015).

- Additional treatment barriers can be readily added to meet pathogen based water quality goals to all treatment trains.

Figure 4.29 Pathogens key conclusions

Organic Carbon

Organic carbon is a key nutrient supporting the growth of microorganisms. Previous studies have demonstrated that recycled waters typically contain higher concentrations of organic carbon than traditional potable waters. TOC has been reported to range from 2 to 80 ppm in recycled water (Storey and Kaucner 2009, Thayanukul et al. 2013). Even when advanced water treatment is applied for water reuse, such as membrane bioreactors and granular activated carbon, organic carbon may be elevated (Thayanukul et al. 2013, Weinrich et al. 2010). While TOC and DOC have traditionally been monitored as a proxy for biostability of treated water during distribution, previous research has demonstrated that TOC and DOC are often not correlated with regrowth of bacteria during distribution. BDOC has been proposed as an alternative parameter for estimating
biostability of treated waters as it more directly represents the fraction of carbon available to microorganisms (Joret and Levi 1986; Ribas et al. 1991; Servais et al. 1987, 1989). TOC, DOC, and BDOC were measured in all blended water scenarios and are shown in Figure 4.30.

Figure 4.30 Total organic carbon (TOC), dissolved organic carbon (DOC), and biodegradable dissolved organic carbon (BDOC) concentrations in treated water for all utilities
For Utility 1, TOC and DOC were significantly higher in the 100 percent O₃-BAC and 90 percent surface/10 percent AWP conditions compared to the traditional potable 100 percent finished scenario (p<0.0001). With the exception of the 90 percent surface/10 percent O₃-BAC scenario, all Utility 1 treatment scenarios yielded significantly higher concentrations of BDOC than the 100 percent finished scenario (p≤0.0366), suggesting that all Utility 1 reuse schemes would create a somewhat greater potential for bacterial regrowth during distribution compared to the traditional potable system. When comparing only the 10 percent reuse blend condition for Utility 1, the biofiltration-based treatment train was more effective at reducing TOC, DOC, and BDOC than the membrane filtration-based treatment (p≤0.0031).

For the biofiltration conditions in Utility 1, decreasing the blend ratio did significantly decrease the TOC and DOC concentration (p<0.0001), but it did not significantly affect the BDOC concentration (p=0.1829). Blend ratio significantly affected the TOC, DOC, and BDOC in Utility 1 membrane filtration-based scenarios, with 90 percent surface/10 percent AWP significantly greater than the 50 percent surface/50 percent AWP scenario in all cases (p≤0.0059). The inverse relationship between ratio of membrane-filtered AWP water and BDOC in Utility 1 was unexpected, as the 90 percent surface/10 percent AWP not only had greater BDOC than the larger reuse blending ratio, but it also had significantly greater BDOC than that measured for the 100 percent surface condition (p<0.0001).

For Utility 2, both groundwater blends that were not pasteurized had significantly greater TOC than all other scenarios, including the 100 percent groundwater blend and both blends treated with pasteurization (p<0.0001). This suggests that the increased TOC resulted from blending with AWP-treated recycled water, but the low TOC measured in pasteurized conditions suggests pasteurization was sufficient to degrade or inactivate excess TOC.

Also for Utility 2, there were no significant differences between any of the scenarios when comparing DOC or BDOC concentrations of treated blends. The low abundance of BDOC suggests that these waters have limited capacity to support microbial regrowth, even when TOC is elevated.

For Utility 3, both reuse schemes produced TOC, DOC, and BDOC concentrations that were not significantly different than the 100 percent finished scenario (p=0.0706-0.9837). Blend ratio at this utility did not significantly impact TOC, DOC, or BDOC (p=0.0686-0.7870). With the exception of BDOC for the 90 percent surface/10 percent tertiary scenario (p=0.0530) at Utility 3 the TOC, DOC, and BDOC resulting from the treatment of 100 percent surface water were significantly greater than corresponding measurements for reuse and traditional potable schemes (p=0.0003-0.0243).

For Utility 4, TOC and DOC were significantly elevated in all conditions that included a surface water blend, compared to Utility 4 groundwater blends and blends containing recycled water (p≤0.0007). Notably, the 100 percent finished water condition at Utility 4 also had more TOC and DOC than both the 100 percent AWP and 100 percent industrial-AWP conditions (p<0.0001). Of the blends containing recycled water at Utility 4, TOC and DOC were both significantly greater in the 100 percent industrial-AWP water than the 100 percent AWP water (p≤0.0040). While the 100 percent finished scenario had the highest average BDOC, followed by the 100 percent industrial-AWP condition, there were no significant differences between BDOC concentrations of the Utility 4 scenarios.

A previous survey of four non-potable recycled water systems found average BDOC concentrations ranging from 400 to 6,200 ppb (Jjemba et al. 2010a). All recycled waters examined in this study were well below 2,000 ppb BDOC and the vast majority of measurements
were below 500 ppb, consistent with the high level of treatment employed, and comparable to concentrations of BDOC documented in the literature for potable waters. Previous studies found average BDOC concentrations for a single potable system ranged from 20 to 930 ppb (Charnock and Kjønno 2000, Ribas et al. 1991).

Key conclusions on organic carbon are shown in Figure 4.31.

Organic Carbon: Key Conclusions

- Reduction of TOC in secondary or filtered secondary effluent by recycled water processes is highly dependent upon the treatment processes employed, with RO-based purification expected to provide much greater removal than other processes. Though the non-RO processes in this study were unable to attain such low TOC levels, the addition of other processes, such as GAC, can lower TOC concentrations.

- In most cases concentrations of TOC, DOC, and BDOC were lower in AWP blends and higher in tertiary and O₃-BAC blends than in the corresponding surface water or groundwater used with each blend.

Figure 4.31 Organic carbon key conclusions

Perfluorinated Chemicals

Perfluorooalkyl substances (PFAs) are anthropogenic contaminants used in a wide array of industrial and consumer processes. They contain a completely fluorinated carbon chain of varying length and typically a charged end-group (e.g. carboxylic acid or sulfonic acid). The fluorinated chain gives the compound high stability and the polar end-group allows for solubility. Thus, PFAs are persistent throughout the aquatic environment and have been detected in wastewaters (Schultz et al. 2006), groundwaters (Post et al. 2009), surface waters (Hansen et al. 2002, Boulanger et al. 2004, Nakayama et al. 2007 and 2010), and drinking waters (Plumlee et al. 2008, Quinones and Snyder 2009, Post et al. 2012).

Wastewaters are sources of PFAs in the environment and in drinking water treatment plants (Sinclair and Kannan 2006). Conventional drinking water treatment is not effective in removing PFAs (Quinones and Snyder 2009), but select AWP treatments can be effective. RO is by far the most effective treatment with removal percentages of more than 90 percent of all chain length PFAs, but new granular activated carbon (GAC) is also effective in sorbing with chain lengths longer than 6 carbons (counted as the carbons with fluorines and not the carbon within the carboxylic acid group to be consistent with the sulfonate group) (Dickenson and Higgins 2016).
GAC treatment is limited by the sorption capacity of the carbon and over time PFAs compounds can be released into the effluent depending on the relative concentrations of the PFAS sorbed to the media and in the aqueous phase. In this project, all the GAC systems had been operated for long enough to become biological and therefore not have the capacity to remove PFAs. RO, and to a large extent nanofiltration, are effective for the removal of PFAs compounds (Tang et al. 2006b, Steinle-Darling and Reinhard 2008, Steinle-Darling et al. 2016).

In 2016, health advisory levels were released by the EPA limiting PFOA, PFOS, and the combined concentration of PFOA and PFOS to 70 ng/L. None of the samples tested in this project were found at concentrations above this limit. In fact, all the samples in Utility 2 and 4 were found at concentrations below the MRL, as observed in Figure 4.32. The 100 percent O₃-BAC samples of Utility 1 and 3 contained concentrations of 25 ng/L and 7.3 ng/L PFOA, respectively (Figure 4.32). Similarly, PFOS concentrations in the 100 percent O₃-BAC waters were 5.7 ng/L and 1.2 ng/L for Utilities 1 and 3, respectively. The surface water from Utility 3 also had detections of PFOS at an average of 0.31 ng/L, just above the MRL of 0.25 ng/L. Adding 5 percent and 10 percent tertiary effluent to the surface water from Utility 3 increased the concentration to 0.42 ng/L and 0.49 ng/L. Blending the 100 percent O₃-BAC waters with surface waters reduced the concentrations of PFOA and PFOS to below the MRL.

![Figure 4.32 PFOA (left) and PFOS (right) concentrations relative to the percentage of recycled water added for the four utilities; the EPA health advisory for PFOA and PFOS is well above the concentrations shown (70 ng/L)](image-url)
Detections of the other PFAs compounds were primarily of chain lengths < C8. Utility 1 100 percent O3-BAC had detections for PFBA (C4) of 7.1 ng/L, PFHxA (C6) of 19 ng/L, PFPnA (C5) of 9.7 ng/L, PFHxS (C6) of 2.2 ng/L, PFHpA (C7) of 3.3 ng/L, PFNA (C9) of 3.9 ng/L, PFDA (C10) of 1.7 ng/L, but PFUdA (C11) and PFDoA (C12) were below the MRL. In Utility 3, the blended O3-BAC at 50 percent SW/50 percent O3-BAC contained PFHxA at 4 ng/L, PFPnA 4.9 ng/L, and PFHpA at 0.74 ng/L. The tertiary effluent blended waters at 90 percent SW/10 percent tertiary had 1.5 ng/L of PFHxA and 0.37 ng/L of PFHxS. The bench-scale simulated treated waters and full-scale waters contained 0.61 and 0.66 ng/L of PFHxA, which means both the source waters and the advanced treated wastewaters contributed to the PFAs concentrations found in blended waters. Within Utilities 2 and 4, all the PFAs were below 1 ng/L and the detections were limited to compounds with chain lengths < C7.

Overall, the AWP treated waters contained lower concentrations of PFAs as compared to the tertiary effluent and O3-BAC treated waters. When blended, the PFAs behaved conservatively and their final concentrations depended on the source water and the advanced treated wastewater.

Key conclusions on PFAs are shown in Figure 4.33.

**PFAs: Key Conclusions**

- PFAs represent an emerging concern for water supplies and can be challenging to remove with conventional water treatment systems. PFAs act conservatively when blended.

- For both AWP applications and conventional drinking water sources, the two PFAs that are regulated (PFOA and PFOS) were well below their EPA health advisory levels.

- PFAs are not well removed by O3-BAC and tertiary waters. However no blend in this study exceeded the EPA health advisory level and additional treatment (e.g., GAC) can be used to further reduce PFAS.

**Figure 4.33 PFAs key conclusions**

**Pharmaceuticals and Personal Care Products, Pesticides, and Steroids**

Trace organic compounds (TOrCs) include PPCPs, pesticides, and steroids and typically occur at ng/L levels in drinking waters and surface waters (Benotti et al. 2009, Kolpin et al. 2002). The presence of TOrCs is often attributed to the influence of wastewater effluents (Barber et al. 2009, NRC 2012). These contaminants are currently not regulated by the EPA, but a few have been included in the Unregulated Contaminant Monitoring Rules, Notification levels, or state guidelines. AWP provides significant removal of TOrCs because of the physical barrier provided by RO for small molecules. UV-AOP adds an additional layer of protection and removal by two
mechanisms: direct photolysis via UV and indirect photolysis via the production of hydroxyl radical, a non-selective oxidant with which many compounds have reaction rates on the order of $10^9 \text{M}^{-1}\text{s}^{-1}$ (Buxton et al. 1988, Lee and von Gunten 2010). O3-BAC systems also provide TOx mitigation by several mechanisms (Gerrity et al. 2011, Reungoat et al. 2012). TOx can be directly oxidized by O3, but also by hydroxyl radicals that are produced by side reactions with the dissolved organic matter (Nöthe et al. 2009). Once the sorption capacity has been exhausted, TOx removal in BAC comes from biodegradation of compounds by bacteria that have colonized the carbon.

In Utilities 1 through 4, of the parameters listed in Table 3.2, acetaminophen, estrone, 17-alpha ethynylestradiol, 17-beta estradiol, ibuprofen, naproxen, progesterone, triclosan, triclocarban, trimethoprim, and testosterone were all below the MRL. Several other compounds were detected infrequently with atenolol only detected in Utility 3 with 3.2 ng/L in 90 percent SW/10 percent O3-BAC, 19 ng/L in the 50 percent SW/50 percent O3-BAC, and 1.2 ng/L in the 90 percent SW/10 percent tertiary effluent. Atenolol is a beta-blocker prescribed to assist cardiac diseases. Atenolol is removed by conventional wastewater treatment anywhere from 50 to 90 percent, (Radjenovic et al. 2009, Drewes et al. 2008), but is effectively degraded by ozonation and removed by RO at rates > 90 percent (Benotti et al. 2009, Drewes et al. 2008, Gerrity et al. 2011). The limited detection of atenolol in Utilities 1, 2, and 4 can be attributed to the efficacy of these treatments.

Caffeine is the most commonly ingested stimulant on the market and is readily biodegraded in conventional wastewater treatment at a rate > 80 percent (Buerge et al. 2003), but can still be found in effluent waters due to the high $\mu$g/L influent levels. Caffeine can be further degraded by ozonation with a removal rate of > 80 percent (Snyder et al. 2006, Broséus et al. 2009). In this project, caffeine was detected in Utility 3 at 8.3 ng/L in 90 percent SW/10 percent tertiary effluent, at 8.4 ng/L in 90 percent SW/10 percent O3-BAC, and 18 ng/L in 50 percent SW/50 percent O3-BAC. Again, these detections are most likely related to the high influent concentrations.

Fluoxetine is an antidepressant commonly known by the trade name Prozac®. Detections were found in the tertiary and O3-BAC blended waters of Utility 3, but were all limited to concentrations below 3 ng/L. Removal efficiencies via biodegradation and ozonation of fluoxetine are expected to be >90 percent (Drewes et al. 2008), which resulted in blended concentrations close to the MRL of 0.5 ng/L.

Gemfibrozil was only detected once in the 100 percent O3-BAC of Utility 1 at a concentration of 1.7 ng/L. Gemfibrozil is prescribed for the treatment of high cholesterol. It is another compound considered to be well-removed by biodegradation and ozonation (Drewes et al. 2008).

Overall, increases in TOx concentrations were attributed to tertiary effluent and O3-BAC treated water and not to the AWP systems. This makes sense given the efficacy of RO in removing many compounds and the high reaction rates of most of these compounds with hydroxyl radicals produced in UV/AOP.

The TOx in this project with multiple detections per utility included DEET, TCEP, primidone, sucralose, and meprobamate. DEET is a common insect repellent. DEET is most likely to enter wastewater systems through surface water runoff or the use of DEET on skin and clothing. Detections of DEET are shown in Figure 4.34 with the highest concentrations observed in Utility 3 with the O3-BAC and tertiary effluent water negatively impacting the water quality of the SW. O3-BBAC treated water had a similar impact for Utility 1. The addition of AWP treated waters from Utility 1 decreased the overall concentration of DEET in the SW blend. However, the AWP
treated waters of Utility 4 contributed to increases in the concentration of DEET in the blended waters. Biodegradation of DEET in activated sludge and ozonation of DEET in wastewater is varied at 50 to 90 percent, but RO and UV/AOP processes are both capable of removing more than > 90 percent (Drewes et al. 2008, Sui et al. 2010). The lower removal from O3-BAC and tertiary wastewater treatment explains the contribution of these waters to increase blended concentrations of DEET.

![Figure 4.34 DEET concentration changes relative to the percentage of recycled water added for the four utilities](image)

TCEP acts as a flame retardant in plastics, foams, and textiles. Due to concerns regarding its carcinogenicity upon ingestion, Minnesota has developed a guidance value of 5,000 ng/L for drinking water (MDH 2011). As compared to other ToxCs, TCEP is recalcitrant to biodegradation and oxidation by ozonation or UV/AOP with removals generally < 25 percent (Benotti et al. 2009, Drewes et al. 2008, Gerrity et al. 2011); however, RO provides a significant barrier with removals expected to be > 90 percent (Drewes et al. 2008). As seen in Figure 4.35, all waters were below the guidance value from Minnesota, but the addition of O3-BAC waters from Utilities 1 and 3 raised the concentration of TCEP in both the surface waters. Although difficult to observe in Figure 4.35, the blended 90 percent SW/10 percent tertiary effluent resulted in a significant increase in the concentration of TCEP from the surface water. The groundwater samples and AWP samples from Utility 2 did not contain any detections of TCEP. In Utility 4, the addition of AWP water from industrial food and beverage production contributed to detections of TCEP in the blended waters. However, a higher percentage of recycled AWP water could be added before increasing the concentration to the same level as the O3-BAC or tertiary effluent.
Primidone is prescribed to treat seizure disorders and acts as an anticonvulsant. It has low activated sludge removals, generally < 25 percent (Yang et al. 2011), but can be removed via O$_3$ with rates ranging from 50 to 90 percent and RO at a rate > 90 percent (Drewes et al. 2008, Gerrity et al. 2011). Because of these removal rates, the concentration of primidone was increased in the surface waters blended with O$_3$-BAC and tertiary effluents in Utility 3, as shown in Figure 4.36. In Utility 1, the concentrations of primidone were similar in the O$_3$-BAC, surface water, and the AWP water meaning that the blended waters did have a change in the primidone concentration (within analytical error of 10 percent).
Sucralose is an artificial sweetener that is not broken down by the body, making it a non-caloric additive. Sucralose is highly recalcitrant to degradation in conventional wastewater, drinking water treatment, and the environment (Scheurer et al. 2010, Torres et al. 2011). This resistance to degradation combined with a near ubiquitous detection in municipal wastewaters (Buerge et al. 2009) allows sucralose to be a good indicator of wastewater-impact (Oppenheimer et al. 2011). O3-BAC systems have a limited capacity to degrade sucralose with a range of 25 to 50 percent removal, but RO has high rejection rates >90 percent (Lee et al. 2010). This is illustrated in Figure 4.37 which shows the sucralose concentrations under various blending scenarios. All the advanced treated wastewaters without RO treatment add significant levels of sucralose to their blends due to the high concentrations found in tertiary and secondary effluents.
Figure 4.37 Sucralose concentration changes relative to the percentage of recycled water added for the four utilities

Meprobamate is an antianxiety drug often found in conventional wastewater effluents because of its resistance to biodegradation (Drewes et al. 2008). Under advanced treatments, it has varied removals for O$_3$-BAC, 50-90 percent (Gerrity et al. 2011), and significant removal during RO and UV/AOP (Drewes et al. 2008). As with sucralose, the addition of Utility 3 tertiary effluent and O$_3$-BAC and the addition of Utility 1 O$_3$-BAC increased the concentration of meprobamate in the blended waters, as shown in Figure 4.38. On their own, groundwaters and surface waters contained low concentrations of meprobamate, AWP treated waters were below the detection limit, and blends with AWP waters improved the water quality with respect to meprobamate.
Figure 4.38 Meprobamate concentration changes relative to the percentage of recycled water added for the four utilities

The concentration of TOréCs in potable recycled waters can vary widely depending on the wastewater effluent and specific behavior in advanced treatment processes. TOréCs in this project all behaved conservatively with dilution dictating the final concentrations in blended waters. Surface waters had detections of sucralose, primidone, meprobamate close to their respective MRLs, but groundwater did not contain any detections of TOréCs. All the TOréCs tested were effectively removed by RO and subsequently improved the water quality of its blend. However, TOréCs were inconsistently removed by O₃-BAC systems, which resulted in blended waters from Utilities 1 and 3 containing increased concentrations as compared to their source water. Despite these increases, TOréC concentrations found were not found above notification levels, guideline values, or regulations in the U.S.

Key conclusions on TOréCs are shown in Figure 4.39.
Figure 4.39 TOrCs key conclusions

- AWP blends had the fewest TOrCs above the detection limit.
- While O₃-BAC blends had the most detections of TOrCs, none were above known health guidance values.
- The TOrCs measured acted conservatively when blended.
CHAPTER 5
DISTRIBUTION SYSTEM

BACKGROUND

This section compares the water quality of each blend as measured both before and after the water blend is conditioned in a pipe rig to simulate premise plumbing. Comparing the water quality of an individual blend both before and after it is conditioned in this simulated premise plumbing provides insight into how a blend's finished water quality may change before it reaches a customer. It also provides insight into how the water blend affects the premise plumbing piping. Furthermore, looking across all water sources and blends, inferences can be made regarding the most effective blend ratios.

RESULTS

For each utility, the blended and unblended simulated treated water was generated each month of the two-month experimental period. Post-disinfection, water was aged in the pipe rigs and changed out with fresh samples every 2 to 3 days. Samples for “pipe-in” and “pipe-out” were collected from duplicate parallel pipe rigs at the beginning and end of weeks 6 and 8. This data will also be referred to as “influent” and “effluent.”

Nutrients

Biological nutrient removal processes during wastewater treatment prevent nitrogen and phosphorus accumulation downstream or in the case of direct potable reuse, in finished drinking water. If biological nutrient removal is not able to reduce the concentrations of ammonia or if chloramines are added and degrade to ammonia, nitrification can happen within the distribution system (Routt and Skadsen 2013). When this occurs, nitrifying bacteria proliferate and use ammonia to produce nitrate and negatively impact water quality by depleting the residual, promoting corrosion issues in pipes, and decreasing pH and DO concentrations (Routt and Skadsen 2013). Additionally, excess nitrate in finished drinking water can have health implications, especially for pregnant women and babies, which is why the EPA established a limit of 10 mg/L of nitrate as N (EPA 1992).

For all utilities, results show the ammonia concentrations between the pipe in and out did not change significantly (Figure 5.1) indicating that nitrification most likely did not occur. In general, Utilities 1, 2, and 4, which all applied chloramines, contained higher concentrations of ammonia as compared to Utility 2 where chlorine was used.
The average nitrate concentrations similarly did not change significantly from pipe-in to pipe-out for all the utility samples, as observed in Figure 5.2. This indicates that no biotic or abiotic conditions were present to significantly impact the concentrations of nitrate.
Phosphorus concentrations also did not change significantly during the incubation period in the pipe rigs, as can be seen in Figure 5.3. The highest observed concentration of phosphorus was found in the Utility 4, 90 percent SW/10 percent AWP sample where zinc orthophosphate was dosed to prevent metal leaching.
Results suggest that blending high ratios of advanced treated wastewater (primarily O3-BAC) can impact the nutrients released into the distribution system. The values were not significantly different than those observed in the blended, simulated treated waters with the differences between utilities attributed to the type of source water, advanced treatment (AWP or O3-BAC), and the upstream wastewater treatment processes. However, elevated ammonia levels could increase for higher retention times; prolonged storage and low flow conditions can be critical factors that cause chloramines to break down into ammonia, potentially leading to nitrification.

Key conclusions on nutrients are shown in Figure 5.4.
Nutrients: Key Conclusions

- No significant changes were observed in nutrient levels pre and post pipe rig simulations for all blends (48 hour retention time).
- No nitrification was seen over this short duration of testing.

Figure 5.4 Nutrients key conclusions

Bulk Water Quality

The bulk water quality parameters measured in the distribution system simulated tests included pH, alkalinity, chloride, sulfate, and ATP. Water released into a drinking water distribution system can influence the types of bacteria that populate it and the levels of corrosion and scale. Within the distribution system, changes in these water quality parameters may be indicators of metal leaching and bacterial proliferation as well as influence the formation of DBPs, including TTHMs and HAAs. However, various water quality parameters can have contrasting effects and thus it is important to consider them individually and as a whole.

Shifts in pH can impact corrosion, leaching of metals, and disinfection efficacy within the distribution system (Singer and Reckhow 2011). However, these changes may also be impacted by alkalinity, chloride, and sulfate due to the changes in protective scale. Lower pH waters can accept electrons from the metals (lead, copper, and zinc) corroded from piping and this process is depressed when the pH is above 9. The pH values seen in Utility 1 for the 90 percent SW/10 percent AWP water had a slight decrease from 7.46 to 7.22 (Figure 5.5). Similarly, in Utility 4 the 90 percent SW/10 percent AWP blended water decreased by 0.2 pH units from the influent to the effluent. In Utility 2, none of the waters had a significant change from the influent to the effluent. The unblended SW sample from Utility 3 was the only test water that increased in the pH from influent to effluent (from 7.6 to 8). Slight changes in three of the pipe rig samples most likely had a limited impact on metal leaching or corrosion. Bacterial growth within distribution systems can potentially reduce the pH, but a significant drop was not observed in any of the test waters. A description of the types of bacteria found in the pipe effluent can be found in the Bacterial Corrosion section.
For chlorine disinfection, which was applied to Utility 3, the optimal pH for germicidal efficacy would be below 7.6, where HOCl is the dominant species. However, Utility 3 was operated at pH > 7.6, which most likely required a longer CT as compared to a system with a lower pH. The preferred species for disinfection with chloramines is monochloramine, which is dominant at a specific chlorine to ammonia ratio and a pH between 7.5 and 9. Utilities 1, 2, and 4 were all within this pH range. The changes in disinfectant residual will be discussed further in the sections below.

Changes in the pH can alter the yield and stability of TTHMs and HAAs (Singer and Reckhow 2011). At higher pH values, the reactive groups within DOM are deprotonated, which leads to increased yields, but the OCl\(^-\) and OBr\(^-\) species present at higher pH values are less reactive than their protonated counterparts. The formation of TTHMs is favored at high pH due to the influence of base hydrolysis whereas HAA formation is favored at lower pH values through oxidation (Singer and Reckhow 2011). In Utility 3 100 percent SW, where the largest shift in pH was observed (from 7.6 to 8.0), a corresponding increase in TTHMs may have occurred relative to the same water with no pH change. However, other factors, such as the reactivity of the DOM present, ratio of DOM and bromide, and the contact time with the disinfectant may have significantly greater impacts on the overall formation of TTHMs and HAAs.

![Figure 5.5 Influent and effluent pH in all four utilities under various blending scenarios; blue, dotted line differentiates between scenarios a and b for Utility 4](image)

Changes in alkalinity, chloride, and sulfate were minimal between the influent and effluent samples as shown in Figures 5.6, 5.7, and 5.8. Changes in drinking water quality from blending
can lead to disruptions in the scale and cause aesthetic changes to the water quality and the release of metals such as lead, copper, and zinc. The metal leaching can change depending on the mass ratio of chloride to sulfate. Additionally, the concentration of alkalinity is an important predictor for the release of lead in the drinking water distribution network (Nguyen et al. 2010). This will be discussed further in the metals section below where these values are used in corrosion indices.

Key conclusions on bulk water quality are shown in Figure 5.9.

