Final Report:
Graham Hill Water Treatment Plant
Source Water Quality Monitoring Study

Prepared for:
City of Santa Cruz

Prepared by:

February 25, 2019

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Executive Summary

Section 1: Introduction

The City of Santa Cruz ("City") began an intensive two-year water quality sampling campaign in October 2016 to characterize its surface water and, to a lesser extent, groundwater sources. The City draws on several sources to supply the Graham Hill Water Treatment Plant (GHWTP): the San Lorenzo River (SLR), Loch Lomond reservoir, three North Coast streams—Laguna Creek, Majors Creek, Liddell Spring—and three groundwater wells. Water treatment takes place at the GHWTP using coagulation, flocculation, sedimentation, filtration, and disinfection. The first goal of the water quality sampling was to develop a comprehensive characterization of the surface water sources to inform future decision-making, including potential GHWTP treatment improvements or the design of a new treatment plant. An additional goal of the water quality monitoring campaign was to better understand the treatability of high-turbidity winter flows from the San Lorenzo River, which are not currently treated at the plant.

Section 2: Source Water Quality Assessment

The water quality monitoring campaign that took place between October 2016 and September 2018 involved data collection for numerous parameters. In addition, historical data going back to 2011 was also included in this assessment to better characterize water quality trends. Many key parameters show increases during winter storms, including total organic carbon (TOC), turbidity, and color. The magnitude of the increase varies between source waters, with the San Lorenzo River typically being the most impacted and the North Coast streams and Loch Lomond reservoir showing smaller variation.

This section also includes a discussion of the microbial contaminants in the source waters. Data on the concentrations of two indicator organisms (total coliform and *E. coli*) and two pathogens (*Cryptosporidium* and *Giardia*) were collected. A review of recent studies was conducted to better understand the origin of the microbial contamination (i.e., human or animal).

Section 3: Treatability Assessment

Source water treatability was analyzed from the standpoint of coagulation performance and solids formation. Treating high-turbidity winter flows from the San Lorenzo River may result in exceedances of the solids disposal limit. Depending on the desired increase in flow to be treated from the San Lorenzo River in the winter, an evaluation of additional solids handling alternatives is recommended to manage the higher solids production from this winter source water blend. Raw blend TOC data were also analyzed for compliance with the enhanced coagulation TOC removal requirements, and over the period analyzed (July 2014 – August 2018), the GHWTP consistently met the TOC removal requirements.

Section 4: Disinfection Byproduct Formation Analysis

Trihalomethanes (THM) formation mitigation strategies need to be considered regardless of whether or not additional winter San Lorenzo River flows are treated at the plant. The modeling presented in this section demonstrates that there may be multiple strategies to meet the THM and haloacetic acids (HAA) limits in the distribution system. Strategies that reduce TOC prior to chlorination are likely to be effective—these strategies include enhanced coagulation, ACTIFLO Carb, and potentially other technologies that can reduce TOC levels below the 2.0-2.5 mg/L threshold. Additional testing of alternative approaches is recommended to evaluate the effectiveness of other control strategies, such as TOC transformation by ozone or ultraviolet (UV) disinfection.

Section 5: Additional Regulatory Considerations

Based on the sampling data, there were no constituents that exceeded either the Maximum Contaminant Levels (MCLs) or secondary MCLs. GHWTP is currently in Bin 2 for *Cryptosporidium* treatment and has an additional 1-log reduction requirement for both virus and *Giardia*. Based on the water quality dataset, it is unlikely that GHWTP would be assigned additional treatment requirements in the near future. Studies to identify the sources of microbiological contamination could be considered.

Section 6: Conclusions and Recommendations

The GHWTP surface source waters exhibit varying degrees of seasonality, with the largest impacts to water quality coming during winter storms. We recommend continued routine water quality monitoring for all surface source waters, and event-based sampling for the San Lorenzo River. Additional routine and event-based sampling would continue to provide useful insight into source water and treated water quality. Event-based sampling for microbial constituents coupled with a targeted analysis with microbial source tracking could help elucidate the sources of increased microbial contamination during winter storms.

Variations in constituents such as alkalinity, pH, and hardness have implications for treatment, but the observed water quality ranges should not result in a raw blend that cannot be treated to meet all regulatory standards. The plant could consider decoupling pH adjustment and coagulant dosing to better optimize control of both parameters. If there are scenarios where coagulant is being over-dosed to achieve pH reduction, there is potential for savings in terms of solids production.

Disinfection byproduct formation, particularly of THMs, is an issue for GHWTP. Under the current operational strategy of chlorinating upstream of the sedimentation basin (in Flash Mixer 2), it may be difficult to consistently meet the plant’s goal of 80% of the THM MCL at terminal locations in the distribution system. Based on the analysis provided in Section 4, the most effective way to control THM formation in the plant’s current configuration will be to limit the raw blend TOC concentration, which is not a feasible option. Alternative solutions include moving the primary chlorination location after the sedimentation basins and using enhanced coagulation to reduce the settled water dissolved organic carbon (DOC), or using an alternative disinfectant strategy involving UV or ozone.

1 Introduction

1.1 Background

The City of Santa Cruz ("City") began an intensive two-year water quality sampling campaign in October 2016 to characterize its surface water and, to a lesser extent, groundwater sources. The City draws on several sources to supply the Graham Hill Water Treatment Plant (GHWTP): the San Lorenzo River (SLR), Loch Lomond reservoir, three North Coast streams—Laguna Creek, Majors Creek, Liddell Spring—and the three groundwater wells at Tait Street. Water treatment takes place at the GHWTP using prechlorination, coagulation, flocculation, sedimentation, filtration, and disinfection. The first goal of the water quality sampling was to develop a comprehensive characterization of the surface water sources and raw water blends to inform future decision-making, including potential GHWTP treatment improvements or the design of a new treatment plant. An additional goal of the water quality monitoring campaign was to better understand the treatability of winter stormwater flows from the San Lorenzo River, which are not currently treated at the plant.

The impetus for this winter water evaluation sprang from the recommendations of the City's Water Supply Advisory Committee (WSAC). From 2014 to 2015, the WSAC undertook an effort to develop strategies for improving both the quantity and reliability of the Santa Cruz water supply. The outcome of the WSAC's effort was the Water Supply Augmentation Strategy (WSAS), which is currently being implemented by the City's Water Department (WSAC 2015). The WSAS identified multiple paths forward, including (1) conservation, (2) aquifer storage and recovery (ASR), (3) in-lieu water transfers to neighboring agencies, and (4) potable reuse or desalination. Two of the proposed strategies, ASR and in-lieu transfers, involve increased treatment of winter water flows from the SLR. In the ASR strategy, additional potable supply (i.e., in excess of the daily demands) would be produced in the winter months and stored in the aquifer for future use. Similarly, the in-lieu option would utilize the additional potable supply during the winter to provide drinking water to neighboring agencies.

Currently, SLR flows are not consistently used throughout much of the winter because storm events lead to elevated turbidity, color, bacteria, and total organic carbon (TOC) levels that, in turn, lead to treatment challenges. The implementation of the ASR and in-lieu transfer strategies, however, hinges on this water being used as the source for the increased potable supply. As a result, extensive sampling was conducted on winter SLR flows to better understand the implications of treating these flows at the GHWTP.

The GHWTP, which was commissioned in 1960, is an aging treatment plant facing several challenges. Over the years, various upgrades have been completed to ensure the plant can continue to meet regulatory requirements. However, without improvements to remove high TOC and safeguard against issues related to high levels of bacteria and parasites, the plant still faces challenges meeting its long-term water quality goals. Two of the biggest treatment challenges are (1) meeting disinfection byproduct (DBP) and (2) solids production limits. Both of these issues are likely to be exacerbated by treating winter water SLR flows, which is why these flows have historically not been treated at the GHWTP. Therefore, the information from this source water characterization study was used to provide a preliminary evaluation on strategies to mitigate these treatment issues, both under normal operating conditions and while treating winter water SLR flows.

1.2 Graham Hill Water Treatment Plant Source Waters

Graham Hill Water Treatment Plant receives a raw blend consisting of varying proportions of multiple surface water and groundwater sources. The plant is fed by the Loch Lomond Reservoir, the San Lorenzo River, three streams located to the northwest of Santa Cruz (the North Coast streams), and three groundwater wells at Tait Street (Tait Wells). Figure 1 shows the breakdown of source water contributions during the water quality monitoring period from October 2016 to September 2017, and from October 2017 to September 2018. A map summarizing the locations of the source waters in relation to the GHWTP is provided in Figure 2.

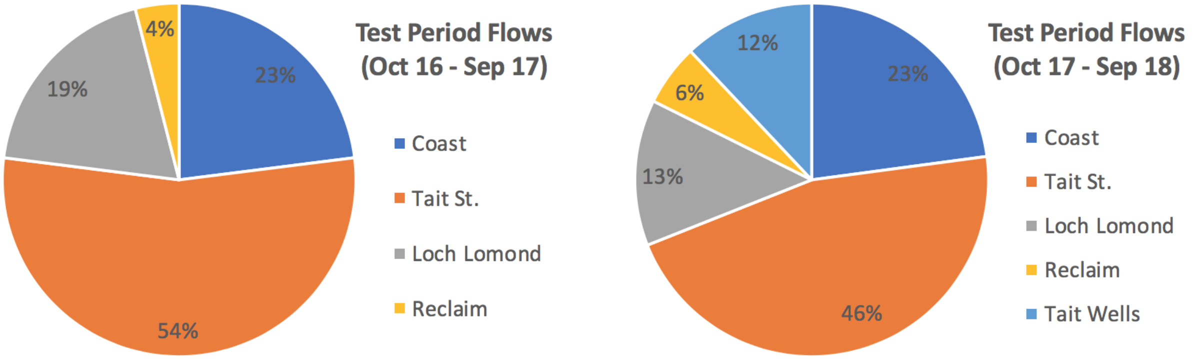

Figure 1. Contributions of various source waters to the raw blend at the Graham Hill Water Treatment Plant during the water quality monitoring period (October 2016 – September 2017 and October 2017 – September 2018).

San Lorenzo River

San Lorenzo River flows are diverted in two locations: Tait Street Diversion and Felton Diversion. The Tait Street Diversion (located in City of Santa Cruz south of the GHWTP—see Figure 2) pumps water from both the river and three groundwater wells located next to the river. These flows are combined in an intake sump; they then join the Coast Pipeline flows (discussed below) and are conveyed to the GHWTP.

San Lorenzo River flow is also diverted about five miles upstream of the Tait Street Diversion in Felton at the Felton Diversion. These flows can be pumped to the Loch Lomond Reservoir for additional reservoir storage and ultimately back to the GHWTP by way of the Newell Creek pipeline.

Loch Lomond Reservoir

Loch Lomond Reservoir is located on Newell Creek, about ten miles north of the City of Santa Cruz. The reservoir's maximum storage capacity is about 8,900 acre-feet. Water is conveyed from Loch Lomond to GHWTP through the Newell Creek Pipeline. Loch Lomond receives primarily local watershed runoff but can also receive a small amount of water diverted from the San Lorenzo River at the Felton Diversion during wet years as allowed under this diversion permit.

North Coast Streams

The three North Coast sources—Liddell Spring, Laguna Creek, and Majors Creek—are transported through the Coast Pipeline to the Coast Pump Station (see Figure 2), where they are then conveyed to the GHWTP. Flows from Liddell Spring and Laguna Creek meet at the Laguna-Liddell 'Y'; flows from Majors Creek enter below this junction. As a result, Majors Creek flows are limited by the flows already in the pipeline from the other two sources. These three source waters exhibit differences in source water quality and are discussed individually throughout the document.

Tait Wells

The Tait Wells are three groundwater wells located near the San Lorenzo River at the Tait Street Diversion. The water drawn from these wells is classified as Groundwater Under Direct Influence of Surface Water (GWUDI), as they are hydraulically connected to the SLR. Water produced by the Tait Wells is delivered to the SLR intake sump at the Coast Pump Station and then pumped to the common transmission pipeline that also conveys the SLR and North Coast water to the GHWTP.

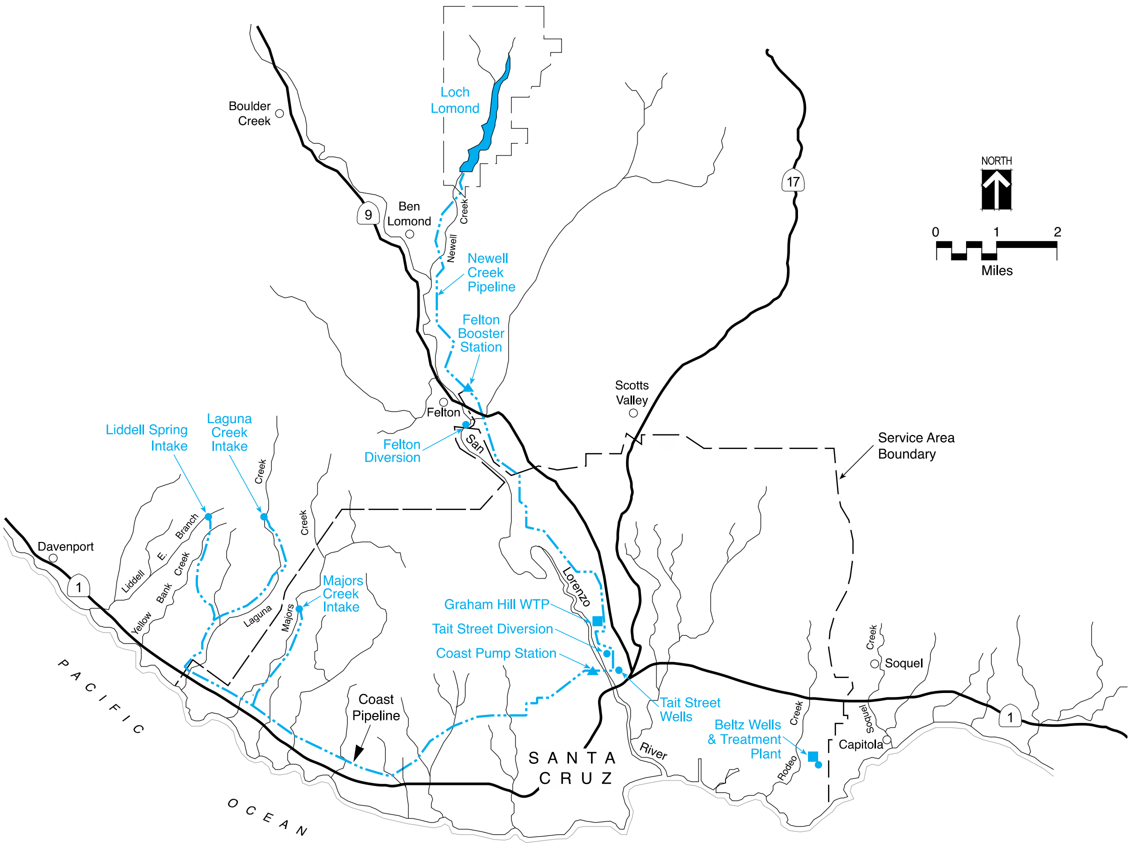

Figure 2. Map of source waters to the Graham Hill Water Treatment Plant. From CDM (2007).

1.3 Sampling Plan Goals and Approach

The City retained Trussell Technologies, Inc., to review historical water quality data and prepare a source water quality sampling plan that would address critical knowledge gaps and inform future decision-making. The goals of the plan were as follows:

- Develop a comprehensive characterization of the water quality of each surface water source, including capturing seasonal trends for key parameters related to treatability and regulatory compliance.
- Assess the potential to treat additional winter San Lorenzo River flows using the winter water quality data.
- Characterize the implications of seasonal source water quality for treatment performance at the GHWTP.

Sampling frequency for key constituents ranged from weekly to quarterly, depending on the source water and constituent. A summary of the constituents measured, and the frequency of measurement, is given in Table 1, along with the relevance of each constituent as it relates to treatability at GHWTP and/or regulatory compliance. Given the periodic frequency of sampling, it was possible that this program alone would not sufficiently capture storm-impacted water quality. Consequently, the City also carried out event-based sampling of the San Lorenzo River to better characterize the impact of winter storms on water quality. Historically the City has not monitored the water quality of the SLR during periods of high turbidity because of their existing treatment limitations. The event-based sampling captured the full range of water quality that occurred in the river during rain events. Constituents measured during event-based sampling are marked with an 'E' in Table 1.

Some modifications were made to the sampling campaign between water year 2017 (WY17) and water year 2018 (WY18).

- Most of the inorganic parameters, e.g. anions panel, metals, had a reduced sampling frequency in Loch Lomond and the North Coast streams in WY18 because it was decided that the existing data set provided a sufficient characterization of seasonality on a monthly basis.
- Sampling for organics in the North Coast streams was increased from monthly to biweekly because of the importance of this parameter for treatment and regulatory compliance, and because of the observed variability in TOC and dissolved organic carbon (DOC) as a result of winter storms.
- Total coliform and *E. coli* were not measured during event-based sampling in WY17 but were added in WY18 to provide more granularity on the microbial water quality of the winter SLR stormwater. Pathogen sampling, which had been carried out in WY17 as part of the updated sanitary survey, was not continued in WY18.
- Sampling for the taste and odor-causing MIB and geosmin was stopped for the SLR and reduced in frequency for Loch Lomond in WY18 because they were not detected in any of the source waters in WY17.

Table 1: Summary of routine monitoring frequency for two-year water quality monitoring campaign. Where two frequencies are shown, the first was for Water Year 17 and the second for Water Year 18.

Constituent	**San Lorenzo River (@ Tait St.)**	**San Lorenzo River (@ Felton)**	**Loch Lomond Reservoir**	**North Coast Streams**	**Raw Blend**	**Relevance**
Alkalinity	Weekly, E	Biweekly	Biweekly	Biweekly	Weekly	Coagulation performance Enhanced coagulation requirements
Conductivity	Weekly, E	Biweekly	Biweekly	Biweekly	Weekly	Indication of water quality changes Coagulation performance
Hardness	Weekly, E	Biweekly	Biweekly	Biweekly	Weekly	Coagulation performance
Total coliform[a]	Weekly, E	Biweekly	Biweekly	Biweekly	Weekly	*Giardia* removal requirements
E. coli[a]	Weekly, E	Biweekly	Biweekly	Biweekly	Weekly	MCL under Revised Total Coliform Rule
Color	Weekly, E	Biweekly	Biweekly	Biweekly	Weekly	Aesthetic acceptability
Dissolved oxygen	Weekly	Biweekly	Biweekly	--	--	Indication of reservoir stratification and/or turnover Can indicate potential for air binding in filters
Odor	Weekly, E	Biweekly	Biweekly	Biweekly	Weekly	Aesthetic acceptability
Total organic carbon	Weekly	Biweekly	Biweekly	Monthly / Biweekly	Weekly	Enhanced coagulation requirements Disinfection byproduct formation

Dissolved organic carbon	Weekly, E	Biweekly	Biweekly	Monthly / Biweekly	Weekly	Disinfection byproduct formation
pH	Weekly, E	Biweekly	Biweekly	Biweekly	Weekly	Coagulation performance Effluent pH limits and finished water stability
Temperature	Weekly, E	Biweekly	Biweekly	Biweekly	Weekly	Coagulation performance DBP formation
Total suspended solids[b]	Weekly, E / None	Biweekly / None	Biweekly / None	Monthly / None	--	Solids generation
Turbidity	Weekly, E	Biweekly	Biweekly	Biweekly	Weekly	Solids generation Coagulant dose and need for polymer use
UV254	Weekly	Biweekly	Biweekly	Monthly / Biweekly	Weekly	Enhanced coagulation requirements
SUVA	Weekly	Biweekly	Biweekly	Monthly / Biweekly	Weekly	Enhanced coagulation requirements
Anions panel[c]	Monthly, E	Monthly	Monthly / Quarterly	Monthly / Quarterly	Monthly / None	General water quality characterization Coagulation performance
Cations panel[d]	Monthly	Monthly	Monthly	Monthly	--	
Total dissolved solids	Monthly, E	Monthly	Monthly / Quarterly	Monthly / Quarterly	--	
Ammonia	Monthly	Monthly	Monthly / Quarterly	Monthly / Quarterly	--	
Iron (total)	Monthly	Monthly	Monthly / Quarterly	Monthly / Quarterly	--	
Iron (dissolved)[b]	Monthly	Monthly	Monthly / Quarterly	Monthly / Quarterly	--	
Manganese (total)	Monthly, E	Monthly	Monthly / Quarterly	Monthly / Quarterly	--	
Manganese (dissolved)[b]	Monthly, E	Monthly	Monthly / Quarterly	Monthly / Quarterly	--	
Bromide	Monthly, E / Quarterly	Monthly / Quarterly	Monthly / Quarterly	Quarterly	Monthly / Quarterly	DBP formation Potential future ozone design
Cryptosporidium[b]	Monthly / None	Monthly / None	Monthly / None	Quarterly / None	Monthly / None	Pathogen removal requirements
Giardia lamblia[b]	Monthly / None	Monthly / None	Monthly / None	Quarterly / None	Monthly / None	
MBAS	Monthly / Quarterly	Monthly / Quarterly	Quarterly	-- / Yearly	--	Coagulation performance
MIB[b]	Monthly / None	Monthly / None	Monthly / Quarterly	--	Monthly / None	Taste and odor issues
Geosmin[b]	Monthly / None	Monthly / None	Monthly / Quarterly	--	Monthly / None	Taste and odor issues

[a] Event based sampling only in October 2017 – September 2018
[b] No sampling in October 2017 – September 2018
[c] Chloride, fluoride, nitrate, nitrite, sulfate
[d] Calcium, magnesium, potassium, sodium

1.4 Additional Data Sources

In addition to the routine and event-based sampling data collected during the test period, several other data sources were included in the treatability assessment. The additional data sources were:

- Historical source water quality data contained in the Laboratory Information Management System (LIMS): five years of historical data (2011 – 2015) were included in the treatability analysis.
- Supervisory Control and Data Acquisition (SCADA) data: several on-line parameters from the SCADA system were included in the analysis, including source water flows, some raw blend water quality characteristics, finished water trihalomethanes (THMs), and others.
- In-stream turbidimeter: the City maintains an in-stream turbidimeter in the San Lorenzo River at the Tait Street Diversion. Data from this meter were used to supplement the routine grab samples collected from the river.
- Stream gauge: the United States Geological Survey (USGS) maintains a stream gauge on the San Lorenzo River at Big Trees. The flow data collected from this gauge were used to determine when storms occurred and how the magnitude of river flow related to changes in certain key water quality parameters.

1.5 Summary

The goal of this report is to summarize the results of the two-year water quality monitoring campaign undertaken to characterize the water sources treated at Graham Hill Water Treatment Plant. Section 2 of this report presents the results of the source water quality monitoring, as well as 5 years of historical data. Trends in key water quality parameters are discussed, along with their implications for treatment at the plant. Section 3 provides more depth on treatability issues, particularly coagulation performance and solids production. Section 4 provides in-depth analysis of disinfection byproduct formation, including descriptions of jar testing and modeling done to evaluate current THM formation at the plant, and the impact of treatment modifications on DBP formation. Section 5 addresses other potential regulatory issues, such as compliance with maximum contaminant levels (MCLs) and pathogen reduction requirements. Conclusions, recommendations, and next steps are summarized in Section 6.

2 Source Water Quality Assessment

Trussell Tech was provided with historical data for all of the parameters presented in Table 2 from 09/07/11 until the beginning of sampling for this project (10/11/16). Sampling specific to this study took place from 10/11/16 through 09/30/18. Key parameters are discussed below, with special attention paid to trends and their implications. It should be noted that Water Year 2017 was an extreme winter in terms of rainfall quantity and storm frequency, while the previous four years (Water Years 2012-2016) experienced drought conditions. In addition, in early January 2017, there was a large landslide of clay soil that washed into the north end of Loch Lomond Reservoir. The suspended clay particles increased the lake turbidity and color above typical levels. The concentrations of these constituents slowly decreased throughout the remainder of the year. Thus, keep in mind that some of the Water Year 2017 data for the reservoir is atypical. The data in Table 2 therefore show ranges of water quality from both dry and wet years, as well as an atypical reservoir year. Subsequent figures show how the key parameters change over time and as a function of storm flows.