**Figure 5.6** Influent and effluent chloride concentrations in all four utilities under various blending scenarios; blue, dotted line differentiates between scenarios a and b for Utility 4.
Figure 5.7 Influent and effluent sulfate concentrations in all four utilities under various blending scenarios; blue, dotted line differentiates between scenarios a and b for Utility 4.
Figure 5.8 Influent and effluent alkalinity concentrations in all four utilities under various blending scenarios; blue, dotted line differentiates between scenarios a and b for Utility 4.
Disinfection Residual Stability

Utilities 1, 2, and 4 applied chloramines as disinfectants and Utility 3 applied chlorine. Despite the differences in starting dose, the consumption of the disinfectant varied across all the waters tested. In addition, ATP concentrations in the influent waters were all <5 pg/mL, which reflects the efficacy of disinfection and the limited active bacteria present in the initial starting water (Figure 5.10). However, post-storage in the pipe rigs, ATP effluent concentrations (with the exception of Utility 2) were significantly higher ($p < 0.001$). In Utilities 2 and 4, the final effluent concentrations were all < 10 pg/mL, indicating a low level of activity and the efficacy of higher chloramine doses applied in these waters. This was also reflected by the small decrease in total chlorine. In Utility 1, the concentration in the blended 90 percent SW/10 percent AWP had a low dose and had a noticeable increase in ATP as compared to the other samples in this utility dosed at the same level. This corresponded to the decrease in total chlorine as observed in Figure 5.10. The 95 percent SW/5 percent tertiary effluent and the 90 percent SW/10 percent tertiary effluent both had elevated levels of ATP in the blends. Again, this increase was reflected by a decrease in the total chlorine concentrations. Most likely this is related to the higher relative demand as compared to other waters in this utility, which is anticipated for the lower level of treatment applied to the tertiary effluent.

Key conclusions on disinfection residual stability are shown in Figure 5.11.
Figure 5.10 Influent and effluent total chlorine and ATP concentrations in all four utilities under various blending scenarios; blue, dotted line differentiates between scenarios a and b for Utility 4.
Disinfection byproducts

Depending on the microbial activity, disinfectant residual, pH, temperature, bromide concentrations, and precursors, DBPs continue to form or may degrade within the distribution system or in storage tanks. Due to the long reaction times between free chlorine and precursors (e.g., DOM), THMs continue to form and have a positive association with residence time and distance traveled within the distribution system (Reckhow and Singer 1990). HAA₅s can follow this trend, but can also biodegrade in distribution systems and have a positive association with low disinfectant residuals (e.g. < 0.3 mg/L) (Baribeau et al. 2006; Tung and Xie 2009).

The concentrations of TTHMs were highest in Utility 3, despite the similar levels of TOC to Utility 1 (Figure 5.12). This can be attributed to the application of free chlorine, which has significantly higher yields for both TTHMs and HAA₅s as compared to chloramines. In all the waters of Utility 3, a significant increase from influent to effluent was in the SW and 95 percent SW/5 percent tertiary effluent (p <0.001). An increase was observed in the two O₃-BAC blended waters in Utility 3 and their final concentrations exceeded the EPA regulation of 80 μg/L (EPA 2010). The effluent concentrations in Utilities 1, 2, and 4 were not significantly different from their respective influent waters. In Utility 1, the AWPAAWP blended waters saw decreases in TOC of approximately 1 mg/L over the storage period. This level of potential biodegradation was consistent with the increases in ATP observed in Figure 5.12.
Figure 5.12 Influent and effluent TTHMs and TOC concentrations in all four utilities under various blending scenarios; blue, dotted line differentiates between scenarios a and b for Utility 4; red dashed line is the maximum contaminant limit for TTHMs; red and blue dots are TOC values for the influent and effluent, respectively

In general, HAA5 concentrations were lower than the TTHM concentrations. Significant changes were observed in several of the waters from Utilities 1 and 3 (p < 0.001). The limited difference can be attributed to the contrasting effects of biodegradation of HAA5 and the increases in concentration due to presence of precursors and disinfectant residual. The biodegradation of HAA5 was observed in the 90 percent SW/10 percent AWP blended water where the ATP concentration was highest (Figure 5.13) and the total chlorine residual was < 0.3 mg/L. This was the only sample in which a decrease in the HAA5 concentration was observed. The waters in Utility 2 did not have a significant change between the influent and effluent samples and once again the blended waters contained lower concentrations of HAA5 as compared to the GW sample. The most significant increase in HAA5 from influent to effluent was observed in the Utility 3
samples containing 100 percent SW and 95 percent SW/5 percent tertiary effluent, in which chlorine was used as the secondary disinfectant.

![Diagram showing HAA5 concentrations for different utilities and blending scenarios.]

**Figure 5.13 Influent and effluent HAA5s in all four utilities under various blending scenarios; blue, dotted line differentiates between scenarios a and b for Utility 4**

The connection between ATP, disinfectant residual, TOC, and HAA degradation was demonstrated in the four utilities. ATP as a surrogate for microbial activity predicted the sample that had biodegradation of HAA5. The TOC concentration was also impacted by microbial activity with mineralization occurring in two of the Utility 1 waters. Utility 3 demonstrated the impact that free chlorine vs. chloramines has on increasing the relative concentration of both regulated DBPs. With the exception of two waters from Utility 3, all of the samples were below the regulations of 80 and 60 \( \mu \text{g/L} \) for TTHMs and HAA5s, respectively.

Key conclusions on disinfection byproducts are shown in Figure 5.14.
Corrosion Potential

The corrosivity of blended waters from the four different utilities to CDA360 was also evaluated. Corrosivity was first screened using imperfect ‘corrosion indices’ as a means of highlighting potentially significant chemical differences in each water (Tables 5.1 through 5.5). Our hypothesis was that most of these indices would not be a good indicator of lead, brass, or dezincification corrosion since they were mostly developed for ductile iron pipe infrastructure. One exception is ‘Turner’s Ratio,’ which has been used to predict hot water brass dezincification and CSMR which sometimes describes trends in cold water brass leaching. Comparisons to experimental results for test-rig lead, copper, and zinc leaching data are then addressed in the sections that follow.

Langelier Saturation Index (LSI)

The most commonly used corrosion index is the Langelier Saturation Index, or LSI (AwwaRF and DVGW-TZW 1996). It is based on the reaction of calcium and bicarbonate to form a calcium carbonate solid. If this solid is predicted to form (i.e., LSI is positive), then the water is considered non-corrosive. A negative LSI indicates the water is corrosive. LSI can be calculated using Equations 5.1 and 5.2:

\[
\text{LSI} = \text{pH}_{a} - \text{pH}_{s} \tag{5.1}
\]

where pH\text{a} is the water pH and pH\text{s} is the pH at calcium carbonate saturation.
\[ \text{pH}_s = \text{pK} - \log[\text{Ca}^{2+}] - \log[\text{HCO}_3^-] - \log \gamma_{\text{Ca}^{2+}} - \log \gamma_{\text{HCO}_3^-} \quad (5.2) \]

where \( \text{pK} \) is \(-\log(K)\) for the reaction of calcium and bicarbonate to form calcium carbonate, \([\text{Ca}^{2+}]\) and \([\text{HCO}_3^-]\) are the molar concentrations of calcium and bicarbonate, and \( \gamma \) is the activity coefficient.

While this approach may sound good in theory, many limitations must be considered (AwwaRF and DVGW-TZW 1996). These include having a good estimate of ionic strength (to accurately estimate the activity coefficients), accounting for the complexation of calcium and/or bicarbonate with other substances in the water, and knowing whether calcite or another polymorph of calcium carbonate is forming. In addition, the deposition of calcium carbonate does not necessarily form a stable, protective scale and therefore may not aid in corrosion prevention.

For Utility 1, all water quality conditions are predicted to be slightly corrosive based on LSI values (Table 5.1), with the 50 percent blend being the most corrosive. For Utility 2, only the 50 percent blend should be slightly corrosive (Table 5.2). None of the conditions at Utility 3 should be corrosive (Table 5.3). At Utility 4a, only the 10 percent blend with surface water is predicted to be slightly corrosive (Table 5.4). Finally, at Utility 4b both water conditions tested should be slightly corrosive (Table 5.5).

**Ryznar Stability Index (RSI)**

The Ryznar Stability Index (RSI) is similar to the LSI in that its basis is the saturation of calcium carbonate. It was developed for mild steel by using a database of scale thickness and associated water quality parameters (Roberge 2007). RSI can be calculated using Equation 5.3:

\[ \text{RSI} = 2(\text{pH}_s) - \text{pH}_a \quad (5.3) \]

The criteria for determining corrosivity are as follows:

- \( \text{RSI} < 6 \) not corrosive
- \( \text{RSI} > 7 \) calcium carbonate may form but not lead to a protective scale
- \( \text{RSI} > 8 \) mild steel corrosion becomes a problem

For Utility 1, all water quality conditions are predicted to be highly corrosive based on the RSI since all values are greater than 9 (Table 5.1). At Utility 2, the two 50 percent blends tested should be slightly corrosive compared to the 0 percent and 10 percent blends (Table 5.2). For Utility 3, all RSI values are just above 7, so corrosion is not expected to be an issue in any of the conditions tested (Table 5.3). At Utility 4a, the 10 percent blend with surface water is predicted to be corrosive (Table 5.4), while at Utility 4b both conditions tested are expected to have corrosion problems (Table 5.5).

**Larson-Skold Index**

The Larson-Skold Index is also widely used to illustrate the corrosiveness of water to mild steel. Larson and Skold found that the corrosion was related to the concentrations of sulfate and chloride relative to bicarbonate (Roberge 2007). This index is calculated using Equation 5.4:

\[ \text{Larson-Skold Index} = \frac{(2 \times [\text{SO}_4^{2-}] + [\text{Cl}^-])}{[\text{HCO}_3^-]} \quad (5.4) \]
where brackets indicate the concentrations in mole/liter. If the index is less than 0.8, the water is non-corrosive; if the index is between 0.8 and 1.2, the water is corrosive; and if the index is greater than 1.2, then the water is highly corrosive (Roberge 2007).

All values of the Larson-Skold Index for Utility 1 are well above the 1.2 threshold for highly corrosive waters, and the index is greater for the two conditions with blended recycled water than for the unblended or the water blended with O₃-BAC treated water (Table 5.1). For Utility 2 all water conditions are predicted to be highly corrosive (Table 5.2). Likewise, all Utility 3 water conditions are predicted to be highly corrosive (Table 5.3). For Utility 4a, the unblended and 10 percent blend with groundwater are predicted to be corrosive, while the blended with 10 percent surface water is predicted to be highly corrosive (Table 5.4). Both waters tested at Utility 4b are predicted to be highly corrosive per the Larson-Skold Index (Table 5.5).

**Chloride-to-Sulfate Mass Ratio**

While not an established corrosion index per se, the CSMR can be an indicator that conditions are right for galvanic corrosion of lead. Galvanic corrosion occurs when dissimilar metals are in contact in the presence of an electrolyte. For premise plumbing, this means lead can corrode from leaded brass or lead-containing solder when the CSMR of the water is high. One study established the threshold of galvanic corrosion at a CSMR of 0.6 mg chloride per mg sulfate (Edwards and Triantafyllidou 2007, Gregory 1990, Oliphant 1983).

All values of the CSMR for Utility 1 are well above the 0.6 threshold for galvanic corrosion, and the CSMR is greater for the two conditions with blended recycled water than for the unblended or the water blended with O₃-BAC treated water (Table 5.1). All water conditions at Utilities 2 and 3 have CSMR values below the 0.6 threshold (Tables 5.2 and 5.3). For Utility 4a, the unblended and 10 percent blend with groundwater should not be susceptible to galvanic corrosion, while the blended with 10 percent surface water is predicted to be highly affected (Table 5.4). Both waters tested at Utility 4b are predicted to be susceptible to galvanic corrosion (Table 5.5).

**Turner’s Ratio**

The ratio of chloride concentration to ‘temporary hardness’ has been widely cited as a key parameter for dezincification to occur (AwwaRF and DVGW-TZW 1996). Dezincification is a form of corrosion where zinc is preferentially removed from brass. ‘Temporary hardness’ refers to the concentration of hardness (i.e., calcium and magnesium) that is removed upon heating, and is a complex function of alkalinity and hardness. Recent work has provided insight into the role of alkalinity and chloride in galvanic corrosion (Edwards and Dudi 2004, Dudi et al. 2005, Triantafyllidou and Edwards 2007). Lead, the less noble metal in brass, is sacrificed and the pH at its surface can drop substantially relative to the bulk water pH. Higher chloride can enhance the lead corrosion as it is drawn to the lead surface to provide electroneutrality, while alkalinity can be beneficial since it can buffer the change in pH at the lead surface.

Turner did not propose a ratio of chloride to alkalinity at which dezincification would likely occur. Rather, he plotted the ratios on a graph and indicated in which water supplies dezincification was occurring. The relationship was not linear; however, at levels of ‘temporary hardness’ less than 100 mg/L, the ratio of chloride-to-‘temporary hardness’ was about 0.5 (i.e., dezincification occurred when ratio was greater than 0.5). At higher levels of ‘temporary hardness’ the ratio
increased towards 1.0. Turner’s results were also based on visual observations within a limited geographical area, and the term, ‘temporary hardness’ is ambiguous as to whether the benefits are from hardness or alkalinity or a combination of the two. Recent work seems to indicate alkalinity is the prime candidate for the reduction of dezincification so we will use that ratio in this study to evaluate the propensity of each water condition for dezincification (Zhang 2009).

Similar to the other corrosion indices discussed above, all Utility 1 waters appear to be susceptible to dezincification due to their Turner Ratios being greater than 0.9 at relatively low alkalinites (Table 5.1). For Utility 2, each water condition has a Turner Ratio of 0.4 at relatively high alkalinity, indicating that dezincification is not likely (Table 5.2). For Utility 3, the Turner Ratios indicate borderline propensity for dezincification (Table 5.3). For Utility 4a, the unblended and 10 percent blend with groundwater should not be susceptible to dezincification, while the blend with 10 percent surface water is predicted to be moderately affected (Table 5.4). Both waters tested at Utility 4b are borderline to be susceptible to dezincification (Table 5.5).

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<tr>
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<td>58.7</td>
<td>40.5</td>
<td>59.8</td>
</tr>
<tr>
<td>T-Phosphate as P</td>
<td>mg/L</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Hardness, Total</td>
<td>mg/L</td>
<td>146.7</td>
<td>130.0</td>
<td>72.5</td>
<td>140.0</td>
</tr>
<tr>
<td>Calcium</td>
<td>mg/L</td>
<td>29.3</td>
<td>23.5</td>
<td>13.5</td>
<td>27.7</td>
</tr>
<tr>
<td>Free chlorine</td>
<td>mg/L</td>
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<td>0.0</td>
<td>0.0</td>
<td>0.9</td>
</tr>
<tr>
<td>Total chlorine</td>
<td>mg/L</td>
<td>1.4</td>
<td>0.9</td>
<td>1.0</td>
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</tr>
<tr>
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<td>-0.9</td>
<td>-1.1</td>
<td>-1.8</td>
<td>-0.9</td>
</tr>
<tr>
<td>RSI</td>
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<td>9.2</td>
<td>9.6</td>
<td>10.7</td>
<td>9.1</td>
</tr>
<tr>
<td>Larson-Skold Ratio</td>
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<td>2.7</td>
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<td>5.1</td>
<td>2.0</td>
</tr>
<tr>
<td>CSMR</td>
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<td>2.6</td>
<td>2.5</td>
<td>1.4</td>
</tr>
<tr>
<td>Turner Ratio</td>
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### Table 5.2
Utility 2 raw water quality parameters and corrosion indices

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<th>Parameter</th>
<th>Units</th>
<th>100% groundwater</th>
<th>90% groundwater / 10% AWP</th>
<th>50% groundwater / 50% AWP</th>
<th>50% groundwater / 50% past-AWP</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH</td>
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<td>7.6</td>
<td>7.6</td>
<td>7.3</td>
<td>7.5</td>
</tr>
<tr>
<td>Conductivity</td>
<td>µS/cm</td>
<td>2,046.7</td>
<td>1,823.3</td>
<td>1,120.0</td>
<td>1,145.0</td>
</tr>
<tr>
<td>Alkalinity, Total</td>
<td>mg/L</td>
<td>283.8</td>
<td>252.0</td>
<td>151.0</td>
<td>158.5</td>
</tr>
<tr>
<td>Chloride</td>
<td>mg/L</td>
<td>105.0</td>
<td>96.3</td>
<td>60.3</td>
<td>59.5</td>
</tr>
<tr>
<td>Sulfate</td>
<td>mg/L</td>
<td>718.3</td>
<td>653.3</td>
<td>358.3</td>
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<td>T-Phosphate as P</td>
<td>mg/L</td>
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<td>mg/L</td>
<td>735.0</td>
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<tr>
<td>Calcium</td>
<td>mg/L</td>
<td>190.0</td>
<td>175.0</td>
<td>95.0</td>
<td>97.5</td>
</tr>
<tr>
<td>Free chlorine</td>
<td>mg/L</td>
<td>0.1</td>
<td>0.1</td>
<td>0.2</td>
<td>0.1</td>
</tr>
<tr>
<td>Total chlorine</td>
<td>mg/L</td>
<td>2.6</td>
<td>2.5</td>
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</tr>
<tr>
<td>LSI</td>
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<td>0.5</td>
<td>-0.2</td>
<td>0.0</td>
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<tr>
<td>RSI</td>
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<td>6.4</td>
<td>6.5</td>
<td>7.7</td>
<td>7.5</td>
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<tr>
<td>Larson-Skold Ratio</td>
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<td>3.2</td>
<td>3.2</td>
<td>3.0</td>
<td>2.9</td>
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<tr>
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<td>0.1</td>
<td>0.1</td>
<td>0.2</td>
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<tr>
<td>Turner Ratio</td>
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### Table 5.3
Utility 3 raw water quality parameters and corrosion indices

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<tr>
<th>Parameter</th>
<th>Units</th>
<th>95% surface / 5% tertiary</th>
<th>90% surface / 10% tertiary</th>
<th>90% surface / 10% O3- BAC</th>
<th>50% surface / 50% O3- BAC</th>
<th>Untreated surface</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH</td>
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<td>8.0</td>
<td>8.0</td>
<td>8.0</td>
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<td>7.9</td>
</tr>
<tr>
<td>Conductivity</td>
<td>µS/cm</td>
<td>1,029.5</td>
<td>1,052.5</td>
<td>1,066.7</td>
<td>1,356.7</td>
<td>972.2</td>
</tr>
<tr>
<td>Alkalinity, Total</td>
<td>mg/L</td>
<td>138.2</td>
<td>138.0</td>
<td>177.2</td>
<td>144.2</td>
<td>142.7</td>
</tr>
<tr>
<td>Chloride</td>
<td>mg/L</td>
<td>95.4</td>
<td>102.8</td>
<td>108.3</td>
<td>161.7</td>
<td>89.2</td>
</tr>
<tr>
<td>Sulfate</td>
<td>mg/L</td>
<td>236.2</td>
<td>243.8</td>
<td>246.7</td>
<td>295.0</td>
<td>229.2</td>
</tr>
<tr>
<td>T-Phosphate as P</td>
<td>mg/L</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Hardness, Total</td>
<td>mg/L</td>
<td>266.7</td>
<td>286.8</td>
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</tr>
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<td>Calcium</td>
<td>mg/L</td>
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<td>73.0</td>
<td>79.5</td>
<td>88.5</td>
<td>73.6</td>
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<tr>
<td>Free chlorine</td>
<td>mg/L</td>
<td>1.3</td>
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<td>0.6</td>
<td>0.2</td>
<td>1.3</td>
</tr>
<tr>
<td>Total chlorine</td>
<td>mg/L</td>
<td>1.5</td>
<td>1.3</td>
<td>1.4</td>
<td>0.9</td>
<td>1.4</td>
</tr>
<tr>
<td>LSI</td>
<td></td>
<td>0.3</td>
<td>0.3</td>
<td>0.5</td>
<td>0.2</td>
<td>0.3</td>
</tr>
<tr>
<td>RSI</td>
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<td>7.4</td>
<td>7.1</td>
<td>7.3</td>
<td>7.4</td>
</tr>
<tr>
<td>Larson-Skold Ratio</td>
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<td>2.9</td>
<td>2.3</td>
<td>3.7</td>
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<tr>
<td>CSMR</td>
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<td>0.4</td>
<td>0.4</td>
<td>0.5</td>
<td>0.4</td>
</tr>
<tr>
<td>Turner Ratio</td>
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<td>0.6</td>
<td>1.1</td>
<td>0.6</td>
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### Table 5.4
Utility 4a raw water quality parameters and corrosion indices

<table>
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<tr>
<th>Parameter</th>
<th>Units</th>
<th>100% groundwater</th>
<th>90% groundwater / 10% AWP</th>
<th>90% surface / 10% AWP</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH</td>
<td></td>
<td>8.0</td>
<td>7.9</td>
<td>8.0</td>
</tr>
<tr>
<td>Conductivity</td>
<td>µS/cm</td>
<td>727.5</td>
<td>701.8</td>
<td>514.8</td>
</tr>
<tr>
<td>Alkalinity, Total</td>
<td>mg/L</td>
<td>201.3</td>
<td>185.3</td>
<td>97.5</td>
</tr>
<tr>
<td>Chloride</td>
<td>mg/L</td>
<td>30.9</td>
<td>40.1</td>
<td>71.4</td>
</tr>
<tr>
<td>Sulfate</td>
<td>mg/L</td>
<td>117.5</td>
<td>104.3</td>
<td>43.9</td>
</tr>
<tr>
<td>T-Phosphate as P</td>
<td>mg/L</td>
<td>0.1</td>
<td>0.0</td>
<td>0.2</td>
</tr>
<tr>
<td>Hardness, Total</td>
<td>mg/L</td>
<td>259.0</td>
<td>259.3</td>
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<td>Calcium</td>
<td>mg/L</td>
<td>85.2</td>
<td>78.6</td>
<td>28.2</td>
</tr>
<tr>
<td>Free chlorine</td>
<td>mg/L</td>
<td>0.0</td>
<td>0.0</td>
<td>0.1</td>
</tr>
<tr>
<td>Total chlorine</td>
<td>mg/L</td>
<td>2.4</td>
<td>2.1</td>
<td>2.3</td>
</tr>
<tr>
<td>LSI</td>
<td></td>
<td>0.6</td>
<td>0.5</td>
<td>-0.2</td>
</tr>
<tr>
<td>RSI</td>
<td></td>
<td>6.8</td>
<td>7.0</td>
<td>8.4</td>
</tr>
<tr>
<td>Larson-Skold Ratio</td>
<td></td>
<td>0.8</td>
<td>0.9</td>
<td>1.5</td>
</tr>
<tr>
<td>CSMR</td>
<td></td>
<td>0.3</td>
<td>0.4</td>
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</tr>
<tr>
<td>Turner Ratio</td>
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</table>

### Table 5.5
Utility 4b raw water quality parameters and corrosion indices

<table>
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<tr>
<th>Parameter</th>
<th>Units</th>
<th>100% finished</th>
<th>90% finished / 10% industrial-AWP</th>
</tr>
</thead>
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<tr>
<td>pH</td>
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<td>7.8</td>
<td>7.4</td>
</tr>
<tr>
<td>Conductivity</td>
<td>uS/cm</td>
<td>517.0</td>
<td>531.8</td>
</tr>
<tr>
<td>Alkalinity, Total</td>
<td>mg/L</td>
<td>107.3</td>
<td>107.5</td>
</tr>
<tr>
<td>Chloride</td>
<td>mg/L</td>
<td>63.3</td>
<td>65.5</td>
</tr>
<tr>
<td>Sulfate</td>
<td>mg/L</td>
<td>47.5</td>
<td>43.9</td>
</tr>
<tr>
<td>T-Phosphate as P</td>
<td>mg/L</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Hardness, Total</td>
<td>mg/L</td>
<td>105.3</td>
<td>104.3</td>
</tr>
<tr>
<td>Calcium</td>
<td>mg/L</td>
<td>24.5</td>
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</tr>
<tr>
<td>Free chlorine</td>
<td>mg/L</td>
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<td>0.1</td>
</tr>
<tr>
<td>Total chlorine</td>
<td>mg/L</td>
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<td>1.2</td>
</tr>
<tr>
<td>LSI</td>
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<td>-0.4</td>
<td>-0.8</td>
</tr>
<tr>
<td>RSI</td>
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<td>8.6</td>
<td>9.0</td>
</tr>
<tr>
<td>Larson-Skold Ratio</td>
<td></td>
<td>1.3</td>
<td>1.3</td>
</tr>
<tr>
<td>CSMR</td>
<td></td>
<td>1.3</td>
<td>1.5</td>
</tr>
<tr>
<td>Turner Ratio</td>
<td></td>
<td>0.6</td>
<td>0.6</td>
</tr>
</tbody>
</table>
Metals

The results from the pipe rig tests at each utility are described in this section. The condition of the brass rods used in these tests is discussed, followed by the metal leaching results. Finally, these results are compared to the corrosion potential indices detailed in the Corrosion Potential Section above.

Brass Corrosion

Plastic pipe rigs containing CDA360 brass rods were used to assess the corrosion potential of multiple water conditions for each utility. The protocol for these tests is presented in Chapter 3. At the end of each test, the pipe rigs were shipped to Virginia Tech for analysis. The brass rods were removed and inspected visually and by X-ray fluorescence.

There were several cases where corrosion was visually observed. One case was Utility 1 90 percent surface/10 percent AWP water where a pit had developed on one of the brass rods (Figure 5.15). ESEM-EDS analysis of the pit showed that it was composed of iron (65 ± 5 weight percent) and oxygen (30 ± 1 weight percent), with only trace amounts of copper (5 ± 4 weight percent).

![Figure 5.15 Pit on the outer surface of CDA360 brass rod exposed to Utility 1 90% surface/10% AWP water](image)

Brass rods from several of the pipe rigs at Utilities 4a and 4b exhibited a blue scale to varying degrees. Most notably, the brass in Utility 4a 90 percent groundwater/10 percent AWP had a fairly extensive coating of blue scale (Figure 5.16a). Brass in Utility 4a 90 percent surface/10 percent AWP also had the bluish colored scale, but to a lesser extent (Figure 5.16b). SEM-EDS was used to try to identify the composition of the blue scale, but results were inconclusive (data not shown). The oxygen:carbon molar ratio was approximately 2.5:1, indicating
that the scale might contain a carbonate species, but the copper was not in a range where it could be definitively classified as malachite \([\text{Cu}_2\text{CO}_3(\text{OH})_2]\).

Figure 5.16 Blue scale on outer surfaces of (a) Utility 4a 90% groundwater/10% AWP, and (b) Utility 4a 90% surface/10% AWP

Metal release from the brass rods in the pipe rigs at Utility 1 is shown in Figure 5.17. The average concentrations shown in the figure are from the final two weeks of testing. Lead release in Utility 1 was higher in the three blended conditions, but was only statistically greater in the 90 percent surface / 10 percent AWP condition \((p=0.002)\). There were no statistical differences in the copper leaching from the four Utility 1 conditions tested \((p>0.05)\). Zinc leaching in Utility 1 blends was greater in both the 90 percent surface / 10 percent AWP and 50 percent surface / 50 percent AWP conditions than in the untreated surface water \((p=0.007\) and 0.0008, respectively). Likewise, zinc leaching in Utility 1 was greater from the 50 percent surface / 50 percent AWP than both the 90 percent surface / 10 percent \(\text{O}_3\)-BAC and 90 percent surface / 10 percent AWP conditions \((p=0.007\) and 0.0009, respectively).

Overall for Utility 1, as shown in Table 5.6, the corrosion indices evaluated in this study were not predictive as far as lead leaching was concerned. The Larson-Skold Index, the CSMR, and the Turner Ratio all had correlation coefficients just below 0.4, while the LSI and Ryznar were around 0.1. Conversely, correlation coefficients were higher for copper and zinc leaching in Utility 1 and these indices could quite possibly be used for prediction of leaching. \(R^2\) values for copper were 0.94 for LSI and Ryznar, 0.7 for Larson-Skold and Turner Ratio, and 0.38 for CSMR. \(R^2\) values for zinc were 0.84-0.91 for LSI, Ryznar, Larson-Skold, and Turner Ratio, and about 0.4 for CSMR.

Metal release from the brass rods in the pipe rigs at Utility 2 is also shown in Figure 5.17. In Utility 2, lead release was generally lower in the three blended conditions compared to the 100 percent groundwater, but was not statistically different from the 100 percent groundwater in any of the conditions tested. Lead leaching in the 50 percent groundwater / 50 percent past-AWP for Utility 2 was greater than lead leaching in the 50 percent groundwater / 50 percent AWP condition \((p=0.02)\) for Utility 2. Generally, copper and zinc leaching were less for the blended conditions in Utility 2 compared to the 100 percent groundwater; however, the Utility 2 pasteurized blend (50 percent groundwater / 50 percent past-AWP) had much greater copper and zinc leaching than the other blended conditions.

As shown in Table 5.6, no corrosion indices were correlated to metal release for Utility 2. All correlation coefficients were less than 0.3, except for the Turner Ratio. Turner Ratio, however, was not useful despite higher correlation coefficients because all four water conditions had similar values (all approximately 0.4) as shown in Table 5.7.

Metal release from the brass rods in the pipe rigs at Utility 3 is shown in Figure 5.17. Lead release was similar in all Utility 3 water quality conditions tested. Copper and zinc concentrations,
however, were greater in the two Utility 3 BAC conditions compared to all other conditions but they were not different than each other despite the vastly different blending ratios (50 percent versus 10 percent).

Figure 5.17 Utilities 1 through 4 metal leaching from CDA360 brass rods; lead concentration is shown on the bottom x-axis, copper and zinc concentrations on the upper x-axis; error bars denote 95% confidence intervals
Table 5.6
Correlation coefficients of corrosion index with metals leaching

<table>
<thead>
<tr>
<th></th>
<th>All Utilities</th>
<th>Utility 1</th>
<th>Utility 2</th>
<th>Utility 3</th>
<th>Utility 4</th>
</tr>
</thead>
<tbody>
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<td><strong>correlation w/ Pb</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>LSI</td>
<td>0.40</td>
<td>0.10</td>
<td>0.25</td>
<td>0.52</td>
<td>0.68</td>
</tr>
<tr>
<td>Ryznar</td>
<td>0.33</td>
<td>0.10</td>
<td>0.23</td>
<td>0.26</td>
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</tr>
<tr>
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<td>0.15</td>
<td>0.37</td>
<td>0.01</td>
<td>0.99</td>
<td>0.16</td>
</tr>
<tr>
<td>CSMR</td>
<td>0.32</td>
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<td>0.19</td>
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<td>0.31</td>
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<td>Turner Ratio</td>
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<td>0.24</td>
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<td><strong>correlation w/ Cu</strong></td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>LSI</td>
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<td>0.10</td>
<td>0.00</td>
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</tr>
<tr>
<td>Turner Ratio</td>
<td>0.00</td>
<td>0.84</td>
<td>0.62</td>
<td>0.26</td>
<td>1.00</td>
</tr>
</tbody>
</table>

Note that grey cells indicate $R^2 > 0.7$ which would indicate a strong correlation.

Table 5.7
Range of corrosion index

<table>
<thead>
<tr>
<th></th>
<th>Utility 1</th>
<th>Utility 2</th>
<th>Utility 3</th>
<th>Utility 4</th>
</tr>
</thead>
<tbody>
<tr>
<td>LSI</td>
<td>-1.8 to -0.9</td>
<td>-0.2 to 0.6</td>
<td>0.2 to 0.5</td>
<td>-0.8 to 0.6</td>
</tr>
<tr>
<td>Ryznar</td>
<td>9.1 to 10.7</td>
<td>6.4 to 7.7</td>
<td>7.1 to 7.4</td>
<td>6.8 to 9.0</td>
</tr>
<tr>
<td>Larson-Skold Ratio</td>
<td>2.0 to 5.1</td>
<td>2.9 to 3.2</td>
<td>2.3 - 3.7</td>
<td>0.8 to 1.5</td>
</tr>
<tr>
<td>CSMR</td>
<td>1.4 to 2.6</td>
<td>0.1 to 0.2</td>
<td>0.4 to 0.5</td>
<td>0.3 to 1.6</td>
</tr>
<tr>
<td>Turner Ratio</td>
<td>0.9 to 2.8</td>
<td>0.37 to 0.40</td>
<td>0.6 to 1.1</td>
<td>0.2 to 0.7</td>
</tr>
</tbody>
</table>

Two corrosion indices evaluated in this study showed promise in being predictive of lead release for Utility 3. As shown in Table 5.6, the Larson-Skold Index exhibited a correlation coefficient of 0.99, while the Turner Ratio had an $R^2$ of 0.94. The best-fit equations are given in Equations 5.5 and 5.6:

\[
(Pb \text{ in } \mu g/L) = 2.4 \times \text{[Larson-Skold Index]} + 2.0 \quad (5.5)
\]

\[
(Pb \text{ in } \mu g/L) = 6.1 \times \text{[Turner Ratio]} + 4.4 \quad (5.6)
\]

Correlations for copper and zinc leaching at Utility 3 were not nearly as good.

Metal release from the brass rods in the pipe rigs at Utility 4 is shown in Figure 5.17. The conditions with Utility 4a water showed similar lead leaching and all the concentrations were very low (< 5 µg/L). Copper and zinc leaching at Utility 4a were similar for the two groundwater
conditions, but were significantly less in the surface water condition. Lead, copper, and zinc concentrations were all similar at the two conditions tested with Utility 4b water.

Grouping these five water quality conditions together for Utility 4, the LSI and Ryznar corrosion indices showed a moderate correlation with lead leaching ($R^2 = 0.68$ and $0.59$, respectively) as shown in Table 5.6. For copper, no corrosion indices were useful in predicting copper corrosion at Utility 4. However, all five corrosion indices showed high correlation ($R^2 \geq 0.97$) for zinc corrosion at Utility 4, although these results may be skewed because the range of zinc concentrations was fairly small (200 to 340 µg/L) as shown in Table 5.7.

One of the overall goals of this study was to determine whether a corrosion index could adequately predict when lead corrosion might be an issue when blending recycled waters. While there was some moderate success at individual utilities, the ultimate goal would be a robust index that could predict lead corrosion across a number of water qualities. In this project that goal could not be achieved satisfactorily as shown through the $R^2$ values for all utilities combined in Table 5.6. Figures 5.18 through 5.22 also show the correlation for each corrosion index evaluated. The best correlation was achieved by the LSI with an $R^2$ value of 0.4 (Figure 5.18). Additional correlations were evaluated for pH, alkalinity, and chloride concentration and the correlation of lead leaching with pH had the highest $R^2$ of 0.42 (generally, lead leaching was greater at lower pH as expected). In previous work, high alkalinity has been shown to lower lead release, but in some cases increase copper corrosion (Dodrill and Edwards 1995, Edwards et al. 1996). However, in this study, no significant correlations with alkalinity could be shown. Dissolved oxygen (DO) is another factor that can contribute to infrastructure corrosion but was not evaluated in this study. Iron metal is not stable in the presence of high levels of DO, meaning a rust layer of either $\alpha$-FeOOH or Fe(OH)$_3$ solids will form as an interior pipe scale (AwwaRF and DVGW-TZW 1996). Little work has been conducted to determine the influence of DO on lead release.

Lead leaching did not increase with blend ratio in most instances. At Utility 1, lead did increase slightly with blending, but at Utility 2, lead leaching actually decreased as recycled water was blended. For Utilities 3 and 4, lead leaching was not really affected by the blend ratio. These results appear to indicate that each facility will need to look at their water chemistry and blending conditions on a case-by-case basis as there is no universal corrosion index that industry can use to determine the extent of lead corrosion. However, as shown in Table 5.6, there are individual instances when certain corrosion indices do work well for lead, copper, and/or zinc corrosion.

Key conclusions on metals and corrosion potential are shown in Figure 5.23.
Figure 5.18 Lead leaching from CDA360 brass rods versus LSI for all participating utilities

\[ y = -6.36x + 11.07 \]
\[ R^2 = 0.40 \]

Figure 5.19 Lead leaching from CDA360 brass rods versus Ryznar Stability Index for all participating utilities

\[ y = 3.39x - 14.96 \]
\[ R^2 = 0.33 \]
Figure 5.20 Lead leaching from CDA360 brass rods versus Larson-Skold Index for all participating utilities

Figure 5.21 Lead leaching from CDA360 brass rods versus CSMR for all participating utilities
Figure 5.22 Lead leaching from CDA360 brass rods versus Turner Ratio for all participating utilities

Figure 5.23 Metals and corrosion potential key conclusions

- Where the water was not stabilized, some metals leaching was observed. This was true for AWP, tertiary, and O₃-BAC applications.

- Metals leaching was not observed in Industrial AWP blends where the water was stabilized.

- No universal corrosion index accurately estimated the extent of lead corrosion in this study.
**Bacterial Corrosion**

Nuisance bacteria in distribution systems can lead to a series of problems that can compromise the aesthetic quality of the finished water. These include discoloration, cloudiness, corrosion and infectious outbreaks (Cullimore 2010). No single analytical test is capable of detecting a broad enough variety of bacteria to sufficiently understand downstream corrosion risks. Hence, the Biological Activity Reaction Tests (BART) were developed to qualitatively and semi-quantitatively estimate the activity or “aggressivity” of nuisance bacteria based on a time lag correlation. BART identifies corrosion, biofouling, or nitrate generation potential. The bacteria are not quantified by counting bacterial colonies on agar plates, but measured in relation to the number of days it takes for the reaction to occur. The longer the time lag to the detection of a reaction, the smaller the aggressive population of the microbial consortium. Results are obtained by monitoring for specific visual indicators to determine corrosion and/or biofouling potential of the target microbial species. A total of six bacterial groups were evaluated as part of these tests: SRB, acid producing bacteria (APB), heterotrophic aerobic bacteria (HAB), slime forming bacteria (SLYM), nitrifying bacteria, and denitrifying bacteria. SRB contribute to pitting corrosion, production of black slimes, black water, and undesirable taste and odor problems (Droycon Bioconcepts Inc. 2004, Little et al. 2007). APB can lead to a localized decrease in water pH that has been associated with pitting corrosion and pinhole leaks (Emde et al. 1992, Little et al. 2007, Rhoads et al. 2015). SLYM bacteria are associated with production of excessive slime which can cause corrosion of metal pipes, plugging, undesirable taste and odor, and cloudiness (Droycon Bioconcepts, Inc. 2004, Little et al. 2007). Nitrifying bacteria (N bacteria) can induce corrosion through nitrite production, lower pH and alkalinity, and decreased dissolved oxygen, as well as lead to undesirable taste and odor and discolored water (Zhang et al. 2009b). Denitrifying bacteria (DN bacteria) have been associated with increased pH, corrosion of metal pipes, and undesirable taste and odors (Masters et al. 2015). Elevated levels of all of these bacteria can also contribute to the degradation of disinfectant residuals (Droycon Bioconcepts, Inc. 2004, Masters et al. 2015, Zhang et al. 2009b). A summary of the results obtained from the BART analysis across all four utilities is presented in Table 5.8.

About 15 mL of blended and unblended samples were collected post-pipe rig incubation in the BART biodetectors. These vials have a conical-shaped base containing crystallized nutrient media specific to the bacterial group being observed and a plastic ball. The rich nutrients dissolve to promote aerobic growth around the ball and anaerobic growth at the base of the vial. The ball floats at the top of the sample to provide a barrier for oxygen diffusion which allows a redox gradient to form within the biodetector.

The interpretation of the color changes is performed using a reference chart that correlates each reaction day to an approximate population size of each bacterial group. The red colored values are indicative of aggressive or rapid growing bacteria, occurring typically within the first two days of reaction. Yellow means the bacteria have a medium growth rate, and green indicates the bacteria are slow growers. The reaction days are different for each test suggesting that the different bacteria have different growth curves. For example, a three day reaction for slime forming bacteria is indicative of aggressive bacteria, but a three day reaction for denitrifying bacteria indicates a more moderate growth rate. Due to an initial misinterpretation of the BART method, Utility 1 was analyzed solely based on a negative or positive test result (absent or present) at the end of the reaction period (after 4 to 8 days).
Table 5.8
WQ Test 4: Qualitative and semi-quantitative* BART results

<table>
<thead>
<tr>
<th>Utilities</th>
<th>Test waters</th>
<th>Sulfate reducing bacteria†</th>
<th>Acid producing bacteria†</th>
<th>Heterotrophic aerobic bacteria‡</th>
<th>Slime forming bacteria§</th>
<th>Nitrifying bacteria§</th>
<th>Denitrifying bacteria‡</th>
</tr>
</thead>
<tbody>
<tr>
<td>Utility 1</td>
<td>SW</td>
<td>Present</td>
<td>Absent</td>
<td>Absent</td>
<td>Present</td>
<td>Absent</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>90%SW/10% O3-BAC</td>
<td>Present</td>
<td>Absent</td>
<td>Absent</td>
<td>Present</td>
<td>Absent</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>90%SW/10% AWP</td>
<td>Present</td>
<td>Absent</td>
<td>Absent</td>
<td>Present</td>
<td>Absent</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>50%SW/50% AWP</td>
<td>Present</td>
<td>Absent</td>
<td>Absent</td>
<td>Present</td>
<td>Absent</td>
<td>N/A</td>
</tr>
<tr>
<td>Utility 2</td>
<td>GW</td>
<td>2,200,000</td>
<td>Absent</td>
<td>Absent</td>
<td>500</td>
<td>Absent</td>
<td>Absent</td>
</tr>
<tr>
<td></td>
<td>90%GW/10% AWP</td>
<td>2,200,000</td>
<td>Absent</td>
<td>6,500</td>
<td>500</td>
<td>Absent</td>
<td>3000</td>
</tr>
<tr>
<td></td>
<td>50%GW/50% AWP</td>
<td>2,200,000</td>
<td>Absent</td>
<td>500</td>
<td>Absent</td>
<td>3000</td>
<td>Absent</td>
</tr>
<tr>
<td></td>
<td>50%GW/50% Past. AWP</td>
<td>2,200,000</td>
<td>Absent</td>
<td>6,500</td>
<td>500</td>
<td>Absent</td>
<td>Absent</td>
</tr>
<tr>
<td>Utility 3</td>
<td>SW</td>
<td>2,200,000</td>
<td>Absent</td>
<td>Absent</td>
<td>1,750,000</td>
<td>Absent</td>
<td>1,800,000</td>
</tr>
<tr>
<td></td>
<td>95%SW/5% Tertiary</td>
<td>2,200,000</td>
<td>Absent</td>
<td>Absent</td>
<td>1,750,000</td>
<td>Absent</td>
<td>1,800,000</td>
</tr>
<tr>
<td></td>
<td>90%SW/10% Tertiary</td>
<td>2,200,000</td>
<td>70,000</td>
<td>6,500</td>
<td>1,750,000</td>
<td>Absent</td>
<td>1,800,000</td>
</tr>
<tr>
<td></td>
<td>90%SW/10% O3-BAC</td>
<td>2,200,000</td>
<td>Absent</td>
<td>6,500</td>
<td>500</td>
<td>Absent</td>
<td>3,000</td>
</tr>
<tr>
<td></td>
<td>50%SW/50% O3-BAC</td>
<td>2,200,000</td>
<td>Absent</td>
<td>500</td>
<td>Absent</td>
<td>Absent</td>
<td>Absent</td>
</tr>
<tr>
<td>Utility 4</td>
<td>GW</td>
<td>2,200,000</td>
<td>Absent</td>
<td>Absent</td>
<td>440,000</td>
<td>Absent</td>
<td>1,800,000</td>
</tr>
<tr>
<td></td>
<td>90%GW/10% AWP</td>
<td>2,200,000</td>
<td>Absent</td>
<td>Absent</td>
<td>440,000</td>
<td>Absent</td>
<td>1,800,000</td>
</tr>
<tr>
<td></td>
<td>90%SW/10% AWP</td>
<td>2,200,000</td>
<td>Absent</td>
<td>Absent</td>
<td>440,000</td>
<td>Absent</td>
<td>1,800,000</td>
</tr>
<tr>
<td></td>
<td>FSW</td>
<td>6,000</td>
<td>Absent</td>
<td>Absent</td>
<td>350,000</td>
<td>Absent</td>
<td>215,000</td>
</tr>
<tr>
<td></td>
<td>90%FSW/10% Industrial AWP</td>
<td>6,000</td>
<td>Absent</td>
<td>Absent</td>
<td>350,000</td>
<td>Absent</td>
<td>215,000</td>
</tr>
</tbody>
</table>

* Bacteria are quantified in cfu/mL and the populations in red are aggressive, those in yellow are moderate, and ones in green are not-aggressive.
† 8-day test.
‡ 4-day test.
§ 5-day test.

SRB were the most aggressive and commonly recurring bacteria detected in all samples. These microbes (i.e., Desulfovibrio, Desulfotomaculum) use sulfate as a substrate for cellular respiration, and release hydrogen sulfide as a metabolic byproduct. The H2S reacts with the ferrous iron present at the base of the vial to form black iron sulfides. Traces of black slime and foul odor are an indication of potential corrosion due to the biofilm present in distribution pipes. With the exclusion of the FSW and 90 percent FSW/10 percent industrial AWP samples, a one day reaction was observed for all samples, which corresponds to an approximate population size of 2,200,000 cfu/mL. The presence of these bacteria even in the 100 percent surface scenario suggests that blending either membrane filtered or biofiltered AWP treated recycled water did not cause any additional classes of corrosion-associated bacteria to become established in the simulated distribution systems. An example of the SRB BART from Utility 2 is shown in Figure 5.24.
The APB were detected less frequently as compared to the SRB. These vials contain a glucose-peptone medium at the base, and bromocresol purple pH indicator beneath the cap. The indicator is dissolved by the sample upon inverting the tube for 30 seconds. The conversion of the purple solution into a yellow color corresponds to the presence of acid producing bacteria, typically occurring in the 3.5 to 5.5 pH range. These microbes produce organic and inorganic acidic products in reductive (no oxygen) environments. Reductive conditions promote the proliferation of both APB and SRB in distribution pipes, which can be highly corrosive to both metal and concrete pipes. A positive test was found only in two of the samples, the Utility 3 90 percent SW/10 percent tertiary water after the second reaction day and the 50 percent SW/50 percent O3-BAC after the seventh reaction day. Where APB were present, the pH did not significantly shift, which is most likely attributed to the presence of high concentrations of alkalinity in these samples. These results indicate that blend ratio appears to be important for limiting the regrowth of corrosion bacteria in tertiary-treated waters. An example of the APB BART from Utility 2 is shown in Figure 5.25.
The HAB test is used to detect the presence of total bacteria that use carbon-containing compounds as a source of energy. Methylene blue dye is used as an indicator for microbial respiration. HAB thrive in oxygen rich environments and as the oxygen is consumed, methylene blue is used as an alternate substrate with a methylene blue reductase enzyme. Bleaching of the blue solution indicates a positive test. Bleaching from the bottom up indicates the presence of aerobic bacteria, and bleaching from the top down is an indication of facultative anaerobic bacteria. The shorter the time lag to color change, the higher the level of respiration, and the larger and more aggressive the bacterial population. Table 5.8 shows moderate HAB activity for the 90 percent GW/10 percent AWP sample and 50 percent GW/50 percent past. AWP sample from Utility 2, and 90 percent SW/10 percent tertiary sample from Utility 3. Note, these detections did not correspond to decreases in the concentration of TOC during the storage in pipe rigs. An example of the HAB BART from Utility 2 is shown in Figure 5.26.
The presence of slime forming bacteria results in the accumulation of slime-like coatings around the surface of pipes and fittings in distribution systems (Cullimore 2010). These conditions can cause engineering problems related to decreased flow efficiency and slime sloughing in the finished water. A variety of visual indicators were used to estimate the dominant slime forming bacteria present among a diverse community of microbes. The presence of these microbes can be indicated by cloudiness, dense slime observed at the base of the vial, and gel-like particulates visible upon gentle stirring. For example, the far left vial in Figure 5.27 shows larger size particulates in the form of floating layered plates, which suggests the sample may contain *Proteus* bacteria. These bacteria tend to aggregate in a disc-like shape and upon mixing make the solution look milky. Slime forming bacteria were active in all samples, but less aggressively in Utility 2 samples where the blends were all treated by AWP and GW-based. Like SRB, the presence of these bacteria even in the 100 percent surface scenario suggests that blending either membrane filtered or biofiltered AWP treated recycled water did not cause any additional classes of corrosion-associated bacteria to become established in the simulated distribution systems. An example of the slime forming bacteria BART from Utility 3 is shown in Figure 5.27.

![Figure 5.27 From left to right, 90% SW/10% O₃-BAC and duplicate, 50% SW/50% O₃-BAC and duplicate from Utility 3; cloudy solution is an indication of the presence of slime forming bacteria](image)

Nitrifying bacteria are responsible for the oxidation of ammonia and organic nitrogenous compounds from ammonium to nitrate (Cullimore 2010). These bacteria are present when the solution transforms to pink or red because of the reaction between the reagent on the cap and the sample and are benchmarked after the fifth day of incubation. The reagent detects for intermediate nitrite products that form during the nitrification process. All the samples tested as a part of this project were negative, which was also reflected in the limited change in concentration from the influent to the effluent of the pipe rig samples of ammonia and nitrate. An example of the nitrifying bacteria BART from Utility 2 is shown in Figure 5.28.
Figure 5.28 From left to right, GW and duplicate, 50% GW/50% past.-AWP and duplicate from Utility 2; pink to red coloration would have occurred in the presence of nitrite products, but was absent at the end of reaction day 5, indicating an absence of nitrifying bacteria

Denitrifying bacteria are essential mediators in the nitrogen cycle that reduce nitrate to nitrogen gas. A positive test resulted from the appearance of foam around the ball. Aggressive activity of these bacteria occurred for all samples in Utility 4, and most samples in Utility 3. More moderate activity was present in the 90 percent GW/10 percent AWP and 50 percent GW/50 percent AWP samples from Utility 2. The concentration of nitrate was not significantly altered by these bacteria, but the bacteria concentration relative to the nitrate may not have been high enough to significantly reduce the concentration during the storage time. In Utility 2, the pasteurization-treated scenario was more effective at controlling DN bacteria compared to the non-pasteurized conditions (3,000 CFU/mL). However, in Utility 4, interestingly, the 90 percent groundwater/10 percent AWP scenario and the 90 percent surface/10 percent AWP scenarios resulted in the same level of regrowth of DN (1,800,000 CFU/mL) as well as SRB (2,200,000 CFU/mL) and SLYM bacteria (440,000 CFU/mL), suggesting that while surface water alone produced less corrosion-associated bacteria, blending this water with AWP treated recycled water did not limit the regrowth of corrosion bacteria, compared to blending with the groundwater. An example of the denitrifying bacteria BART from Utility 2 is shown in Figure 5.29.
Figure 5.29 From left to right, GW and duplicate, 50% GW/50% past.-AWP and duplicate from Utility 2; the absence of foam around the ball indicates a negative test for denitrifying bacteria

Understanding the presence of nuisance bacteria is important because of the potential impact they have on the water quality. Of all the blended waters tested, the most prevalent bacteria were the SRBs, slime forming bacteria, and denitrifying bacteria. However, the presence of these bacteria did not impact the bulk water quality parameters. All reuse schemes effectively prevented regrowth of N bacteria. While blend ratio was important in controlling regrowth in tertiary-treated recycled waters, overall the results suggest that water source and treatment scheme are more important factors affecting the extent to which corrosion-associated bacteria are able to grow in simulated distribution systems.

Key conclusions on biological corrosion are shown in Figure 5.30.
Figure 5.30 Biological corrosion key conclusions

Metals Released and BART Results

The release of metals during the pipe rig tests could be caused by several factors, including chemical water quality and the presence of nuisance bacteria. Data generated as a part of the BART test provides a semi-quantitative assessment of the concentrations and thus no significant correlations were generated. As shown in Figure 5.31, the concentration of metals released during the experiment did not appear to be linked to one type of bacteria.
The impact of microbiological activity was further evaluated using 16S ribosomal RNA (16S rRNA) as a surrogate for bacteria counts as presented below. 16S rRNA was measured in the effluent and biofilm of each pipe rig at the end of the 8-week testing period for each pipe rig. Utility 4 was excluded from this analysis because a corrosion inhibitor was included in the testing matrix and visual evidence of scale formation on several of the brass rods were observed at the completion of the experiment. For the remaining blend conditions of the other three Utilities, the correlation between lead leaching and the logarithm of 16S rRNA for the bulk water (pipe rig out) concentrations indicated a weak correlation ($R^2 = 0.49$). However, if we consider the 16S rRNA in the biofilm, the apparent correlation with lead leaching increases to a $R^2$ of 0.72 (Figure 5.32). No apparent correlation ($R^2 < 0.1$) between copper and zinc leaching with the logarithm of 16S rRNA was observed. Confirmation of this observation was not possible as the experimental matrix was not designed to evaluate a potential mechanism. Further study is needed to evaluate lead leaching with microbiological activity beyond this correlation.
Key conclusions on metals and pipe rig biofilms are shown in Figure 5.33.

![Graph showing lead leached versus mean log 16S gene copies in biofilm for all pipe rigs tested at all utilities.]

**Figure 5.32** Lead leached versus mean log 16S gene copies in biofilm for all pipe rigs tested at all utilities
Opportunistic Pathogens (OPs)

OPs have been implicated as the primary source of waterborne disease in the U.S. (Pruden et al. 2013a). Unlike fecal pathogens that typically infect the gastrointestinal system, OPs infect via other “opportunist” routes. For example, *Legionella pneumophila* and *Mycobacterium avium* may infect hosts’ lungs when inhaled (Fraser et al. 1989, Horsburgh 1991) and *Pseudomonas aeruginosa* can infect hosts’ bloodstream, eyes, ears, skin, or lungs (Bodey et al. 1983). These alternative routes of infection are relevant to recycled water use in which aerosols may be generated during spray irrigation and dermal contact with irrigated surfaces may occur (Garner et al. 2016). OPs possess several distinct properties that make them well suited for growth in distribution systems. OPs tend to be resistant to disinfection, often reside in biofilms, grow well at low organic carbon concentrations, and during periods of stagnation (Falkinham 2015). OPs are prevalent in drinking water, for example, a survey of 272 water samples collected from cold water taps in the U.S. revealed the presence of *L. pneumophila* gene markers in 47 percent of taps (Donohue et al. 2014). OPs have also been widely detected in non-potable recycled water systems (Jjemba et al. 2010a,b).