In addition to examining the water quality of the individual source waters, the trends in the raw blend entering the GHWTP were also examined. The composition of the raw blend changes throughout the year depending on source water quality and availability, and it can even change from day to day. Trends in raw blend quality are discussed where relevant in relation to treatability and source water composition.

2.1 Water Quality Summary

Table 2: Summary of water quality measured in source waters. Values presented are average (minimum – maximum). Data presented are from the Laboratory Information Management System maintained by the City of Santa Cruz and include all available data collected between September 2011 and September 2018.

Constituent	Units	MCL	Raw Blend	San Lorenzo River (@ Tait St.)	San Lorenzo River (@ Felton)	Newell Creek Reservoir	Liddell Spring	Laguna Creek	Majors Creek	Tait Wells
Alkalinity	mg/L	--	126 (74 – 146)	108 (40 – 146)	114 (40 – 134)	109 (70 – 154)	206 (102 – 234)	130 (44 – 164)	104 (32 – 134)	118 (100 – 134)
Conductivity	µmhos/cm	--	395 (270 – 550)	343 (35 – 490)	368 (145 – 500)	371 (290 – 480)	499 (390 – 785)	277 (130 – 365)	298 (120 – 405)	446 (395 – 490)
Hardness	mg/L	--	164 (100 – 270)	139 (60 – 188)	148 (64 – 188)	155 (110 – 186)	267 (223 – 400)	136 (56 – 174)	132 (44 – 178)	159 (142 – 180)

Constituent	Units	MCL	Raw Blend	San Lorenzo River (@ Tait St.)	San Lorenzo River (@ Felton)	Newell Creek Reservoir	Liddell Spring	Laguna Creek	Majors Creek	Tait Wells
Total Coliform	MPN/100 mL	--	1.3 x10^{3} (14.5 – 2.4x10^{4})	4.2 x10^{3} (276 – 8.7x10^{4})	8.8 x10^{3} (345 – 3.1x10^{5})	4.8 x10^{2} (2 – 8.4x10^{3})	2.0 x10^{1} (<1 – 770)	4.5 x10^{2} (53 – 9.9x10^{3})	1.3 x10^{3} (122 – 3.4x10^{4})	<1
E. coli	MPN/100 mL	--	62 (<1 – 1.7x10^{3})	276 (5.2 – 7.1x10^{3})	698 (31 – 2.5x10^{4})	3.3 (<1 – 26)	2 (<1 – 3)	25 (<1 – 687)	51 (<1 – 1.4x10^{3})	<1
Color	Color units	11[a]	17 (2 – 210)	53 (8 – 1900)	42 (8 – 800)	25 (8 – 120)	3 (1 – 28)	6 (2 – 60)	14 (3 – 200)	3 (2 – 6)
Dissolved Oxygen	mg/L	--	--	10.1 (8.3 – 13.1)	9.9 (8.3 – 12.7)	7.9 (3.6 – 11.8)	--	--	--	--
Odor	TON	3[a]	6 (2 – 12)	7 (2 – 70)	8 (2 – 50)	6 (1 – 17)	2 (1 – 5)	4 (1 – 10)	6 (1 – 12)	1 (1 – 2)
Total Organic Carbon	mg/L	--	2.2 (1.0 – 6.6)	3.0 (1.3 – 17)	3.1 (1.4 – 16.6)	4.4 (3.1 – 5.8)	0.2 (0.1 – 0.4)	0.7 (0.3 – 4.8)	1.7 (0.6 – 13)	1.3 (1.1 – 1.8)
Dissolved Organic Carbon	mg/L	--	2.0 (1.0 – 5.8)	3.4 (0.1 – 12)	2.9 (0.1 – 9.2)	4.6 (4.1 – 5.9)	0.3 (<0.3 – 0.5)	0.9 (0.5 – 4.6)	1.9 0.7 – 8.8)	--
Orthophosphate	mg/L	--	0.3 (0.06 – 0.4)	0.3 (0.07 – 0.8)	0.5 (0.1 – 1.1)	0.1 (0.08 – 0.2)	0.09	0.09 (0.04 – 0.14)	0.19 (0.07 – 0.32)	--
pH	pH units	--	7.6 (7.0 – 8.1)	7.9 (7.4 – 8.2)	7.8 (7.4 – 8.2)	7.3 (6.9 – 8.4)	7.3 (6.8 – 7.8)	8.1 (7.5 – 8.3)	7.9 (7.2 – 8.2)	6.9 (6.8 – 7.2)
Temperature	°C	--	14.5 (6.7 – 20.6)	13.4 (3.9 – 20.4)	12.7 (3.2 – 19.1)	12.9 (9.6 – 19.5)	14.3 (7.3 – 15.7)	11.4 (6.3 – 15.9)	11.7 (5.9 – 16.8)	15.4 (10.6 – 18.7)
Total Suspended Solids	mg/L	--	--	187 (0.2 – 2870)	4.4 (0.3 – 29.3)	12 (<0.1 – 49.3)	2.1 (<0.1 – 16.8)	1.8 (<0.1 – 9.8)	11.1 (0.5 – 68.1)	--
Turbidity	NTU	--	2.1 (0.05 – 40.9)	23.8 (0.47 – 1590)	7.7 (0.53 – 352)	5.6 (0.3 – 72.4)	0.4 (0.07 – 17)	0.6 (0.09 – 28.1)	2.9 (0.18 – 110)	0.5 (0.06 – 2.2)
UV254	cm^{-1}	--	0.06 (0.02 – 0.20)	0.12 (0.05 – 3.55)	0.15 (0.05 – 3.89)	0.14 (0.06 – 0.21)	0.001	0.03	0.07 (0.02 – 0.32)	--

Constituent	Units	MCL	Raw Blend	San Lorenzo River (@ Tait St.)	San Lorenzo River (@ Felton)	Newell Creek Reservoir	Liddell Spring	Laguna Creek	Majors Creek	Tait Wells
							(<0.001 – 0.006)	(0.01 – 0.17)		
SUVA	L/mg-m	--	3.0 (2 – 3.7)	3.2 (2.8 – 4.1)	3.3 (2.1 –4)	3.1 (1.4 – 3.7)	0.3 (<0.003 – 2)	2.9 (2.2 – 4.1)	3.2 (2.7 – 4.4)	--
Ammonia	mg/L	--	<0.05	0.05 (<0.05 – 0.05)	0.13 (<0.05 – 0.21)	<0.05	<0.05	0.06 (<0.05 – 0.08)	<0.05	--
Calcium	mg/L	--	--	34.6 (13 – 48)	43.8 (30 – 49)	43.4 (30 – 57)	94.9 (78 – 130)	42.3 (15 – 57)	40.1 (12 – 60)	--
Chloride	mg/L	250[a]	19.7 (13 – 26)	21 (5.6 – 37.2)	25.3 (9.9 – 36.2)	15.0 (7.3 – 19)	10.7 (8.5 – 12)	10.4 (7.1 – 12.2)	16.2 (9.2 – 17.8)	26.7 (21 – 34)
Fluoride	mg/L	2	0.16 (0.11 – 0.23)	0.17 (0.12 – 0.20)	0.18 (0.14 – 0.22)	0.26 (0.21 – 0.31)	0.09 (0.07 – 0.14)	0.08 (0.05 – 0.11)	0.08 (0.06 – 0.13)	0.22 (0.18 – 0.25)
Total Iron	µg/L	300[a]	89	244 (63 – 1700)	292 (110 – 1600)	365 (<20 – 3500)	304 (<20 – 1200)	109 (<20 – 860)	612 (21 – 4400)	465 (46 – 1000)
Dissolved Iron	µg/L	--	--	67 (39 – 120)	62 (37 – 100)	90 (<20 – 120)	<20	43 (<20 – 56)	78 (25 – 180)	--
Magnesium	mg/L	--	--	8.9 (7 – 11)	9.0 (7.2 – 10)	10 (7.4 – 14)	10.7 (8.5 – 16)	5 (2.8 – 6.6)	3.6 (2.8 – 4.7)	--
Total Manganese	µg/L	50[a]	<2	43.6 (9 – 290)	32.5 (14 – 80)	48.8 (<2 – 533)	17.5 (<2 – 56)	15.5 (<2 – 140)	11.2 (3.1 – 39)	131.6 (32 – 200)
Dissolved Manganese	µg/L	--	--	6.0 (<2 – 21)	12.4 (2.1 – 20)	4.2 (<2 – 7.3)	<2	<2	6.0 (<2 – 9.7)	--
Nitrate	mg/L as NO_3	45	1.5 (0.7 – 2.4)	1.4 (0.2 – 2.3)	2.1 (0.8– 3.2)	1.1 (<0.2 – 1.8)	1.3 (1.1 – 1.9)	0.4 (<0.2 – 0.8)	1.3 (<0.2 – 2)	<0.2
Nitrite	mg/L as N	1	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	--
Potassium	mg/L	--	--	2.2 (1.8 – 3)	2.0 (1.8 – 2.6)	2.3 (1.7 – 2.8)	2.0 (1.7 – 2.5)	2.0 (1.7 – 2.5)	1.8 (1.4 – 2.7)	--

Constituent	Units	MCL	Raw Blend	San Lorenzo River (@ Tait St.)	San Lorenzo River (@ Felton)	Newell Creek Reservoir	Liddell Spring	Laguna Creek	Majors Creek	Tait Wells
Sodium	mg/L	--	--	23 (13 – 33)	23.7 (13 – 30)	19.9 (13 – 31)	11.9 (10 – 14)	10.2 (7 – 13)	15.0 (8.6 – 20)	--
Sulfate	mg/L	--	56.4 (38 – 87.9)	48.2 (18 – 81)	52.6 (32.8 – 84.6)	70.9 (52 – 83)	71.6 (43.1 – 210)	14.0 (5.7 – 17.6)	34.9 (10 – 54)	59.3 (52 – 66)
Total Dissolved Solids	mg/L	--	290	220 (21 – 320)	258 (200 – 310)	250 (180 – 310)	364 (293 – 540)	183 (80 – 250)	199 (90 – 276)	--
Bromide	µg/L	--	61.4 (45 – 72)	46.5 (19 – 110)	67.3 (34 – 120)	45 (22 – 66)	40.2 (35 – 47)	27 (24 – 36)	49.6 (33 – 61)	--
Cryptosporidium	oocysts/L	--	--	0.18 (<0.10 – 0.50)	0.20 (<0.10 – 0.50)	<0.10	<0.10	<0.10	<0.10	--
Giardia	cysts/L	--	--	0.20 (<0.10 – 0.40)	0.17 (<0.10 – 0.30)	<0.10	<0.10	0.13 (<0.10 – 0.20)	0.17 (<0.10 – 0.30)	--
MBAS	mg/L	0.5[a]	--	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	--
MIB	ng/L	--	<5	<5	<5	<5	--	--	--	--
Geosmin	ng/L	--	<3	<3	<3 (<3 – 3.3)	<3	--	--	--	--

[a] Secondary MCL

2.2 Analysis of Key Parameters

Stream Flow Reference

To aid in the visualization of trends in water quality for key parameters, many of the figures shown in this section have been overlaid on a plot of San Lorenzo River stream flow data (see Figure 3). These data are valuable as a reference point because the stream flow provides a quick reference for the frequency, magnitude, and duration of storm events. The stream flow data clearly shows the drought conditions in Water Years 2012 – 2016, the more extreme wet conditions in Water Year 2017, and the dry conditions in Water Year 2018. A close look at the river flow during the water quality sampling campaign (Water Years 2017 and 2018) is given in Figure 4. Implications of the rainfall differences for key water quality parameters between the two years will be discussed in the following sections.

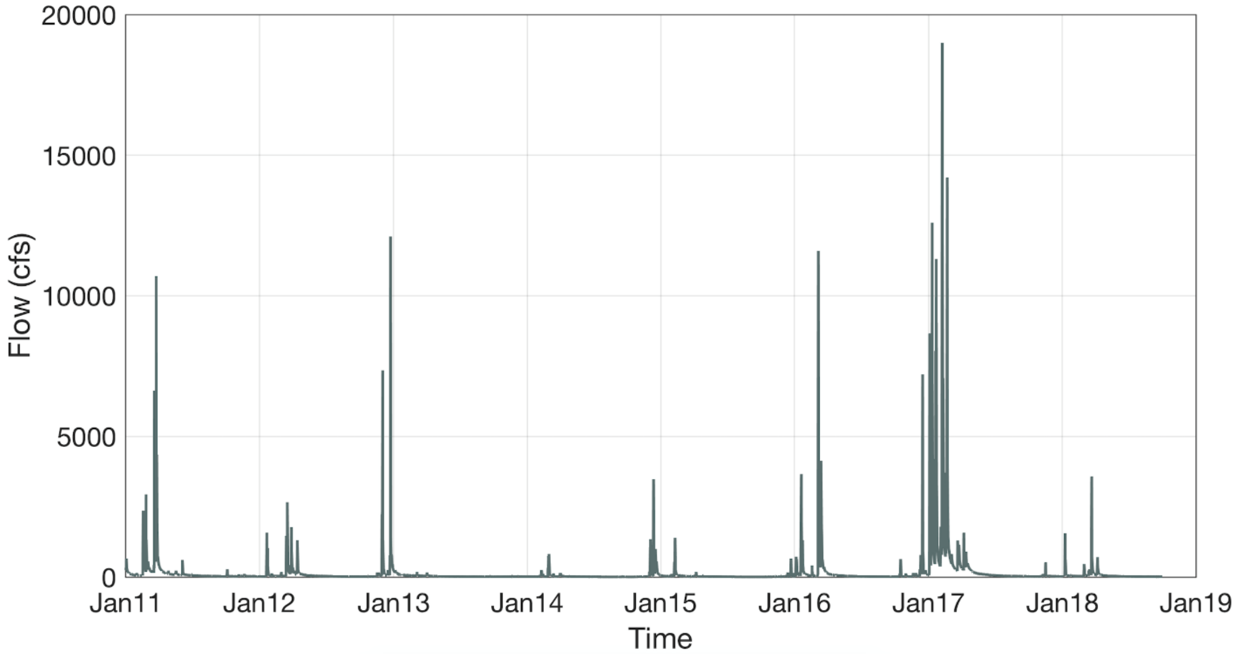

Figure 3. Historical flows in the San Lorenzo River. Data obtained from the USGS stream gage at Big Trees (USGS gage number 11160500) between January 2011 and September 2018.

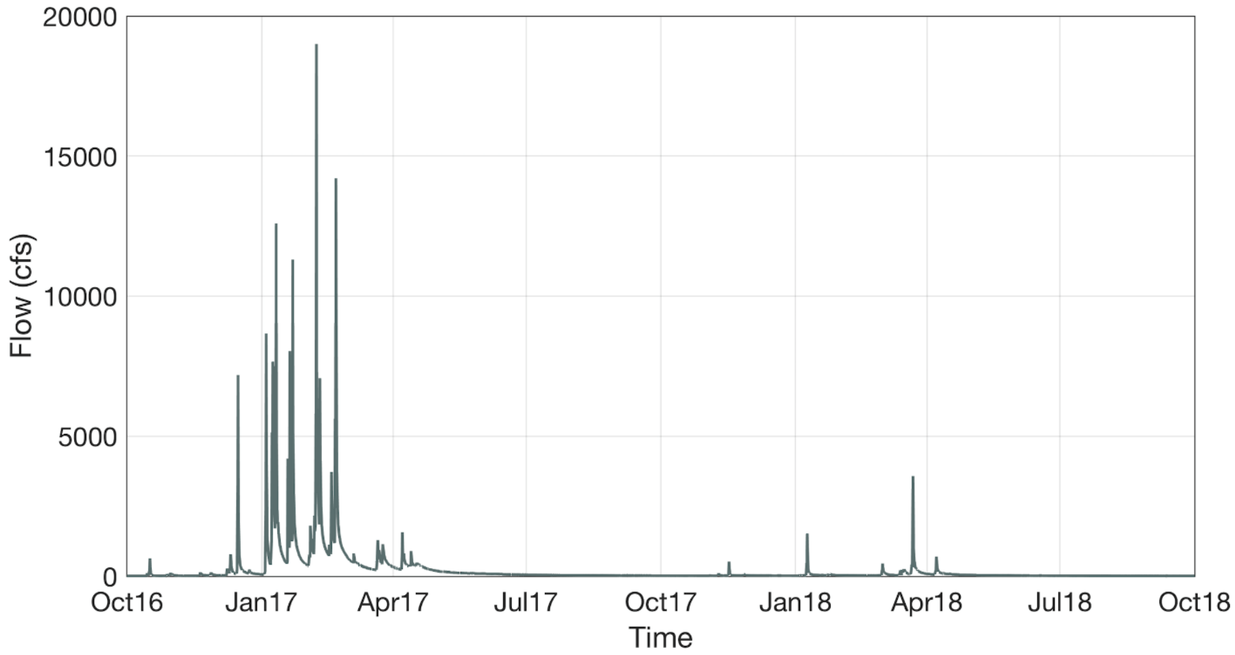

Figure 4. Recent historical flows in the San Lorenzo River. Data obtained from the USGS stream gage at Big Trees (USGS gage number 11160500) between October 2016 and October 2018.

Alkalinity

Alkalinity is a measure of the ability of a water to resist changes in pH (Crittenden et al. 2012). Alum, the coagulant used at the GHWTP, consumes alkalinity when added to water (0.5 mg/L of alkalinity as $CaCO_3$ consumed per 1 mg/L of alum), and it is important for coagulation performance to have sufficient alkalinity to prevent the pH from dropping below acceptable ranges. Excessively low pH values can lead to the formation of soluble aluminum species that can pass through the water treatment plant and possibly lead to a regulatory compliance issue (the primary MCL for aluminum is 1 mg/L). As aluminum is amphoteric, the same is true at elevated pH's above 7.5 or so. The ideal pH range varies by season: it is generally preferable to keep the pH during coagulation around 6 in the summer and 7 in the winter[1]. To maintain this pH range and ensure adequate coagulation, the operators at GHWTP aim to keep the alkalinity of the influent water at or above 80 mg/L as $CaCO_3$.

Alkalinity exhibits a seasonal trend for all sources, as well as the Raw Blend—variations over time are shown in Figure 5. In general, the alkalinity of the source waters and Raw Blend decreased in the winter due to storms. The alkalinity of most sources (including the Raw Blend) varied between 100 and 150 mg/L as $CaCO_3$, though lower alkalinities did occur during storm events, with values dropping into the 50 to 100 mg/L range (see Figure 5). This phenomenon was the result of low-alkalinity rain water diluting the source waters. Liddell Spring exhibits much smaller changes during the winter than the other sources, which is expected given that it is a spring rather than a surface water creek and is therefore less influenced by rainfall. For the

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[1] Aluminum solubility is impacted by temperature; at lower temperatures, the optimal pH for coagulation is higher.

same reason, the average alkalinity of Liddell Spring is much higher than that of the other sources, with values around 200 mg/L as $CaCO_3$.

The alkalinity of Laguna Creek is slightly higher than that of the other sources at times (except Liddell Spring, see Figure 6).[2] The alkalinity of Laguna Creek surpasses the Raw Blend and the remaining sources at the 30th percentile value, meaning that 70% of the alkalinity measurements at Laguna Creek were higher than that of the other sources (except Liddell Spring).

The alkalinity of the various source waters decreased below 80 mg/L as $CaCO_3$ only during winter storms. Between 2012 and 2015, storms were smaller and less frequent, resulting in only a few instances where alkalinity dropped below 80 mg/L in the San Lorenzo River and Majors Creek. In the winter of 2015/2016, storms were larger and more frequent than those in years past, resulting in a higher number of measurements below 80 mg/L as $CaCO_3$, and more measurements dipping farther below 80 mg/L than in previous years. Alkalinities in the San Lorenzo River, Majors Creek, and Laguna Creek dropped below 80 mg/L as $CaCO_3$ between November 2015 and March 2016 with a minimum of 40 mg/L measured in the San Lorenzo River at Tait St. on 1/19/16 (during a storm). The Raw Blend was not impacted to a large extent during this storm, as it was buffered by the high alkalinity of Liddell Spring; however, the Raw Blend alkalinity did drop more than usual during storms in the winter of 2016/2017.

The storms in the winter of 2016/2017 were much larger than those of the previous five years, as is evidenced by the higher flows in the San Lorenzo River over longer durations (Figure 3). The alkalinity of the San Lorenzo River, Majors Creek, and Laguna Creek all dropped below 80 mg/L as $CaCO_3$ at multiple points between October 2016 and April 2017. In the Loch Lomond Reservoir, the alkalinity dropped below 80 mg/L in October 2016 and did not recover to typical levels until July 2017. The minimum alkalinity during this period (and during the entire period analyzed) was 32 mg/L and occurred in Majors Creek on 1/10/17 and 3/21/17. As mentioned above, variations in alkalinity in the Raw Blend were generally buffered by the flows of higher alkalinity source waters. The minimum alkalinity of the Raw Blend was 74 mg/L as $CaCO_3$, which occurred on 2/08/17.

The storms in the winter of 2017/2018 were smaller and less frequent than those in the 2016/2017 winter. As a result, there were fewer and less dramatic drops in alkalinity during the 2017/2018 winter compared to the 2016/2017 winter. The minimum alkalinity during this period was 40 mg/L, which was measured in SLR at Felton on 3/22/18.

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[2] Throughout this report, data for certain key parameters are presented both over time, and in probability plot format. The probability plots characterize the variability of water quality parameters for each source water and may be useful for future treatment design.

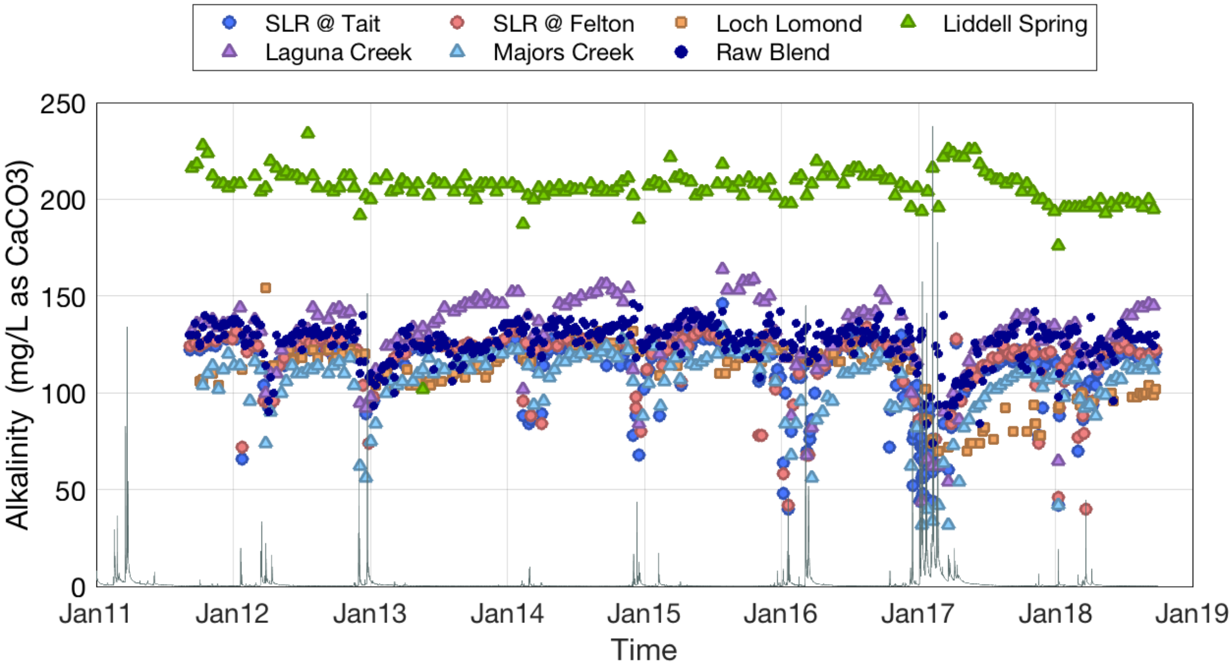

Figure 5: Alkalinity of source waters between September 2011 and September 2018.