*Legionella* spp., *Mycobacterium* spp., and *P. aeruginosa* gene markers were monitored in the influent, effluent, and biofilm of simulated distribution systems. 16S rRNA genes, which are universal to all bacteria, were also monitored as a proxy for total bacteria count.

In Utility 1 scenarios (shown in Figure 5.34), there were no significant differences between gene abundances in influent for the various scenarios for 16S rRNA genes ($p \geq 0.3123$) or any OPs ($p \geq 0.4533$). There were also no significant differences between scenarios for 16S rRNA genes in biofilm samples ($p \geq 0.2453$) and OPs were not detected in any biofilm samples.
Figure 5.34 16S rRNA genes and gene markers associated with opportunistic pathogens in the effluent of simulated distribution system rigs for Utility 1; hollow bars indicate measurements below the limit of quantification

Blend ratio in Utility 1 appears to be more important for controlling regrowth of total bacteria than whether pipe rig influent water was treated via biofiltration or membrane filtration. Effluent 16S rRNA genes were greater in both the 90 percent surface/10 percent O₃-BAC and 90 percent surface/10 percent AWP scenarios compared to the 50 percent surface/50 percent AWP scenarios \((p \leq 0.0028)\). 16S rRNA genes in rig effluent for 90 percent surface/10 percent O₃-BAC and 90 percent surface/10 percent AWP scenarios were not significantly different. The surface water appears to better support regrowth of total bacteria than the membrane-filtered AWP water, so increasing AWP blend ratio actually resulted in less effluent total bacteria regrowth than the 100 percent surface condition.

Effluent *Legionella* spp. and *Mycobacterium* spp. were only detected at sub-quantifiable levels in the 100 percent surface and 90 percent surface/10 percent O₃-BAC conditions in Utility 1. There were no significant differences in *Legionella* spp. levels for Utility 1 scenarios \((p \geq 0.1709)\). *Mycobacterium* spp. were significantly greater in effluent of the 90 percent surface/10 percent O₃-BAC scenario than the 90 percent surface/10 percent AWP scenario \((p=0.0044)\), suggesting that membrane filtration treatment was more effective at limiting regrowth than biofiltration in Utility 1. The 100 percent surface condition in Utility 1 resulted in more effluent *Mycobacterium* spp. than the 50 percent surface/50 percent AWP and 90 percent surface/10 percent AWP blends \((p=0.0013)\). This suggests that blending with this highly treated recycled water actually results in less regrowth of *Mycobacterium* spp. than traditional potable water sourced from local surface water. *P. aeruginosa* was not detected in any Utility 1 simulated distribution system samples.
In Utility 2 scenarios, shown in Figure 5.35, there were no significant differences between scenarios for influent \((p \geq 0.3123)\) or biofilm \((p \geq 0.2453)\) concentrations of total bacteria represented by 16S rRNA genes. Both the 90 percent groundwater/10 percent AWP and 50 percent groundwater/50 percent AWP scenarios resulted in significantly less regrowth of total bacteria \((p \leq 0.0136)\) than the traditional potable scenario, 100 percent groundwater. Pasteurization treatment did not significantly impact regrowth of total bacteria, as the 50 percent groundwater/50 percent AWP and 50 percent groundwater/50 percent past.-AWP scenarios were not significantly different \((p=0.0661)\).

![Figure 5.35 16S rRNA genes and gene markers associated with opportunistic pathogens in the effluent of simulated distribution system rigs for Utility 2; hollow bars indicate measurements below the limit of quantification](image)

Figure 5.35 16S rRNA genes and gene markers associated with opportunistic pathogens in the effluent of simulated distribution system rigs for Utility 2; hollow bars indicate measurements below the limit of quantification

Legionella spp. gene markers were detected in the effluent of all Utility 2 scenarios at relatively low concentrations, and there were no significant differences between Legionella spp. concentrations in effluent concentrations across all Utility 2 scenarios \((p \geq 0.1047)\). Mycobacterium spp. regrew in the 100 percent groundwater and 50 percent groundwater/50 percent past.-AWP Utility 2 scenarios, but no regrowth was detected in the other Utility 2 scenarios. Pasteurization treatment did not limit regrowth of Mycobacterium spp. as the 50 percent groundwater/50 percent past.-AWP scenario produced significantly more Mycobacterium spp. than the Utility 2 50 percent groundwater/50 percent AWP scenario \((p=0.0046)\). In general, Legionella spp. were able to grow in all Utility 2 AWP conditions, while pasteurized waters also supported regrowth of Mycobacterium spp. There were no significant differences between OPs in biofilms across scenarios \((p \geq 0.2453)\). P. aeruginosa was not detected in any bulk water or biofilm samples collected from Utility 2’s simulated distribution systems.
In Utility 3 scenarios, shown in Figure 5.36, there were no significant differences between scenarios for influent (p≥0.0545) or biofilm (p≥0.6985) concentrations of total bacteria represented by 16S rRNA genes. Effluent 16S rRNA gene concentrations were greater in both the 95 percent surface/5 percent tertiary and 90 percent surface/10 percent tertiary blend scenarios than in the 100 percent surface condition (p≤0.0028). The 90 percent surface/10 percent tertiary blend scenario resulted in greater regrowth of total bacteria than either the 90 percent surface/10 percent O₃-BAC of the 50 percent surface/50 percent O₃-BAC scenarios (p≤0.0128). Blend ratio significantly affected the regrowth of total bacteria in tertiary blends as the 90 percent surface/10 percent tertiary produced significantly more 16S rRNA gene in the effluent than the 95 percent surface/5 percent tertiary scenario (p=0.0181). These results suggest that tertiary treated waters have the potential to support more bacterial regrowth, but selection of appropriate blend ratio can significantly limit regrowth.

![Figure 5.36 16S rRNA genes and gene markers associated with opportunistic pathogens in the effluent of simulated distribution system rigs for Utility 3; hollow bars indicate measurements below the limit of quantification](https://example.com/figure536.png)

*Figure 5.36 16S rRNA genes and gene markers associated with opportunistic pathogens in the effluent of simulated distribution system rigs for Utility 3; hollow bars indicate measurements below the limit of quantification*

*Legionella* spp. and *P. aeruginosa* gene markers were not detected in either the effluent or the biofilm of any Utility 3 scenarios. *Mycobacterium* spp. was detected in several Utility 3 scenarios and both the 90 percent surface/10 percent O₃-BAC and the 50 percent surface/50 percent O₃-BAC Utility 3 scenarios produced more regrowth of *Mycobacterium* spp. than the traditional Utility 3 potable condition, 100 percent surface (p≤0.0323). *Mycobacterium* spp. was detected in the biofilm of 90 percent surface/10 percent tertiary and 50 percent surface/50 percent O₃-BAC rigs, but concentrations were not significantly greater than non-detect Utility 3 scenarios (p≥0.2207).
In Utility 4 scenarios, shown in Figure 5.37, there were no significant differences between influent 16S rRNA gene concentrations between scenarios ($p \geq 0.0518$), with the exception that 100 percent finished was significantly elevated over 100 percent groundwater ($p=0.0304$). There were also no significant differences between 16S rRNA gene concentration in biofilm samples ($p \geq 0.2453$). Effluent 16S rRNA gene concentrations in Utility 4 scenarios were significantly lower in the 90 percent surface/10 percent AWP scenario than the 100 percent groundwater and 100 percent finished scenarios ($p=0.0014, 0.0009$), demonstrating that blending surface water with this highly treated recycled water actually reduced the capacity for regrowth of total bacteria in the distribution system compared to traditional potable water sources. Similarly, the Utility 4 100 percent groundwater effluent 16S rRNA genes were significantly greater than the 90 percent groundwater/10 percent AWP scenario ($p=0.0136$). The Utility 4 90 percent finished/10 percent industrial-AWP scenario supported more regrowth, as this scenario resulted in more effluent 16S rRNA genes than either the 90 percent surface/10 percent AWP or 90 percent groundwater/10 percent AWP scenarios ($p=0.0009, 0.0406$). The 90 percent finished/10 percent industrial-AWP effluent 16S rRNA gene concentrations were not significantly different from the potable water scenarios, 100 percent finished and 100 percent groundwater ($p=0.0014, 0.0009$). No OP gene markers were detected in the influent, effluent, or biofilm of any Utility 4 scenarios.

![figure](image.png)

**Figure 5.37** 16S rRNA genes and gene markers associated with opportunistic pathogens in the effluent of simulated distribution system rigs for Utility 4

Recycled waters from Utilities 1 and 4 were both highly effective at controlling distribution system regrowth of OPs. Results from Utility 1 indicate that blends containing recycled water did not result in significantly more OP regrowth than treated surface water. Membrane filtration, in particular, resulted in recycled water that did not support regrowth of OPs during eight weeks of
simulated distribution testing. While both *Legionella* spp. and *Mycobacterium* spp. regrew during simulated distribution testing in Utility 2 scenarios, none of the scenarios produced significantly more regrowth than the treated groundwater condition. *Mycobacterium* spp. regrew in Utility 3 simulated distribution systems, but use of a 5 percent blend for tertiary treated waters or 10 percent blend for advanced oxidation treated waters, as opposed to 10 percent and 50 percent, respectively, did significantly reduce the regrowth of *Mycobacterium* spp.

Results from utilities 1, 2, and 4 indicate that the tested reuse scenarios do not appear to produce more regrowth of OPs during simulated distribution than corresponding traditional potable source waters. Results from Utility 3 indicate that regrowth of *Mycobacterium* spp. can be limited by selecting an appropriate blend ratio.

Key conclusions on opportunistic pathogens are shown in Figure 5.38.

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**Figure 5.38 Opportunistic pathogens key conclusions**

**Antibiotic Resistant Genes (ARGs)**

Antibiotic resistance among human pathogens is emerging as a critical concern for public health. The U.S. Centers for Disease Control has estimated that antibiotic resistant bacteria cause at least two million infections and 23,000 deaths each year (CDC 2013). Treated wastewater has been identified as a key source of increased ARGs in receiving surface waters and multiple studies of conventional wastewater treatment plants have demonstrated that ARGs are often not removed during treatment, and in some cases, are even amplified (Munir et al. 2011, Rizzo et al. 2013, Schlüter et al. 2007, Zhang et al. 2009a). Advanced treatment processes that are often used in reuse applications have not been thoroughly investigated with respect to removal of ARGs. Three genes were monitored throughout the study: *intI1*, *qnrA*, and *vanA*. *intI1* is a class 1 integron gene and
while it is not an ARG, *intI1* may serve as a proxy for the presence of many ARGs as it is typically genetically linked to various ARGs on mobile genetic elements and has been proposed as an indicator of anthropogenic pollution (Gillings et al. 2014). *qnrA* is a quinolone resistant gene that is typically plasmid-mediated, making it readily available for propagation between microbial cells via horizontal gene transfer (Poirel et al. 2005). *vanA* confers resistance to vancomycin, which is a last-resort antibiotic of critical importance for human health.

As shown in Figure 5.39, there were no significant differences between influent ARGs among Utility 1 scenarios (p ≥ 0.4533). *vanA* was widely detected at sub-quantifiable concentrations in Utility 1 samples, but the only quantifiable concentration of *vanA* occurred in the biofilm of the 90 percent surface/10 percent AWP scenario. However, there were no significant differences in *vanA* concentration between Utility 1 scenarios in either the effluent (p = 0.1709) or the biofilm (p ≥ 0.4142). While *qnrA* was not detected in any Utility 1 effluent samples, *intI1* was only detected at sub-quantifiable concentrations in the 90 percent surface/10 percent O₃-BAC Utility 1 scenario, though differences of *intI1* between scenarios were not significant (p ≥ 0.1700). Both *qnrA* and *intI1* were not detected in any Utility 1 biofilm samples.

![Figure 5.39 Antibiotic resistant gene markers in the effluent of simulated distribution system rigs for Utility 1; hollow bars indicate measurements below the limit of quantification](image-url)
Results for Utility 2 are shown in Figure 5.40. Although all three target genes were detected sporadically in samples collected from the simulated distribution systems of Utility 2, no genes were present above quantifiable levels, and there were no significant differences between gene abundances in the influent ($p \geq 0.1814$), effluent ($p \geq 0.0764$), or biofilm samples ($p \geq 0.4795$).

![Figure 5.40 Antibiotic resistant gene markers in the effluent of simulated distribution system rigs for Utility 2; hollow bars indicate measurements below the limit of quantification](image)

Results for Utility 3 are shown in Figure 5.41. No ARGs were present at quantifiable levels in any distribution system samples from Utility 3. There were no significant differences between gene abundances in the influent ($p \geq 0.4533$), effluent ($p \geq 0.3816$), or biofilm samples ($p \geq 0.6171$).

No ARGs were detected in the influent, effluent, or biofilm of any Utility 4 scenarios.

Together these results suggest that all utilities and tested scenarios were highly effective at limiting regrowth of bacteria containing the ARGs of interest during eight weeks of simulated distribution. An ARG was only detected in a single sample at above quantifiable levels: the $\text{vanA}$ resistant gene in the biofilm of Utility 1’s 90 percent surface/10 percent AWP condition. While these results are very promising indications that the DPR waters do not pose added risk in terms of proliferating ARGs in the distribution system, it is important to consider that these were short term studies with young biofilms.

Key conclusions on ARGs are shown in Figure 5.42.
Figure 5.41 Antibiotic resistant gene markers in the effluent of simulated distribution system rigs for Utility 3; hollow bars indicate measurements below the limit of quantification.

Bacteria Containing ARGs: Key Conclusions

- All utilities and tested scenarios were highly effective at limiting regrowth of bacteria containing the antibiotic resistance genes of interest during 8 weeks of simulated distribution. However these were short terms studies with young biofilms.

Figure 5.42 ARGs key conclusions
Antibiotic Resistant Bacteria

While molecular methods are highly accurate at identification and quantification of ARGs, these methods do not provide information about whether these genes are contained within live bacteria, as opposed to being associated with dead cells, occurring extracellularly, or within the biofilm in the form of extracellular polymeric substances. These methods also do not indicate whether the ARGs are being expressed. Culture-based methods provide information about the viability and expression of cells carrying ARGs, however culture-based methods present many challenges. Less than one percent of environmental bacteria are currently culturable using standard methods (Allen et al. 2010), so our capacity to examine the resistome among live bacteria via culture is highly limited. HPC is a standard method for culturing water bacteria aimed at cultivating diverse microorganisms on nutrient media. To investigate the viability and potential for expression of ARGs in collected samples, we cultured water samples on the standard HPC agar supplemented with nine different antibiotics to examine the occurrence of antibiotic resistance among HPCs.

Interestingly in Utility 1 scenarios, as shown in Figure 5.43, the 90 percent surface/10 percent AWP condition resulted in more regrowth of HPCs than the 50 percent surface/50 percent AWP, though there were more than two orders of magnitude of variation in total HPCs measured between the two rigs. HPCs capable of growth on media containing ciprofloxacin (80, 99 percent of total HPCs for rigs 1, 2 respectively), chloramphenicol (86, 89 percent), sulfamethoxazole (110, 97 percent), and vancomycin (110, 97 percent) were consistently detected at high relative abundances in the Utility 1 90 percent surface/10 percent AWP rigs. Rig 2 also had large proportions of HPCs capable of growth on media containing ampicillin (100 percent) and tetracycline (97 percent). The increased abundance of total HPCs and HPCs capable of growth on media containing certain antibiotics in the Utility 1 90 percent surface/10 percent AWP suggests that the surface water is contributing to conditions that are ideal for regrowth as decreasing the surface water blend ratio resulted in less regrowth of total HPCs.

![Figure 5.43 Heterotrophic plate count bacteria grown on media supplemented with antibiotics from Utility 1 simulated distribution system effluent from Utility 1 rigs](image-url)
Utility 2 scenarios were all extremely effective at controlling regrowth of total HPCs, as shown in Figure 5.44. With the exception of one 50 percent groundwater/50 percent past.-AWP rig, all conditions produced undetected or below quantification levels of HPCs.

For the Utility 3 scenarios, as shown in Figure 5.45, both 90 percent surface/10 percent tertiary rigs produced approximately two orders of magnitude greater total HPCs than the 95 percent surface/5 percent tertiary scenario rigs. Of the HPCs cultured from 90 percent surface/10 percent tertiary rigs, ampicillin (90, 72 percent of total HPCs for rigs 1, 2 respectively), gentamicin (98, 99 percent), oxacillin (97, 99 percent), rifampin (94, 97 percent), tetracycline (58, 43 percent), and vancomycin (84, 92 percent) resistant HPCs were found in both rigs. Ciprofloxacin (72 percent), chloramphenicol (49 percent), and sulfamethoxazole (94 percent) resistant HPCs were also detected in rig 2.

**Figure 5.44 Heterotrophic plate count bacteria grown on media supplemented with antibiotics from Utility 2 simulated distribution system effluent from Utility 2 rigs**
Figure 5.45 Heterotrophic plate count bacteria grown on media supplemented with antibiotics from Utility 3 simulated distribution system effluent from Utility 3 rigs

For the Utility 4 scenarios, shown in Figure 5.46, the 100 percent groundwater and 90 percent groundwater/10 percent AWP conditions both resulted in regrowth of total HPCs and HPCs capable of growth on media containing ampicillin, oxacillin, and rifampin. In the Utility 4 100 percent groundwater condition, 69 to 95 percent of total HPCs grew on media containing ampicillin, 89 to 99 percent grew on media containing oxacillin, and 89 to 100 percent grew on media containing rifampin. From the Utility 4 90 percent groundwater/10 percent AWP scenario, 78 to 91 percent of total HPCs grew on media containing ampicillin, 90 to 97 percent grew on media containing oxacillin, and 94 to 99 percent grew on media containing rifampin. These results suggest that groundwater better facilitates the regrowth of total and resistant HPCs than AWP water.
Figure 5.46 Heterotrophic plate count bacteria grown on media supplemented with antibiotics from Utility 4 simulated distribution system effluent from Utility 4 rigs

Together these results demonstrate that, with few exceptions the potable and reuse scenarios tested were highly effective at limiting regrowth of total HPCs and HPCs capable of growth on media containing antibiotics, though some regrowth of HPCs in simulated distribution systems was consistently observed. Results from Utilities 1 and 4 indicate that in scenarios where RO and UV/AOP processes are used to achieve very high quality recycled water, using higher blends of recycled water actually resulted in less regrowth of total and resistant HPCs than use of traditional potable water sources. Results from Utility 3 indicate that tertiary treated waters have the potential to support regrowth of HPCs in distribution systems, however limiting the blend ratio to 5 percent was effective at reducing regrowth of total HPCs by approximately two orders of magnitude compared to a 10 percent blend ratio. These results are also striking because the pattern of resistant HPCs was largely variable among utilities, but seemed to be relatively consistent within utility conditions, suggesting that source water and geographic factors are important in selecting for HPCs capable of growth on media containing antibiotics. HPCs capable of growth in the presence of several antibiotic classes, including ampicillin and oxacillin, were detected consistently across utilities. These results also support the results obtained from molecular quantification of ARGs. The highest levels of vanA, a vancomycin resistant gene, detected throughout this study were in the biofilm of Utility 1 90 percent surface/10 percent AWP rigs. Vancomycin-resistant HPCs were also present in this scenario at high relative concentrations (97 to 110 percent total HPCs). Detection of qnrA, a quinolone resistant gene, did not correlate with detection of ciprofloxacin, a quinolone antibiotic. This either suggests that a different
quinolone resistant gene conferred the observed ciprofloxacin resistance among HPCs, or \textit{qnr}A was present at below-detection quantities.

While these studies are promising in their indication that these DPR blending scenarios do not present added risk in terms of enhancing re-growth of ARGs or ARBs, it is important to consider that these were short-term studies. Longer term studies with mature biofilms would be of value.

It has been hypothesized that BDOC offers a more accurate indicator of regrowth potential than TOC or DOC. When we correlated rig influent TOC, DOC, and BDOC concentrations with two measurements of bacterial regrowth in rig effluent (HPC and 16S rRNA genes), conflicting trends were observed as shown in Figure 5.47 and Figure 5.48. BDOC was only slightly more strongly correlated with bulk water HPC regrowth (R$^2=0.1058$) than TOC (R$^2=0.0087$) or DOC (R$^2=0.0942$). However, 16S rRNA genes were slightly more strongly correlated with DOC (R$^2=0.1557$) than TOC (R$^2=0.0014$) or BDOC (R$^2=0.1174$). However, all correlations were very weak.

Key conclusions on total and resistant HPCs are shown in Figure 5.49.

Figure 5.47 Correlations between heterotrophic plate count (HPC) in rig effluent and total organic carbon (TOC), dissolved organic carbon (DOC), and biodegradable dissolved organic carbon (BDOC) concentrations in rig influent
Figure 5.48 Correlations between 16S rRNA genes in rig effluent and total organic carbon (TOC), dissolved organic carbon (DOC), and biodegradable dissolved organic carbon (BDOC) concentrations in rig influent

![Graph showing correlations between 16S rRNA genes and organic carbon concentrations.](image)

**Total and Resistant Heterotrophic Plate Counts: Key Conclusions**

- Less regrowth of total HPCs and HPCs capable of growth on media containing antibiotics (resistant HPCs) was seen in AWP blends than in the corresponding surface water or groundwater used with each blend.
- Tertiary Applications have the most regrowth of total and resistant HPCs in distribution systems.

Figure 5.49 Total and resistant HPCs key conclusions
CHAPTER 6
IMPLICATIONS FOR ENGINEERED STORAGE DESIGN

A part of the scope of this project was to determine the implications of its results on design considerations for engineered storage buffers (ESBs), including the impacts on monitoring, hold time requirements, and/or treatment needs. As noted above, an ESB is designed to retain advanced treated recycled water for a sufficient period of time to be able to detect and react to a process failure before the water is used for raw or finished water augmentation of conventional supplies. This is referred to as the Failure Response Time, or FRT (Salveson et al. 2015). The FRT is dictated by how quickly monitoring systems will detect any process failure; and the longer the FRT, the larger the ESB must be (Salveson et al. 2015).

The results from this study, while critically important to understanding the implications of blending on downstream water quality for potable reuse projects, do not directly affect this sizing paradigm. However, the results do affect the design of ESBs in two general ways: (1) some general takeaways from the testing can be extended to the ESB as a storage vessel that is in some aspects (mainly, materials) similar to a pipe, and (2) pathogen data collected for this project provides additional insight into what the minimum treatment goals should be for these kinds of projects, which affects the calculation of how the ESB is sized.

Regarding the more general observations of the implications of this work on an ESB as a storage vessel, corrosivity concerns are particularly acute for the ESB, as it will typically not have the benefit of blending with higher-alkalinity water prior to storage. No blend scenario evaluated in this study looked specifically at corrosion in the pipe rigs from 100 percent advanced treated wastewater, so the extent of corrosion from a 100 percent blend of the advanced treated wastewaters evaluated in this study is not known. However, inferences as to the corrosion potential can be made from the blends evaluated. Utilities 1, 2, and 3 did not chemically stabilize their blends and are thus more representative of the water quality stored in an ESB. At Utility 1, higher blends of AWP water saw higher lead and zinc leaching; however, this was not observed at Utility 2 where there was little change in metals leaching as the blend of AWP water increased. Utility 2 did see higher copper, zinc, and lead leaching in pasteurized-AWP blends though. For non-RO advanced treated wastewater (O₃-BAC), copper and zinc leaching increased at higher blend ratios of O₃-BAC water. To address these corrosivity concerns, a compatible tank material or coating will be needed in combination with chemical stabilization of that water. Utility 4 chemically stabilized their water blends and thus provides insight into the corrosivity potential of advanced treated wastewater that is stabilized prior to ESB storage. At Utility 4, metals leaching was generally low (less than 5 µg/L for lead) and the presence of AWP water either did not affect or decreased the concentration of metals in the blends measured. While the dataset analyzed is limited, it is consistent with expected results and lends further credence to the need for chemical stabilization prior to ESB storage to limit corrosion.

Additionally, when considering an ESB as a storage vessel similar to the pipe rigs used in this study, the potential for DBP formation needs to be considered. While storage of high DBP formation potential water is only a problem if a chlorine residual is maintained, DBP formation potential is particularly important if an ESB with free chlorine is used to provide additional log reduction credits for pathogen removal. Storage of waters with higher DBP formation potentials needs to be considered with respect the amount of time available for DBP formation, and, as noted for distribution systems, longer storage times may also lead to additional regrowth and biological
corrosion. The pipe rig data collected for this study looked at the change in water quality after storage in a pipe rig for two days. Every two to three days water in the pipe rig was replaced and after 8 weeks of this, water samples exiting the pipe rig were analyzed. From discussion in Chapter 4, this study concluded that for the samples analyzed, AWP waters tended to have lower TTHM FP than conventional drinking water sources while O₃-BAC waters tended to have higher TTHM FP than conventional drinking water sources. Thus, if 100 percent O₃-BAC water was stored in an ESB with free chlorine we would expect the concentration of THMs to increase. The results from Utility 3, where chlorine was used for disinfection instead of chloramines, confirm this. In fact, exceedances of the TTHM limit were observed for Utility 3 blends with O₃-BAC waters. Additionally, Utility 3 blends with O₃-BAC also saw increased HAA5 concentrations, but no exceedances were observed. For the other three utilities where chloramines were used for disinfection, no increases in TTHM or HAA5 were observed.

Regarding influent pathogen concentrations, the comparison of secondary or tertiary effluent concentrations to goals listed in Table 4.1 did not explicitly list the LRV requirements implied by this direct comparative approach. Taking non-detects at their detection limits, these averaged 5.8-log, 4.4-log, and 8.2-log for *Giardia*, *Cryptosporidium*, and enteric virus, respectively, across all the secondary and tertiary effluent samples, with maximum respective values of 7.5-log, 4.6-log and 12.6-log as shown in Table 6.1. These data represent one or two samples from each facility (total of 8 samples), and should thus be taken not as a rigorous characterization of any facility. However, the results in aggregate provide some valuable insights. The span of this data set in particular is interesting, as it implies less treatment is necessary for *Cryptosporidium* for every scenario studied than required by any current permitting process (Texas requires a minimum 5.5-log from wastewater treatment effluent to purified water, and California requires 10-log from raw wastewater to purified water), whereas the maximum value for enteric virus (based on the sample of secondary effluent collected from Utility 3 with a concentration of 820,000 GC/L) implies a greater treatment need at 12.6-log than California's 12-log standard for IPR (Texas does not provide a cap). Note that this value is calculated based on gene copy results, and the relationship of these data to viability is an area where more research is needed. Additionally, it is important to note the great variability across the enterovirus dataset and even within the utility that observed the high concentration of 820,000 GC/L.