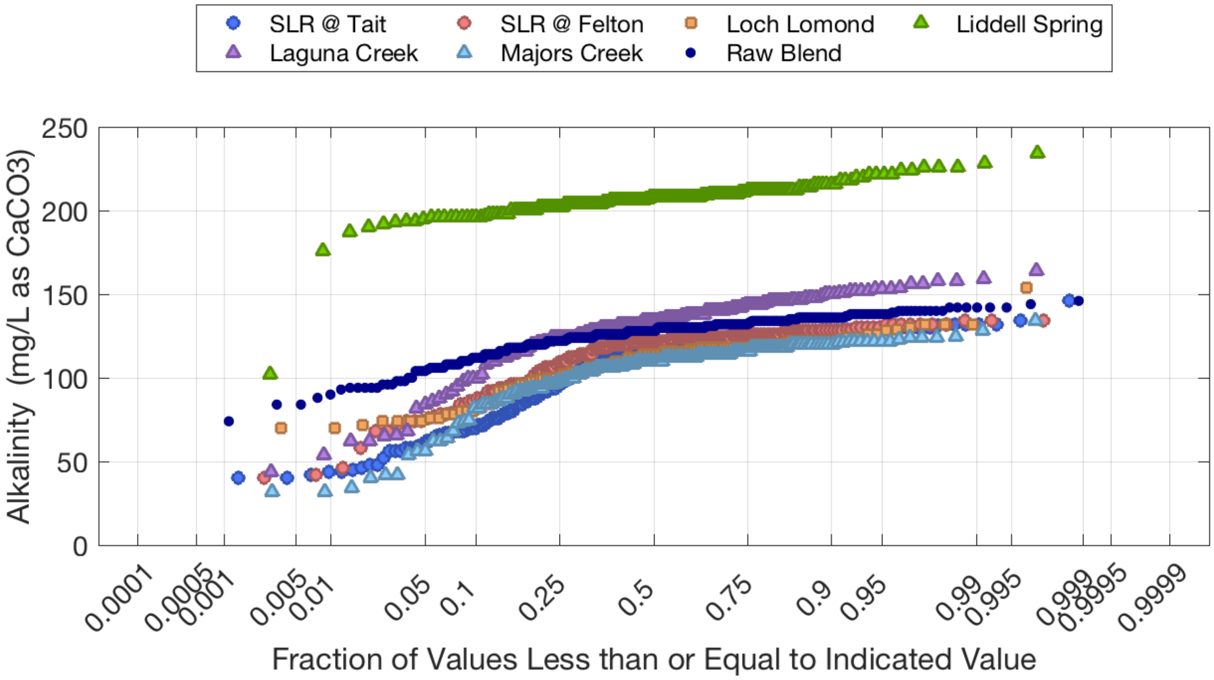

Figure 6: Probability plot of alkalinity of source waters for data collected between September 2011 and September 2018.

Total Organic Carbon

TOC is a measurement of the dissolved and particulate carbon in a water. TOC is an important water quality parameter because it has implications for a number of issues, including coagulation treatment requirements and DBP formation. In the source waters studied, essentially all (97-100%) of the TOC is present in dissolved form (DOC), so DOC and TOC can be used interchangeably.

TOC concentrations in most of the source waters displayed seasonal trends, with TOC increasing in the winter (see Figure 7). While many other constituents experience decreases in concentration in the winter because of dilution by rainwater, TOC increases because the storms wash organic material into the river. The San Lorenzo River showed the greatest variation in TOC concentration during storms, with values as high as 17 mg/L. The TOC concentrations of the North Coast sources generally remained below 2 mg/L, even during storm events. Loch Lomond generally had the highest TOC, ranging between 2.5 and 6 mg/L, and did not exhibit a high degree of variability as a result of storm events. The TOC of the Raw Blend ranged from 0.96 to 6.64 mg/L, with 50% of values less than 2 mg/L (see Figure 8).

The lowest concentration of TOC was observed in Liddell Spring, which sometimes had values dropping below the detection limit. Over the course of this study, Liddell Spring always had TOC values below 0.5 mg/L.

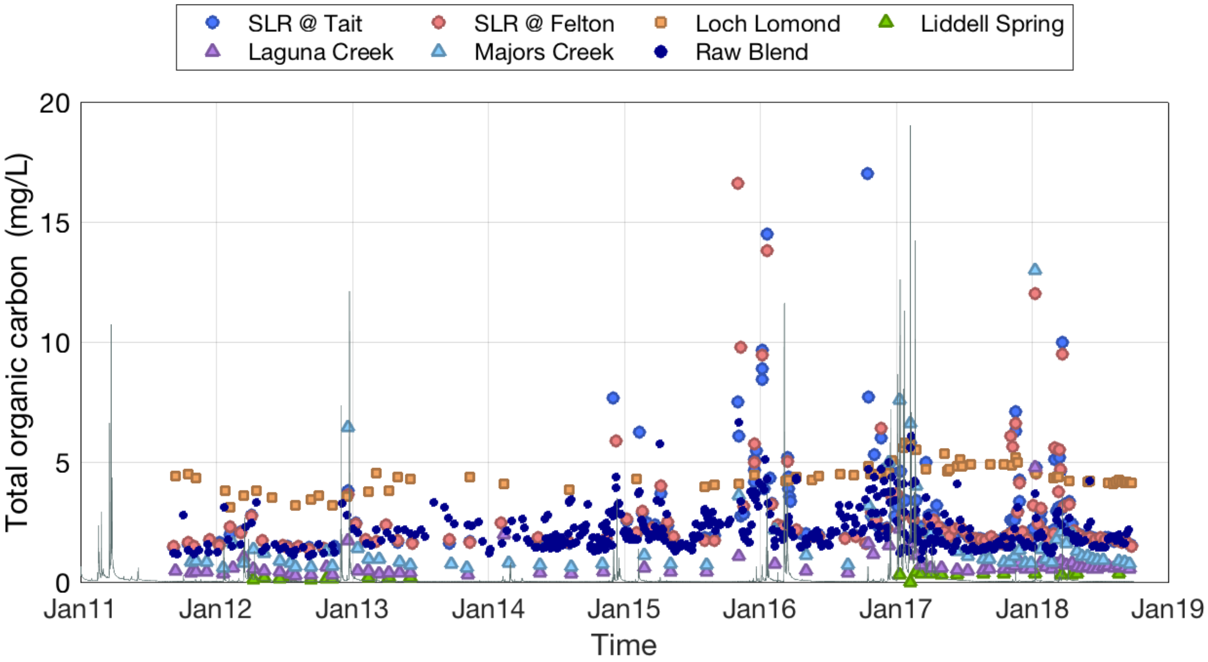

Figure 7: TOC of source waters between September 2011 and September 2018. Event-based sampling data for San Lorenzo River are not shown because the event-based samples collected dissolved organic carbon rather than TOC.

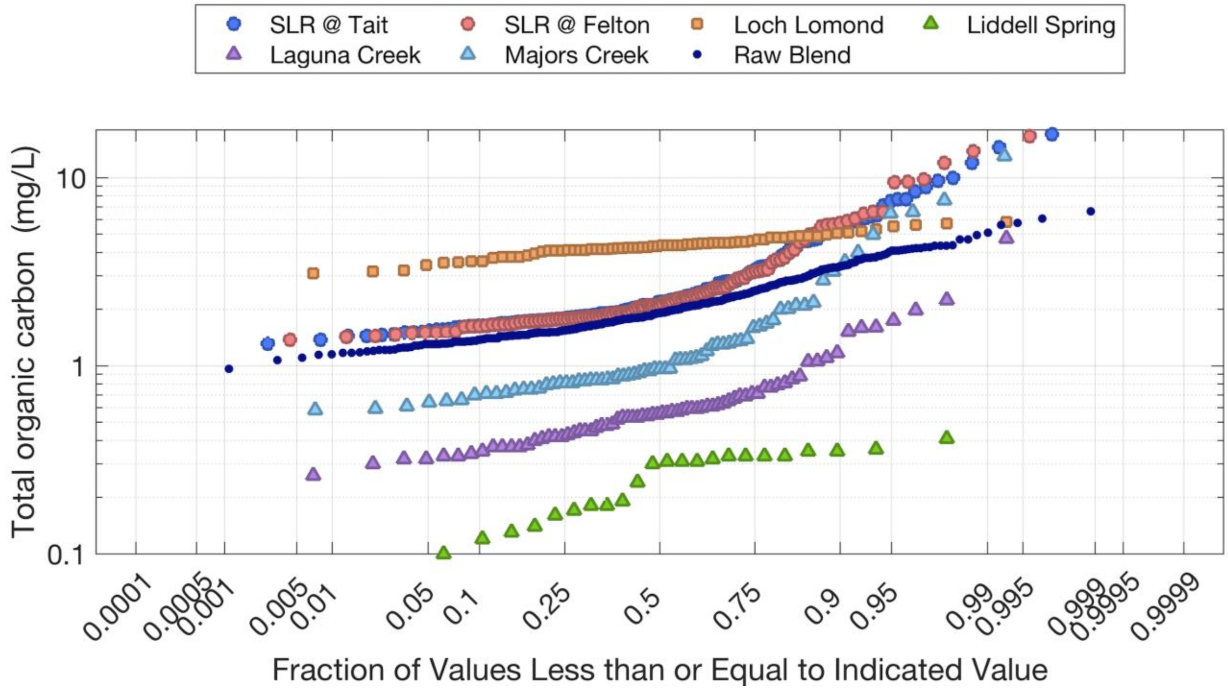

Figure 8: Probability plot of TOC of source waters for data collected between September 2011 and September 2018. Data shown are grab samples collected by the Santa Cruz water quality laboratory; raw blend data also include grab samples collected by operators.

First Flush Analysis

‘First flush’ is a phenomenon where the first storm(s) of the season result in a disproportionately high level of contamination to a water body. With the increased frequency of sampling that occurred in the SLR during storm events, the impacts of first flush storms were captured in Water Year 2017 and 2018 in a way that had not been previously observed. While it is likely that this pattern also occurred in previous years, the effect of the first flush was less evident in the historical data, likely due to the lower sampling frequency. To assess the first flush impacts on organic carbon concentrations in the SLR, DOC data from the SLR Tait Street Diversion were used. DOC was used in this analysis rather than TOC because DOC was measured by operators during the 2016/17 event-based sampling (during the winter of 2017/18, both TOC and DOC were measured during event-based sampling). As mentioned above, DOC may be used interchangeably with TOC because essentially all of the TOC is present as DOC.

Trends in DOC were analyzed in relation to storms that occurred in the winter of 2016/17 (see Figure 9). In the winter of 2016/17, DOC spiked during the first storm of the season, quickly reaching its maximum value of 12 mg/L from an initial concentration of 1.6 mg/L. Even though it was a relatively small storm (as evidenced by the small increase in river flow), the first storm of the season often washes large amounts of organics into the river, resulting in the observed DOC spike. The subsequent two storms (Storms 2 and 3 in Figure 9) continued to result in DOC concentrations above 8 mg/L, and then the concentrations leveled out in the following storms, with the maximum DOC values ranging between 6 and 8 mg/L.

First flush impacts were also observed in the winter of 2017/18, but because of the differences in storm patterns, they don't follow the same trend as those observed in 2016/17. As shown in Figure 9, the DOC spiked similarly for each of the storms in 2017/18; unlike the 2016/2017 winter, the organics concentration did not level out for the storms later in the season. The observed difference between the two winters may be due to the fact that there were fewer storm events and the dry periods between storms were longer during the winter of 2017/18 compared to 2016/17. The longer dry periods between storms may have caused a build-up of organics such that there was not just one first flush for the season but a 'first' flush for each storm. In addition, the magnitude of the storms in 2017/18 was smaller than that in 2016/17 and thus the first few storms may not have been large enough to completely wash away the accumulated surface organics.

Due to the high organics loads in the first flush flows, operators currently avoid taking San Lorenzo River flows during these periods even though the turbidities may be relatively low. As will be discussed later, high organic carbon concentrations result in high DBP formation.

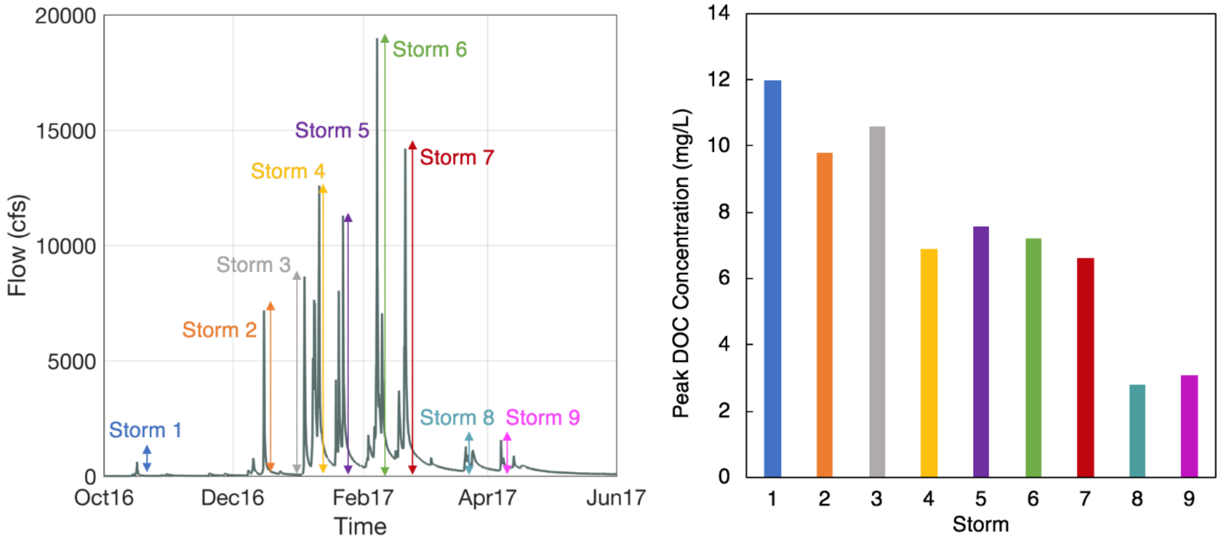

Figure 9: Assigned storms of winter 2016/2017 and the peak DOC in SLR at Tait that corresponded with these storms. River flow data were obtained from the USGS stream gage at Big Trees (USGS gage number 11160500). DOC data are grab samples collected by GHWTP operators during storm events.

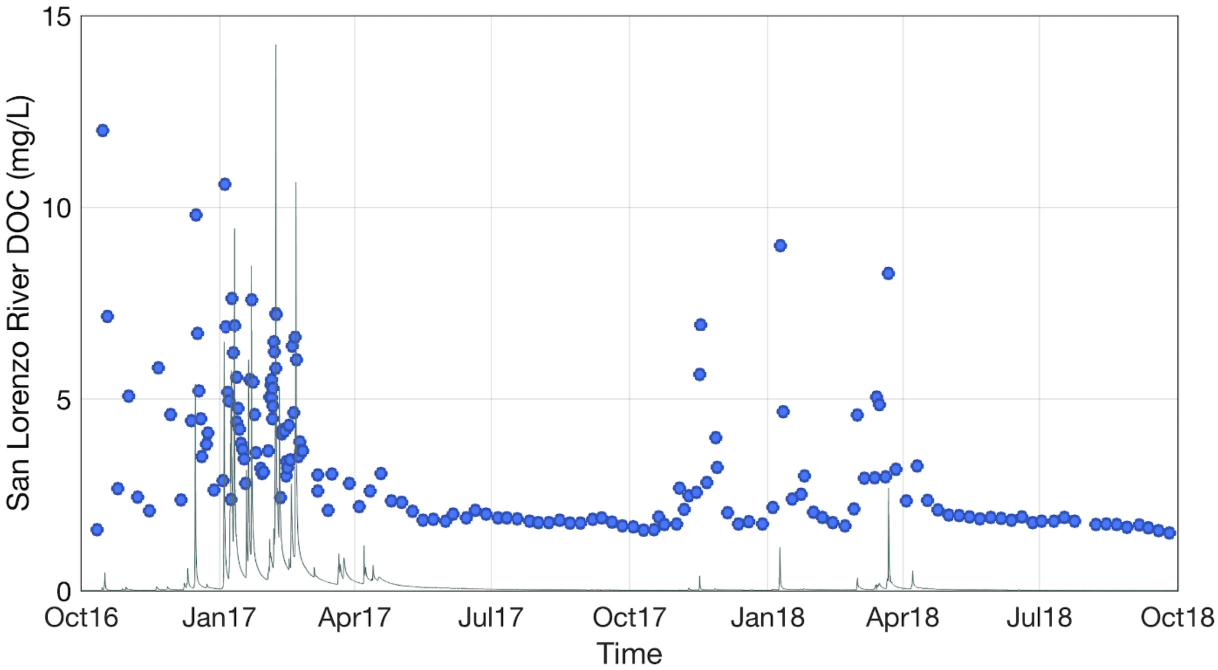

Figure 10. San Lorenzo River (at Tait) DOC between October 2016 and October 2018. The DOC data includes grab samples collected by GHWTP operators during storm events.

Hardness and Conductivity

Both hardness and conductivity provide information on the amount and type of ions present in the water. Hardness specifically measures the concentration of divalent cations (typically dominated by calcium and magnesium), whereas conductivity is a bulk measurement that quantifies ions in solution. The concentration and type of ions in the water may have implications for coagulation performance. Generally, clarification processes are improved with higher ion concentrations due to their role in destabilizing particles.

Both the hardness and conductivity of the source waters and the Raw Blend exhibit seasonal trends (see Figure 11 and Figure 12). Most of the source waters experience a decrease in both hardness and conductivity in the winter due to dilution from rain water. In comparison, Liddell Spring increases in both hardness and conductivity during the winter. The increase in Liddell Spring's hardness and conductivity was large enough to cause an increase in the hardness and conductivity of the Raw Blend in some years. As with the other parameters, the response of source water hardness and conductivity to rainfall was more dramatic during winter 2016/17 compared to the other winters, likely due to the large magnitude of the storms in 2016/17.

In addition to this seasonal fluctuation, there has been an overall upward trend for the conductivity in all source waters over the course of the time period analyzed (see Figure 12). For example, the monthly average conductivity for the Raw Blend in September 2011 was 382 µmhos/cm, while the averages in September 2016, 2017, and 2018 were 446 (17% increase), 414 (8% increase), and 418 (9% increase) µmhos/cm, respectively. This upward trend is not reflected in the hardness measurements and thus appears to be caused by an increase in monovalent cations (e.g., chloride, sodium, etc.).

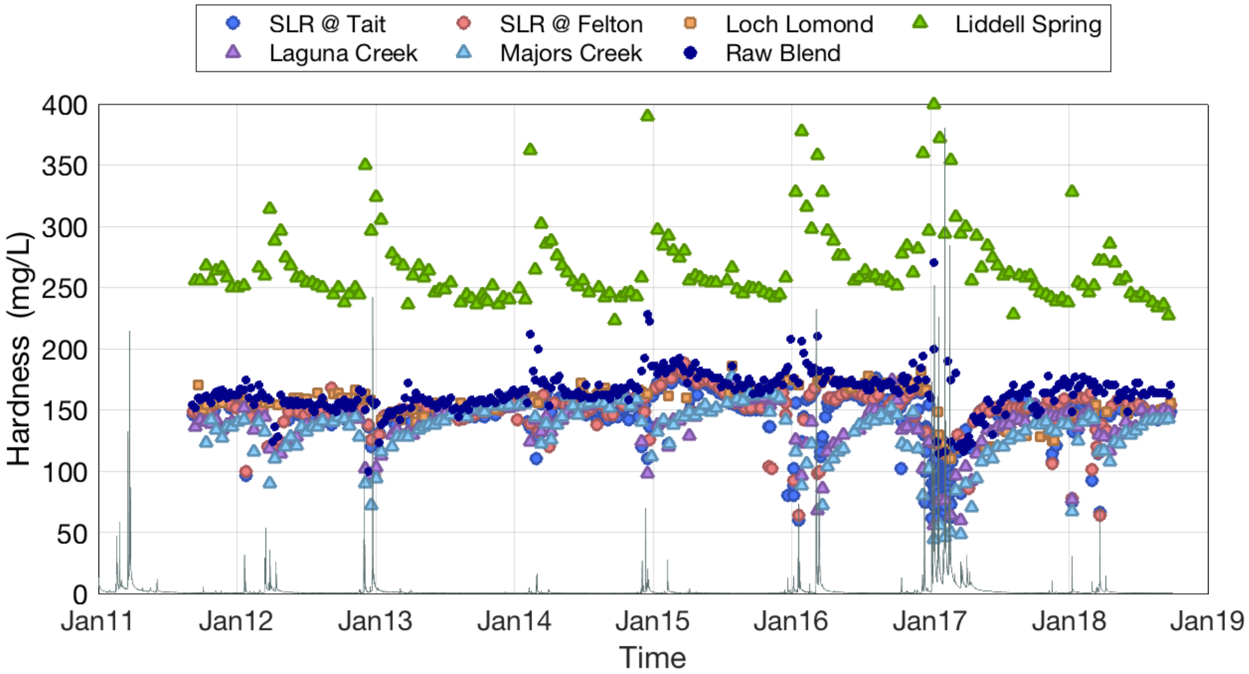

Figure 11: Hardness of source waters between September 2011 and September 2018.

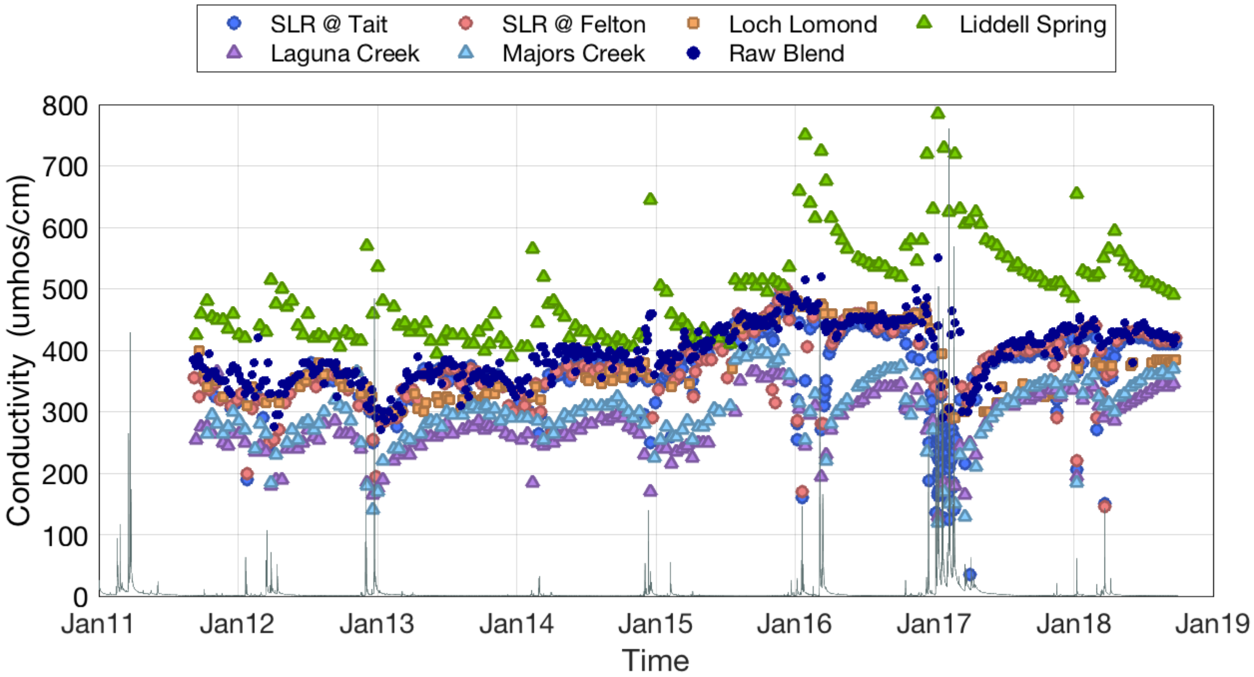

Figure 12: Conductivity of source waters between September 2011 and September 2018.