Permitting paradigms aside, this dataset illustrates the need for pathogen characterization of the specific water to be used for potable purposes (be it at the raw wastewater or treated effluent stage) in order to ensure a design that meets the 1:10,000 risk target. Furthermore, this dataset lends some credence to the need for monitoring requirements in the water feeding a DPR system to better characterize the variability and concentrations of pathogens. With this pathogen source water characterization, a probabilistic quantitative microbial risk assessment can be performed to ensure the EPA risk levels are met (Independent Advisory Panel 2015). If through this risk assessment it is determined that less log removal credit is needed to meet EPA risk levels than is currently required, the ESB size and detention time may decrease, thus lowering the cost and infrastructure impact of a DPR facility. A shorter detention time in the ESB would also limit DBP formation and corrosion concerns discussed above. Based on the limited data collected in this study, it appears that, at least for *Cryptosporidium* this may be the case.
<table>
<thead>
<tr>
<th>Utility</th>
<th>Water source</th>
<th>Measured concentration</th>
<th>Required log removal values to achieve risk goals</th>
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</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Giardia</td>
<td>Cryptosporidium</td>
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<td>Utility 1</td>
<td>Secondary Effluent</td>
<td>7.9</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>Secondary Effluent</td>
<td>217</td>
<td>1</td>
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<td>Utility 2</td>
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<tr>
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<td>Secondary Effluent</td>
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<tr>
<td>Utility 4b</td>
<td>Secondary Effluent</td>
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<td>1</td>
</tr>
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</table>

*Note that this value is calculated based on gene copy results, and the relationship of these data to viability is an area where more research is needed.*
CHAPTER 7
CONCLUSIONS AND RECOMMENDATIONS

OBJECTIVES

Our goal for this project is to examine how reclaimed water quality, blend ratio, blend location and blend water quality impact the treatment and distribution systems and water quality. The research project examined the impact of the following three variables:

- Different qualities of recycled water (ranging from tertiary effluent to purified water from various advanced treatment trains).
- Different proportions of recycled water to raw water (ranging from 0 percent to 50 percent).
- Different points of introduction/blending (prior to a water treatment plant and direct injection into the distribution system).

Bench-scale, pilot-scale, and full-scale testing were employed to examine these three variables and related water quality. Importantly, premise plumbing rigs were used to simulate worst-case (stagnant conditions) of the blended waters in water distribution systems. A general review of reclaimed water treatment systems and blending water quality is shown in Tables 7.1 and 7.2.

For the different reclaimed water treatment trains, there are reasonable expectations of water quality that can be made based upon the selected treatment processes. For example, water that has been treated by RO is expected to be low in TOC and aggressive in nature. Generally, the results can be correlated to three different types of reclaimed water:

- Municipal AWP Applications (AWP) – Very low organics, nutrients, salts, and microbiology. More aggressive water due to the lack of constituents.
- Industrial AWP Applications (Industrial AWP) – Similar to municipal AWP applications, but with a higher nutrient and organic load in the raw wastewater resulting in higher nutrient and organic load in the finished water.
- Tertiary and O₃-BAC Applications (Tertiary and O₃-BAC) – Similar water chemistry to raw surface waters. Higher TOC, nutrients, and microbiology.

The results are presented below in three categories (Treatment Systems, Distributions Systems, and Engineered Storage) with targeted subcategories (Recycled Water Quality, Blend Ratio, Blend Location and Conventional Blend Water Quality).
## Table 7.1

Tested reclaimed water treatment systems

<table>
<thead>
<tr>
<th>Utility</th>
<th>Process identification</th>
<th>Secondary treatment</th>
<th>Tertiary filtration</th>
<th>Tertiary chlorination†</th>
<th>O₃</th>
<th>BAF</th>
<th>Pasteurization</th>
<th>MBR</th>
<th>MF/UF</th>
<th>RO</th>
<th>UV AOP</th>
<th>Stabilization</th>
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<tr>
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</tbody>
</table>

*AWP includes low pressure membrane filtration (MF or UF), reverse osmosis (RO), and ultraviolet light advanced oxidation process (UV AOP).

†BNR is a moderate to high SRT activated sludge process that nitrifies (removes ammonia) and for some facilities denitrifies (removes nitrate).

‡P connotes phosphorus removal in the biological treatment process.

§Tertiary chlorination consisted of a combination of total and free chlorine.

**Purified waters stabilized with zinc orthophosphate and calcium chloride in accordance with client preferences for other potable water reuse groundwater recharge projects.
### Table 7.2

<table>
<thead>
<tr>
<th>Utility</th>
<th>Process identification</th>
<th>Blend water</th>
<th>Treatment post blending</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>O₃-BAC</td>
<td>Raw Surface Water</td>
<td>WTP*</td>
</tr>
<tr>
<td>1</td>
<td>AWP</td>
<td>Raw Surface Water</td>
<td>WTP*</td>
</tr>
<tr>
<td>2</td>
<td>AWP</td>
<td>Groundwater</td>
<td>-</td>
</tr>
<tr>
<td>2</td>
<td>Past-AWP</td>
<td>Groundwater</td>
<td>-</td>
</tr>
<tr>
<td>3</td>
<td>Tertiary</td>
<td>Raw Surface Water</td>
<td>WTP*</td>
</tr>
<tr>
<td>3</td>
<td>O₃-BAC</td>
<td>Raw Surface Water</td>
<td>WTP*</td>
</tr>
<tr>
<td>4a</td>
<td>AWP</td>
<td>Raw Surface Water and Groundwater</td>
<td>WTP for blend with surface water only</td>
</tr>
<tr>
<td>4b</td>
<td>Industrial AWP</td>
<td>Groundwater</td>
<td>-</td>
</tr>
</tbody>
</table>

*WTP is a bench-scale water treatment plant, providing filtration and disinfection of the blended water.

### TREATMENT SYSTEMS

Treatment of reclaimed water to potable standards can be engineered with or without RO and can be equally protective of public health. A primary exception is salt, which requires RO or similar (e.g., nanofiltration) treatment if feed concentrations are high. For this project, the team examined RO-based purification trains that were intended to meet all regulated and health standards for potable water reuse, whereas the non-RO treatment trains represent “tertiary” or advanced tertiary (O₃-BAC) treatment. In parallel with this publication is the completion of extensive non-RO direct potable reuse demonstration efforts, which couple the O₃-BAC technologies studies for this project with UF, granular activated carbon (GAC), and UV (Altamonte Springs 2017). The testing of the tertiary and O₃-BAC treatment trains represent a much lower level of purification compared to the O₃/BAC/UF/GAC/UV train referenced above.

### Recycled Water Quality

Samples were analyzed to assess recycled water quality across the advanced treatment trains and blend ratios (post blending and treatment but pre-pipe rig simulation). In general, AWP waters were of a higher quality than the O₃-BAC waters across blend ratios, primarily due to the use of RO. Both AWP and O₃-BAC blends had contaminant levels below regulated limits for most parameters considered, summarized below:

1) **Nutrients**
   - Municipal AWP Applications: Total nitrogen, ammonia, nitrate, nitrite, and phosphorous were measured in this study and samples in the blended waters were below regulated limits.
   - Industrial AWP Applications: Similar to above, with the exception of high nitrate due to high nitrate levels in the feed water and incomplete removal by RO. Typically RO can remove more than 85 percent of nitrate (Metcalf & Eddy 2007), so high feed concentrations can result in RO permeate with measurable and impactful levels.
2) Bulk Water Quality

- Tertiary and O₃-BAC Applications: Similar to above, with sufficiently high nitrate levels in secondary effluent in some cases that can pass through the tested processes. For non-RO municipal projects, and depending upon the blend ratio, nitrate removal through biological processes may be required.

- Municipal AWP Applications: pH, turbidity, TOC, DOC, UV at 254 nm, conductivity, alkalinity, sulfate, and chloride were measured in this study. Generally, blends with AWP had lower levels of these parameters than conventional drinking water sources. This is as expected with RO treatment.

- Industrial AWP Applications: Unlike municipal AWP applications, concentrations of turbidity, chloride, conductivity, and pH were higher in industrial AWP blends than in conventional drinking water sources. DOC and alkalinity concentrations were similar. Despite these higher concentrations, none of these blends exceeded regulated limits.

- Tertiary and O₃-BAC Applications: As expected, unlike municipal AWP applications concentrations of pH, turbidity, TOC, DOC, UV at 254 nm, conductivity, alkalinity, sulfate, and chloride were generally higher than conventional drinking water sources. With the exception of sulfate and conductivity at specific blends, these higher levels in O₃-BAC blends did not exceed regulatory limits. For sulfate, a 50 percent SW / 50 percent O₃-BAC blend exceeded EPA's secondary standard. For conductivity, 100 percent O₃-BAC water exceeded the CA secondary maximum contaminant limit (900 µS/cm). For O₃-BAC type purification, clearly, salt removal may be required depending upon feed concentrations and finished water quality goals.

3) Disinfection Byproducts (DBPs)

- Municipal AWP Applications: NDMA, TTHM, HAA5, and TTHM FP were measured in this study. For municipal AWP applications, TTHM FP tended to be lower than conventional drinking water sources, which is consistent to findings presented in past drinking water studies and industry experience. With respect to TTHM and HAA5 formation, all blends were below the regulated values. Blending had an inconsistent impact, in some cases the reclaimed water reduced the blended concentrations, and in others it did not. NDMA was partially removed by AWP trains; however, high levels of NDMA in the source water caused an exceedance above the 10 ng/L State of California notification level when 100 percent AWP water was used (CDPH 2013). NDMA feed concentrations vary based upon wastewater feed concentrations, formation through purification (e.g., chloramination, ozonation), and removal through purification (e.g., RO, UV, BAC). Both RO and non-RO treatment trains can be engineered to maintain NDMA at low levels. Reformation of NDMA has also been shown to occur after purification. RO allows NDMA precursors to pass through (most likely low molecular weight amines: dimethylamine or trimethylamine) that are able to reform NDMA in the presence of chloramines that was subsequently applied in the lab, or ultraviolet advanced oxidation (UV/AOP) degraded NDMA into nitrite and dimethylamine which then similarly allowed for NDMA to reform when chloramines were added (Plumlee and Reinhard 2007, Plumlee et al. 2008, Xu et al. 2009).
- Industrial AWP Applications: Similar to above. The 100 percent industrial AWP water also exceeded the 10 ng/L State of California notification level for NDMA due to high levels in the source water (CDPH 2013).

- Tertiary and O₃-BAC Applications: Higher TTHM FP was observed in tertiary and O₃-BAC blends, which is consistent to findings presented in past drinking water studies and industry experience. O₃-BAC could reduce TTHM FP through use of additional treatment, such as GAC (Altamonte Springs 2017). Like AWP applications, concentrations of TTHM and HAA5 were below regulated values. NDMA was observed in the tertiary effluent, and was partially removed by O₃-BAC; however, no NDMA concentration above 10 ng/L was observed in tertiary and O₃-BAC applications.

- SUVA₂₅₄, was also calculated as part of this study to correlate SUVA₂₅₄ with TTHM FP. SUVA₂₅₄ is the ratio of UV₂₅₄ to DOC. In past work with drinking waters and surface waters, SUVA₂₅₄ has been positivity correlated to the TTHM FP and this was attributed to the presence of reactive aromatic moieties (Reckhow et al. 1990; Liang and Singer 2003; Weishaar et al. 2003). However, this study found that caution must be applied when using SUVA₂₅₄ to predict the TTHM FP of blended waters with low DOC. For this work, TTHM formation under FP test conditions was significantly negatively correlated with SUVA₂₅₄ (R² = 0.79, p < 0.001). For these waters, low DOC values drove up the SUVA₂₅₄, but also limited the formation of TTHMs by the low concentration of total precursors present. This result is not surprising as organic matter character is important in DBP formation.

4) Pathogens: A direct measure of the pathogen levels in the advanced treated wastewater of interest is not sufficient to ensure the recycled water meets the EPA's allowable risk levels, as the detection levels for pathogens are not sufficiently low. The result is the use of a log removal approach to evaluate a water's safety. This log removal approach is based on the treatment train used and is intended to be conservative.

- Municipal AWP Applications: Applying a strict log removal approach to the advanced treated wastewaters considered in this study, we see that only the pasteurized-AWP water meets a 12/10/10 log removal goal set by CA DDW for IPR projects (CDPH 2015), given the treatment trains employed. Clearly, further log removal credits could be claimed by adding additional treatment, such as the environmental buffer for IPR applications. For this study, pathogen levels in the advanced treatment source waters were also measured. Starting with this data and then applying assumed log reduction credits of the treatment trains considered, we see that both the pasteurized-AWP water and AWP waters meet allowable risk levels discussed in the NWRI DPR Framework (Independent Advisory Panel 2015). Note that the pathogen sampling is limited, and is not intended to represent a broad level of confidence in pathogen concentrations at each site.

- Industrial AWP Applications: Similar to above, the industrial AWP application met EPA's allowable risk levels when evaluating the pathogen levels in the advanced treatment source waters; however, using a strict log removal approach industrial AWP applications did not meet a 12/10/10 log removal goal set by CA DDW for IPR projects (CDPH 2015). Again, as stated above, further log removal credits could be claimed by adding additional treatment, such as the environmental buffer for IPR applications.
• Tertiary and $O_3$-BAC Applications: Both approaches for measuring pathogen levels in the advanced treated wastewaters indicated that tertiary and $O_3$-BAC applications did not meet the allowable risk levels, which is expected, given the treatment methods employed. Other disinfection technologies could be added to different process trains to attain a similar level of disinfection, such as a downstream water treatment plant.

5) Organic Carbon: TOC reduction is an important treatment target to reduce DBP precursors in particular and broad range organic pollutants in general. Additionally, TOC regulations are under consideration or have been adopted in certain states (e.g., California [0.5 mg/L] and Florida [3 mg/L]) for potable reuse applications. The basis for the difference between California and Florida is not justified in the literature, though recent extensive studies in Florida indicate that TOC levels in the range of 2.0 to 3.5 mg/L do not result in DBP exceedances (Altamonte Springs 2017). The Water Research Foundation is currently funding research investigating appropriate TOC guidelines for potable reuse applications.

• Municipal AWP Applications: Reduction of TOC in secondary or filtered secondary effluent by recycled water processes is highly dependent upon the treatment processes employed, with RO-based purification expected to provide much greater removal than other processes. In general, the RO process is a robust barrier to TOC, readily dropping TOC to <0.3 mg/L in RO permeate. In most cases, in this study, concentrations of TOC, DOC, and BDOC were lower in municipal AWP blends than conventional drinking water sources. Additionally, in all blends BDOC concentrations were within the range of potable water.

• Industrial AWP Applications: Industrial AWP applications did see reductions of TOC, DOC, and BDOC with the addition of industrial AWP water, as expected.

• Tertiary and $O_3$-BAC Applications: Concentrations of TOC, DOC, and BDOC in tertiary and $O_3$-BAC applications were similar or slightly higher than concentrations of these chemicals in conventional drinking water sources. This result is also expected, given the non-RO treatment train employed. Though non-RO processes are unable to attain such low TOC levels, work in Altamonte Springs (2017) reduced TOC to as low as 2 mg/L after $O_3$-BAC/GAC.

6) PFAs: Perfluoroalkyl substances (PFAs) represent an emerging concern for water supplies and can be challenging to remove with conventional treatment systems.

• Municipal AWP Applications: This study saw little change in concentrations of PFAs for municipal AWP applications when compared to conventional drinking water sources. This indicates that concentrations of PFAs were low in the AWP source waters and/or RO sufficiently removed PFAs during treatment. For both municipal AWP applications and conventional drinking water sources the concentrations of PFOA and PFOS were well below the EPA health advisory levels of 70 ng/L.

• Industrial AWP Applications: Similar to above.

• Tertiary and $O_3$-BAC Applications: This study found that PFAs are not well removed by $O_3$-BAC and tertiary treatment, resulting in higher concentrations of PFOA and PFOS in $O_3$-BAC and tertiary treated waters. No blend in this study, however, exceeded the EPA health advisory level. If treatment is needed, additional treatment barriers beyond $O_3$-BAC based systems would be required. Recent work
in Altamonte Springs (2017) indicated broad level PFAs removal by GAC in the Altamonte Springs purification train of O₃/BAC/UF/GAC/UV.

7) TOrCs: Trace organic compounds (TOrCs) represent a broad range of hormones, pharmaceuticals, and personal care products. While typically well below any health threshold, TOrCs represent a public perception concern for potable water reuse projects.

- Municipal AWP Applications: The majority of the TOrCs, which can include hormones, pharmaceuticals, and personal care products, were sampled in all blends (post treatment, but pre-pipe rig conditioning). Of the advanced treatment trains considered, AWP blends had the fewest TOrCs above the detection limit. These chemicals included: DEET and primidone.
- Industrial AWP Applications: Similar to above; however, industrial AWP applications also had detections of TCEP.
- Tertiary and O₃-BAC Applications: O₃-BAC blends had the most detections of TOrCs. The chemicals detected in O₃-BAC blends included DEET, primidone, TCEP, caffeine, fluoxetine, gemfibrozil, TCEP, sucralose, and meprobromate. None of the TOrC concentrations found were above known health guidance values.

**Blend Ratio**

For this project, a blend number, such as 75 percent, refers to the percentage of other raw or finished water supply (e.g. 75 percent) that is blended with 25 percent recycled water.

For the chemicals found at higher concentrations in recycled water where a larger safety factor may be desired, either the addition of further purification or limiting the blend ratio may be effective strategies. In some cases, the advanced treated wastewater had lower levels of chemicals then the natural blend water, so lower blend ratios may not produce higher quality water. Many of these chemicals were found to act conservatively (e.g., concentration based directly upon the concentration in the two waters and the blend ratio) and thus can be easily predicted for different blend ratios. Below is a summary of these findings.

1) Nutrients: The primary nutrient of concern is nitrate.

- Municipal AWP Applications: As described in the section above on nutrient concentrations in municipal AWP applications, nitrate was below regulated limits for all blends. Thus, for the waters used in this study, limiting blend ratio was not necessary to control nutrient concentrations. Nitrate (and total nitrogen for that matter) in municipal secondary effluent is well treated by RO-based purification.
- Industrial AWP Applications: For industrial AWP applications, high nitrate levels in the industrial AWP source water caused concentrations above the EPA regulated value of 10 mg/L (EPA 1992), highlighting the need for blending with low nitrate source water, if nitrate removal through biological processes is not done. For the waters considered, a blend of 88 percent recycled water would be sufficient to limit high nitrate levels.
- Tertiary and O₃-BAC Applications: Like industrial AWP applications, O₃-BAC applications measured in this study would need to be blended with low nitrate source water to meet EPA limits. At Utility 1, a blend of 50 percent recycled water would be sufficient and at Utility 3, a blend of 60 percent recycled water would be
sufficient to limit high nitrate levels. Alternatively, nitrate removal through biological processes may be required.

2) Bulk Water Quality: Bulk water quality comparisons focused upon parameters that may be predictive of corrosion, such as pH and alkalinity.
   • Municipal AWP Applications: As described above, blends with higher percentages of municipal AWP water had lower concentrations of turbidity, conductivity, and sulfate when compared to existing potable water supplies, thus improving existing water quality. However, blends with higher percentages of AWP water also had lower concentrations of alkalinity and pH when compared to existing potable water supplies, thus making the water more corrosive. Many of these parameters were found to act conservatively, so expected constituent concentrations can be determined for a utility's unique water profile on a case-by-case basis when choosing an appropriate blend percentage. These changes to water quality with the introduction of AWP must be balanced by adjusting the blend ratio to find the optimal water quality for the potable water distribution system.
   • Industrial AWP Applications: Industrial AWP waters saw higher concentrations of turbidity, chloride, and conductivity than conventional drinking water sources and lower sulfate concentrations than conventional drinking water sources. These parameters acted conservatively, so blended water quality can be balanced by adjusting blend ratio.
   • Tertiary and O₃-BAC Applications: Blends with higher percentages of tertiary and O₃-BAC water had higher levels of pH, turbidity, alkalinity, conductivity, and sulfate. These elevated levels of sulfate in O₃-BAC water exceed the EPA secondary standard at a 50 percent blend at Utility 3 and elevated levels of conductivity in O₃-BAC exceeded the CA secondary maximum contaminant level at a 100 percent blend in Utility 1. It was also found that the tertiary blend used in this study would exceed the turbidity limit at a blend of 30 percent. Again, many of the bulk water quality parameters tested in this study were conservative and thus utilities considering DPR can determine expected bulk water quality parameters for a particular blend by measuring parameter concentrations in the two waters considered for DPR.

3) DBPs: Disinfection byproducts (DBPs) can be formed by reclaimed water treatment (e.g., bromate by ozone, NDMA by chloramination) and reduced by reclaimed water treatment (e.g., NDMA by BAC, NDMA by UV). Further, reclaimed water processes that result in less TOC are assumed to contain less DBP precursors.
   • Municipal AWP Applications: Generally speaking, RO-based purification systems (e.g., AWP) have low DBP levels and low DBP formation potential. Thus, higher percentages of blend water are anticipated to improve water quality pertaining to DBPs. However, NDMA is only partially removed by AWP trains so high levels of NDMA in the source water can cause high levels of NDMA in the blended water if high percentages of municipal AWP water are used.
   • Industrial AWP Applications: Similar to municipal AWP applications.
   • Tertiary and O₃-BAC Applications: The opposite is true for O₃-BAC water which tends to have similar or higher TTHM FP to conventional raw water supplies (noting that further purification of O₃-BAC effluent with GAC substantially reduces both DBPs and DBP formation potential). In determining an appropriate
blending ratio, the concentration and character of TOC in an AWP (low TOC concentration) versus an O$_3$-BAC treated water (typically higher concentration) needs to be considered.

4) Organic Carbon: Organic carbon concentrations are assumed to be linked to DBP formation.
   - Municipal AWP Applications: As expected, blends with higher concentrations of municipal AWP water generally had lower concentrations of TOC, DOC, and BDOC, due to RO treatment. Thus, blends with higher concentrations of this municipal AWP would lower concentrations of organic carbon in the blended water.
   - Industrial AWP Applications: Similar to above.
   - Tertiary and O$_3$-BAC Applications: The opposite is true for the tertiary and O$_3$-BAC applications where RO was not used. As concentrations of organic carbon are higher in these waters, limiting the blend ratio would be one way to control blended water organic carbon concentrations.

5) PFAs
   - Municipal AWP Applications: PFAs act conservatively when blended. However, since concentrations of PFAs for municipal AWP applications were comparable to conventional drinking water sources, the impact of blending was minimal.
   - Industrial AWP Applications: Similar to above.
   - Tertiary and O$_3$-BAC Applications: PFAs are not well removed through O$_3$-BAC treatment and thus their concentration in the advanced water source water should be considered when picking an appropriate blend ratio. As PFAs act conservatively, their concentration at a certain blend ratio can be estimated for a proposed DPR system by measuring the PFA concentrations in the two source waters.

6) TOrCs
   - Municipal AWP Applications: The TOrCs measured all behaved conservatively and thus their concentration at a certain blend ratio can be estimated for a proposed DPR system. However, given their low levels in the waters considered in this study and the general lack of evidence of health effects at ng/L levels, blending to control TOrCs was not necessary for these source waters.
   - Industrial AWP Applications: Similar to above.
   - Tertiary and O$_3$-BAC Applications: O$_3$-BAC blends had the most detections of TOrCs. However, like municipal and industrial AWP applications, the low levels of TOrCs measured in this study made blending to control TOrCs unnecessary.

**Blend Location and Conventional Blend Water Quality**

The scenarios tested also varied in both their point of blending and the source of the conventional drinking water used for blending. The different scenarios considered are summarized below:

- Utility 1 - Surface water blended with advanced treated wastewater, then subjected to treatment via conventional water treatment techniques.
• Utility 3 - Surface water blended with advanced treated wastewater pre-treatment and then subjected to treatment via conventional water treatment techniques.
• Utility 4a - Two source waters: 1) Groundwater blended with advanced treated wastewater post-treatment and 2) surface water blended with advanced treated wastewater pre-treatment and then subjected to treatment via conventional water treatment techniques.
• Utility 4b - Surface water blended with advanced treated industrial wastewater post-treatment.

With these different scenarios, information on how blending with groundwater versus surface water and blending pre- versus post-treatment were considered. In general, groundwater versus surface water quality greatly impacted the blended water quality for some parameters. Additionally, pre- versus post-treatment affected pathogen and TOC concentrations.

1) Bulk Water Quality:
• Surface Water Blended Pre-Treatment: Generally surface water had low concentrations of conductivity. However, at Utility 3, surface water had conductivity concentrations above the CA Secondary maximum contaminant level resulting in all blends exceeding the CA secondary maximum contaminant level.
• Groundwater Blended Post-Treatment: Groundwater blends typically had higher levels of alkalinity, sulfate, and chloride than the surface water blends considered in this study. For Utility 2 (blends with groundwater), all blends had concentrations of sulfate above the EPA's limit and concentrations of conductivity above the CA secondary maximum contaminant level. In this case, conventional source water quality is important, as even a 50 percent blend with low conductivity, AWP water did not result in a conductivity level below the CA limit.

2) DBPs
• Surface Water Blended Pre-Treatment: This study found that concentrations of TTHM and HAA5 in blends with surface water were similar or higher than concentrations of TTHM and HAA5 in blends with groundwater blended post-treatment. This is likely due to lower DOC concentrations in groundwater sources.
• Groundwater Blended Post-Treatment: Same as above.

3) Pathogens
• Surface Water Blended Pre-Treatment: The point of introduction for advanced treated wastewater is important for determining pathogen log removal credits. If blending occurs prior to water treatment, additional log removal credits can be obtained, with the level defined by the treatment techniques (e.g., 2/2.5/3 log removal credit for virus/Giardia/Cryptosporidium, respectively [Texas Commission on Environmental Quality 2016]). The additional log removal credits obtained for water treatment at Utility 2 were sufficient for the pasteurized AWP blend to meet the 12/10/10 log removal goal.
• Groundwater Blended Post-Treatment: While the municipal AWP water at Utility 1 and Utility 4a were blended post-treatment and thus did not receive log removal credits for water treatment, based on measured pathogen data in the source water, we see that these blends meet the EPA's allowable risk level without the additional water treatment step. As mentioned above, however, the source water data collected
in this study represents limited data for each source water considered, and is not sufficient for determining a recycled water's safety.

4) Organic Carbon
- Surface Water Blended Pre-Treatment: For Utility 1 and Utility 3 where advanced treated wastewater was blended with surface water prior to treatment, the concentration of TOC, DBPs, and alkalinity found in the surface water can affect the required treatment technique in the final water treatment plant (as required by the Stage 1 Disinfectants and Disinfection Byproduct Rule to reduce DBP precursors). This rule was implemented to balance the risks between microbial pathogens and DBPs, as disinfectants used to control pathogens can react with naturally occurring organic and inorganic matter to form DPBs. Systems using surface water and conventional filtration are required to remove specified percentages of organic materials (measured as TOC) unless they meet alternative compliance criteria. All the blends in Utility 1 and Utility 3 met alternative compliance criteria and thus not subject to specified TOC removal requirements. In most cases, the Utility 1 and Utility 3 blends were exempt because of their low TOC, TTHM, and HAA5 concentrations. However, in 5 cases, where either the TOC or TTHM concentrations were higher, the blends' low SUVA_{254} values exempted these blends from this rule. Even though blending surface water with advanced treated wastewater can increase (or decrease) TOC, TTHM, and HAA5, the blend amounts of advanced treated wastewater considered in this study were not significant enough to trigger the Stage 1 Disinfectants and Disinfection Byproduct Rule.
  - Groundwater Blended Post-Treatment: All of the groundwater blends had TOC levels that were below the Stage 1 Disinfectants and Disinfection Byproduct Rule threshold of 2 mg/L.