Turbidity

Turbidity is an optical assessment of water clarity; it measures the scattering of light by suspended particles – a phenomenon which causes water to appear cloudy (Crittenden et al. 2012). Turbidity is thus a measurement of the suspended particles in water and has implications for the solids handling (discussed further in Section 3) and disinfection. The Interim Enhanced Surface Water Treatment Rule (IESWTR) requires that systems sourcing surface water and using conventional filtration must maintain turbidity levels in their combined filter effluent no higher than 0.3 NTU at least 95% of the time, and never exceed 1 NTU. GHWTP has historically stopped taking San Lorenzo River water when the turbidity rises above 10 NTU in order to ensure this goal is met and to avoid issues with solids production.

As shown in Figure 13, the turbidity of the source waters only increased above the 10 NTU threshold during winter storms. Prior to winter 2015/2016, these exceedances occurred almost exclusively in the San Lorenzo River and typically no more than five times per year. Starting in the winter of 2015/2016, the exceedances increased in frequency and duration, as the storms were stronger than preceding years. The San Lorenzo River exceeded 10 NTU repeatedly between November 2015 and March 2016, reaching a maximum of 222 NTU on 1/19/16 at the Tait Street Diversion (as recorded by routine grab sampling). While the San Lorenzo River water had very high turbidity during this period, that source water is not used during periods of high turbidity and thus the Raw Blend never exceeded 10 NTU.

However, during the 2017 Water Year, the Raw Blend exceeded 10 NTU a total of 11 times and reached a maximum of 40.9 NTU on 6/14/17 (see Figure 13). This increase in Raw Blend turbidity is due to the fact that many of the source waters exceeded 10 NTU in 2016/17 (including Laguna Creek, Loch Lomond, Liddell Spring, and Majors Creek, in addition to the San Lorenzo River). The San Lorenzo River turbidity peaked at 1,590 NTU on 2/7/17 during the largest storm of the year (indicated by the highest river flow during this period). In addition, 2016/17 was the first period (during the time analyzed) that Loch Lomond turbidity exceeded 10 NTU. The turbidity of Loch Lomond did not return to below 10 NTU until October 2017. This elevated turbidity was a result of the clay landslide that occurred in January 2017. Given that the winter of 2016/2017 included unusually large storms and a clay landslide, it is likely not representative of typical winter conditions. The implications of San Lorenzo River winter turbidity for solids production are further discussed in Section 3.

During the 2017/18 winter, the turbidity exceeded 10 NTU in three of the source waters: San Lorenzo River, Loch Lomond, and Majors Creek. The highest turbidity measurement of 2017/18 was 357 NTU, which occurred on 3/22/18 in the San Lorenzo River at Tait. Despite these events, the raw blend was managed so that the turbidity never exceeded 10 NTU.

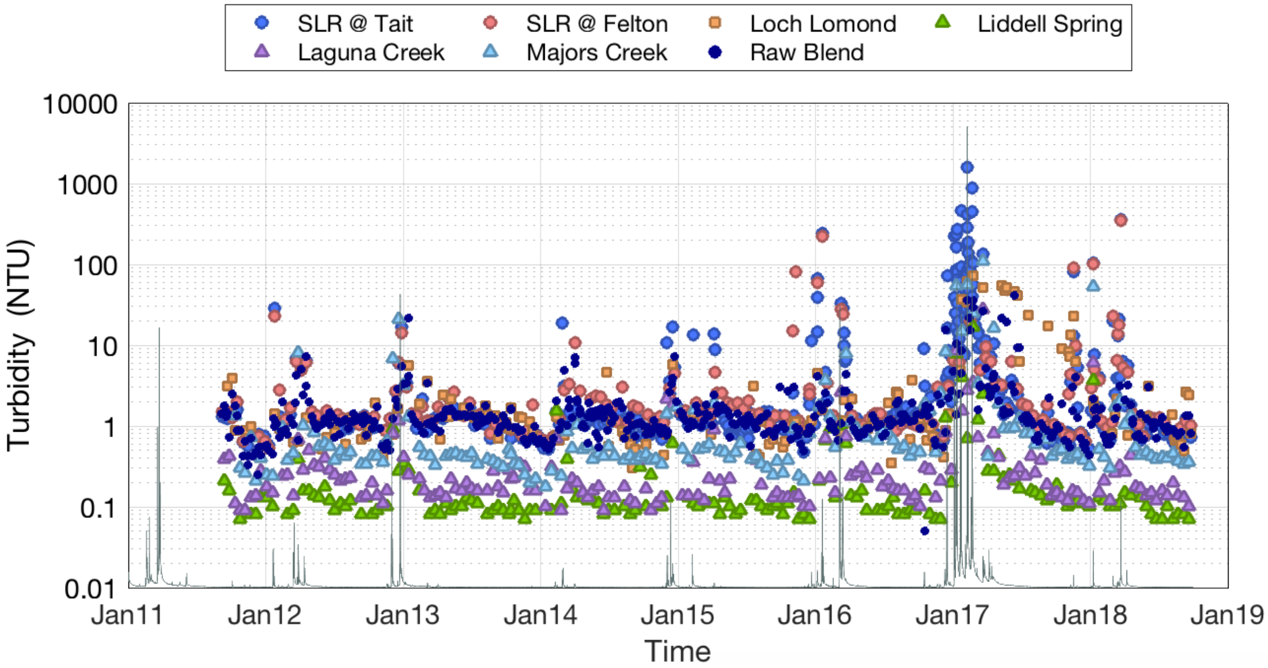

Figure 13: Turbidity of source waters between September 2011 and September 2018. Turbidity data for the San Lorenzo River include routine samples and event-based samples collected in the winters of 2016/17 and 2017/18; in-stream turbidimeter data are not included.

Turbidity data were collected in several ways. Turbidity grab samples were collected as part of routine source water monitoring, as well as during storm events in the San Lorenzo River. The City of Santa Cruz also maintains an in-stream turbidimeter in the river that continuously monitors turbidity; this turbidimeter logs data every five minutes, and records the minimum, maximum, median, and mean values recorded over the previous five minutes. A comparison of the median in-stream turbidity data and the grab sample data is shown in Figure 14. The data were cut off above turbidities of 50 NTU for ease of viewing the data and comparing the results at relevant turbidity values (i.e. those most likely to be treated at GHWTP). As can be seen, the in-stream and grab sample data agree well, although the grab samples are frequently slightly lower than the in-stream values. In the range of relevant turbidities, this discrepancy is not likely to have a large impact on SLR treatability.

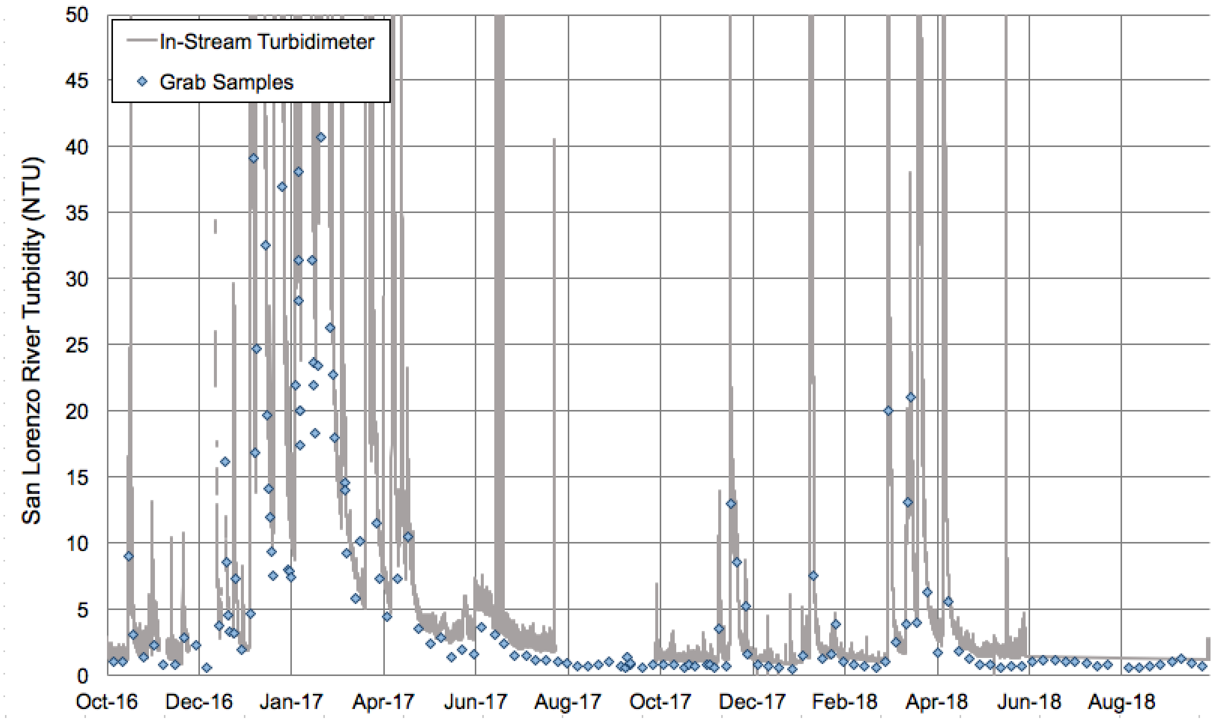

Figure 14. Comparison of grab sample and in-stream turbidimeter turbidity data for the San Lorenzo River at Tait St. Grab samples were collected as part of routine monitoring, as well as during storm events.

Total Suspended Solids

The total suspended solids (TSS) concentration is a direct measurement of the quantity of suspended particles (solids) in a water sample. It is quantified as the dry weight of solids captured through a specified filter size. Turbidity is a measure of optical clarity, which is generally controlled by the concentration of suspended particles, but not in a simple linear way. In terms of solids handling implications at GHWTP, TSS is a more directly relevant parameter than turbidity. As will be discussed later, it is useful to develop a relationship between these two parameters because turbidity is much simpler to measure.

Prior to this study, TSS data were not routinely collected. Consequently, only data from 10/11/16 through 9/9/17 are presented (TSS was not measured in Water Year 2018). The TSS exhibits the same trends as the turbidity, increasing dramatically in the source waters during winter storms. As expected, the TSS is highest in the San Lorenzo River during storms and, similar to the turbidity, the TSS in Loch Lomond remained elevated through the end of the period of data analyzed in this study (see Figure 15). The implications of the TSS levels in the source waters with respect to solids handling will be discussed in Section 3.

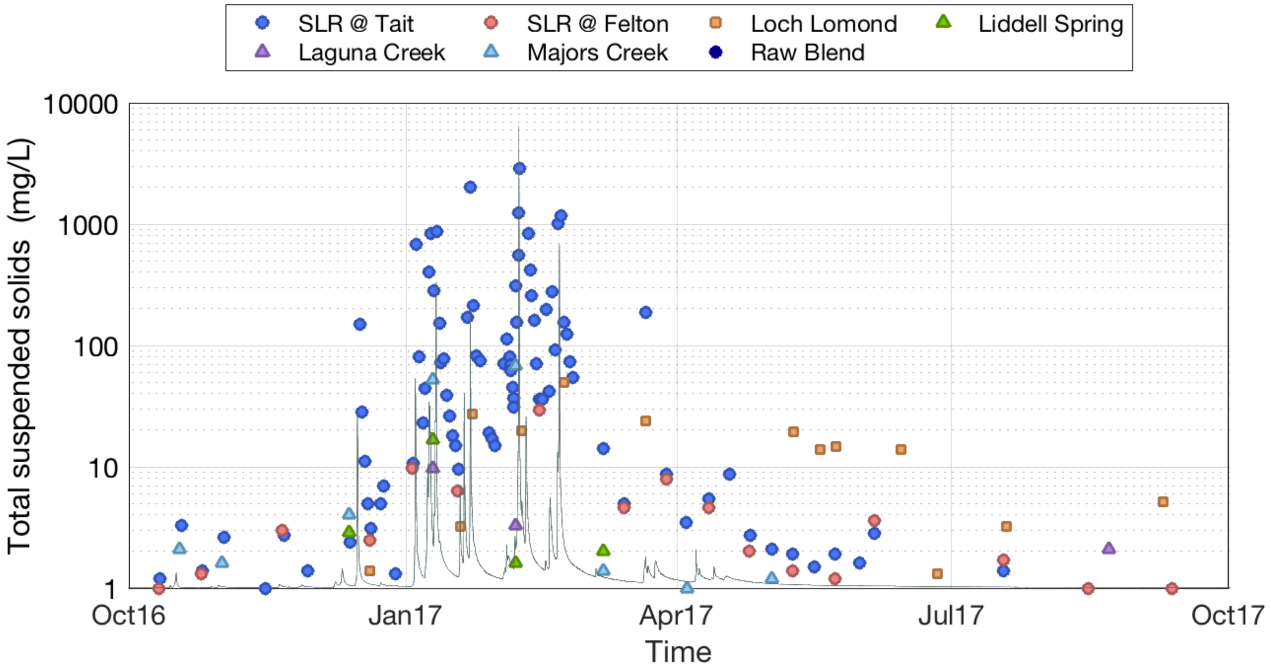

Figure 15: Total suspended solids concentrations of source waters between October 2016 and September 2017 (no TSS data were collected prior to the source water monitoring campaign and no TSS data were collected in Water Year 2018).

Color

Color is a parameter used to define the aesthetic quality of water resulting from the presence of certain dissolved species. It is also an indication of water's organic content, which includes constituents such as humic and fulvic acids (Crittenden et al. 2012). Additionally, color can be an indication of the presence and level of other constituents, including iron and manganese. The secondary MCL for color in finished water is 15 units. Color can be expected to follow the same trends as TOC and turbidity.

As shown in Figure 16, the spread and trends of color follow the same as those shown for turbidity (Figure 13), i.e., color increases during storm events. Because there were more storm events during the 2016/2017 winter, there were more peaks in color compared to the years prior to 2016 or the 2017/2018 winter. The North Coast sources consistently have the lowest color. Consequently, the Raw Blend color is dominated by the San Lorenzo River and Loch Lomond. All of the sources experience an increase in color during storms, with Liddell Spring impacted the least and San Lorenzo River the most (and having the highest values overall). The Raw Blend is managed to buffer these fluctuations. Recall that the contribution of color from the San Lorenzo River water is limited by the fact that it is not used at the plant if the turbidity was above 10 NTU.

It is clear from the probability plot (Figure 17) that the color of the Raw Blend is slightly lower than that of the San Lorenzo River and Loch Lomond at essentially all times. The North Coast streams generally have lower color than the Raw Blend, although the color in Majors Creek does exceed that of the Raw Blend about 5% of the time. In general, the North Coast streams provide a buffer against the color fluctuations of San Lorenzo River and Loch Lomond to reduce the color of the Raw Blend.

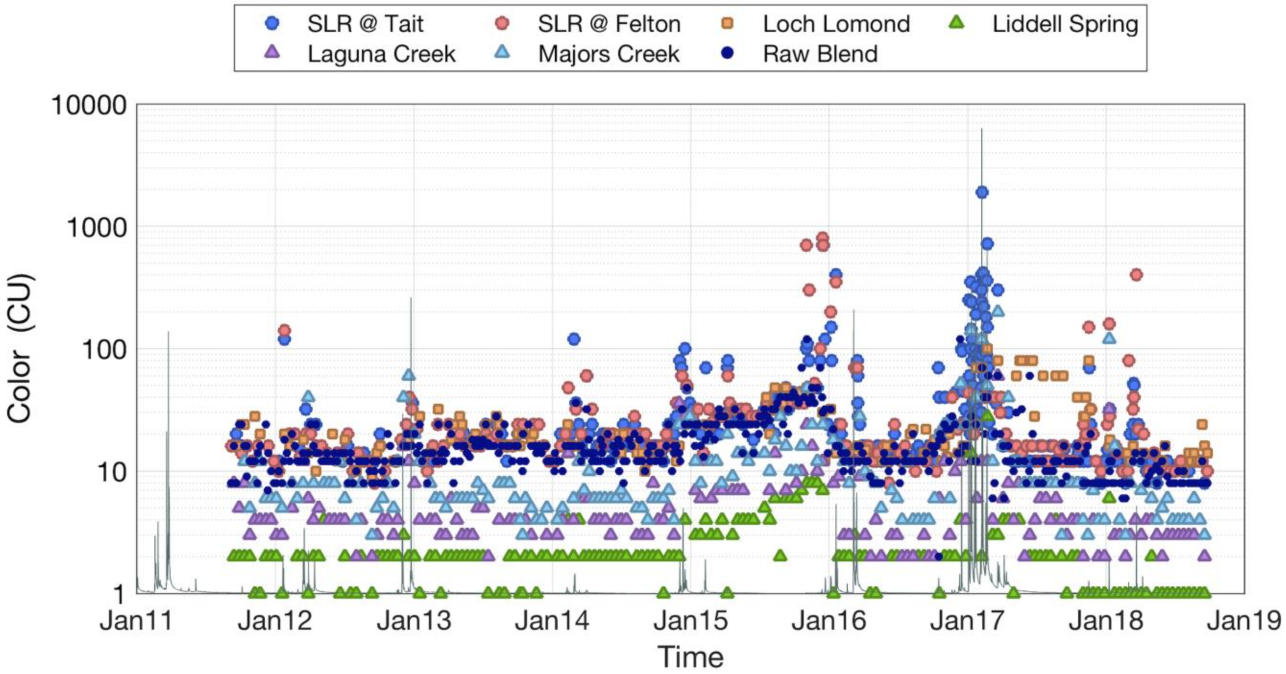

Figure 16: Color of source waters between September 2011 and September 2018.

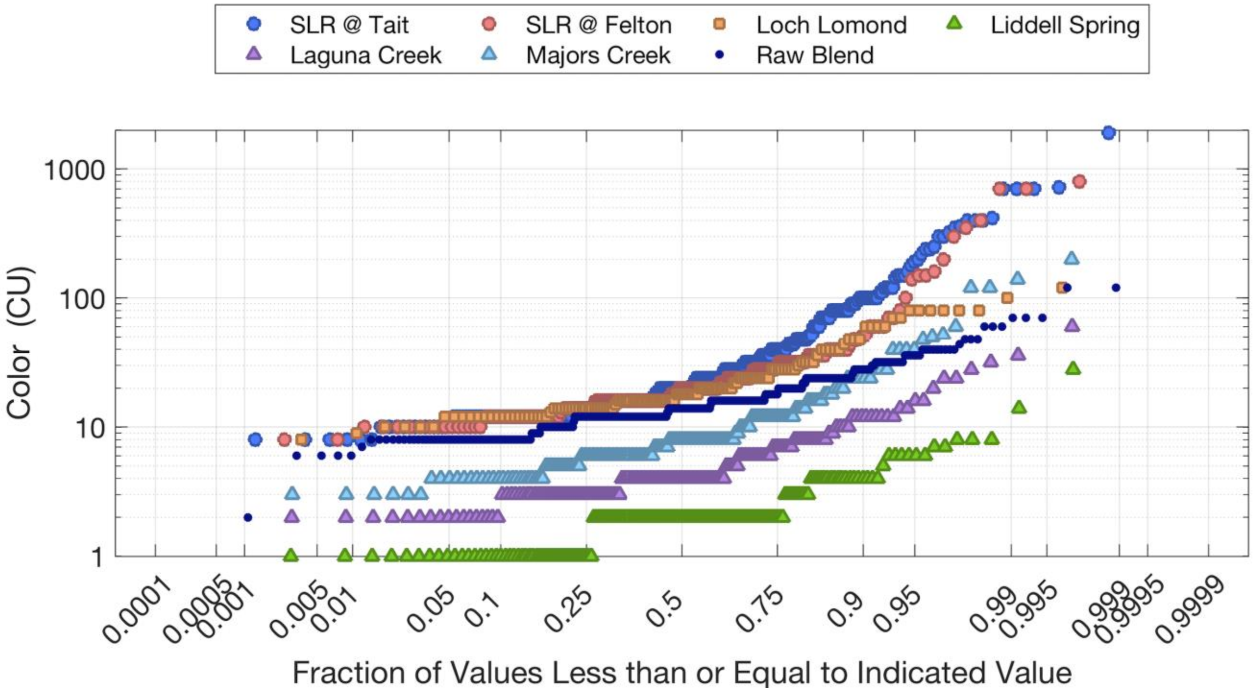

Figure 17: Probability plot of color of source waters for data collected between September 2011 and September 2018.

Bromide

Bromide concentrations were measured in the source waters to assess the potential for forming brominated DBPs. Brominated DBPs are a concern for both chlorine and ozone disinfection approaches. Based on our engineering experience, brominated DBPs become a concern if the bromide concentrations exceed 300 μg/L. Bromide has been monitored in the source waters since 2014. The measured concentrations in all source waters remain below 300 μg/L (see Figure 18). In fact, all source waters (including the Raw Blend) have remained below 50% of this limit, with the highest bromide concentration reaching only 120 μg/L in the San Lorenzo River (on 9/16/14). It appears that the bromide concentrations in the San Lorenzo River have been decreasing over time, though it is difficult to assess definitively with the limited data available. Overall, bromide does not pose any major issues for treatment at GHWTP and is not expected to become an issue.

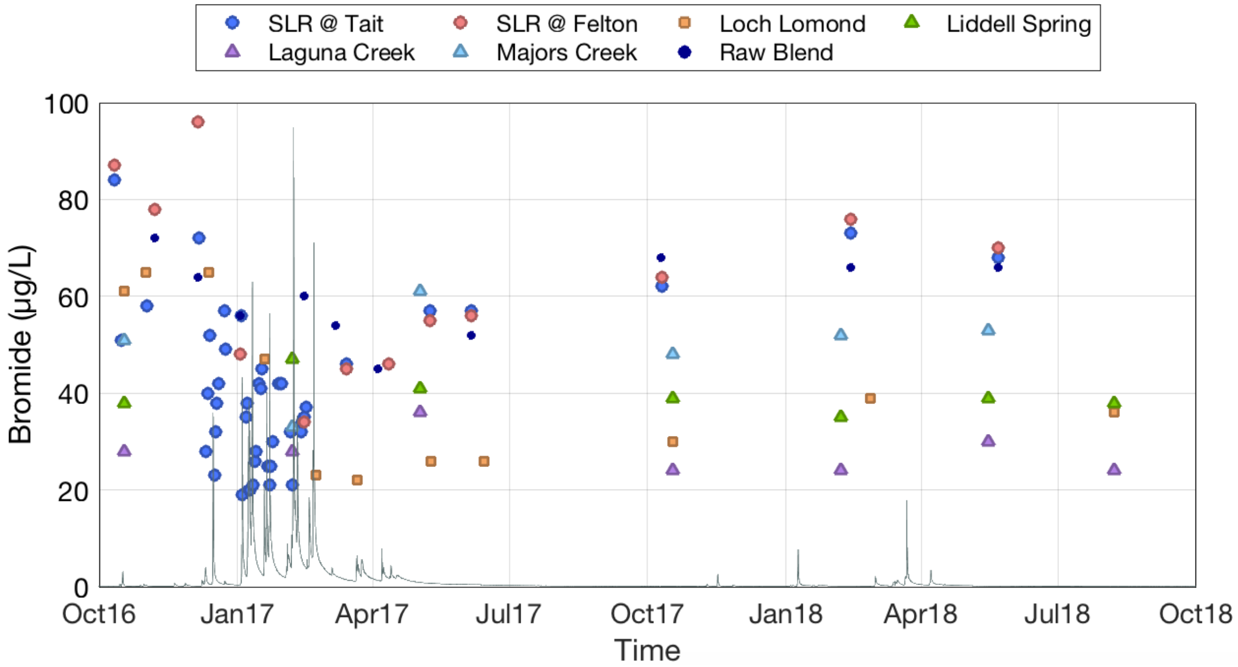

Figure 18: Bromide concentrations of source waters between August 2014 and September 2018. No data were available prior to August 2014.