5) PFAs
- Surface Water Blended Pre-Treatment: PFAs pass through conventional water treatment so the point of blending is not relevant for these parameters. No differences were seen in PFA concentrations for 100 percent groundwater and 100 percent surface waters.
  - Groundwater Blended Post-Treatment: Same as above.

6) TOrCs
- Surface Water Blended Pre-Treatment: Only sucralose, primidone, and meprobamate were measured at levels close to their respective MCLs in 100 percent surface water blends.
  - Groundwater Blended Post-Treatment: Of the TOrCs measured in this study, none were present in 100 percent groundwater blends.

**DISTRIBUTION SYSTEMS**

**Recycled Water Quality**

A primary goal of this study was to determine the effect of blending advanced treated wastewaters on distribution system water quality. Not only did this study look at the water quality immediately after blending and treatment, as described in the section above, but this study also quantified changes in water quality after spending time in a simulated premise plumbing system.
Results are positive for both AWP and O₃-BAC blends. The key findings from this analysis are summarized below.

1) Nutrients
   - Municipal AWP Applications: No significant changes were observed in nutrient levels pre- and post-pipe rig simulation for municipal AWP applications. Additionally, during the course of the pipe rig simulations for all utilities, results show the ammonia concentrations through the system did not change significantly indicating that nitrification most likely did not occur. However, where chlorine was used for disinfection, ammonia concentrations were higher than where chloramines were used for disinfection.
   - Industrial AWP Applications: Similar to above. No significant changes were observed in nutrient levels pre- and post-pipe rig simulation for industrial AWP applications.
   - Tertiary and O₃-BAC Applications: Similar to above. No significant changes were observed in nutrient levels pre- and post-pipe rig simulation for tertiary and O₃-BAC applications.
   - While these results are promising, in real distribution systems containing higher retention times, prolonged storage, and low flow conditions, elevated ammonia levels, if present, could still result in nitrification.

2) Bulk Water Quality
   - Municipal AWP Applications: No significant changes in pH, alkalinity, chloride, conductivity, or sulfate were observed over the course of the pipe rig testing for municipal AWP applications. This is true for tests with and without stabilization (Utility 4a only).
   - Industrial AWP Applications: Similar to above. No significant changes were observed in pre- and post-pipe rig simulation for industrial AWP applications. All industrial AWP applications included stabilization prior to pipe rig testing.
   - Tertiary and O₃-BAC Applications: Similar to above. No significant changes were observed in pre- and post-pipe rig simulation for tertiary and O₃-BAC applications.

3) Disinfection Residual Stability: The initial disinfection dose of either chlorine or chloramines was added to inactivate pathogens in the blended and treated waters and residuals were maintained. Of the four utilities, Utilities 1, 2, and 4 applied an initial dose of chlorine followed by ammonia at their full-scale facilities to produce chloramines for disinfection with target doses of 1.8 to 2.2 mg/L, 3.8 mg/L, and 2.6 mg/L total chlorine, respectively. Utility 3 had a chlorine target dose of 1.5 mg/L free chlorine.
   - Municipal AWP Applications: All municipal AWP applications used chloramines for disinfection. In general, almost all municipal AWP applications had limited or no increase in ATP concentrations post-pipe rig simulation. In most cases, these elevated ATP concentrations were still below 10 pg/mL, indicating a low level of activity and the efficacy of higher chloramine doses applied in these waters.
   - Industrial AWP Applications: Similar to above. Limited increases in ATP concentration were observed in post-pipe rig simulation for industrial AWP applications. For all industrial AWP applications chloramines were used for disinfection.
Tertiary and $O_3$-BAC Applications: The two tertiary blends, which used chlorine instead of chloramines for disinfection, had a much larger ATP increase and total chlorine decrease, most likely due to the higher relative demand as compared to other waters, as expected. It is also possible that the tertiary applications had a larger ATP increase post-pipe rig simulation because the tertiary applications had a lower amount of disinfection prior to blending and a higher organic load (food source) for microbiological activity.

4) DBPs

- Municipal AWP Applications: No significant changes in DBPs were observed over the course of the pipe rig testing for municipal AWP applications. This is expected, given that chloramination was used for disinfection.
- Industrial AWP Applications: Similar to above. No significant changes were observed in DBPs pre- and post-pipe rig simulation for industrial AWP applications. This is expected, given that chloramination was used for disinfection.
- Tertiary and $O_3$-BAC Applications: Utility 3, where chlorine was used for disinfection, had post-pipe rig exceedances for TTHMs for the $O_3$-BAC blends. Utility 3 also saw a significant increase in TTHM for tertiary blends post-pipe rig simulations. Also, where chlorine was used, HAA5 tended to increase (significant for a few samples) but otherwise no significant changes occurred for HAA5. No HAA5 exceedances were observed.

5) Metals and Corrosion Potential

- Municipal AWP Applications: Given the use of RO for municipal AWP applications, it is expected that some corrosion and metals leaching would occur in these lends. Metals leaching was observed in this study. At Utility 1, blends with higher concentrations of AWP water tended to have higher concentrations of lead and zinc in the pipe rig effluent than conventional drinking water sources, but no significant change was observed in the concentration of copper. However, at Utility 2, there was not a significant change in lead concentration over the duration of premise plumping testing for AWP blends. Pasteurization-AWP blends at Utility 2 did have greater concentrations of copper, zinc, and lead than AWP blends at a 50 percent blend ratio.
- Industrial AWP Applications: While industrial AWP applications also used RO treatment, the industrial AWP applications stabilized their water. Thus, an increase in lead, zinc, and copper leaching over conventional drinking water sources were not observed for industrial AWP applications.
- Tertiary and $O_3$-BAC Applications: Even though RO was not used for tertiary and $O_3$-BAC applications, some metals leaching was observed. $O_3$-BAC blends tended to have higher concentrations of copper and zinc in the pipe rig effluent than other advanced water treatment blends.

To predict these increases in metal concentration, corrosion indices were calculated for all blends tested in this study. However, based on the results of this study, there was no universal corrosion index based on water quality parameters that accurately estimate the extent of lead corrosion that might occur after DPR implementation. There are individual instances when corrosion indices do work well for predicting lead, copper, and/or zinc corrosion. These instances
appear to be case specific and indicate a need for a utility-specific study prior to DPR implementation.

In some instances, the microbial gene density correlated well with lead corrosion from CDA360 brass although there was no correlation with copper or zinc corrosion. Considering Utilities 1, 2, and 3, the logarithm of 16S rRNA genes (universal to all bacteria and measured as a proxy for total bacteria count) in the biofilm correlated well with lead leaching in the pipe rigs ($R^2 = 0.72$). However, the experimental plan for this study was not designed to evaluate the mechanisms of this correlation. Therefore, this observation should be evaluated further in future studies. Utility 4 was excluded from this correlation because it was the only utility in this study that employed corrosion control.

1) Biological Corrosion: Biological corrosion was also measured for all blends, and in general advanced treated wastewater blends had similar levels of biological corrosion as groundwater and surface water sources. All water blends saw moderate or aggressive sulfate reducing bacteria (SRB) and sulfate forming bacteria (SFB) biological corrosion.
   - Municipal AWP Applications: Municipal AWP applications also inconsistently saw acid producing bacteria and denitrifying bacteria; however, these occurrences were typically moderate or non-aggressive occurrences.
   - Industrial AWP Applications: No additional biological corrosion was observed.
   - Tertiary and O₃-BAC Applications: Tertiary and O₃-BAC applications also inconsistently saw acid producing bacteria and denitrifying bacteria; however, these occurrences were typically moderate or non-aggressive occurrences.

2) Opportunistic Pathogens: Advanced treated wastewater did not appear to produce more regrowth of opportunistic pathogens (OPs) during simulated distribution than the corresponding traditional potable water source. This was true across municipal AWP, industrial AWP, tertiary, and O₃-BAC applications. As part of the study of OPs, 16S rRNA genes were also monitored. These genes are universal to all bacteria and included as a proxy for total bacteria count. In all scenarios regrowth of 16S rRNA genes consistently occurred.

3) Bacteria Containing Antibiotic Resistant Genes: All utilities and tested scenarios were highly effective at limiting regrowth of bacteria containing the antibiotic resistant genes (ARGs) of interest during 8 weeks of simulated distribution. An ARG was only detected in a single sample at above quantifiable levels: the vanA resistant gene in the biofilm of Utility 1’s 90 percent surface/10 percent AWP condition. While these results are very promising indicating the DPR waters do not pose added risk in terms of proliferating ARGs in the distribution system, it is important to consider that these were short term studies with young biofilms.

4) Total and Resistant Heterotrophic Plate Counts
   - Municipal AWP Applications: In this study, less regrowth of total and resistant heterotrophic plate counts (HPCs) was seen in AWP blends than in traditional potable water sources. However, some regrowth of HPCs in simulated distribution systems was consistently observed. HPCs capable of growth in the presence of several antibiotic classes, including ampicillin and oxacillin, were detected consistently across utilities.
• Industrial AWP Applications: Like municipal AWP applications, industrial AWP applications had less regrowth of total and resistant HPCs than in traditional potable water sources.
• Tertiary and O₃-BAC Applications: As expected, results from this study indicate that tertiary applications have the potential to support regrowth of total and resistant HPCs in distribution systems.

Blend Ratio

Many of the parameters measured did not change during premise plumbing testing, thus the effects of blend ratio on post-pipe rig water quality are no different than the effects outlined in the Treatment Systems section above. However, where changes were observed in post-pipe rig water quality, blend ratio was considered. The results of that analysis are shown below.

1) Total and Resistant Heterotrophic Plate Counts
• Municipal AWP Applications: As described above, in this study, less regrowth of total and resistant HPCs was seen in AWP blends than in traditional potable water sources. Thus, it is possible that higher blends of municipal AWP applications could help reduce regrowth of HPCs. However, this should be studied further.
• Industrial AWP Applications: Similar to municipal AWP applications.
• Tertiary and O₃-BAC Applications: As described above, tertiary applications have the potential to support regrowth of HPCs in distribution systems. However, limiting the blend ratio to 5 percent was effective at reducing regrowth of total HPCs by approximately two orders of magnitude compared to a 10 percent blend ratio.

Blend Location and Conventional Blend Water Quality

Many of the parameters measured did not change during premise plumbing testing, thus the effects of blending location and conventional blend water quality on post-pipe rig water quality are as outlined in the Treatment Systems section above. Additionally, no significant and universal correlation between blend location or conventional blend water quality and metals leaching or corrosion was observed in this study. While conventional water treatment removes metals, the post-pipe rig lead, copper, and zinc concentrations were not significantly different when waters were blended pre- versus post- water treatment.

ENGINEERED STORAGE

Recycled Water Quality

Engineered storage was not used for the treatment systems evaluated in this work. It is expected that engineered storage will be an important part of any DPR system to provide Failure Response Time (FRT) and additional treatment for an advanced treated wastewater prior to blending. The time in storage is assumed to range from 30 minutes to as much as a day, though 2 hours of storage appears to be a number that is preferred by utility partners. As it is expected that engineered storage would be added to the system prior to blending with other water supplies, only the effect of recycled water quality on engineered storage was considered. It was assumed that in
many ways an engineered storage buffer (ESB) would act similarly to a pipe rig. The results from this study, while critically important to understanding the implications of blending on downstream water quality for potable reuse projects, do not directly affect the sizing paradigm for ESBs. However, the results do affect the design of ESBs as described below.

1) Metals and Corrosion Potential: Corrosivity concerns are particularly acute for the ESB, as it will typically not have the benefit of blending with higher-alkalinity water prior to storage, so a compatible tank material or coating will be needed in combination with chemical stabilization of that water. Utilities 1, 2, and 3 did not chemically stabilize their blends and saw mixed corrosion results in AWP blends. In some cases, higher blends of AWP water increased lead and zinc leaching and in other cases no changes were observed. Pasteurized-AWP water did increase lead, zinc, and copper leaching and O₃-BAC water increased copper and zinc leaching. Utility 4 was the only utility to stabilize their water blends. At this utility, metals leaching was generally low and the presence of AWP water either did not affect or decreased the concentration of metals in the blends measured.

2) DBPs: It was found that for O₃-BAC waters where TTHM FP was higher than conventional drinking water sources, disinfecting with free chlorine caused the concentrations of both TTHM and HAA5 to increase in the pipe rig. This trend was not observed in blends where chloramine disinfection was used. This finding, while not new, is important when considering using an ESB with free chlorine to achieve additional log reduction credits for pathogen removal.

3) Pathogens: Based on the limited data collected, log removal value requirements from secondary effluent to finished water were determined for Giardia, Cryptosporidium, and enteric virus with maximum values of 7.5-log, 4.6-log, and 12.6-log, respectively. This dataset is particularly interesting because it implies less treatment is necessary for Cryptosporidium for every scenario studied than required by any current permitting process, whereas the maximum value for enteric virus implies a greater treatment need than California's 12-log standard for IPR. The results of this study lend some credence to the need for monitoring requirements in the water feeding a DPR system to better characterize the variability and concentrations of pathogens. With this pathogen source water characterization, a probabilistic quantitative microbial risk assessment (QMRA) can be performed to ensure the EPA risk levels are met (Independent Advisory Panel 2015). Such an analysis will document the value of free chlorination of virus, which could occur in the ESB should DBP formation not be a concern. If through this risk assessment it is determined that less log removal credit is needed to meet EPA risk levels than is currently required, the ESB size and detention time may decrease, thus lowering the cost and infrastructure impact of a DPR facility. A shorter detention time in the ESB would also limit DBP formation and corrosion concerns discussed above.

SUMMARY

The first goal of this study was to examine the impact of reclaimed water quality on the downstream treatment and distribution systems. From this examination it can be concluded that in general, most of the reclaimed water blends considered met the majority of water quality regulations indicating that reclaimed water quality did not detrimentally impact overall water quality. In particular, none of the TO Côs measured in any blend were above notification levels,
guidance values, or regulations in the U.S. and in fact the majority of TOrCs were below the detection limit. This may be due to low levels of TOrCs in the advanced treated wastewaters' source water quality or the effectiveness of treatment and blending. Regarding OPs and ARGs, the reclaimed water blends in this study did not produce more regrowth of OPs or ARGs during the simulated distribution than corresponding traditional potable water sources. Furthermore, this study found that reclaimed water blends that used RO and UV/AOP resulted in less regrowth of total and resistance HPCs in the simulated distribution system than the use of traditional potable water sources. While this is promising, more research is needed using older biofilms. Unstabilized reclaimed water did negatively affect corrosion in the simulated distribution systems, even after blending with conventional water supplies. Thus, stabilization of reclaimed water, prior to blending, is recommended.

The second goal of this study was to examine the impact of blend ratio on the downstream treatment and distribution systems. As expected, many measured chemicals behaved conservatively when blended. For these parameters, it should be sufficient to use the blend ratio to estimate expected blended concentration and determine the optimal blending percentage. If blending municipal AWP water, one should keep in mind that this water is generally very low in organics, nutrients, salts, and microbiology. This water also tends to be more aggressive. If blending industrial AWP water, one should keep in mind that, while similar to municipal AWP, such waters may be higher in nutrient and organic load. If blending tertiary or \( \text{O}_3 \)-BAC water, one should keep in mind that its water chemistry tends to be similar to raw surface waters with higher TOC, nutrients, and microbiology. Furthermore, particular attention should be paid to nitrate and NDMA, as no nitrate or NDMA removal was seen in the \( \text{O}_3 \)-BAC process and only partial removal was observed in the AWP trains. PFAs are also of concern as limited to no removal of PFAs was observed during \( \text{O}_3 \)-BAC. For these chemicals, limiting blend ratio or removing them with additional advanced treatment processes (e.g., GAC) may be necessary.

The third goal of this study was to examine the impact of blend location and conventional blend water quality on the downstream treatment and distribution systems. In general blend location did not significantly impact downstream treatment or distribution systems except for meeting pathogen reduction requirements. Conventional blend water quality was important and, in some cases, high levels of chemicals in the conventional blend water caused regulation exceedances. Generally, it was found that groundwater had higher concentrations of conductivity, alkalinity, sulfate, and chloride than surface water but surface water had higher DOC concentrations than groundwater. This should be taken into consideration when determining an appropriate blend ratio.

**NEXT STEPS**

Several recommendations for future studies were drawn from this project and are presented below:

1) Better characterization of pathogen densities in source water (secondary effluent) is needed. This would help in developing uniform water recycling criteria for DPR that uses a probabilistic approach in applying log reduction credits. This approach is further discussed in the California State Water Resources Control Board's Expert Panel Report (Olivieri et al. 2016).

2) To calculate virus log removal credits, this study conservatively assumed that all gene copies were viable pathogens. However, gene copy numbers do not necessarily
correlate to viable pathogens. More research is needed to determine the best approach for measuring accurate virus log removal credits given this limitation.

3) Further studies are needed to fully explore the role of biological corrosion in premise plumbing systems from all types of water blends.

4) Further study on the biofilm and lead leaching in premise plumbing systems is needed. One observation of this study was that 16S rRNA correlated fairly well with lead leaching. However further consideration should be devoted to this observation in future studies.

5) While these studies are promising in their indication that these DPR blending scenarios do not present added risk in terms of enhancing re-growth of ARGs or ARBs as compared to conventional drinking water sources, it is important to consider that these were short-term studies. Longer term studies with mature biofilms would be of value.

6) This study limited the blend ratio to 50 percent. Future studies could look at higher blend ratios as it applies to corrosion. Those studies should include stabilization.
APPENDIX A
WATER QUALITY GOALS FOR DPR

WATER QUALITY GOALS FOR DPR

One of the main criteria in judging water quality is public health protection, and the two principal groups of contaminants that may present health threats are microbial pathogens and toxic chemicals.

Chemicals of concern include both regulated and unregulated chemicals, including chemicals of emerging concern (CECs), trace organic compounds (TORCs), and pharmaceuticals and personal care products (PPCPs) (Trussell et al. 2013). Regulators typically differentiate risks presented by the presence of chemicals from that associated with pathogens due to the nature of their effects. Unlike the chemicals, which largely present a chronic health concern, pathogens cause acute health effects with a wide range of potential clinical manifestations (Regli et al. 1991). The severity of these effects varies depending on both the microorganism and the level of exposure.

During the development of the Surface Water Treatment Rules, the U.S. Environmental Protection Agency (EPA) evaluated the risks from both the microbial and chemical contaminants, and concluded that “an immediate implication of this work is the realization that microbial pathogens continue to dominate the comparative waterborne human health risks” (Macler and Regli 1993). The focus of this portion of the literature review is therefore largely on the threat from microbial contaminants.

Regulatory Frameworks to Control Risk

The first U.S. drinking water standards, developed in 1914, sought to achieve microbiological safety through the reduction of coliform bacteria (Crittenden et al. 2012). The idea was that a water that a combination of (1) treatment to achieve a low turbidity and (2) measurement of indicator bacteria levels at or below the detection limit could be used to demonstrate that the water was virtually free of pathogens, and therefore appropriately safe for consumption. This general strategy—creating a water that is demonstrably “free of pathogens”—remained largely in place for almost a century of regulations, and is credited as one of the great successes in the protection of public health (Cooper et al. 2012).

As the fields of public health and microbiology continued to grow, however, the drinking water industry became aware that there were other pathogens that are less sensitive to treatment than bacteria. These pathogens—including viruses and protozoa—showed increased resistance to common disinfection techniques and higher ability to pass through physical treatment processes, such as media filtration. In addition, coliforms are not conservative indicator microorganisms for viruses and protozoa. Moreover, the infectivity of the viral and protozoan pathogens is often much higher, meaning that lower doses can cause infections. For some of these pathogens, it is believed that even a single viral particle or protozoan cyst or oocyst can cause infection.

One of the problems in understanding the risk from these viral and protozoan pathogens was the inability of epidemiological studies to detect infection at low risk levels. Furthermore, it is not possible to measure pathogens in water directly to determine risk. This limitation stems from multiple factors including the difficulty in accurately quantifying occurrence, uncertainties in pathogen infectivity and virulence, variability and diversity in pathogen occurrence, and the large volumes of samples required to demonstrate negligible risk levels (Regli et al. 1991).
Consequently, EPA turned to risk assessment methods to help understand the public health threat from viral and protozoan pathogens, and to develop regulations that would provide public health protection at targeted levels of risk. This was an important shift in that it moved away from the concept that a safe water was “free of microbes,” and provided the ability to understand risks at levels beyond those detectable with less sensitive tools (e.g., epidemiological studies). The main benefit of moving to this framework was that it could now address low levels of contamination, and improve the development of policies and regulations that manage these risks.

The Surface Water Treatment Rule (SWTR; EPA 1989) brought a movement toward the incorporation of risk into US regulations concerning microbial contaminants. In 1996, amendments to the Safe Drinking Water Act (SDWA) required that EPA consider a detailed risk and cost assessment, in addition to the best peer-reviewed science, when developing standards (EPA 1996). The level of microbial risk that is generally cited as “negligible” or de minimis is based on achieving an annual risk of infection of less than one person in 10,000. This is often referred to in shorthand as the “1 in 10,000 risk level.” As described by the EPA, these de minimis levels are not always strictly science-based, but rather risk management decisions that also factor in other elements such as feasibility, and increased sensitivity of certain subpopulations (including elderly, young, and immunocompromised) that are more apt to experience both infection and disease when exposed to a lower pathogen dose (Macler and Regli 1993).

Thus, the 1 in 10,000 risk level forms the Texas Commission on Environmental Quality's (TCEQ's) basis of permitting Texas DPR projects, which it developed during permitting of the Raw Water Production Facility in Big Spring, the first DPR project in the US, which came online in spring 2013.

In addition, WateReuse Research Foundation Project 11-02 (Trussell et al. 2013) evaluated and compared a number of risk frameworks, and concluded the 1 in 10,000 risk level was more conservative than others. In agreement with the TCEQ's approach, Trussell et al. (2013) also posit that potable reuse projects should seek to achieve the 1 in 10,000 risk level, with the logic that potable reuse projects need to provide equal (or better) protection than existing drinking water regulations. The use of the 1 in 10,000 risk level was also endorsed by an NWRI Expert Panel that was convened to provide input on appropriate public health targets for direct potable reuse (Crook et al. 2013). The resulting recommendations for pathogen levels in potable water are summarized in Table A.1 below.

<table>
<thead>
<tr>
<th>Pathogen</th>
<th>Concentration</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Viruses</td>
<td>$2.2 \times 10^{-7}$ enteric virus infectious units per liter</td>
<td>Regli et al. (1991)</td>
</tr>
<tr>
<td>Giardia</td>
<td>$6.8 \times 10^{-6}$ cysts per liter</td>
<td>Regli et al. (1991)</td>
</tr>
<tr>
<td>Cryptosporidium</td>
<td>$3 \times 10^{-5}$ oocysts per liter</td>
<td>Haas et al. (1996)</td>
</tr>
<tr>
<td>Salmonella</td>
<td>$5.3 \times 10^{-5}$ enteric virus IU/L</td>
<td>Trussell et al. (2013)</td>
</tr>
</tbody>
</table>

Source: Adapted from Trussell et al. 2013.
Risk-Based Drinking Water and Potable Reuse Regulations

Microbial Constituents

One of the top goals of the US drinking water regulations is to make treated waters microbiologically safe. For the EPA, this means that the water must be treated to prevent the outbreak of illnesses, but also to minimize any effect of drinking water on endemic level of disease (Macler and Regli 1993). The main regulations that pertain to microbial contaminants in potable water derive from the 1989 Surface Water Treatment Rule (SWTR), which assumed that the 1 in 10,000 risk levels would be met for virus and *Giardia* if a specific level of treatment were provided. By introducing source water monitoring, the Long-Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR), requires augmented treatment if the mean source water *Cryptosporidium* levels correspond to an estimated annual risk of infection of 2 in 1,000 people or greater (Cooper et al. 2012, EPA 2006). Detailed information about microbial removal targets and their rationale from a potable reuse perspective is presented in Trussell et al. (2013) and NWRI (2013a).

Most potable reuse guidelines and regulations aim to achieve similar risk levels as those used in drinking water. In the US, both California and Texas have developed pathogen goals based on the 1 in 10,000 risk level, with analogous goals used internationally (Natural Resource Management Ministerial Council et al. 2008, WHO 2011, McDonald and Nellor 2015).

Chemical Constituents

Chemicals are more diverse and are often present at higher concentrations in wastewaters, so additional guidance is frequently provided. Besides meeting the chemical criteria regulated under the Safe Drinking Water Act, there are often additional criteria to ensure removal and control of otherwise unregulated trace level pollutants (CDPH 2014, Crook et al. 2013, NWRI 2013a, Trussell et al. 2013).

The EPA sets legally enforceable standards for drinking water in the form of National Primary Drinking Water Regulations, or primary standards. The purpose of these standards is to protect public health by setting limits on the levels of contaminants that are permissible in drinking water. Standards are created by first determining a goal for each contaminant. This maximum contaminant level goal (MCLG) is based on assessing the concentration of the contaminant at which no adverse health effects are anticipated, and includes a margin of safety. With this MCLG as a target, the EPA selects a standard, known as a maximum contaminant level (MCL), which balances the benefits of achieving the MCLG while accounting for technical, analytical, economic, and other limitations.

The EPA requires compliance with these regulations in one of two forms. For most chemicals, compliance requires demonstrating that treated water quality is at or below the concentrations specified by the MCL. For other constituents, water systems must provide a treatment technique (TT), i.e., an enforceable treatment process or procedure that is required for removal of a measurable surrogate contaminant that is used to ensure effective removal of other contaminants. TT’s are used when a reliable, economical, and technically feasible method to measure a contaminant’s level of concern cannot be determined. Currently, there are eighty-eight constituents with primary standards, broken down into six main groups: disinfectants (3), disinfectant byproducts (DBPs) (4), inorganic chemicals (16), microorganisms (8), organic chemicals (53), and radionuclides (4) (EPA 2016a).
In addition to the primary standards, EPA has also developed non-enforceable secondary standards that address aesthetic issues, such as taste, odor, and appearance of water. Currently, there are 15 secondary standards specified by the EPA (EPA 2016b).

States may establish additional constituents for regulation, or impose more stringent requirements, but must meet the federal requirements at minimum. California, for example, has both additional requirements—including perchlorate and hexavalent chromium—as well as more stringent standards for some chemical compounds (DDW 2017).

CHARACTERIZATION OF WATER QUALITY AFTER VARIOUS LEVELS OF TREATMENT

The industry’s literature on treatment of recycled water to various standards is extensive. Within the sections below are summaries of pathogen and pollutant concentrations, and their removal through various treatment processes. As an overview, Table A.2 below provides a qualitative summary of treatment expectations as well as data gaps to be filled as part of this project. This table highlights the need to understand the corrosion impacts and premise plumbing issues with blending different qualities of recycled (and purified recycled) water with other more “conventional” raw water supplies. This project investigated these data gaps, and provided detailed analysis of pathogens, disinfection byproducts (DBPs), and trace pollutants as they apply to DPR.