Iron and Manganese

The presence of iron and manganese can be a concern in drinking water due to aesthetic issues, particularly with respect to color and taste. Iron and manganese are regulated with secondary MCLs (sMCLs) of 0.3 and 0.05 mg/L, respectively. These metals can be present in both particulate and dissolved forms. While the sMCLs are for the total concentration (do not differentiate between forms), the implications for removal may be different for the particulate versus dissolved forms.

Though both iron and manganese exceeded these sMCLs in most of the source waters, the majority of the measured iron and manganese existed in the particulate form. Because the particulate forms of iron and manganese are easily removed via treatment, they are not expected to be a concern. The dissolved fractions of both metals did not exceed the sMCLs in any of the sources; the highest dissolved iron concentration was 0.18 mg/L in Majors Creek, and the highest dissolved manganese concentration was 0.02 mg/L in the San Lorenzo River. As a result, iron and manganese are not expected to pose an issue for treatment or compliance.

Ammonia, Nitrate, and Nitrite

Ammonia is not regulated in drinking water. Ammonia is typically found below the detection limit or at very low concentrations in source waters for drinking water facilities and in finished drinking water. In the source waters analyzed, ammonia was typically not detected, and in the instances where it was detected, it was present at very low concentrations (<0.1 mg/L in all sources expect the San Lorenzo River; <0.25 mg/L in the San Lorenzo River).

Nitrate and nitrite can cause health impacts in humans and are thus regulated in drinking water. In some surface waters, the presence of nitrate is indicative of the potential for algae in stagnant areas and, in turn, the potential for associated taste and odor events. The MCL for nitrate is 10 mg/L as nitrogen (or 45 mg/L as NO_3) and the MCL for nitrite is 1 mg/L as nitrogen; the sum of both nitrate and nitrite is regulated at 10 mg/L as nitrogen. The concentration of nitrate and nitrate was far below the MCLs in all sources; in fact, nitrite was typically not detected and, when it was, it below 0.1 mg/L as N. Nitrate was detected in all sources expect the Tait Wells but did not exceed 3.2 mg/L as NO_3, which is well below the MCL of 45 mg/L as NO_3. The sum of nitrate and nitrite was also far below the MCL of 10 mg/L as nitrogen. Nitrate and nitrite are not expected to be issues for the GHWTP.

Fluoride

Fluoride has an MCL of 2 mg/L in California. According to recommendations from the U.S. Department of Health and Human Services, the optimal fluoride level for drinking water systems practicing fluoridation is 0.7 mg/L (U.S. DHHS 2015); this recommendation has also been adopted by the Division of Drinking Water (DDW). Although fluoride was detected in all of the source waters, it was present at levels far below the MCL, never exceeding 0.3 mg/L in any source. Fluoride is not expected to be a regulatory issue for GHWTP.

Total Dissolved Solids

The total dissolved solids (TDS) is a measurement of the ion content of a water, and is often correlated with conductivity and to some extent hardness. There is an sMCL for TDS of 500 mg/L because it contributes to aesthetic issues, such as deposits, color, and taste. TDS exhibits the same trend as many of the inorganic constituents in the water, such as calcium, sulfate, chloride, etc. Less data were gathered for TDS when compared with hardness and conductivity, but the event-based sampling in 2016/2017 captured the winter storms well (see Figure 19). During the winter of 2016/2017, TDS followed the same trend in relation to storm events as both hardness and conductivity: most water sources experienced a decrease in TDS during storms except Liddell Spring, which experienced an increase. The TDS in all of the source waters remained far below the sMCL of 500 mg/L except for in Liddell Spring, where the TDS spiked once to 540 mg/L during the winter of 2016/2017. Because of the blending with other source waters, this spike did not likely cause the Raw Blend TDS to exceed the sMCL. Though there was only one TDS measurement in the Raw Blend (on 6/7/17), it is likely that the TDS of the Raw Blend was similar to that of the source waters and comfortably below the sMCL.

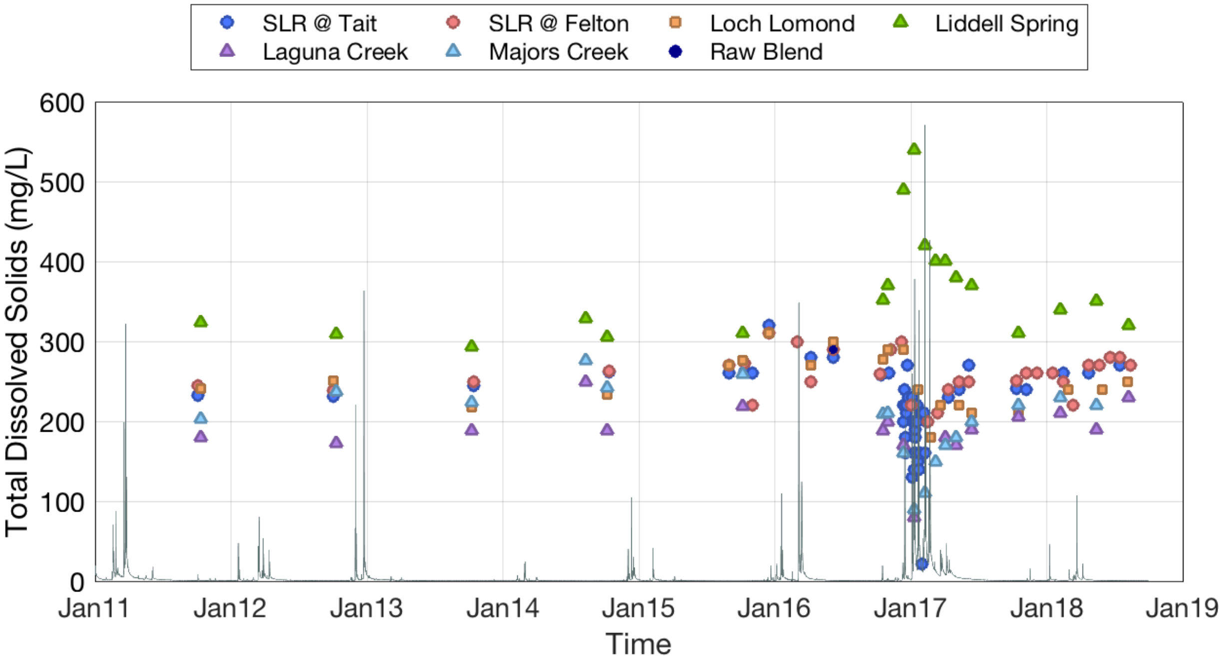

Figure 19: Total dissolved solids concentrations of source waters between September 2011 and September 2018.

Microbial Constituents

Total Coliform

The Surface Water Treatment Rule (SWTR) requires surface water treatment plants to provide a 4-log reduction in virus concentrations and 3-log reduction in *Giardia* concentrations. Additional guidelines in Appendix B of the California DDW's Surface Water Treatment Rule Guidance Document (DDW Appendix B) provide for additional *Giardia* and virus reduction requirements based on the source water total coliform concentration. According to these guidelines, the monthly median total coliform concentration is used to determine the specific virus and *Giardia* reduction requirements, as summarized in Table 3 below. Although DDW often follows these guidelines, it is important to note that these are not regulations as published in Title 22 CCR.

Table 3: Treatment guidelines for *Giardia* and virus reduction (DDW Appendix B).

Monthly Median Total Coliform Concentrations (MPN/100 mL)	*Giardia* Reduction (Log Reduction)	Virus Reduction Guidelines (Log Reduction)
<1000	3	4
>1000 – 10,000	4	5
>10,000 – 100,000	5	6

Because they have reported monthly median concentrations of total coliform between 1,000 and 10,000 MPN/100 mL, current treatment requirements at GHWTP for virus and *Giardia* are 5-logs and 4-logs, respectively. This concentration range is consistent with the concentrations observed over the past seven years where Raw Blend total coliform concentrations were generally between 100 and 10,000 MPN/100 mL (Figure 20). While there were a few points both above and below, the monthly median concentrations always fell in this range. Of the source waters, San Lorenzo River generally has the highest concentration of total coliform and Liddell Spring has the lowest concentration of total coliform. As shown in Figure 21, only 5 – 10% of total coliform measurements in San Lorenzo River were above 10,000 MPN/100 mL. Based on the results from this study, it is unlikely that the GHWTP will be moved into the highest *Giardia* and virus treatment bin, especially if the first flush and highest turbidity San Lorenzo River flows are not treated at the plant.

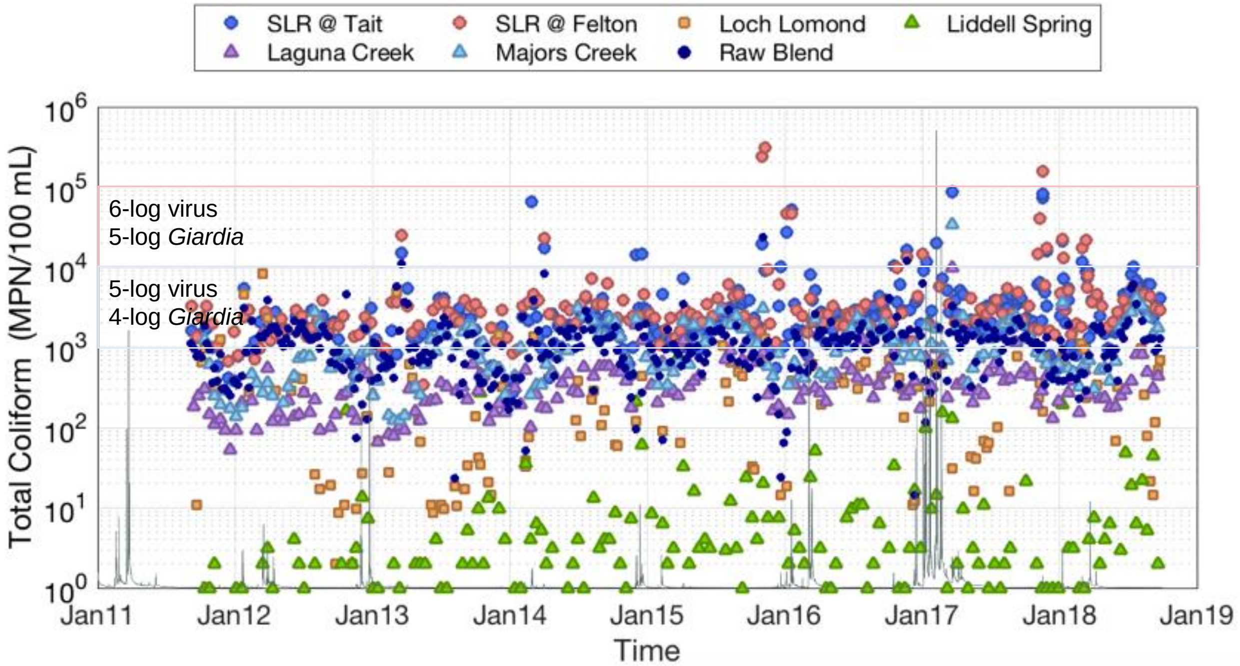

Figure 20: Total coliform concentrations of source waters between September 2011 and September 2018. Shaded bands show potential pathogen treatment requirements for monthly median total coliform concentration ranges.

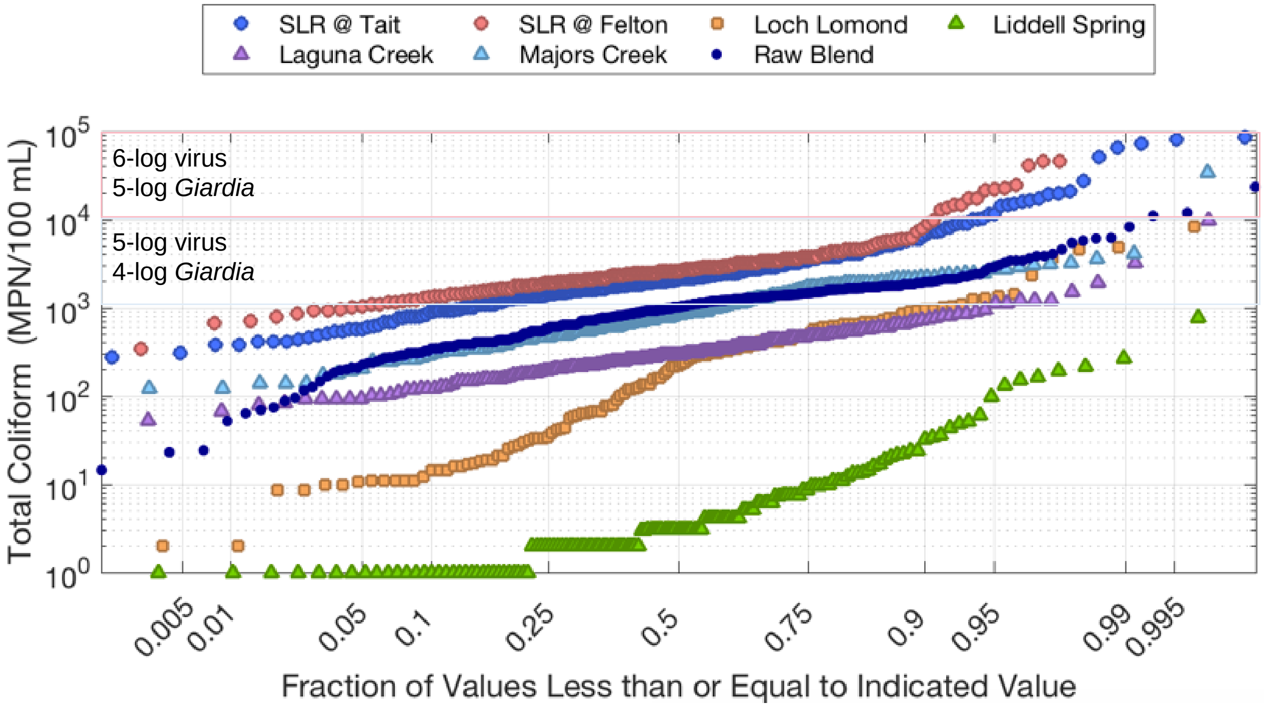

Figure 21. Probability plot of total coliform concentrations in the source waters and raw blend for data collected between September 2011 and September 2018. Shaded bands show potential pathogen treatment requirements for monthly median total coliform concentration ranges.

E. coli

The concentration of *E. coli* provides an indication of the extent of human and animal fecal contamination of a watershed. For drinking water supplies, the common guidance is that fecal coliform levels above 200 MPN/100 mL signifies a source with potentially large contamination from human sources (NRC 2004). There are no formal *E. coli* limits in this watershed, so the fecal coliform Total Mass Daily Load (TMDL) is used here as a point of comparison. The San Lorenzo River Watershed has a total maximum daily load (TMDL) for fecal coliform of 200 MPN/100 mL (30-day log-mean limit), with 90% of samples below 400 MPN/100 mL (Kennedy/Jenks 2013).

Similar to total coliform, the *E. coli* concentrations were highly variable between different source waters (Figure 22). Liddell Spring and Loch Lomond had the lowest concentrations of *E. coli*; concentrations were <10 MPN/100 mL in 100% and 95% of samples in these sources, respectively (Figure 23). The San Lorenzo River had the highest concentrations of *E. coli*, exceeding the log-mean limit of 200 MPN/100 mL in over 95% of samples (Figure 23).

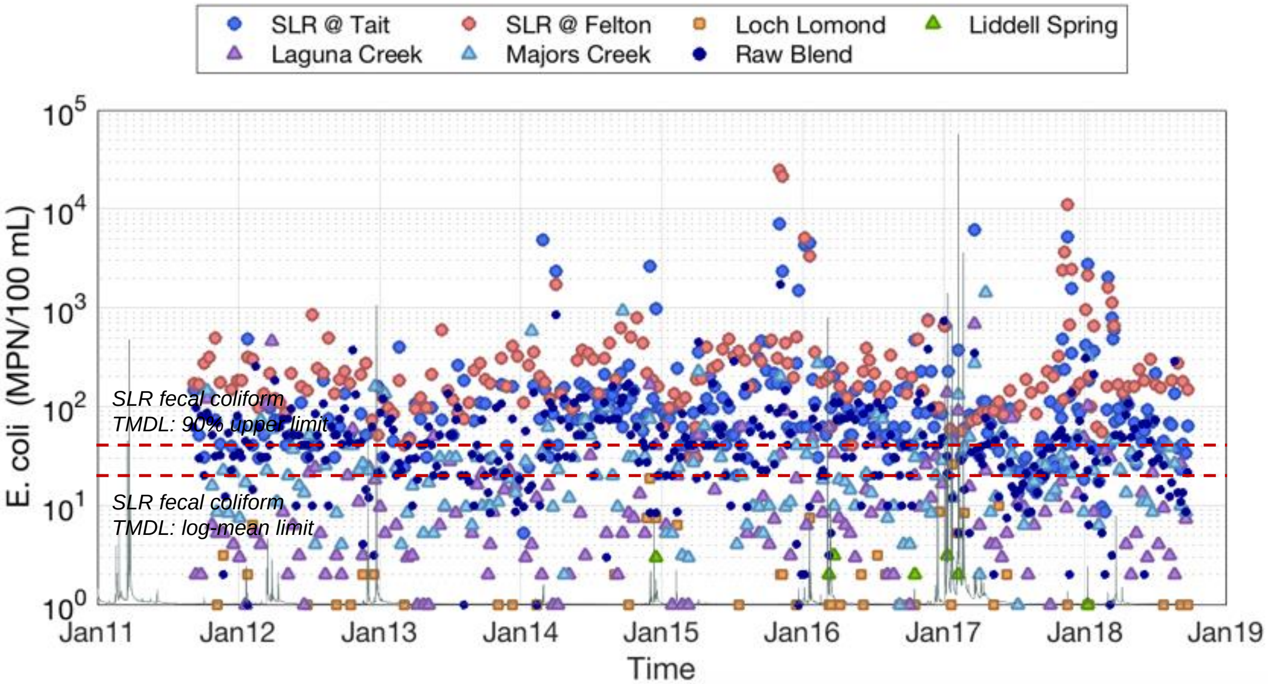

Figure 22: *E. coli* concentrations of source waters between September 2011 and September 2018.

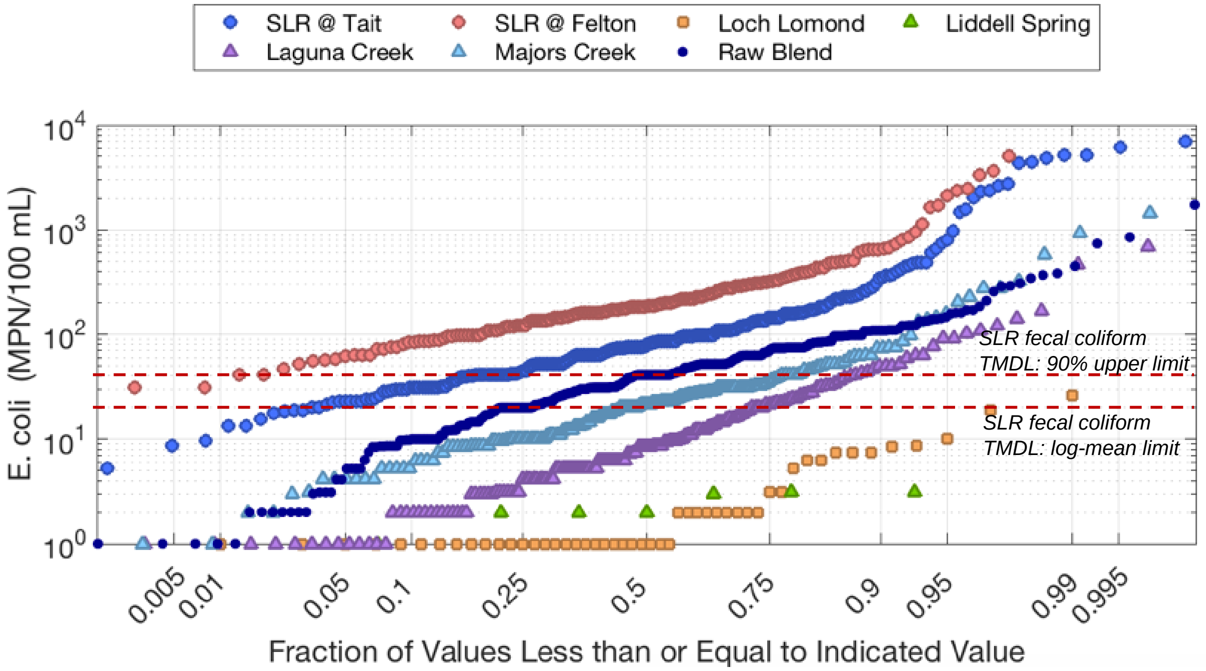

Figure 23. Probability plot of *E. coli* concentrations in source waters and raw blend for data collected between September 2011 and September 2018. Red lines indicate Total Mass Daily Load (TMDL) limits for fecal coliform in the San Lorenzo River (SLR) watershed.

Event-Based Sampling: Total Coliform and *E. coli*

Event-based sampling for total coliform and *E. coli* in the SLR started in October 2017. The goal of this additional sampling was to capture how storms influence the microbial contamination of the SLR. The sampling in Water Year 2018 showed that the concentration of total coliform and *E. coli* increases by at least an order of magnitude during winter storms (Figure 24 and Figure 25). Even small storms (corresponding to river flows <1000 cfs) resulted in more than a 10-fold increase in total coliform and *E. coli* concentrations. The same increase in total coliform and *E. coli* during winter storms was not observed in Water Year 2017; this may be because event-based sampling was not conducted during that period for these constituents. This data shows that event-based sampling is valuable for capturing the full influence of storms on water quality. As discussed in Section 6 below, we recommend continuing event-based sampling of total coliform and *E. coli* in the future, particularly to help evaluate whether the microbiological quality of the river water is progressively worsening over the years.

While any conclusions from this data are preliminary because it is limited to one year of event-based sampling, it appears that storms can increase the microbial contamination of the SLR. The primary source of this contamination cannot yet be determined based on the data available. Potential sources of contamination are discussed in the following section. Regardless of the source of contamination, disinfection performance should be a priority in order to ensure the microbiological safety of the water.

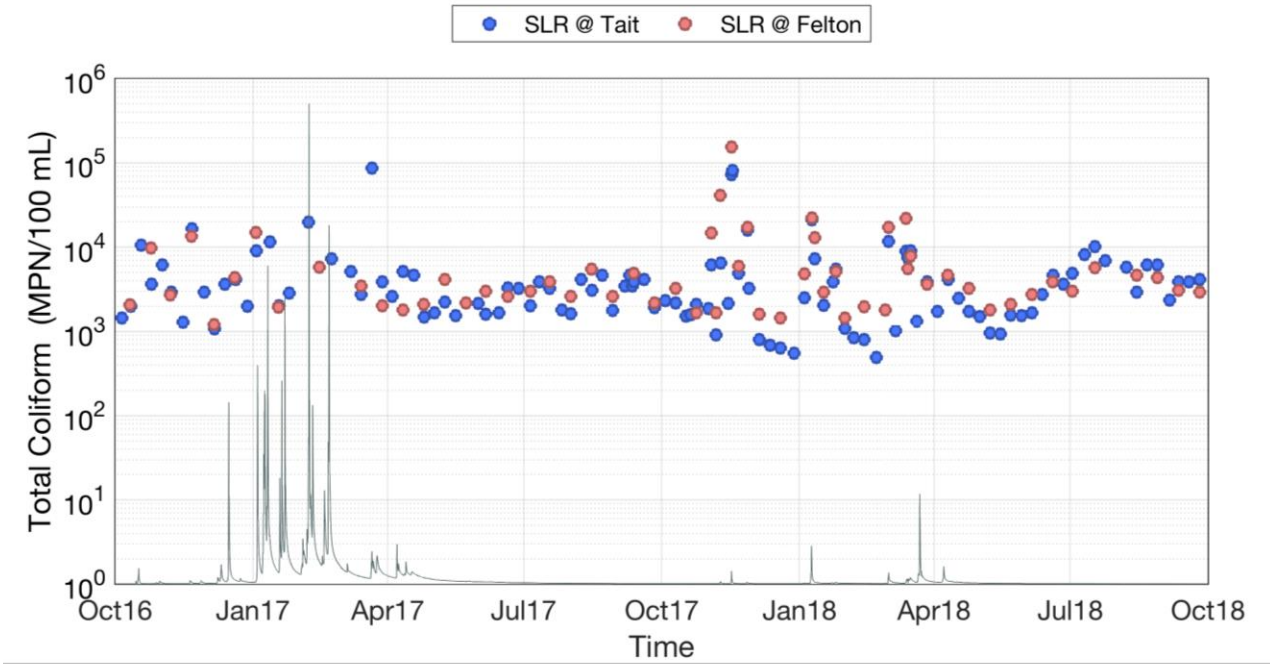

Figure 24. Total Coliform at San Lorenzo River between October 2016 and October 2018. 2017/2018 included event-based sampling while 2016/2017 did not event-based sampling.

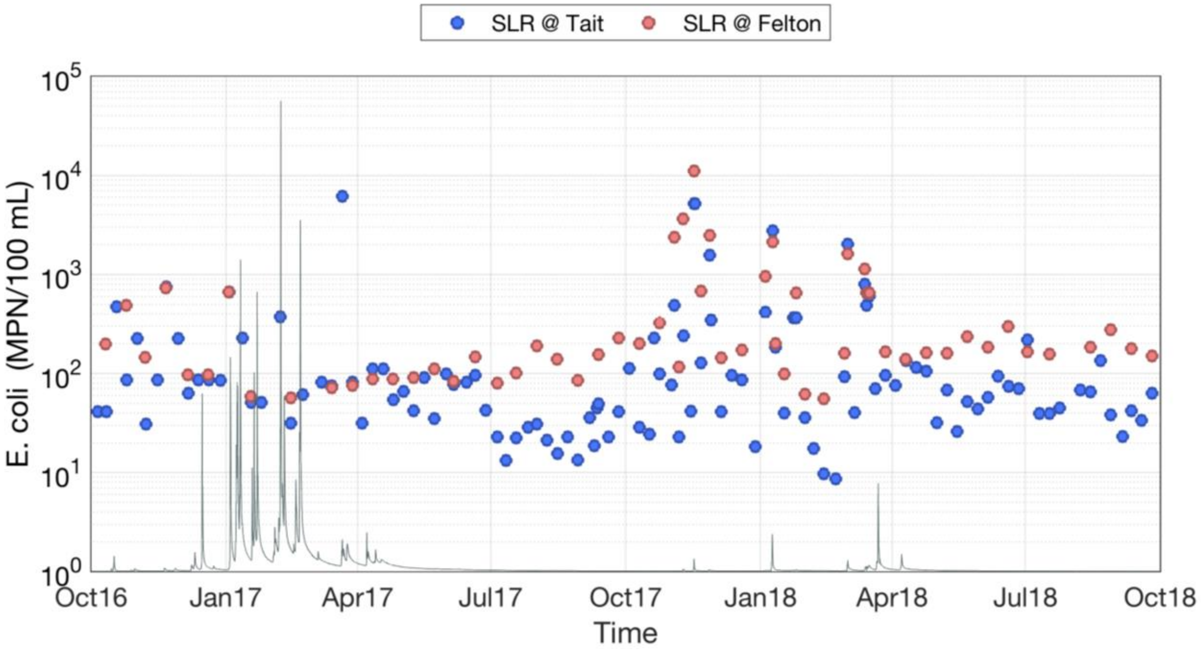

Figure 25. *E. coli* at San Lorenzo River between October 2016 and October 2018. 2017/2018 included event-based sampling while 2016/2017 did not event-based sampling.

Giardia and *Cryptosporidium*

Giardia and *Cryptosporidium* are pathogens of concern that are regulated under the Surface Water Treatment Rule (SWTR, EPA 1989), Interim Enhanced Surface Water Treatment Rule (IESWTR, EPA 1998), and Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR, EPA 2006), respectively. The minimum treatment requirement for *Giardia* is 3-logs, with at least 0.5-log coming from disinfection. The treatment requirement for *Giardia* at the GHWTP is higher (4-logs) than this minimum because of the high monthly median total coliform concentrations (as discussed above). In this water quality monitoring campaign, the source waters were only sampled for *Giardia* during Water Year 2017. During this sampling, *Giardia* was often not detected in the individual source waters; the maximum concentration was 0.4 cysts/L in the San Lorenzo River (Figure 26).

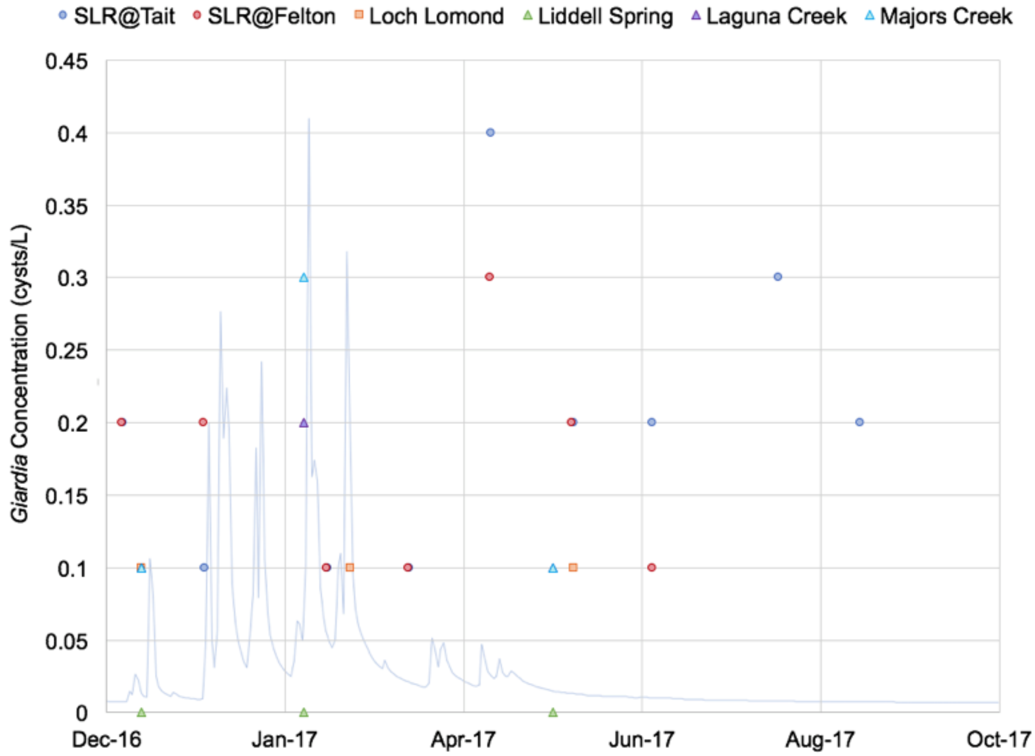

Figure 26: Giardia concentrations of source waters between December 2016 and September 2017.

Cryptosporidium removal requirements are determined from the observed *Cryptosporidium* concentrations in the source water, using the maximum running annual average (RAA) to determine a ‘bin,’ as described in the LT2ESWTR (see Table 4 for bin classifications and associated treatment requirements). Sanitary survey sampling conducted in 2016-2017 resulted in GHWTP being placed in Bin 2, thus requiring a total of 3-log reduction in *Cryptosporidium*. The sampling conducted in the 2016-2017 source water quality monitoring campaign resulted in similar findings: the maximum concentration of *Cryptosporidium* was 0.5 oocysts/L and was detected in SLR on 12/16/16 (Figure 27).

Table 4. *Cryptosporidium* bin classification based on concentration. Adapted from EPA 2006.

Bin	*Cryptosporidium* Concentration (oocysts/L)	*Cryptosporidium* Reduction Requirement
1	<0.075	2-log
2	0.075 – <1.0	3-log
3	1.0 – <3.0	4-log
4	≥3.0	4.5-log

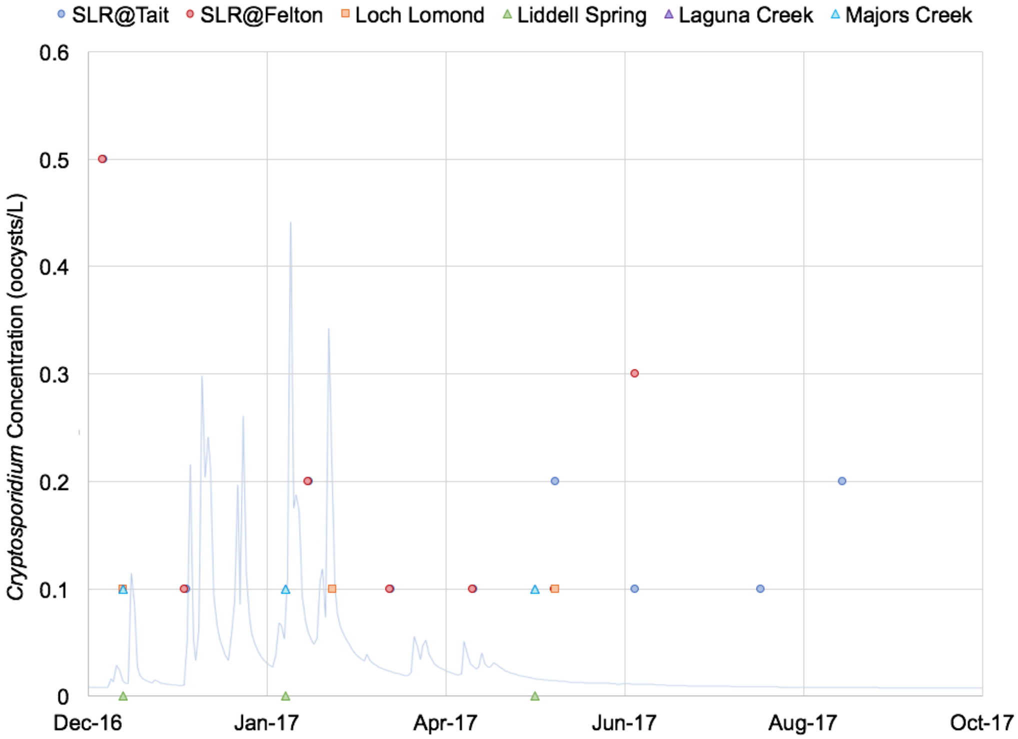

Figure 27: Cryptosporidium concentrations of source waters between December 2016 and September 2017.

2.3 Sources of Microbial Contamination in the San Lorenzo River

To better understand the source of microbial contamination in the SLR, a literature review was conducted. The SLR watershed has several potential sources of contamination, including septic systems, urban runoff, livestock/stables, and unauthorized activity (e.g. homeless encampments) (Kennedy/Jenks 2018). The watershed was mapped for land use in the 2018 Sanitary Survey (Kennedy/Jenks 2018) (Figure 28). Agriculture is a minor land use in the watershed, and most of the agricultural activity tends to be along the coastal ridges and away from the major streams; therefore, agriculture has a minor influence on the water quality. Grazing livestock (e.g., cattle and dairy cows) are also not expected to be large contributors of bacteria and pathogens to the SLR due to the low number of grazing livestock in the area. Historically, septic systems have been identified as a source of pollutants to the river. However, the 1995 Wastewater Management Plan implemented practices to prevent degradation of surface water quality from septic systems and introduced corrective measures to improve existing systems and reduce contaminant loading to the river. As of the 2018 Sanitary Survey, septic system failure rates were estimated at less than 0.5%.

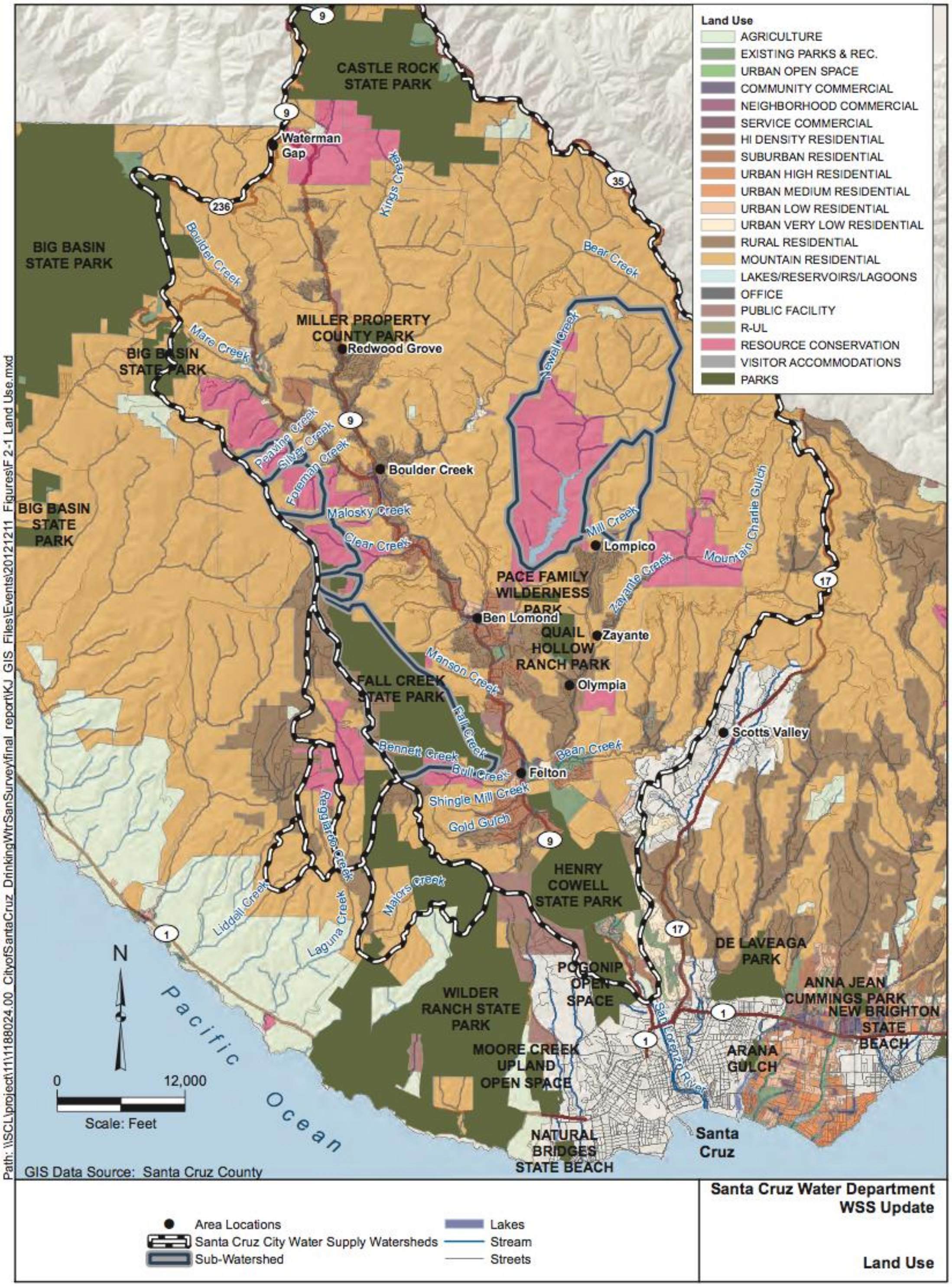

Figure 28: Land use in the San Lorenzo River Watershed, copied from the 2018 Sanitary Survey.

It is interesting to note that the event-based sampling conducted in the present study indicated that even the initial small storms in November 2017 led to large increases in total coliform and *E. coli* levels in the SLR. Given their small size, it seems unlikely that these storms would sufficiently saturate the ground to allow for the subsurface transport of total coliform and *E. coli* from the septic systems to the SLR. In line with this reasoning, the 2018 Sanitary Survey concludes that bacterial contamination of SLR is unlikely a result of groundwater contamination by properly functioning septic systems. The Sanitary Survey does state, however, that it could be the result of septic failures that release sewage to the ground surface, which is then transported into the surface water. Additional studies would be needed to definitively determine the sources of these microbial indicators.

One method for determining the origin of the fecal contamination includes monitoring for chemicals that are unique to human fecal contamination. Two such chemicals of potential interest are sucralose (an artificial sweetener) and caffeine. By measuring the concentration of these indicators along with the total coliform and *E. coli* concentrations, additional insight could be gained into the origin of the contamination during storms.

Another strategy to identify human vs. non-human contamination is the use of microbial source tracking that utilizes microbial “signatures” to differentiate between human and animal fecal contamination. A microbial source tracking study of the SLR watershed was conducted from 2002-2004 (Ricker and Peters 2006). This study used a method called “ribotyping” to differentiate between the *E. coli* from humans and the *E. coli* from various other warm-blooded animals. Ribotyping involves extracting the DNA from the *E. coli* in a water sample and comparing it to DNA of known sources (e.g., human, bird). The study included two sampling campaigns that evaluated multiple locations in the watershed; the relevant sample stations are shown in Figure 29. In the first of these sampling campaigns, approximately 100 samples were collected from four locations along the SLR (Boulder Creek, Felton, Sycamore Grove, and Rivermouth) during the winter of 2002-2003 (12 different sampling days). In the second of these sampling campaigns, approximately 100 samples were collected from 11 locations (including two locations on the SLR as well as various beaches and creeks) over the course of one year (Fall 2003 – Fall 2004).

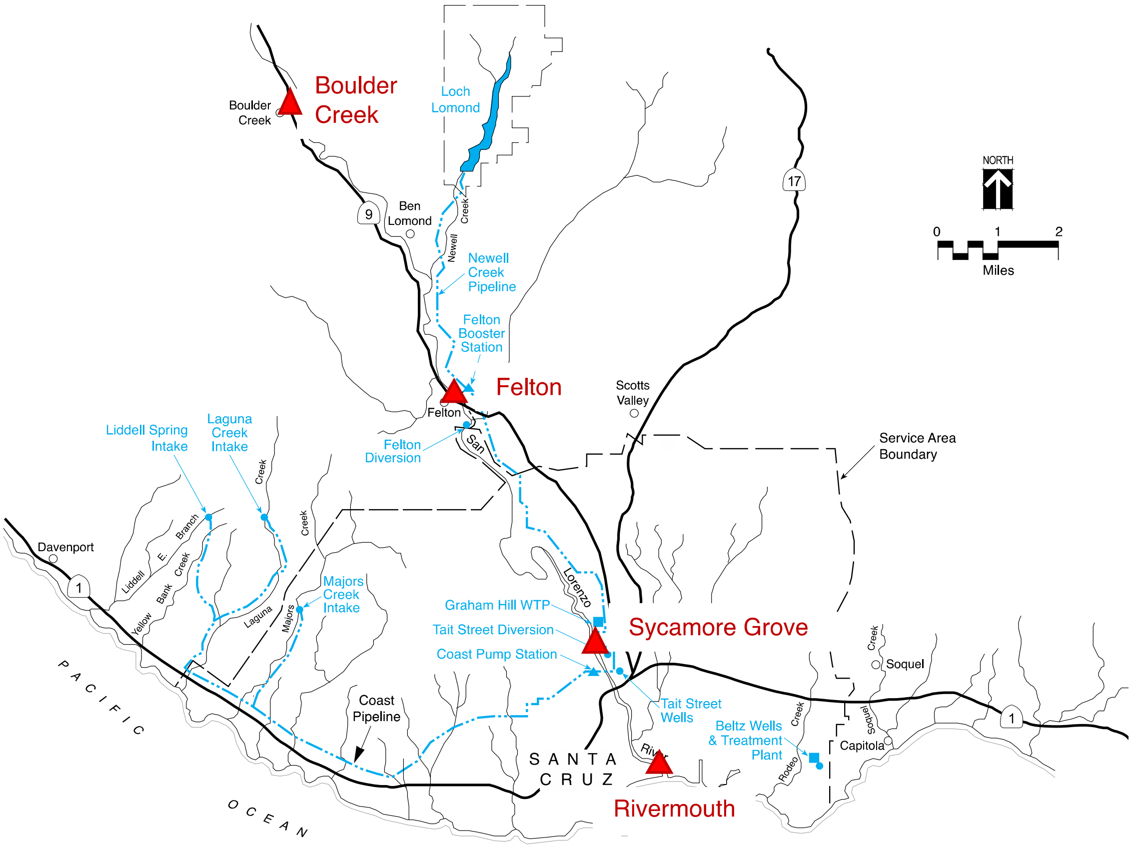

Figure 29: Sample locations along the SLR in the 2002-2004 microbial source tracking study. Modified from CDM (2007).

Table 5 shows the *E. coli* contribution from humans and various animals at four locations along the SLR during winter conditions (i.e., the first sampling campaign conducted during winter 2002-2003). Based on this data, birds and humans were the predominant sources of the *E. coli* in the SLR during the winter. The percent of *E. coli* from each of the sources is similar among the four stations, though the total concentration of *E. coli* in the Rivermouth is higher than in the upstream stations. Of these four sampling locations, the Rivermouth is least relevant to the current discussion because it is downstream of the GHWTP.

Table 6 shows the *E. coli* contribution from humans and various animals at two locations along the SLR in both the summer and winter. The results in this table are based on the combined dataset from the two sampling campaigns. In the winter, the largest contributors of *E. coli* are birds (42% at the Rivermouth and 32% at Sycamore Grove), followed closely by humans (28% at the Rivermouth and 25% at Sycamore grove). In the summer, the predominant contributor of *E. coli* are birds (40% at the Rivermouth and 65% at Sycamore Grove); humans only contribute 0-4% of the *E. coli* in the summer. Other minor contributors are wildlife, dogs, cows, rodents, and horses. A small portion (15-20%) of the *E. coli* could not be clearly linked to any of these animals.

Table 5: Human and Animal *E. coli* Contributions at Four Stations along the SLR (Winter 2002-2003 data, Ricker and Peters 2006)

	Rivermouth	**Sycamore Grove**	**Felton**	**Boulder Creek**
Log mean E. coli (cfu/100 mL)	434	84	181	117
Avian (Bird)	39%	29%	23%	34%
Human	30%	21%	21%	26%
Wildlife	4%	11%	15%	13%
Canine (Dog)	5%	6%	12%	9%
Bovine (Cow)	1%	5%	1%	0%
Rodent	4%	9%	7%	5%
Horse	1%	1%	10%	1%
Feline (Cat)	0%	1%	1%	1%
Unknown	15%	17%	11%	10%

Table 6: Seasonal Human and Animal *E. coli* Contributions at Two Stations along the SLR (Winter 2002-2003 data and 2003-2004 data, Ricker and Peters 2006)

	Rivermouth		**Sycamore Grove**	
	Winter	**Summer**	**Winter**	**Summer**
Log mean E. coli (cfu/100 mL)	236	194	82	64
Avian (Bird)	42%	40%	32%	65%
Human	28%	4%	25%	0%
Wildlife	6%	12%	4%	10%
Canine (Dog)	3%	4%	5%	0%
Bovine (Cow)	2%	4%	7%	0%
Rodent	6%	16%	7%	10%
Horse	2%	0%	1%	0%
Unknown	14%	20%	17%	15%

The results from Ricker and Peters suggest that humans are one of the major sources of microbial contamination in the winter and not in the summer. However, several notable improvements have occurred that may have further reduced the human contribution of *E. coli* since 2002. Specifically, at the time of the study, septic failures occurred at a rate of 1-5% (Ricker and Peters 2006) whereas in recent years the failure rate has been less than 0.5% (Kennedy/Jenks 2018). The reduction in septic system failure rates is likely the result of the Wastewater Management Plan of 1995. In addition, sewer system improvements have been implemented since 2002 (Kennedy/Jenks 2018), which could reduce the contribution of *E. coli* from sewage leaks. As a result of these improvements, the percent of *E. coli* from humans may have decreased since 2002-2004.