<table>
<thead>
<tr>
<th>Pathogens</th>
<th>Nutrients</th>
<th>DBPs</th>
<th>Trace pollutants</th>
<th>Corrosion impacts</th>
<th>Premise plumbing</th>
</tr>
</thead>
<tbody>
<tr>
<td>Secondary effluent</td>
<td>X</td>
<td>XX</td>
<td>X</td>
<td>X</td>
<td>Data Gap</td>
</tr>
<tr>
<td>O3/BAF</td>
<td>X</td>
<td>--</td>
<td>O</td>
<td>XX</td>
<td>Data Gap</td>
</tr>
<tr>
<td>Chlorination</td>
<td>X</td>
<td>--</td>
<td>O</td>
<td>X</td>
<td>Data Gap</td>
</tr>
<tr>
<td>MF/UF</td>
<td>X</td>
<td>--</td>
<td>--</td>
<td>--</td>
<td>Data Gap</td>
</tr>
<tr>
<td>RO/NF</td>
<td>X</td>
<td>XX</td>
<td>--</td>
<td>X</td>
<td>Data Gap</td>
</tr>
<tr>
<td>UV AOP</td>
<td>XX</td>
<td>--</td>
<td>X</td>
<td>XX</td>
<td>Data Gap</td>
</tr>
</tbody>
</table>

X = Good performance.
XX = Excellent performance.
-- = Limited performance.
O = Variable impact from positive to negative, depends on location in treatment train.
To determine the level of risk in a potable application, a few pieces of information are needed:

1) Concentrations of pathogens and chemical contaminants in the source water;
2) Reductions in contaminant concentrations through treatment;
3) Target contaminant concentrations in the final treated product water, recognizing that in a DPR situation, contaminant removals should ultimately satisfy drinking water regulations; and
4) Known relationships between contaminant concentrations and risk.

Various water sources could be used as the starting point in these calculations. For conventional drinking water scenarios, source waters are typically surface waters or groundwaters. Potable reuse scenarios begin these calculations with either raw wastewater or secondary effluent. A third scenario exists, which is a mixture of the previous two, also called *de facto* reuse. Water qualities of these water types are discussed here.

Trussell et al. (2013) provides detailed information about pathogens that can be found in raw wastewater, along with their respective concentrations and associated risks. However, quantifying raw wastewater pathogen concentrations presents several challenges given the complexity of the wastewater matrix. This may also be an issue for primary effluents that may contain a high degree of particulate and organic matter. Primary effluents may also lead to lower recoveries of microorganisms when compared to secondary effluents, which may have fewer matrix interferences.

As a result, some institutions prefer to begin their risk calculations starting with the pathogen loads in the secondary effluent, with the logic that pathogen data in secondary effluents is of higher quality. Cooper et al. (2012) tabulated average pathogen concentrations in secondary effluents. Due to the diversity of processes currently being used in wastewater treatment, combined with the potential variability in influent concentrations, concentrations of pathogen in effluent may vary significantly. As a result, treatment plants that use secondary effluents as their starting point should first characterize their effluents for a period of time prior to the calculation of removal requirements.

This is exactly the approach taken by the TCEQ to permitting DPR in Texas, which requires an extensive characterization of the effluent proposed as a source water for DPR to determine the level of treatment needed (McDonald and Nellor 2015). Through its permitting efforts for existing and proposed DPR projects, the TCEQ is also rapidly accumulating a sizeable dataset on pathogen concentrations in effluent, and indeed, pathogen concentrations, and the associated microbial LRV targets, have varied significantly, with at least a 2-log range spanned by the effluents characterized to date.

Surface waters are receiving increasing amounts of wastewater effluent discharges, and these blended waters are subsequently used as drinking water sources. As mentioned at the beginning of this literature review, this scenario refers to *de facto* reuse. De degree of *de factor* reuse is growing, along with its study (EPA 1980, NRC 2012, Rice et al. 2013, Brooks et al. 2006). The National Research Council (NRC) evaluated a number of reuse scenarios to estimate the risk from both planned and *de facto* reuse. Based on their analysis, which consisted of a series of risk exemplars evaluating bacteria, protozoa, and virus, they concluded that planned reuse is as safe or safer (lower risk) than drinking water systems that are subject to *de facto* reuse (NRC 2012). Given
existing data needs, many assumptions were required in these calculations, but the results underscore the importance of better quantifying the extent and risk of *de facto* reuse.

Regarding the second item listed above, advanced treatment trains are technically capable of transforming raw sewage into high quality water. Trussell et al. (2013) provide detailed information about removal of microbial, chemical, and aesthetic contaminants by various treatment trains used for potable reuse applications, along with their equivalency. WateReuse Research Foundation Project 13-12 also provides additional information about the impact of treatment processes. Among others, this project aimed at evaluating upstream wastewater treatment impacts on DPR source water and downstream advanced treatment processes.

The microbial indicators that are used to assess treatment performance in DPR applications are *Cryptosporidium*, *Giardia*, and enteric viruses. Trussell et al. (2013) proposed that non-typhoid *Salmonella* be added as a bacterial indicator of acute gastrointestinal illnesses. Regarding chemical contaminants, NWRI (2013a) proposes chemicals of potential health concern that should be used to evaluate the effectiveness of chemical removal by treatment trains. Ultimately, public health protection is the most critical factor in characterizing the performance of a treatment strategy. However, cost- and energy efficiency of the various treatment processes also drive project decisions.
APPENDIX B
RESEARCH METHODS

ORGANIZATION OF TEST PROTOCOL

The standard operating procedure (SOP) outlined for the simulated water treatment tests were based upon processes used at the participating utility from which the test waters are received. All laboratory work in this protocol was conducted using appropriate preventive protection equipment (PPE) including lab coats, safety goggles, and appropriate gloves.

PREPARATION OF STORAGE CONTAINERS AND GLASSWARE

Unless otherwise stated, one-gallon (1 GAL) amber glass jugs and glassware used in this study were cleaned following the USEPA’s cleaning procedures listed in the OSWER Directive 9240.0-05A “Specifications and Guidance for Contaminant-Free Sample Containers”:

1) Wash with Equinox non-residue detergent/ultrapure water mixture.
2) Rinse thoroughly with tap water, followed by ultrapure Milli-Q water.
3) Acid rinse bottles in 10% HNO₃/10%HCl acid bath solution.
4) Rinse in de-ionized water.
5) Add methanol to rinse and then air-dry until bottles are completely dried.

BLENDING PROTOCOL FOR SOURCE WATERS

Table B.1
Materials for blending source waters

<table>
<thead>
<tr>
<th>Item</th>
<th>Size/Quantity</th>
</tr>
</thead>
<tbody>
<tr>
<td>Glass beaker mugs</td>
<td>1 L</td>
</tr>
<tr>
<td>Graduated measuring cylinder</td>
<td>2 L</td>
</tr>
<tr>
<td>Trace clean amber jugs</td>
<td>1 GAL</td>
</tr>
<tr>
<td>Stirring rod</td>
<td>1</td>
</tr>
<tr>
<td>Wypall L30 White paper wipes</td>
<td>1 box</td>
</tr>
<tr>
<td>33-gallon Uline stainless steel container</td>
<td>2</td>
</tr>
<tr>
<td>Roll of white paper labels</td>
<td>1</td>
</tr>
<tr>
<td>Black sharpie markers</td>
<td>2</td>
</tr>
</tbody>
</table>

Non-Blended Water

1) Prepare labels (name of sample, initial of sampler, date of sampling) for amber glass jugs and stainless steel containers using black sharpie marker and roll of paper labels. Stick labels on each amber glass jug.
2) Empty water samples from plastic cubitainers as received from Utility into 30 GAL U-line stainless steel container.
3) Using 1 L glass beaker mugs, dispense water samples into the graduated measuring cylinders. Measure exactly 3785 ml of water sample (1 US GAL = 3785 ml) into the measuring cylinders and dispense into each 1-GAL amber glass jug.
4) Wash the stainless containers with soap and rinse with tap water, and ultrapure water.
5) Proceed to the proposed simulation tests outlined in this protocol for participating utility.

Blended Water

1) Repeat step 1 protocol for the non-blended water.
2) Measure out required volumes for water to be blended and mix in a 30 gallon stainless steel container. For example, for 20 gallons of 10% blended biofiltration water: mix 2 US gallons (3785 mL x 2) of biofiltration water with 18 US gallons of raw surface water.
3) Repeat steps 3 – 5 protocol for the non-blended water.

Proposed Simulation Tests

- Ozonation
- Coagulation/flocculation/Sedimentation
- Filtration
- Chlorination
  - Pasteurization
  - Chloramination
OZONATION: STANDARD OPERATING PROCEDURE (SOP)

Indigo II Solution

The indigo method for determining dissolved ozone residual requires a stock indigo solution. Fresh indigo stock solution should be made within 24 h of the pre-ozonation tests and the solution can be used within 1 – 2 weeks from preparation.

Table B.2
Materials for Indigo II solution

<table>
<thead>
<tr>
<th>Item</th>
<th>Size/Quantity</th>
</tr>
</thead>
<tbody>
<tr>
<td>Volumetric flask with stopper</td>
<td>1 L</td>
</tr>
<tr>
<td>Spatula</td>
<td>1</td>
</tr>
<tr>
<td>Graduated cylinder</td>
<td>10 mL</td>
</tr>
<tr>
<td>Mettler balance</td>
<td>1</td>
</tr>
<tr>
<td>Fume hood</td>
<td>1</td>
</tr>
<tr>
<td>Weigh boat</td>
<td>2</td>
</tr>
<tr>
<td>Wypall paper wipes</td>
<td>1 box</td>
</tr>
<tr>
<td>Amber bottle with dispenser</td>
<td>1 L</td>
</tr>
<tr>
<td>Phosphoric acid (H₃PO₄, Fisher Scientific, 85% w/w)</td>
<td>7 mL</td>
</tr>
<tr>
<td>Sodium phosphate monobasic (Fisher Scientific, 98-102%)</td>
<td>10 g</td>
</tr>
<tr>
<td>Potassium indigotrisulfonate (Sigma Aldrich, 60%)</td>
<td>77 mg</td>
</tr>
<tr>
<td>Ultrapure deionized water (Milli-Q Reference Water Purification System)</td>
<td>1</td>
</tr>
</tbody>
</table>

Indigo II Solution Preparation

1) Add 500 mL of Milli-Q ultrapure water to a clean 1000 mL volumetric flask
2) Under a fume hood measure 7 mL of phosphoric acid into a 10 mL graduated cylinder. Pour acid into the center of the volumetric flask. Invert the flask with stopper to mix the acid.
3) On the Mettler balance, zero out a weigh boat and measure 10 g of mono sodium phosphate. Record the actual amount in lab notebook. Add to the volumetric flask and completely mix by inverting the flask with a stopper.
4) Zero out another weigh boat on the Mettler balance and measure 77 mg (0.077 g on the scale display) of the Indigo crystals. Record the value. Carefully mix the indigo solution into the flask using ultrapure water to wash all the indigo off the boat.
5) Fill the volumetric flask with reagent ultrapure water to the 1000 mL mark and mix the solution.
6) Transfer this solution into the amber bottle with a dispenser.
7) Clean all equipment and place the flask, spatula, and graduated cylinder in the appropriate spot for glassware.
8) Record the date and the Indigo ID# on the amber bottle. ID should consist of the six-digit date – 01 for the first batch of the day, - 02 for the second batch etc.
9) Store the indigo solution in the refrigerator when not in use.
10) Record the following information in the “Solutions Logbook:” date, initials, amount of phosphoric acid, mono sodium phosphate, and indigo used, batch number, indigo information.

**Potassium Iodide Solution**

Fresh potassium iodide solution should be made within 24 h of the pre-ozonation tests and the solution can be used within 1 – 2 weeks from preparation.

<table>
<thead>
<tr>
<th>Table B.3</th>
<th>Materials for potassium iodide solution</th>
</tr>
</thead>
<tbody>
<tr>
<td>Item</td>
<td>Size/Quantity</td>
</tr>
<tr>
<td>Gas washing bottles</td>
<td>500 mL</td>
</tr>
<tr>
<td>Potassium iodide (Acros Organics, 99+%): 50 g</td>
<td></td>
</tr>
<tr>
<td>Graduated cylinder</td>
<td>500 mL</td>
</tr>
<tr>
<td>Mettler balance</td>
<td>1</td>
</tr>
<tr>
<td>Weigh boat</td>
<td>1</td>
</tr>
<tr>
<td>Ultrapure deionized water (Milli-Q Reference Water Purification System): 1</td>
<td></td>
</tr>
<tr>
<td>HDPE Plastic container (Nalgene): 500 mL</td>
<td></td>
</tr>
</tbody>
</table>

**Potassium Iodide Solution Preparation**

1) Measure 500 mL of ultrapure water into a 500 mL plastic container.
2) Weigh out 50 g Potassium iodide (KI) solution on a Mettler balance.
3) Add 50 g KI into the 500 mL plastic container containing the ultrapure pure reagent water.
4) Shake the mixture and allow for the crystals to dissolve completely.
5) Store the solutions at room temperature until ready for use.
6) Transfer the KI solution into a gas washing bottle when ready for pre-ozonation.
Sodium Thiosulfate Solution

Table B.4
Materials for sodium thiosulfate solution

<table>
<thead>
<tr>
<th>Item</th>
<th>Size/Quantity</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mettler balance</td>
<td>1</td>
</tr>
<tr>
<td>Weigh boat</td>
<td>1</td>
</tr>
<tr>
<td>Ultrapure deionized water (Milli-Q Reference Water Purification System)</td>
<td>1 L</td>
</tr>
<tr>
<td>HDPE plastic container (Nalgene)</td>
<td>1 L</td>
</tr>
<tr>
<td>Sodium thiosulfate anhydrous (Fisher Scientific, 100%)</td>
<td>20 g</td>
</tr>
</tbody>
</table>

Sodium Thiosulfate Solution Preparation

1) Measure 1000 mL of ultrapure reagent water into a 1 L plastic container.
2) Weigh out 20 g Sodium thiosulfate (Na₂S₂O₃) solution on a Mettler balance.
3) Add 20 g Na₂S₂O₃ into the 1 L plastic container containing the ultrapure pure reagent water.
4) Shake the mixture and allow for the crystals to dissolve completely.
5) Store the solution at room temperature until ready for use.

Table B.5
Equipment and Materials for Bench-scale pre-ozonation

<table>
<thead>
<tr>
<th>Item</th>
<th>Size/Quantity</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ozone generator (Ozonia, CSF-1A)</td>
<td>1</td>
</tr>
<tr>
<td>Jacketed batch ozone reactor flask (Allen Glassware, Boulder, CO)</td>
<td>4 L</td>
</tr>
<tr>
<td>Recirculating chiller for batch ozone reactor</td>
<td>1</td>
</tr>
<tr>
<td>Recirculating chiller for ozone generator</td>
<td>1</td>
</tr>
<tr>
<td>Gas washing bottles</td>
<td>2 x 500 ml</td>
</tr>
<tr>
<td>Glass beakers</td>
<td>100 mL, 250 ml, 1 L</td>
</tr>
<tr>
<td>Eppendorf pipette and tips</td>
<td>1 mL</td>
</tr>
<tr>
<td>Graduated measuring cylinders</td>
<td>50ml, 250 mL</td>
</tr>
<tr>
<td>Spectrophotometer (Hach DR 4000)</td>
<td>1</td>
</tr>
<tr>
<td>Erlenmeyer flasks</td>
<td>10 x 125 mL</td>
</tr>
<tr>
<td>Oxygen gas tank</td>
<td></td>
</tr>
<tr>
<td>Indigo II solution</td>
<td></td>
</tr>
<tr>
<td>Potassium iodide (Acros Organics, 99+%)</td>
<td>100 g</td>
</tr>
<tr>
<td>Sodium thiosulfate anhydrous (Fisher Scientific, 100%)</td>
<td>250 g</td>
</tr>
<tr>
<td>Stir plate</td>
<td>1</td>
</tr>
</tbody>
</table>
Batch ozone test set-up

Glassware Preparation for Ozone Tests

Glassware in the ozone lab should only be rinsed or cleaned using reagent water and then air-dried. Avoid soap or methanol for cleaning glassware to be used for ozone testing.

Bench-Scale Pre-Ozonation Tests

A. Batch Ozone Reactor
   1) Fill the jacketed ozone reactor flask with 1 L of ultrapure Milli-Q water. If more than 1 L of ozone solution will be required for the tests, fill required volume of ultrapure Milli-Q water (typically 8 – 10 L for pre-ozonating 40 – 45, 1 GAL test water samples) in one or more glass bottles and chill in ice or refrigerate overnight prior to running pre-ozonation tests. Fill the jacketed flask with the chilled Milli-Q water up to the 4 L mark for the first 20 – 25 GAL samples.
   2) Verify the temperature setting on the recirculating chiller for the jacketed batch ozone reactor is set at 2°C.
   3) Turn on the chiller for the jacketed ozone flask.
   4) Check and ensure that water is circulating through the flask.
   5) Turn on stir plate for the jacketed reactor (300 rpm). Position the jacketed flask on the stir plate to ensure that the stirring rod inside the reactor is at the center of the flask and mixing is uniform. Adjust speed setting accordingly.
   6) Fill two gas washing bottles with 500 ml of KI solution (i.e., 50 g KI dissolved in 500 ml DI water). Each solution should last for 1 – 2 days of testing.
7) Connect the two gas washing bottles in series by connecting them with Teflon tube to the batch ozone flask.

B. Ozone Generator

8) Open the valve on the oxygen cylinder. Check that the pressure of the oxygen gas is ~2000 psi. Replace with new oxygen cylinder from warehouse if pressure drop indicates that gas cylinder is empty.

9) Oxygen regulator pressure gauge should be at 40 – 50 psi.

10) Turn on the chiller for the ozone generator. Verify that the external pump is on and no recirculation to the ozone generator is occurring. Make sure valves to the ozone generator are closed.

11) Record all values (oxygen gas pressure, oxygen regulator pressure, etc.) in the ozone generator log book.

12) Allow the chillers (ozone reactor chiller and ozone generator chiller) to run for 1 h. After 1 h, the temperature in the ozone generator chiller should be at 4°C. Place clip on tubing to stop re-circulation in the chiller and open valves to the ozone generator to allow re-circulation to the generator.

13) Turn the ozone generator on by switching MAINS ON. This turns the power on to the ozone generator and does not produce any oxygen or ozone gas.

14) After 30 min, turn on the oxygen gas flow to the batch ozone reactor by switching PURGE ON. Check and ensure that fume hood is ON.

15) Normal operating gas flow rate is 0.5 on the flow meter scale on the ozone generator from 0 – 10 (or 5% on the scale from 0 to 100%). Read the top of the float. Lower gas flow rates are better for increased transfer efficiency.

16) Record all values observed on the ozone log book [generator run hours, gas flow rate (typically 0.5 – 1), power setting (50 – 60 %), power reading in KW, generator start and stop times for purge and ozone, etc.].

17) Oxygen flow should start to flow immediately. Oxygen flow should be visible in the rotameter of the generator, the batch ozone reactor, and the 2 gas wash bottles (there should be bubbling inside the gas wash bottles). If oxygen flow is not visible in the gas wash bottles. Adjust tubing connecting the jacketed ozone reactor to the 2 gas wash bottles to ensure there are no leaks occurring.

18) Allow the ozone generator to purge for 1 h with oxygen gas.

19) After the purge has been completed, the generator can be turned on for ozone generation. Turn on the ozone generator, by switching PSU ON, and then turn off the oxygen flow by switching PURGE OFF. This should be done simultaneously.

20) After 20 – 60 min, the ozone stock solution in the jacketed ozone flask should have a purple/blue hue and the concentration should be stable.

21) At this point, the concentration of the ozone stock should be at ~60 - 80 mg/L.

22) While the ozone is being generated, observe the color of the two KI solutions in the gas washing bottles.

23) The solution in the first bottle will turn dark brown or amber, and the solution in the second bottle should remain yellow.

24) When KI solution has been exhausted, the first bottle will lose its color and become clear, while the second bottle will then turn brown or amber in color.

25) When testing is complete, the solution in the second bottle should be transferred in the first bottle. A freshly prepared solution should be placed in the second bottle.
C. Indigo Method: Measurement of Ozone Residual in Stock Solution

26) Switch on spectrophotometer DR 4000 and let it warm up for 1 h (this can be done while the ozone generator is purging).

27) Remove the cuvette from the spectrophotometer and allow the instrument to perform self-check.

28) After 1 h, place the cuvette inside the spectrophotometer aperture and set wavelength to 600nm, using option tab.

29) Select exit, and switch SIPPER ON by pressing the SIPPER ON ICON.

30) Log-on to the lab computer and open the Excel spreadsheet used to calculate ozone residual concentration levels in grab samples from the jacketed reactor.

31) Obtain 10; clean 125 mL-Erlenmeyer flasks with numbered labels from #1 to #10.

32) Weigh each Erlenmeyer flask on a Mettler balance. Record the weights of the empty Erlenmeyer flasks in the opened spreadsheet for Flasks #1 to #10.

33) Remove freshly prepared Indigo solution from refrigerator. Using the dispenser on the 1 L amber bottle, dispense 10 ml of indigo solution into each of the 10 Erlenmeyer flasks.

34) Using the 50 mL graduated cylinder, add 40 mL of ultrapure Milli-Q water to the indigo solution in Erlenmeyer flask #1, and add 39.5 ml of ultrapure Milli-Q water to the remaining 9 Erlenmeyer flasks (#2 to #10) with indigo solutions.

35) Erlenmeyer flask #1 will be designated as the control solution and flasks #2 – #10 will be used for measuring ozone residuals in the stock solution over the course of the tests.

36) Add 50 mL ultrapure Milli-Q water to a 100 mL beaker. Place the spectrophotometer sipper inside the ultrapure Mill-Q water in the 100 mL beaker. Press ZERO to zero the instrument.

37) Weigh Erlenmeyer Flask#1 (control) solution on the Metler balance. Record the weight of the flask on the ozone spreadsheet. Remove the sipper from the ultrapure water and place the sipper inside Erlenmeyer Flask #1. Press READ and record the absorbance of the control indigo solution in the spreadsheet.

38) Add a small amount (~5 – 10 mL) of prepared Na₂S₂O₃ into a 1 L beaker pre-labeled as “waste.” This will be used for quenching ozone in any “waste” ozonated stock solution before disposal.

39) Once the concentration in the jacketed ozone reactor appears to be stable (purple/blue hue). Open the reactor valve and dispense a small amount of ozone stock solution into a 250 mL beaker (pre-labeled “ozone stock”) to flush the line for dispensing ozonated stock solution. Discard this ozone solution into a 1 L “waste” glass beaker with Na₂S₂O₃.

40) Dispense another small amount of the ozonated stock solution. Using the 1 mL Eppendorf pipette, immediately pipette 0.5 mL of the ozonated stock solution into the Erlenmeyer flasks #2 with indigo solution. This will be an 80:1 dilution of the stock solution.

41) Swirl the mixture to mix. Depending on the stock ozone solution concentration levels, the dark blue Indigo solution should turn pale or light blue.

42) Weigh the Erlenmeyer flask #2 again, and record the weight of the flask #2 on the ozone residual spreadsheet.
43) Remove the sipper from the control (Erlenmeyer Flask#1) solution, and place the sipper inside the weighed Erlenmeyer #2 flasks containing the stock ozone and indigo solution.

44) Press READ to measure ozone concentration in the grab sample from the ozone reactor. Record the absorbance in the spreadsheet, and obtain the grab residual concentration in mg/L calculated in spreadsheet.

45) Multiply the grab residual concentration result by 80 to account for the dilution factor. This is the stock ozone concentration. If the calculated stock concentration is low (< 60 mg/L), allow the ozone to run for another 20 - 30 min. After 30 min, determine the concentration of the ozone stock solution using Erlenmeyer flasks #3, and repeating steps 40 – 44 in the protocol.

46) Dosing of the samples into the amber glass jugs containing the test water can begin once the stock solution is at least 60 mg/L.

47) Target dose for Utility 1 is 2 mg/L. Determine the volume of stock ozone solution from stock solution concentration, required to achieve 2 mg/L ozone dose in 3785 mL water sample.

48) Target dose for Utility 4a is 0.5 mg/L. Determine the volume of stock ozone solution from stock solution concentration, required to achieve 0.5 mg/L ozone dose in 3785 mL water sample.

49) Dispense the ozone stock solution in the 250 ml beaker and immediately transfer the required volume into a measuring cylinder. Discard any extra volume into the 1 L waste beaker with Na₂S₂O₃. Add the measured volume of ozonated stock solution into the 1 GAL amber jug with test water sample. Shake the jug to mix.

50) For every ozonated sample, add a label to the 1-GAL amber jugs to indicate sample has been ozonated include date and time.

51) Continue to measure out and add the required volumes of ozonated stock solution to the remaining 1 GAL jugs until the volume of ozonated stock solution in the reactor is <100 mL or below the diffuser. Check the ozone stock solution concentration levels after every 8 samples and adjust the measured volumes accordingly.

D. Refilling Ozone Reactor

52) When the volume of ozone stock solution remaining in the batch reactor decreases to <100 mL, turn the ozone generator off by switching PSU OFF and turning on the oxygen flow simultaneously by switching PURGE ON.

53) Allow 5 – 10 mins to purge the system of ozone gas.

54) Open the batch reactor and re-fill with chilled/refrigerated Milli-Q water to the 4 L mark while the oxygen flow is on. Cap reactor tightly, but do not crack the lid.

55) Turn on the ozone generator by switching PSU ON.

56) After 20 – 30 mins, the ozone solution should have a purple/blue hue and should be ready for use. If the concentration is too low allow the ozone to run for another 30 - 60 min until ozone stock concentration is at least 60mg/L. Repeat steps 40 – 44 in the protocol using the Indigo solutions in the remaining Erlenmeyer flasks.
57) Dosing of the remaining 20 – 25 water gallons can continue using steps 46 – 49 of the protocol.
58) Repeat steps 50 – 55, if the volume of ozone stock solution remaining in the batch reactor decreases to less than 100 mL.