If the City is interested in gaining an updated understanding of the impact of humans on the microbial source water quality, it would be beneficial to conduct another microbial source tracking study. An understanding of who is contributing to the *E. coli* in the SLR can help guide efforts to control or mitigate the issue. For example, if the results show that humans are no longer a major contributor, control efforts could be refocused to address point and non-point sources from the most relevant animal species. If the results show that humans are still a major contributor of *E. coli* to the SLR in the winter, the City could investigate if there are additional infrastructure improvements that could be made (e.g., sewer upgrades) or whether the human *E. coli* is associated with unauthorized activity (e.g., homeless encampments).

2.4 Tait Wells

The monitoring campaign during Water Year 2018 also included additional sampling of the Tait Wells. The Tait Wells were generally found to have superior water quality compared to SLR (Figure 30). For example, total coliform and *E. coli* were never detected in the Tait Wells, whereas in the SLR the median concentrations for total coliform and *E. coli* were 2100 MPN/100 mL and 79 MPN/100 mL, respectively. In addition, the median TOC of the Tait Wells (1.3 mg/L) is lower than that of the SLR (2.2 mg/L). The median turbidity of the Tait Wells (0.2 NTU) is also much lower than that of the SLR (1.3 NTU). Consequently, the operations staff utilizes the Tait Wells to improve the Raw Blend water quality by reducing TOC, turbidity, and indicator organism concentrations.

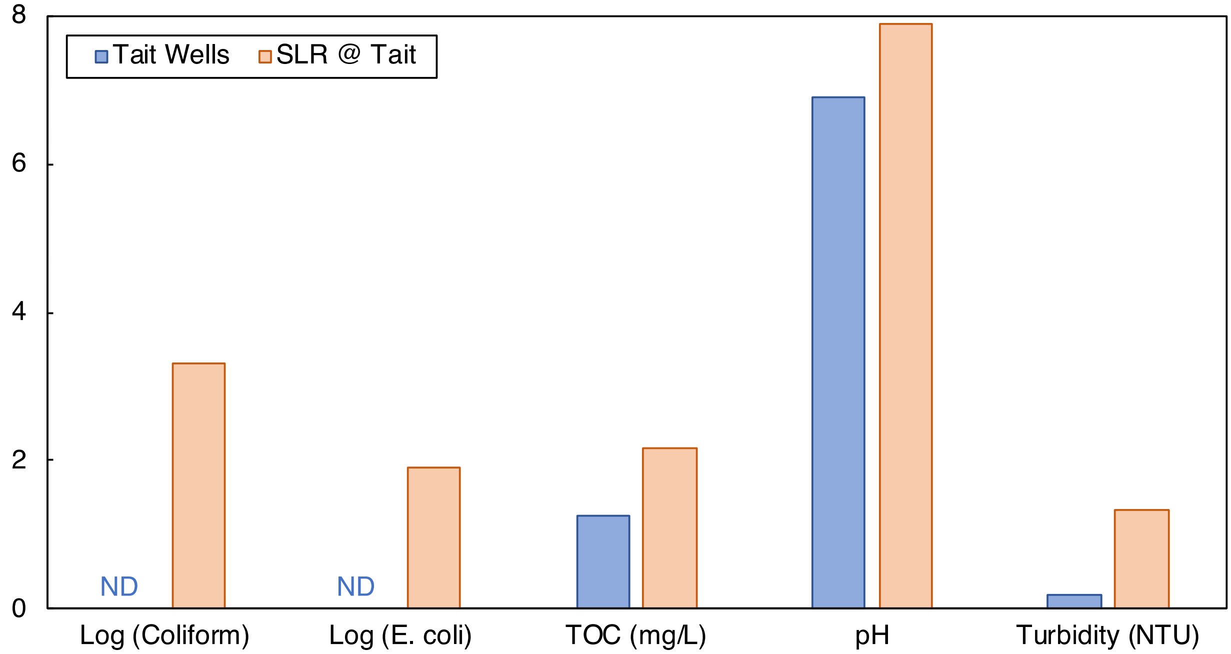

Figure 30: Comparison of the Tait well and SLR (at Tait St.) water quality. Median values for Log(Coliform, MPN/100mL), Log(*E. coli*, MPN, 100mL), TOC, pH, and turbidity are shown.

2.5 Source Water Quality Summary

San Lorenzo River

The SLR generally exhibited larger variations in water quality than the other source waters. The river is highly impacted by winter storms, which can result in up to a thousand-fold increase in turbidity and TSS, as well as a doubling or tripling of TOC. These changes have implications for treatment at GHWTP, as will be discussed in the subsequent section. Winter storms also lead to a reduction in several water quality constituents, including alkalinity, hardness, and many inorganics. The concentrations of microbial indicator organisms and pathogens (total coliform, *E. coli*, and *Cryptosporidium*) show potential microbial contamination in the San Lorenzo River watershed. Ensuring adequate disinfection performance should be a priority at GHWTP, particularly in the context of resolving other treatment issues, including DBP reduction and solids formation.

North Coast Streams

Liddell Spring exhibits the least variation in water quality, which is expected given that it is a natural spring source. It does experience increases in TDS, conductivity, and hardness, providing some buffer against the decrease in these constituents in the other source waters. It also maintains consistently high alkalinity throughout the year, helping to maintain the raw blend alkalinity above the 80 mg/L target for coagulation.

Majors and Laguna Creeks are generally less impacted than the San Lorenzo River by winter storms but do experience variations in several constituents. Both sources experience decreases in alkalinity, conductivity, and hardness, and increases in turbidity, TSS, and TOC during winter storms. The solids concentration and turbidity in these sources remain ten-fold lower than those observed in the San Lorenzo River, and thus are generally not an issue for treatment at GHWTP. High TOC concentrations were observed in Major Creek in the winter of 2016/17 (up to almost 8 mg/L) and the winter of 2017/18 (up to 13 mg/L). However, these levels were rare, and the TOC was below 4 mg/L 90% of the time over the past seven years. The TOC in Laguna Creek spiked to 4.8 mg/L once during the winter of 2017/18 but was below 2 mg/L 95% of the time over the past seven years.

Loch Lomond Reservoir

The Loch Lomond reservoir also experiences less variation in water quality than the San Lorenzo River but does display changes in some parameters. Turbidity and TSS both increased during winter storms but remained well below the concentrations observed in the river (except after the clay landslide in January 2017). The baseline TOC concentration in the reservoir is higher than that in the river—the median reservoir TOC is about 4 mg/L, compared to a median of about 2 mg/L in the river. During storm events, the reservoir TOC does not spike as high as the river, although it does increase slightly. This difference in TOC trends is relevant for DBP formation, as will be discussed in the subsequent section. In the winter, the river may be the largest contributor of TOC, but in the summer, it is the reservoir.

Tait Wells

The Tait Wells experience minimal seasonal variation in water quality, which is expected given that they are groundwater sources. The Tait Wells have lower total coliform, *E. coli*, TOC, and turbidity than SLR, and thus can be used to improve Raw Blend water quality if needed.

3 Treatability Assessment

3.1 Coagulation Performance Assessment

Coagulation performance is dependent on various water quality parameters, including temperature, pH, alkalinity, and hardness. In this section, the impact of the source water quality on coagulation performance at GHWTP will be discussed.

Influence of pH, Temperature and Alkalinity

One of the primary factors affecting alum coagulation performance at GHWTP is pH, given its impacts on the solubility and speciation of alum. The optimal pH for coagulation using alum is about 6 in the summer and 7 in the winter (based on seasonal differences in temperature) (Crittenden et al. 2012). To optimize coagulation, pH adjustment is frequently employed. The alkalinity of the water then becomes important given that it is more difficult to adjust pH when the alkalinity is high. Conversely, low alkalinity waters are easier to adjust, but offer less pH buffering capacity, which allows for more fluctuations in pH. Alum consumes alkalinity and decreases pH when it is added to water; however, the pH decrease is buffered by the alkalinity until it is depleted.

The pH of the Raw Blend fluctuated between 7 and 8.1 for the duration of the sampling period, decreasing slightly with the increased river flow (stronger storms) in the winter of 2016/2017 (see Figure 31). Winter storms rarely caused the pH of individual source waters to move outside of the typical range of the raw blend pH. Accordingly, treating winter water is not expected to result in large changes to the raw blend pH and thus will not likely require additional considerations with respect to pH adjustment.

Acidified alum is used at GHWTP to lower the pH closer to the optimal range for coagulation. The pH range of settled water is 6.5 – 7.5. Using the acidified alum for pH adjustment reduces the operational flexibility to adjust pH independently of the coagulant dose. Optimizing pH adjustment and coagulant dose could potentially allow for reductions in solids production in certain scenarios (e.g., if coagulant is being over-dosed to achieve pH reduction). Further evaluation of the benefits of decoupling pH adjustment and coagulant addition is recommended.

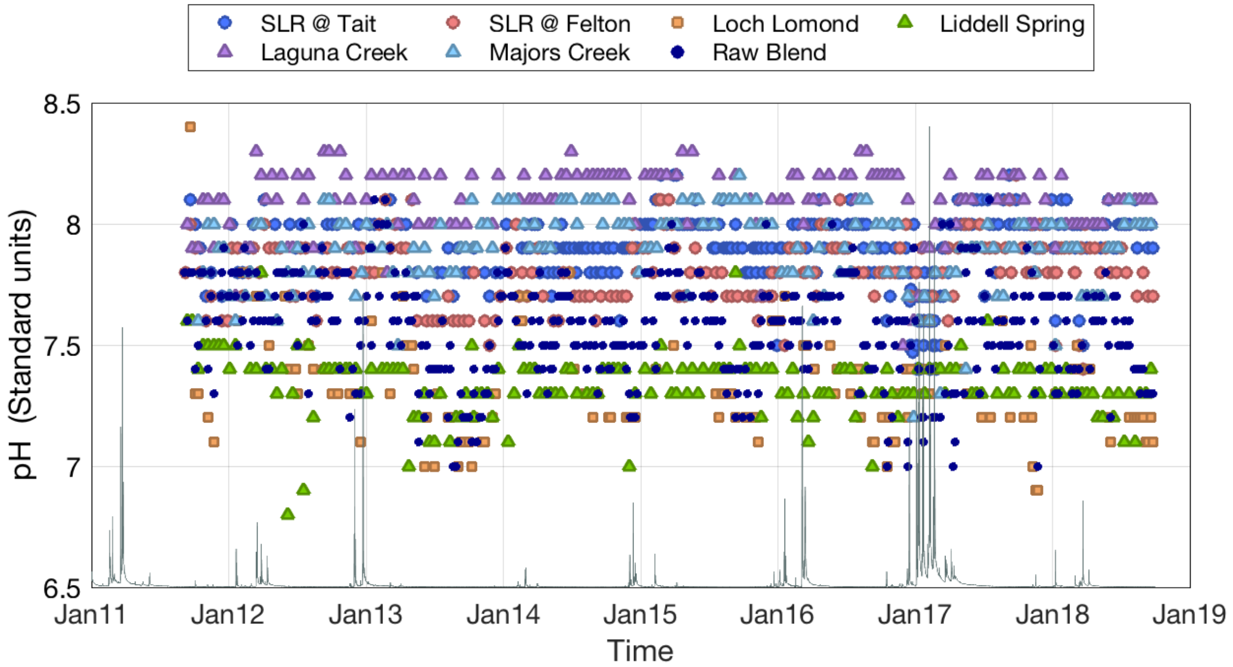

Figure 31: pH of the source waters and Raw Blend between September 2011 and September 2018.

Influence of Ion Concentrations

The ionic composition of the water, measured in terms of hardness, conductivity, and TDS, may also affect coagulation. A higher ion concentration increases coagulation efficiency because it decreases the repulsion of the colloids in the water, thereby aiding in the attachment of colloids to one another and the formation of flocs. The variables influencing ionic composition, i.e., hardness, conductivity, and TDS, fluctuate seasonally and decrease with winter storms. Therefore, coagulation could be more difficult in the winter, which would be problematic because turbidity is highest in the winter.

Enhanced Coagulation Requirements

The Stage 1 Disinfection and Disinfection Byproducts Rule (D/DBP Rule) was promulgated to minimize disinfection byproducts in finished water. One of the ways that this goal is accomplished is by requiring the removal of TOC, a DBP precursor indicator. The treatment technique is referred to as ‘enhanced coagulation’, and the required amount of TOC removal is based on the alkalinity and TOC of the source water (refer to Table 7).

Table 7: TOC Removal Required by Stage 1 D/DBP Rule

Source Water TOC (mg/L)	Source Water Alkalinity (mg/L as $CaCO_3$)		
	0 – 60	>60 – 120	>120
>2.0 – 4.0	35%	25%	15%
>4.0 – 8.0	45%	35%	25%
>8.0	50%	40%	30%

During the time period analyzed, the TOC of the Raw Blend was below 4 mg/L in about 95% of the samples (see Figure 8) and below 8 mg/L in all of the samples. The alkalinity of the Raw Blend (see Figure 6) was greater than 120 mg/L as $CaCO_3$ in about 80% of the samples. Based on these concentrations, the required TOC removal is likely to be 15% most of the time, but the exact removal requirements can change as the raw water quality changes.

In order to calculate the specific TOC removal required, the TOC and alkalinity of the source water and the TOC of the finished water must be measured monthly for at least a year (EPA 1999). The required TOC removal is then determined on an ongoing basis using the source water TOC and alkalinity for each month. The actual TOC removal is also calculated for each month (average in the case that there are multiple measurements). The actual percent removal is then divided by the required removal, yielding a ratio of the actual:required removals with the goal of obtaining a value of one or greater. Compliance is determined as a running annual average (RAA), which is accomplished by averaging the most recent 12 months of actual:required ratios. The RAA is calculated quarterly after the first year of measurements. Compliance is achieved if this RAA is greater than or equal to one, which has been the case for GHWTP extending back to June of 2014 (the duration for which TOC grab sample data was analyzed).

Table 8 shows the compliance data for the months for which data were provided. Compliance with the enhanced coagulation regulations of the Stage 1 D/DBP Rule is not expected to be an issue if additional San Lorenzo River winter flows are treated. The TOC of the raw blend may increase during the winter with additional river flow; during storm events, TOC values above 4 mg/L were routinely measured (see Figure 7). These increases in TOC resulted in three months during the winter of 2016/17 when the plant had an actual:required removal ratio less than one. However, because compliance is based on a running annual average, GHWTP remained within compliance because of the averaging provided by months where the TOC removal achieved was much higher than what is required (up to 2.9 times higher – see Table 8). The highest TOC values observed in SLR were during first flush events; if these flows are avoided, future compliance with the enhanced coagulation requirements should not be an issue for GHWTP.

Table 8: Compliance with the Enhanced Coagulation Regulation of the Stage 1 D/DBP Rule from 2014 to 2018. Required TOC removals were determined using monthly average alkalinity and TOC grab sample data. TOC removals were calculated from operator paired raw and effluent TOC grab samples.

Month	Required TOC Removal (%)	Actual TOC Removal (%)	Monthly Actual:Required Ratio	RAA Actual:Required Ratio
Jul 2014	15	30	2.0	--
Aug 2014	15	39	2.6	--
Sept 2014	15	30	2.0	--
Oct 2014	15	21	1.4	--
Nov 2014	15	32	2.1	--
Dec 2014	15	33	2.2	--
Jan 2015	15	31	2.1	--
Feb 2015	15	36	2.4	--
Mar 2015	15	37	2.5	--
April 2015	15	43	2.9	--
May 2015	15	33	2.2	--
Jun 2015	15	32	2.1	2.2
Jul 2015	15	31	2.1	--
Aug 2015	15	34	2.3	--
Sept 2015	15	29	1.9	2.2
Oct 2015	--*	--	--	--
Nov 2015	15	33	2.2	--
Dec 2015	15	35	2.3	2.3
Jan 2016	25	31	1.3	--
Feb 2016	--^	32	--	--
Mar 2016	15	35	2.3	2.2
April 2016	15	43	2.9	--
May 2016	15	34	2.3	--
Jun 2016	15	35	2.4	2.2
Jul 2016	15	34	2.3	--
Aug 2016	15	28	1.9	--
Sept 2016	15	27	1.8	2.1
Oct 2016	15	10	0.7	--
Nov 2016	15	37	2.4	--
Dec 2016	15	25	1.6	2.0
Jan 2017	25	24	0.9	--
Feb 2017	25	22	0.9	--
Mar 2017	25	33	1.3	1.8
Apr 2017	25	23	0.9	--
May 2017	25	36	1.4	--

Month	Required TOC Removal (%)	Actual TOC Removal (%)	Monthly Actual:Required Ratio	RAA Actual:Required Ratio
Jun 2017	25	42	1.7	1.5
Jul 2017	15	33	2.2	--
Aug 2017	15	26	1.7	--
Sep 2017	15	28	1.9	1.5
Oct 2017	25	27	1.1	--
Nov 2017	15	27	1.8	--
Dec 2017	15	32	2.1	1.5
Jan 2018	25	35	1.4	--
Feb 2018	15	27	1.8	--
Mar 2018	25	36	1.4	1.6
Apr 2018	15	33	2.2	--
May 2018	15	32	2.1	--
Jun 2018	15	30	2.0	1.8
Jul 2018	15	30	2.0	--
Aug 2018	15	30	2.0	--

*no finished water TOC data
^no source water alkalinity data

3.2 Solids Formation Analysis

GHWTP has three limits on solids discharged to the sewer:

- Concentration: 3,000 mg/L wet solids
- Flow: 50 gpm
- Total solids: 2,085 lbs/day dry solids

The high solids loading in winter water flows from the San Lorenzo River poses a challenge to meeting these solids limits. The relationship between turbidity and TSS in the SLR (based on 2016/17 data) is presented in Figure 32. When the whole dataset is used, the correlation is dominated by the rare, high-turbidity events. Because we are more interested in understanding the relationship between TSS and turbidity in the range of water quality likely to be treated at the GHWTP, an additional correlation was developed for turbidities less than 100 NTU. This correlation is compared with those of the other source waters in Figure 33. As shown, for a given turbidity, the SLR is likely to contain higher suspended solids concentrations than the other sources. In other words, the light-scattering characteristics of the particulates shows some variation across source waters.

Currently, if the SLR exceeds a turbidity 10 NTU, the water will not be treated at the GHWTP. However, there is interest in raising this limit to be able to treat more winter water SLR flows at the GHWTP. To assess the impact of raising this turbidity limit, the correlation between TSS and turbidity below 100 NTU in the San Lorenzo River was used to estimate solids production from treating SLR water with different turbidities. The results for twelve scenarios are shown in Table

9; these include four turbidity levels (10, 20, 50, and 100 NTU), and three flow rates (4, 5, and 8 mgd).

In addition to the predicted solids production, the amount of additional flow that could be taken from the SLR for the different turbidity levels was also estimated, using probability plots of SLR turbidity and flow (Figure 34 and Figure 35). As shown, the turbidity observed in the river varies from year to year. In WY 2017, which had an exceptionally wet winter, higher turbidities were observed for more of the year. In contrast, the other three years shown (2013-2015) exhibit relatively similar turbidity levels. Thus, the potential to treat additional winter flow by increasing the maximum treatable turbidity limit will vary over time. To estimate this flow "gain," the turbidity limits were translated into river flow rate limits based on a correlation between turbidity and flow; the corresponding flow limits are shown in Figure 35. As an estimate, increasing the turbidity limit to 20 NTU would result in up to a 5% potential increase in treatable river flows; increasing it to 50 NTU could increase the availability by up to 20% in wet years, and up to 7% in drier years; and increasing it to 100 NTU could increase availability by up to 30% in a wet year and 9% in a dry year.

Increasing the turbidity limit would come with a corresponding increase in solids production. The values shown in Table 9 are for solids coming from the river water TSS, the addition of alum coagulant (at an assumed dose of 70 mg/L), and the addition of polymer (at an assumed dose of 3 mg/L). The sludge mass was estimated using the following relationships: 0.33 kg dry sludge/kg alum coagulant, and 1 kg dry sludge/kg polymer (Crittenden et al., 2012). The solids production estimates for the higher turbidity thresholds are likely an underestimate given the fact that higher coagulant and polymer doses may be needed to achieve sufficient turbidity (and TOC) reduction for these conditions.

Under all three flow scenarios, treating water up to 100 NTU will result in exceedances of the solids discharge limit of 2,035 lb/day. Increasing the turbidity limit to 20 NTU may be possible at low flows; additional jar testing is recommended to better clarify the necessary coagulant dose to treat water of this turbidity and thus predict the total solids that would be produced. Depending on the desired increase in flow to be treated from the San Lorenzo River in the winter, an evaluation of additional solids handling alternatives is recommended to manage the higher solids production from this source water.

Table 9. Predicted solids formation and potential flow gains from treating San Lorenzo River water of varying turbidity. Bold, italicized values exceed the discharge limit of 2,035 lb/day. Wet year flow gain estimates are based on Water Year 2017; dry year estimates are based on WY 2013-2015.

SLR Turbidity (NTU)	% Flow Gain	Assumed Flow (mgd)	Solids Production			Total Solids (lb/day)
			TSS (lb/day)	Alum, 70 mg/L (lb/day)	Polymer, 3 mg/L (lb/day)	
10	0%	4	545	770	100	1,414
		5	681	962	125	1,768
		8	1,090	1,539	200	***2,829***
20	5% (wet year) 2-5% (dry year)	4	1,105	770	100	1,975
		5	1,382	962	125	***2,469***
		8	2,211	1,539	200	***3,950***
50	20% (wet year) 2-7% (dry year)	4	2,786	770	100	***3,656***
		5	3,483	962	125	***4,570***
		8	5,573	1,539	200	***7,312***
100	30% (wet year) 3-9% (dry year)	4	5,588	770	100	***6,458***
		5	6,985	962	125	***8,072***
		8	11,177	1,539	200	***12,916***

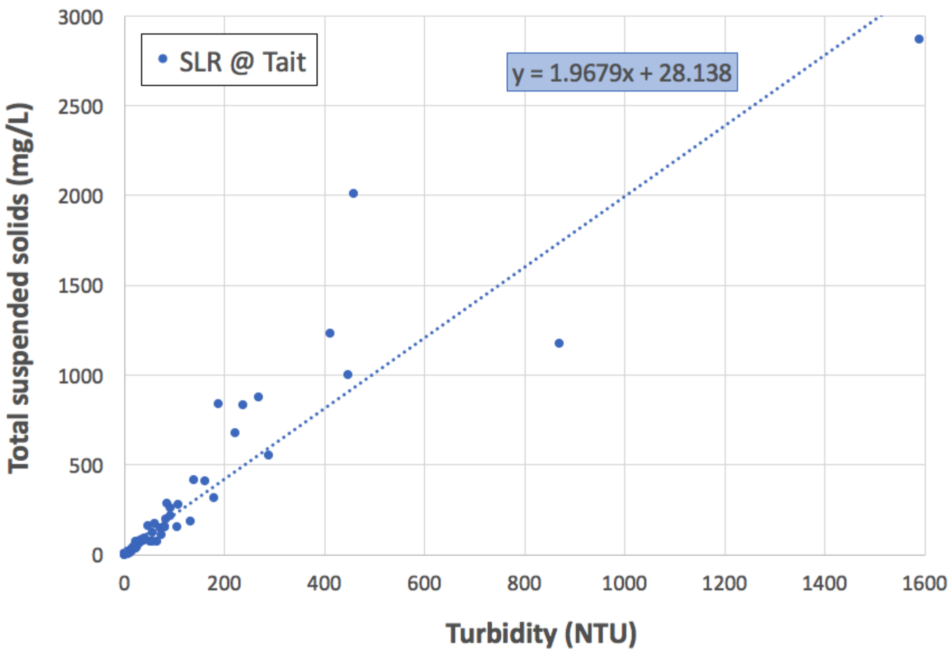

Figure 32. Correlation between turbidity and total suspended solids in San Lorenzo River water from the Tait Street Diversion. Turbidity data are grab samples collected between October 2016 and September 2017; in-stream turbidimeter data are not included.