E. Shutting Off Ozone Generator
59) After pre-ozonation of all the samples, turn off ozone generator by switching PSU OFF and turn on oxygen purge by switching PURGE ON simultaneously.
60) Empty the reactor of any of the remaining ozone stock solution while the vessel is still pressurized. Discard the remaining ozone stock solution into the waste beaker and quench with the Na₂S₂O₃.
61) After approximately 15 minutes, turn off the oxygen flow by switching PURGE OFF.
62) Turn OFF the power to the ozone generator by switching MAINS OFF.
63) Close the valve of the oxygen cylinder.
64) Turn off recirculating chillers for the ozone generator and ozone reactor.
### Table B.6

**Materials for coagulation/flocculation/sedimentation tests**

<table>
<thead>
<tr>
<th>Item</th>
<th>Size/Quantity</th>
</tr>
</thead>
<tbody>
<tr>
<td>2100N HACH Laboratory turbidimeter</td>
<td>1</td>
</tr>
<tr>
<td>Liquid Alum obtained from utility, 44% purity</td>
<td>1 x 500 mL</td>
</tr>
<tr>
<td>Cationic polymer obtained from utility, 20% purity</td>
<td>1 x 500 mL</td>
</tr>
<tr>
<td>Non-ionic polymer flocculant aid obtained from utility, 100% purity</td>
<td>1 x 500 mL</td>
</tr>
<tr>
<td>HACH Turbidity Gel standards</td>
<td>1 x 2000 NTU, 1 x 1000 NTU</td>
</tr>
<tr>
<td>Amber jugs with test water samples</td>
<td>45 x 1 GAL</td>
</tr>
<tr>
<td>Phipps and Bird PB-900 programmable jar testers with 6 paddles</td>
<td>2</td>
</tr>
<tr>
<td>Phipps and Bird BKER 2 square acrylic jars</td>
<td>24 x 2 L</td>
</tr>
<tr>
<td>Eppendorf pipettes with tips</td>
<td>10 - 100µL, 5 – 20 µL</td>
</tr>
<tr>
<td>Glass syringe</td>
<td>10 µL</td>
</tr>
<tr>
<td>Stainless steel drums</td>
<td>2 x 30 GAL</td>
</tr>
</tbody>
</table>

---

**Glassware Preparation for Coagulation/Flocculation/Sedimentation Tests**
Two liter acrylic jars were washed with soap and rinsed with tap water and additionally rinsed in ultrapure water. Steel paddles for the jar testers will be rinsed with ultrapure water.

**Jar Tests Protocol**

1) Turn on turbidimeters and allow instrument to warm up for 1 h.
2) After 1 h, calibrate the 2100 N turbidimeter using the different gel standards per the instructions outlined in the instrument manual.
3) Turn on the pH meter.
4) Calibrate pH meter with pH 4, 7, 10 buffer standards.
5) Begin with first test water (e.g., 20 – 25 GAL of non-blended raw water). Empty the water samples from the 1-GAL amber jugs into 12 clean jars (6 jars for each Jar tester). Fill the test water to the 2 L mark in the jars.
6) Insert the stainless-steel paddles into the test water.
7) For Utility 1, 38 mg/L of 44% liquid alum solution and 1.5 mg/L of 20% cationic polymer were each added to 2 L acrylic jars and rapidly mixed at 100 rpm for 2 min to promote coagulation. Next, 0.18 mg/L of 10% non-ionic polymer was added after 2 min to promote flocculation, and mixed at 30 rpm for 26 min.
8) For Utility 2, no coagulation, sedimentation, and flocculation was undertaken.
9) For Utility 3, 0.6 mg/L of 40% FeCl$_3$ was added to 2 L acrylic jars and samples were rapidly mixed at 100 rpm for 2 min to promote coagulation, and then mixed at 30 rpm for 26 min to promote flocculation.
10) For Utility 4, 1.5 mg/L of 40% FeCl$_3$ and 1.2 mg/L of 20% cationic polymer were added to the 2 L acrylic jars simultaneously. The samples were rapidly mixed at 100 rpm for 2 min to promote coagulation, and then mixed at 30 rpm for 30 min to promote flocculation.
11) Samples were allowed to settle for 15 minutes prior to moving on to the next step.
12) Replace 12 jars with water samples with another 12 set of BKER2 jars, and fill the remaining test water samples to the 2 L mark in each of the jars.
13) Repeat steps 5 – 13 until treatment of the first test water (20 – 25 GAL) is completed.
14) Open the valve on the BKER2 jars and slowly dispense the treated water into the clean 30-GAL stainless steel container used for the blending protocol.
15) Discard water with flocs below the valve into drain. Clean the 2 jars using cleaning protocol outlined for the coagulation, sedimentation, and flocculation tests.
16) Proceed to the next test water samples and repeat steps 1 – 15 for the second test water samples (e.g., 20 – 25 GAL of 10% Biofiltration blended water).
17) Store the two 30-GAL stainless steel containers with the test waters in refrigerator at 4°C until ready for filtration.
18) Clean the empty 1-GAL amber jugs containing test water using soap, rinse with tap water and ultrapure water.
Stored and refrigerated water from jar tests
### Table B.7
Materials for filtration tests

<table>
<thead>
<tr>
<th>Item</th>
<th>Size</th>
</tr>
</thead>
<tbody>
<tr>
<td>Whatman 90mm diameter glass microfiber filters</td>
<td>1.5 µm</td>
</tr>
<tr>
<td>Whatman 90mm diameter glass microfiber filters</td>
<td>0.7 µm</td>
</tr>
<tr>
<td>Millipore Glass Vacuum filters with 90mm base and cap</td>
<td>3 x 1 L</td>
</tr>
<tr>
<td>Graduated measuring cylinders</td>
<td>3 x 1 L</td>
</tr>
<tr>
<td>Vacuum pumps</td>
<td>3</td>
</tr>
<tr>
<td>Stainless steel drum with test water samples</td>
<td>2 x 30-GAL</td>
</tr>
<tr>
<td>Amber glass jugs</td>
<td>45 x 1-GAL</td>
</tr>
<tr>
<td>Glass beaker mugs</td>
<td>3 x 1 L</td>
</tr>
<tr>
<td>Clamps for 90mm vacuum filters</td>
<td>3</td>
</tr>
<tr>
<td>Stainless steel tweezers</td>
<td>3</td>
</tr>
<tr>
<td>Beakers</td>
<td>2 x 250 mL</td>
</tr>
<tr>
<td>Amber glass bottles</td>
<td>6 x 1 L</td>
</tr>
<tr>
<td>Funnel</td>
<td>2</td>
</tr>
<tr>
<td>Glass vacuum flasks</td>
<td>3 x 4 L</td>
</tr>
</tbody>
</table>

**Filtration Protocol**


1) Remove the 30-GAL drums with each test water sample from the refrigerator.
2) Begin with the first set of test water samples.
3) Set up three filter stations each with 1 vacuum pump, paper wipes, 1 Millipore Glass vacuum filter with 90mm base and cap, 1 pack of Whatman 1.5 µm (Utility 1 and 4a) or 0.7 (Utility 3), 90mm glass fiber filter, 1 clamp, 1 L glass beaker mug, 1 funnel, 1 L graduated measuring cylinder and 4 L Vacuum flask.
4) For each filter station, plug the vacuum pumps to a power source. Place the vacuum filter base over the 4 L flask.
5) Using a clean tweezer, carefully remove a microfiber filter from the filter pack and place over the vacuum filter base.
6) Place the 1 L cap over the vacuum filter, and firmly clamp the cap to the base of the vacuum filter.
7) Connect the tubing on vacuum pump to the vacuum filter.
8) Set the glass beakers on 2 or 3 thick sheets of paper wipes to avoid contamination from work surface during filtration.
9) For each filter station, fill the 1-L glass beaker with water from the 30-GAL stainless steel drum and empty into the 1 L vacuum filter cap.
10) Switch the vacuum pump ON. Allow the filtrate to fill the 4 L flask. Continue filtering until the 4 L flask is almost full.
11) Using the 1 L graduated cylinder, measure exactly 3.785 L of the filtrate and dispense into the corresponding 1-GAL amber glass jug using a funnel.
12) Remove the clamp and disconnect the vacuum filter set. Replace the microfiber filter and resume filtration for the next 1-GAL jug.
13) Continue filtration until all the 20 – 25, 1-GAL amber glass jugs are filled. Change filter paper as often as necessary.
14) Clean the vacuum filters with soap, rinse with tap water and ultrapure water. Allow the vacuum filters to air-dry.
15) Proceed to the next set of test water samples and repeat steps 4 – 13 until all the test water samples have been filtered.
16) Using a clean beaker, collect 100 ml filtered water sample for each test water to determine the pH and turbidity of the filtered samples.
17) Measure pH and turbidity and record in lab notebook.
18) Where required, collect 2 - 3 L of sample for each filtered test water in 1 L amber bottles for trihalomethane formation potential (THMFP) tests. Store in refrigerator. Proceed to chlorination.
### Chlorination and Chloramination: SOP

#### Table B.8

<table>
<thead>
<tr>
<th>Item</th>
<th>Size/Quantity</th>
</tr>
</thead>
<tbody>
<tr>
<td>Erlenmeyer flasks</td>
<td>4 x 50 mL</td>
</tr>
<tr>
<td>Pre-labeled beaker (for waste)</td>
<td>1 x 1 L</td>
</tr>
<tr>
<td>Eppendorf pipette and tips</td>
<td>1 x 5 mL</td>
</tr>
<tr>
<td>Volumetric burette</td>
<td>1 x 50 mL</td>
</tr>
<tr>
<td>5 – 6% sodium hypochlorite, NaOCl (Fisher Scientific, 5 – 6%)</td>
<td>1 x 500 mL</td>
</tr>
<tr>
<td>Potassium iodide (Acros Organics, 99%)</td>
<td></td>
</tr>
<tr>
<td>Sodium thiosulfate anhydrous (Fisher Scientific, 100%)</td>
<td></td>
</tr>
<tr>
<td>Potassium iodate (KIO₃) (Fisher Scientific, 0.1 N)</td>
<td></td>
</tr>
<tr>
<td>Hydrochloric acid (Fisher Scientific, concentrated)</td>
<td></td>
</tr>
<tr>
<td>0.5 M Sodium hydroxide (NaOH)</td>
<td></td>
</tr>
<tr>
<td>1 M Phosphate buffer</td>
<td></td>
</tr>
<tr>
<td>Amber glass jugs with test water</td>
<td>45 x 1-GAL</td>
</tr>
<tr>
<td>Amber glass bottles</td>
<td>60 mL</td>
</tr>
<tr>
<td>Orbital shaker</td>
<td>1</td>
</tr>
<tr>
<td>20-22% as ammonia, Aqua ammonia (NH₄OH) solution from utility</td>
<td>1 x 250 mL</td>
</tr>
<tr>
<td>Ammonium chloride (Fisher Scientific, &gt;99.5%)</td>
<td></td>
</tr>
<tr>
<td>Glass syringes</td>
<td>10 µL, 25 µL, 1mL</td>
</tr>
<tr>
<td>Graduate measuring cylinder</td>
<td>1 x 100 mL</td>
</tr>
<tr>
<td>HACH DR 890 colorimeter kit</td>
<td>1</td>
</tr>
<tr>
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**Chlorination Protocol**

Determine stock concentration of 5 – 6% NaOCl using iodometric titration method.

**Iodometric Titration**

1) Empty 100 – 150 mL of stock 5 – 6% NaOCl as received from manufacturer into 250 mL amber glass container. Label bottle with sample name, your name/initial, and date. Store in NaOCl stock solution in amber bottle in refrigerator when not in use.

2) Make-up 100 g/L KI, 0.2 M Na₂S₂O₃, 0.1 M KIO₃, 3 M HCl, 0.5 M NaOH, and 1 M Phosphate buffer solutions. Use hypo-titration Excel spreadsheet to ensure accurate calculations of mass and volumes for each reagent. It is important to be very accurate with the KIO₃ solution. This is used to standardize the Na₂S₂O₃ titrant, which is then used to standardize the NaOCl.

3) Fill burette with >40mL of Na₂S₂O₃ solution and waste until it reaches 40mL.

4) Record starting volume of Na₂S₂O₃ in burette.

5) Add 6mL of HCl and 6mL of KI to Erlenmeyer flask#1.

6) Swirl Erlenmeyer flask and slowly add the amount of KIO₃ dictated in the spreadsheet (0.4mL). KIO₃ is used to determine an accurate concentration for the Na₂S₂O₃.
7) Swirl mixture as you add Na₂S₂O₃ solution from the burette. Add Na₂S₂O₃ slowly, a drop could cause the color change.
8) Record amount needed to turn color from dark red/brown to clear. Enter this value on the spreadsheet.
9) Waste the Na₂S₂O₃ to the next gradation on the burette so that you have a new starting volume and record this number.
10) Empty flask#1 contents into the waste container.
11) Repeat process (steps 3-9) for a total of three test runs using Erlenmeyer flasks#2, and #3. If coefficient of variation, CV% <1, continue with standardization of NaOCl.
12) Wash the 3 Erlenmeyer flasks (#1 to #3) with soap and rinse with tap water and ultrapure Milli-Q water. Allow to air dry.
13) Record starting volume of Na₂S₂O₃ in burette.
14) Add 6mL of HCl and 6mL of KI to Erlenmeyer flask#4.
15) Swirl Erlenmeyer flask #4 and slowly add the amount of NaOCl dictated in the spreadsheet (0.4mL).
16) Swirl mixture as you add Na₂S₂O₃ solution from the burette. Add Na₂S₂O₃ slowly, a drop could cause the color change.
17) Record amount needed to turn color from dark red/brown to clear. Enter this value on the spreadsheet.
18) Repeat process for a total of three test runs using cleaned Erlenmeyer flasks #1 and #2. If CV% <1, record concentration (mg/L as free Cl₂) and today’s date on the bottle for the NaOCl stock solution.
19) Proceed to chlorine and ammonia dosing of test water samples.

**Chlorine and Ammonia Dosing**

1) To determine the initial dose/concentration, run preliminary chlorine or chloramines decay tests for each post-filtration test water using 5 initial doses.
2) For Utility 1, the target is 1.8 – 2.2 mg/L chlorine after 11 minutes with subsequent quenching with aqua ammonia.
3) For Utility 2, a target of 3.8 mg/L total chlorine was targeted by addition of chlorine and then quenching with aqua ammonia.
4) For Utility 3, a free chlorine residual of 1.5 mg/L after 2 hours was targeted.
5) For Utility 4, 2.6 mg/L of pre-formed monochloramines were targeted after 16.5 minutes.
6) Using the hypo-titration spreadsheet, determine the NH₃ concentration required to obtain free Cl₂:NH₃ mass ratio of 4.5:1 for Utility 1 and 2. For example, for 1.8 mg/L free chlorine residual after 11 min, ammonia dose will be 0.4 mg/L.
7) To make-up a stock solution of pre-formed monochloramines, ammonium chloride was dissolved in a pH 8 solution and NaOCl was slowly added at a 3.5:1 Cl₂:NH₃ mass ratio. This stock solution concentration was targeted to be 1.4 g/L of as total chlorine.
8) Dose 60 mL amber bottles with 5 initial doses and measure free or total Cl₂ residual with HACH colorimeter in each solution to verify starting doses.
9) Place 60 mL amber bottles with 5 doses on an orbital shaker. Shake for appropriate amount of time.
10) Retrieve sample bottles from shaker and measure the free or total Cl₂ residual with HACH colorimeter.

11) Measure free chlorine and total chlorine residual in solution. If the combined chlorine (total chlorine – free chlorine = combined chlorine) is within the targeted range, proceed to chlorinating the 1-GAL test water samples.

12) Determine the required volume of NaOCl or monochloramines to spike into 3785 mL (1 GAL) to target required dose that resulted in the appropriate residual.

13) Determine the volume of stock NH₄OH solution to spike into 1-GAL amber jug to obtain for Utilities 1 and 2.

14) Obtain 1-GAL amber jugs with test water samples and spike in required volume of NaOCl or preformed monochloramines in 1-GAL jug.

15) Place the 1-GAL jug in the orbital shaker and shake for appropriate amount of time. Next, remove the sample from shaker and add the required volume of NH₄OH to 1-GAL test water solution. Swirl and mix the solution.

16) Measure and record free and total chlorine solution.

17) Add label “chlorine added” to the amber jug, write date and combined chlorine residual concentration on the label.

18) Store chlorinated samples in refrigerator at 4°C until ready for pasteurization, or needed for pipe rig conditioning.
PASTEURIZATION: SOP

Pasteurization tests were performed for 10 – 12 GAL of each test waters to be used for conditioning the pipe rigs in Weeks 3 and 4 within the first month of conditioning, or in Weeks 7 and 8 during the second month of conditioning. For each test water, select 10 to 12 GAL out of the 20 to 25 GAL chlorinated water samples stored in the refrigerator and set aside for pasteurization tests.

### Table B.9
<table>
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<tr>
<td>Water baths</td>
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<td>Amber glass jugs with test water</td>
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<tr>
<td>Glass thermometer</td>
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**Pasteurization Protocol**

1) Begin with first test water samples and load 10 to 12, 1-GAL amber jug with the test water samples on a cart. Allow the 10 – 12 GAL of the test water to sit on cart at room temperature. This will raise from the temperature of the water samples from 4 °C to ~20 °C, and reduce heating time for the pasteurization tests.
2) Wash the 10 L stainless steel container and 50 mL graduated cylinder with soap and rinse with tap water. Rinse again with ultrapure Milli-Q water.
3) Fill the stainless steel container with ultrapure Milli-Q water. Dispense the ultrapure water into each water bath and fill the water baths until almost full.
4) Fill 50 mL measuring cylinder with Milli-Q ultrapure water. Rinse the thermometer with ultrapure water. Place the clean thermometer in the graduated cylinder with ultrapure water. This will ensure the thermometer is clean every time it is dipped into the test water samples.
5) Plug water baths to power source and switch ON. Adjust temperature setting on each water bath to 70 °C.
6) Allow water in water baths to heat up to 70 °C. This should take about 1 h. Check the temperature of the water in each bath every 20 min using a thermometer and record the temperature in lab note book.
7) Once the water in the bath reaches 70 °C. Place the 1-GAL amber jugs in each water bath.
8) Allow the water in each amber jug to heat up to 70 °C. This should take about 3 h. Using the thermometer, check the temperature of the water every 30 min and record the temperature and time at which the temperature was measured. This will help keep track of the water temperature in the jugs within the 3 h period.
9) Once the water in the jugs reaches 70 °C. Allow the water to heat for 30 min at 70 °C. Set timer for 30 min from the time the water temperature in the jug reaches 70 °C.

10) After 30 min, remove the samples from the bath. At this point, the amber jugs are hot to touch. Allow the jugs too cool at room temperature. Add label “pasteurized” to the amber jug, write date of pasteurization on the label.

11) Fill 3 large coolers with raw ice and place coolers inside the refrigerator at 4 °C. Once the amber jugs are cool to the touch, place the jugs inside the coolers with raw ice. Ensure that the jugs are covered with ice inside the coolers. Note the time the samples are placed inside coolers.

12) Allow samples to sit in ice for 30 min. Remove samples from coolers or ice bath after 30 min and store in refrigerator at 4 °C until ready for chloramination.

13) Proceed to the second test water sample, e.g., 10 % Biofiltration blend water once pasteurization of the 10 – 12 GAL for first test water is completed.

14) Load 10 to 12, 1-GAL amber jug with the second test water samples on a cart. Allow the 10 – 12 amber jugs to sit on a cart at room temperature.

15) Repeat steps 7 – 12 until all 10 – 12 GAL of water samples are pasteurized.

**ORGANIC CARBON: SOP**

Samples (250 mL) were shaken vigorously before opening, then one 30 mL aliquot was transferred to an amber glass vial for TOC analysis and two mL were transferred to a 250 mL amber glass bottle. The remaining volume was passed through a 0.22 µm pore size mixed cellulose esters membrane filter (Millipore, Billerica, MA). Filters were rinsed with 600 mL of Nanopure (Thermo Scientific, Waltham, MA) purified water prior to sample filtration to avoid carbon leaching. A 30 mL aliquot was transferred into an amber glass vial for DOC analysis. The remaining filtered volume was added to the amber glass bottle, which was covered with foil and stored in a dark area for 45 days. After 45 days, each sample was mixed and a 30 mL sample was taken for determination of BDOC, as previously described (Servais et al. 1987). Samples were analyzed on a Sievers 5310C portable TOC analyzer (GE, Boulder, CO) according to Standard Method 5310C (APHA 2005). Briefly, immediately after collection, samples were preserved with 100 µl of phosphoric acid and then sparged with ultrapure nitrogen gas for three minutes at ten psi prior to analysis. Each sample was analyzed three times and the average of the second two measurements was reported to avoid potential carryover from the previous sample. BDOC was determined from the difference between the DOC measurement on day 0 and day 45. All glassware was baked before use at 550°C for five hours. Statistical differences among conditions were tested using a Tukey HSD test using JMP.

**OPPORTUNISTIC PATHOGEN (OP) GENE MARKERS AND ANTIBIOTIC RESISTANT GENES (ARG): SOP**

One liter samples were collected for molecular analyses. Samples were concentrated onto 0.22 µm pore size mixed cellulose esters membrane filter (Millipore, Billerica, MA). Filters were torn into ~1 cm² fragments using sterile forceps, transferred to extraction tubes, and frozen at -20°C until processing. DNA was extracted using a FastDNA SPIN kit (MP Biomedicals, Solon, OH) according to manufacturer instructions. OP gene markers and ARGs were quantified by quantitative polymerase chain reaction using previously published primers and thermocycler conditions. The following genes were targeted for quantification of OPPPs: a highly specific
region of the 23S rRNA gene for *Legionella* spp. (Nazarian et al. 2008), a highly specific region of the 16S rRNA gene for *Mycobacterium* spp. (Radomski et al. 2010), and the *ecfX* and *gyrB* genes for *Pseudomonas aeruginosa* (Anuj et al. 2009). Two ARGs were also quantified: a quinolone resistant gene, *qnrA* (Colomer-Lluch et al. 2014), and a vancomycin resistant gene, *vanA* (Dutka-Malen et al. 1995), along with the class 1 integron integrase gene *intI1* (Hardwick et al. 2008). The universal bacterial gene, 16S rRNA, was also quantified (Suzuki et al. 2000). All non-probe assays (16S rRNA genes, *vanA*) were performed in triplicate 10 µl reactions that included 5 µl SsoFast EvaGreen SuperMix (Bio-Rad, Hercules, CA), 0.8 µl of forward and reverse primers at 5 µM, 2.4 µl molecular grade water, and 1 µl sample. All probe assays (*Legionella* spp., *Mycobacterium* spp., *P. aeruginosa*, *qnrA*) were performed in triplicate 10 µl reactions that included 5 µl SsoFast Probes SuperMix (Bio-Rad), 0.5 µl of each forward and reverse primer at 5 µM, 0.19 µl of each probe at 10 µM, 1 µl sample, and molecular grade water to reach the total reaction volume. Prior to all analyses, 16S rRNA genes were quantified in a representative subset of samples diluted ten-, 20-, 50-, and 100-fold to identify the minimum concentration at which inhibition was negligible. A ten-fold dilution was selected and applied to all samples. All qPCR runs included a triplicate negative control and triplicate standard curve consisting of ten-fold serial dilutions ranging from $10^7$-$10^1$ gene copies/µl for all genes except 16S rRNA, for which $10^8$ to $10^2$ gene copies/µl were used. The limit of quantification for all genes was 10 gene copies per reaction. Statistical differences between samples were tested by Kruskal Wallis rank sum test with a posthoc pairwise Wilcoxon test using JMP.

**ANTIBIOTIC RESISTANT HETEROTROPHIC PLATE COUNT (HPC): SOP**

The fraction of the total HPC capable of growth in the presence of antibiotics was determined by plating samples on R2A media with and without one of nine antibiotics added. Media was supplemented separately with ampicillin (4 µg/mL), ciprofloxacin (0.5 µg/mL), chloramphenicol (4 µg/mL), gentamicin (2 µg/mL), oxacillin (1 µg/mL), rifampin (0.5 µg/mL), sulfamethoxazole (128 µg/mL), tetracycline (2 µg/mL), and vancomycin (0.5 µg/mL). Culturing was performed according to standard method 9215C (APHA 2005). Briefly, four ten-fold serial dilutions of water sample were prepared and 0.1 mL of each was spread onto prepared R2A agar. Plates were incubated for seven days at 37ºC, then colonies were counted. Only plates that produced between 20 and 200 colonies were considered quantifiable.
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Texas Commission on Environmental Quality. 2016. Meeting treatment technique requirements for Cryptosporidium, Giardia, and viruses. PTR Section Staff Guidance.


### ABBREVIATIONS

<table>
<thead>
<tr>
<th>Abbreviation</th>
<th>Definition</th>
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<tr>
<td>µg/L</td>
<td>micro grams per liter</td>
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<tr>
<td>µm</td>
<td>micro meter</td>
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<tr>
<td>ACU</td>
<td>apparent color units</td>
</tr>
<tr>
<td>AI</td>
<td>Aggressiveness Index</td>
</tr>
<tr>
<td>AOB</td>
<td>ammonia oxidizing bacteria</td>
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<td>AOC</td>
<td>assimilable organic carbon</td>
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<td>APB</td>
<td>acid producing bacteria</td>
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<tr>
<td>ARG</td>
<td>antibiotic resistant genes</td>
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<td>ASR</td>
<td>aquifer storage and recovery</td>
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<td>ATP</td>
<td>adenosine triphosphate</td>
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<td>AWP</td>
<td>advanced water purification</td>
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<td>BAM</td>
<td>bulk-advective mixing</td>
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<td>BART</td>
<td>Biological Activity Reaction Tests</td>
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<td>BSF</td>
<td>bromide substitution factor</td>
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<td>°C</td>
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<tr>
<td>Ca(OH)\textsubscript{2}</td>
<td>calcium hydroxide</td>
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<td>CaO</td>
<td>calcium oxide</td>
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<tr>
<td>CCPP</td>
<td>Calcium Carbonate Precipitation Potential</td>
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<td>CDPH</td>
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<td>CECs</td>
<td>chemicals of emerging concern</td>
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<td>cm</td>
<td>centimeter</td>
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<tr>
<td>CO\textsubscript{2}</td>
<td>carbon dioxide</td>
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<td>CSMR</td>
<td>Chloride to Sulfate Mass Ratio</td>
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<td>Clean Water Act</td>
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<td>DBP</td>
<td>disinfection byproduct</td>
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<td>Division of Drinking Water</td>
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<tr>
<td>DEET</td>
<td>N,N-diethyl-meta-toluamide</td>
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<tr>
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<td>DNA</td>
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EPS extracellular polymeric substances
ESB engineered storage buffer
FP formation potential
FRT failure and response time
GAC granular activated carbon
GW groundwater
H₂S hydrogen sulfide
H₃PO₄ phosphoric acid
HAAs haloacetic acids
HAB heterotrophic aerobic bacteria
HPCs heterotrophic plate counts
HSD honest significant difference
IPR indirect potable reuse
LSI Langelier Saturation Index
LT2ESWTR Long Term 2 Enhanced Surface Water Treatment Rule
MCL maximum contaminant limit
MF microfiltration
µg/L microgram per liter
mg/L milligrams per liter
µm micrometer
MGLG maximum contaminant limit goal
MIC microbiologically-induced corrosion
mL milli-liter
MRL method reporting limits
Na₂CO₃ sodium carbonate
NaCl sodium chloride
NaHCO₃ sodium bicarbonate
NaOH sodium hydroxide
N-DBPs nitrogenous DBPs
NDMA N-nitrosodimethylamine
NF nanofiltration
ng/L nanogram per liter
nm nanometer
NMOR N-nitrosomorpholine
NOB nitrite oxidizing bacteria
NOM natural organic matter
NSF National Science Foundation
NTU Nephelometric Turbidity Unit
NWRI National Water Research Institute
<table>
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<tr>
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<td>ozone and biological aerated filters</td>
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<td>OP</td>
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<td>ORP</td>
<td>oxidation-reduction potential</td>
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