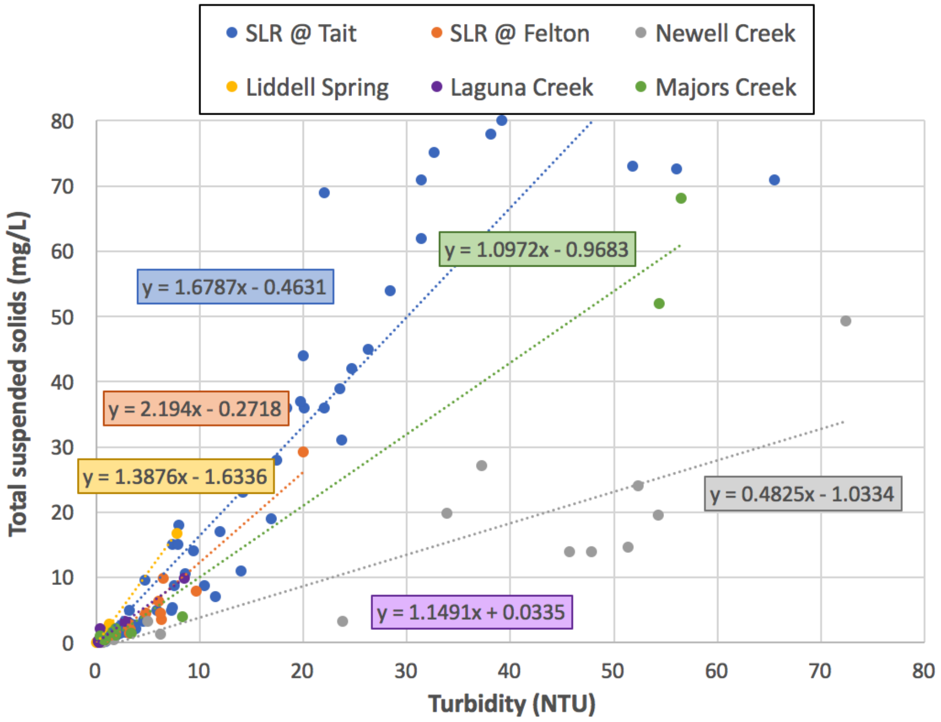

Figure 33. Correlation between turbidity and total suspended solids in all GHWTP surface water sources. The correlation for San Lorenzo River at Tait Street includes only data for turbidity less than 100 NTU. All data are grab samples collected between October 2016 and September 2017.

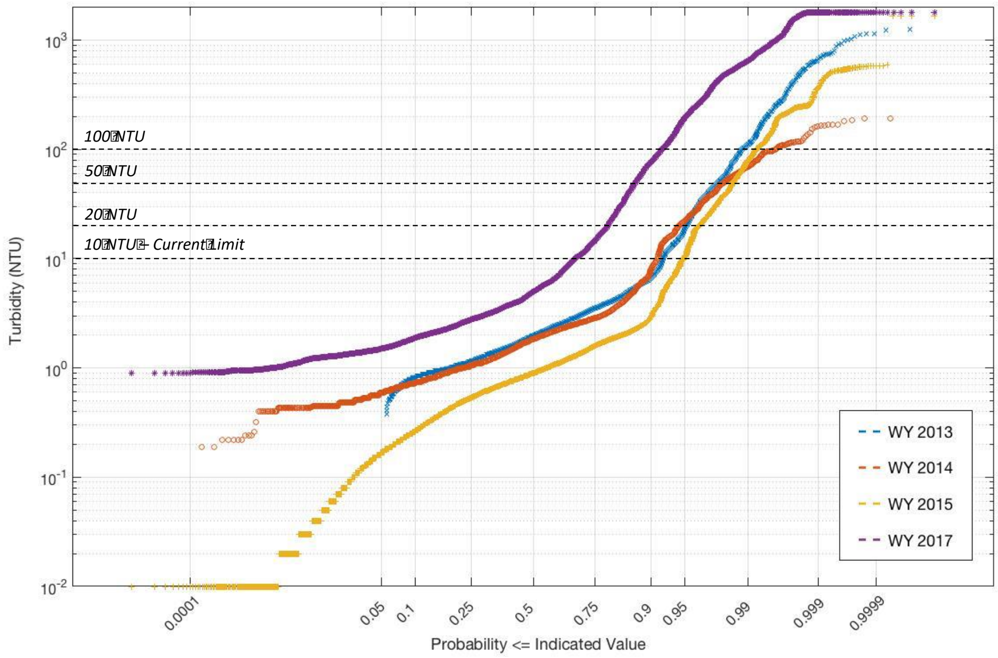

Figure 34. Probability plot of turbidity in the San Lorenzo River at Tait Street. Data are from in-stream turbidimeter at 5-minute increments. Three turbidity limits are shown, along with the corresponding percent of flow that would be available to take at the GHWTP.

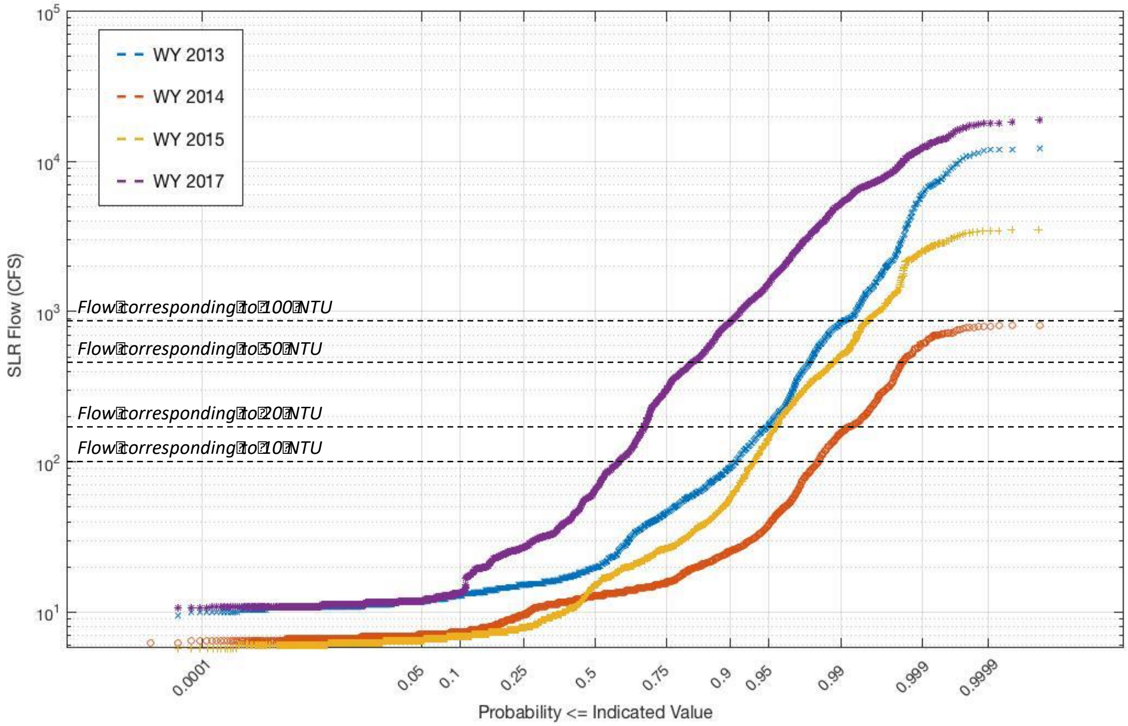

Figure 35. Probability plot of flow in San Lorenzo River for multiple water years (WY). Also shown are the flows corresponding to multiple potential turbidity cutoffs. River flow data obtained from the USGS stream gage at Big Trees (USGS gage number 11160500)

4 Disinfection Byproduct Formation Analysis

Disinfection byproduct formation is a critical issue for the GHWTP. Currently, chlorine is dosed prior to flocculation and the sedimentation basins are used to achieve the majority of the chlorine contact time, as the facility was not designed with a separate chlorine contact basin. This strategy results in extended contact times (i.e., longer than the minimum values needed for CT compliance) between chlorine and DBP precursors, leading to high DBP formation. The goal of this analysis is to assess how key parameters impact DBP formation, develop a predictive model that can allow for better DBP control, and evaluate potential mitigation strategies with the available data.

4.1 Relationship of Key Parameters with DBP Formation

GHWTP must comply with all DBP MCLs, including meeting the limits of 80 μg/L for total trihalomethanes (TTHMs) and 60 μg/L for haloacetic acids (HAA5). Samples must be collected four times per quarter, with at least 25% of samples each quarter representing maximum residence times in the distribution system (Title 22 CCR). A locational running annual average (LRAA) value for both TTHM and HAA5 is used to determine compliance with the MCLs. The LRAA is computed quarterly using the previous four quarters of data; each quarter's data is first averaged, and then the four quarterly values are averaged to obtain the LRAA.

TOC and THMs

Because organic carbon species can act as DBP precursors, DBP formation is often strongly correlated with TOC levels (Crittenden et al. 2012). The relationship between raw blend TOC concentrations and THM formation through the sedimentation basins is illustrated in Figure 36. These two variables track well in the winter, as can be seen from the period from October 2016 to March 2017. During the summer months, however, the TOC is lower on average and does not track as well with THM formation. The same general trends are observed for Water Year 2018: TOC and THM track well from November 2017 to April 2018 and do not track as well during the summer months. These observations demonstrate that while TOC is an important factor for considering THM formation, it is not the only factor.

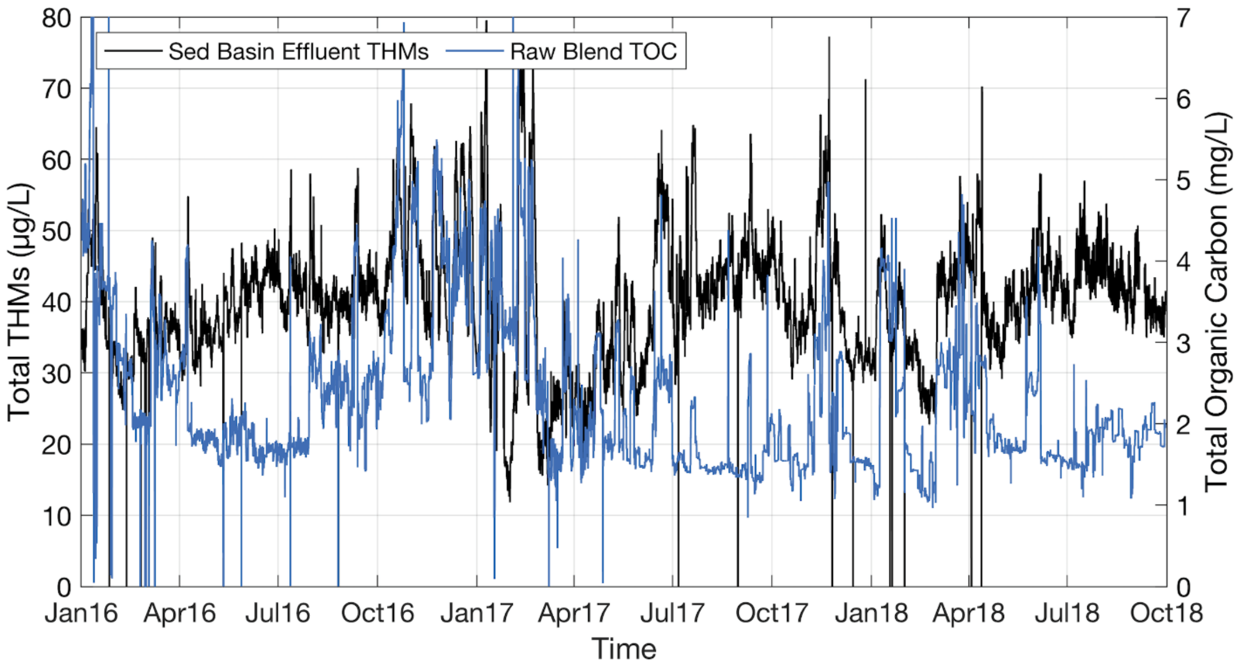

Figure 36. Relationship between raw blend total organic carbon and trihalomethane formation through the sedimentation basins. THM data are taken at hourly increments from the SCADA system; measurement occurs at the end of the sedimentation basin. Raw blend TOC data were available only periodically; the data presented here are calculated based on the flow contribution of each source water and the closest available TOC concentration data point.

Several other water quality parameters are also relevant for THM formation, including temperature, pH, and specific UV absorbance (SUVA). To evaluate the impact of these water quality parameters on THM formation, each parameter was plotted versus the ratio of THM formation to raw blend TOC (THM:TOC). This comparison allows for the identification of factors that could cause the THM:TOC ratio to differ over time (rather than remain constant, as would be expected if TOC were the only important contributing factor).

SUVA and THMs

One hypothesis for the divergence between the TOC and THM profiles in the summer is that the *quality* of the organic matter in the summer is different and more likely to form DBPs. One variable that provides insight into the quality of the organic matter is SUVA, i.e., the UV absorbance at 254 nm divided by DOC. SUVA provides a measure of the fraction of the total organics that are aromatic in nature. Because aromatic organics are generally more reactive than other organics, the aromaticity of a water can frequently be correlated with the extent of THM formation. For example, for two waters with the same TOC concentration, the water with the higher SUVA would be expected to form higher concentrations of DBPs. SUVA data for the raw blend is available between October 2016 and September 2018. These data show that the raw blend SUVA generally remains between 2.5 and 3.5 L/mg-M during both the winter and summer, indicating that the quality of the TOC in the water is consistent across the year. Because SUVA does not increase as the ratio of THM:TOC increases, the changes in the

THM:TOC ratio do not appear to be a function of the *quality* (i.e., aromaticity) of the raw blend TOC (Figure 37).

pH and THMs

Within the peer-reviewed literature, it has been widely reported that pH plays an important role in THM formation. Nevertheless, there was no clear trend between the settled water pH and THM:TOC (Figure 38). One explanation for this apparent lack of correlation was that the pH did not vary enough to observe a trend (the pH is controlled between a narrow range of 6.7 and 7.3).

Temperature and THMs

In contrast, the raw blend temperature tracks well with the THM:TOC ratio in both Water Year 2017 and 2018: as temperature increases, so does the THM:TOC. This trend helps explain why the concentration of THMs is still high in the summer even though the TOC is generally lower in the summer.

The comparison of SUVA and temperature to THM:TOC demonstrates that the concentration of TOC and the temperature are more important factors to THM formation than the quality of the TOC. Thus, the relationships and trends observed are independent of the composition of source waters making up the raw blend.

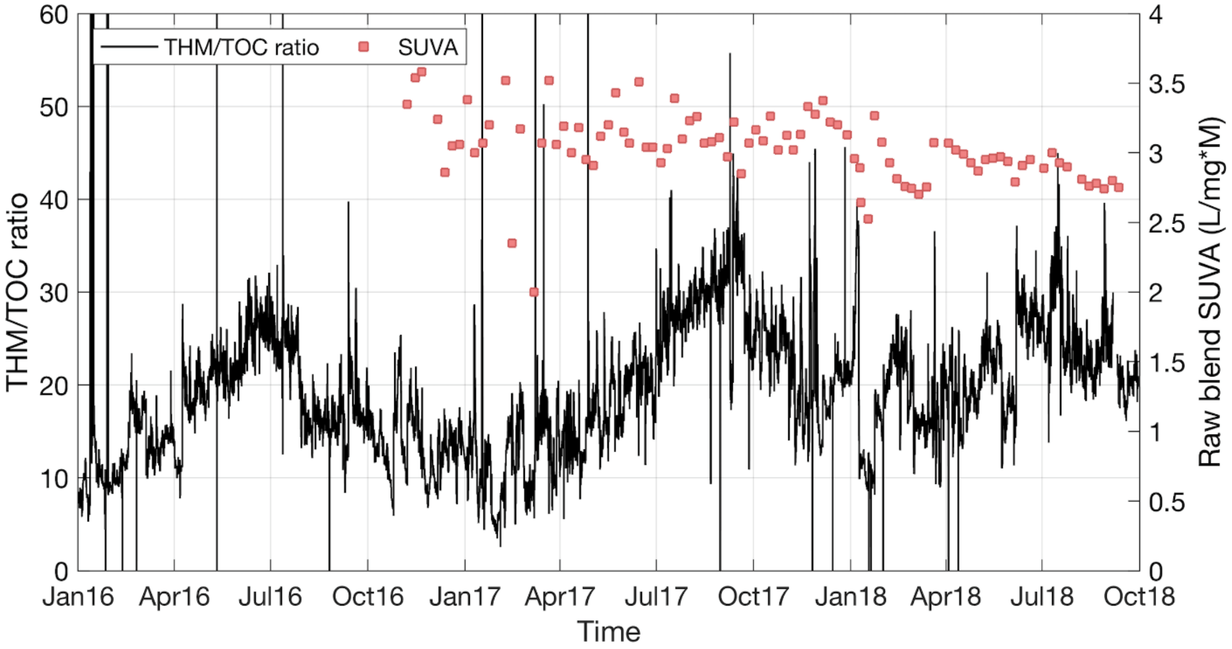

Figure 37. Comparison of raw blend specific UV absorbance (SUVA) with the THM to TOC ratio. The THM/TOC ratio is calculated on an hourly basis with THM data from the SCADA system and a calculated raw blend TOC. SUVA values are calculated from DOC and UV254 absorbance grab sample data. No data were available prior to November 2016.

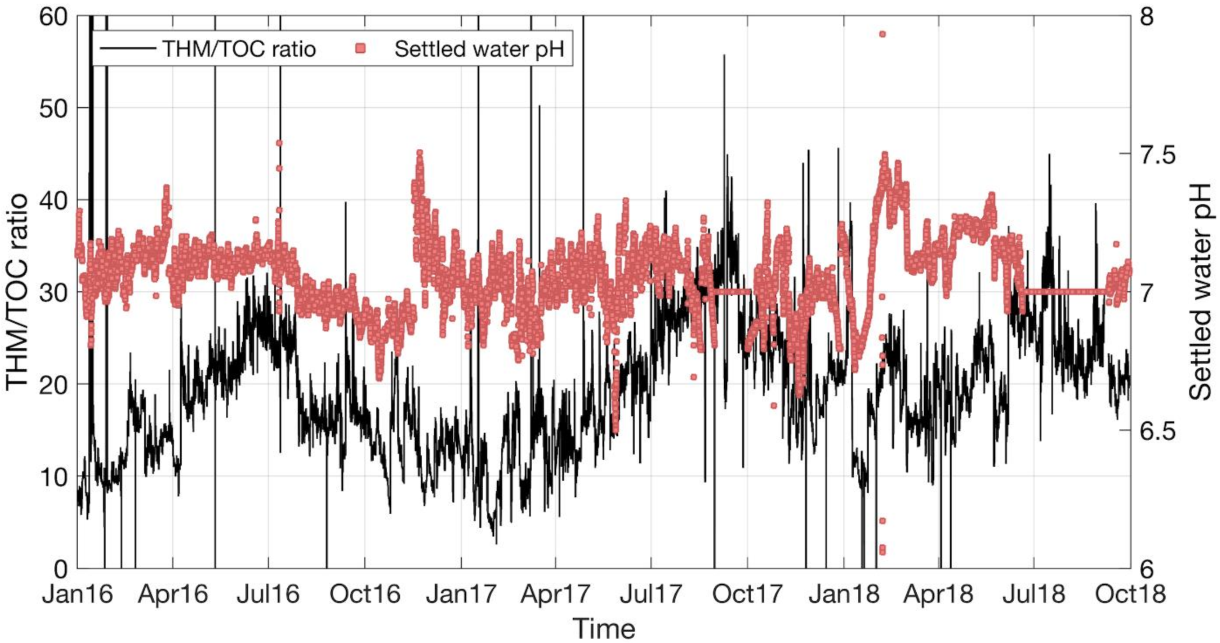

Figure 38. Comparison of settled water pH to the THM to TOC ratio. The THM/TOC ratio is calculated on an hourly basis with THM data from the SCADA system and a calculated raw blend TOC. Settled water pH data are taken at hourly increments from the SCADA system.

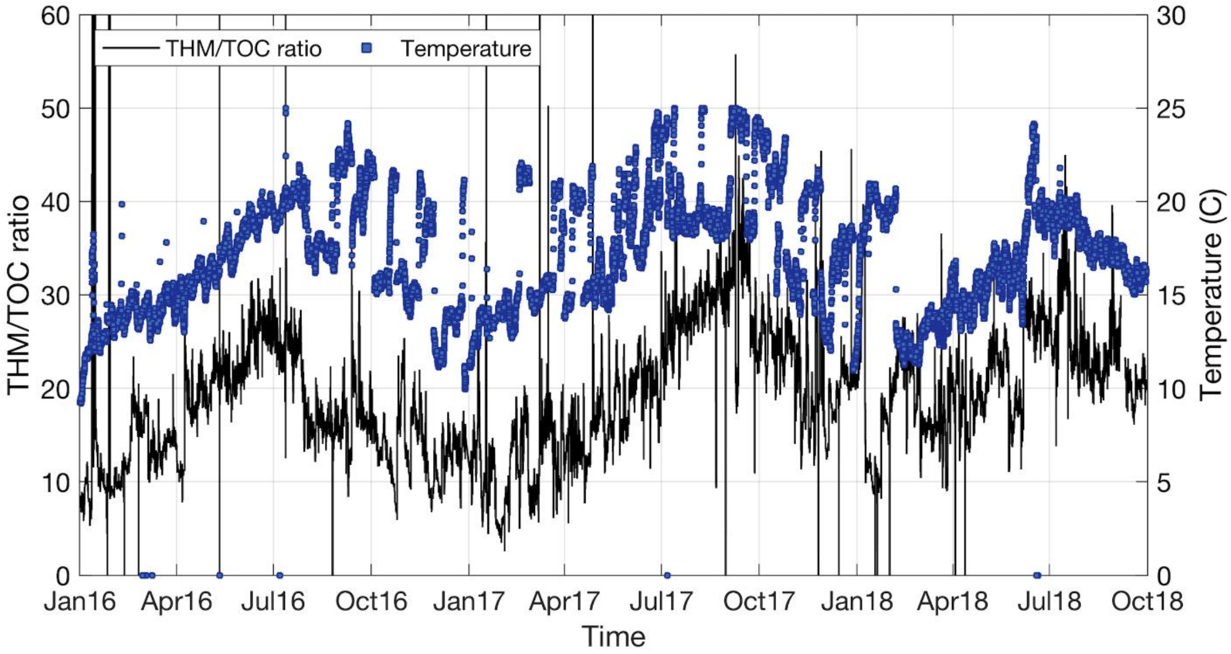

Figure 39. Comparison of raw blend temperature with THM to TOC ratio. The THM/TOC ratio is calculated on an hourly basis with THM data from the SCADA system and a calculated raw blend TOC. Temperature data are grab samples collected during routine monitoring.

4.2 DBP Formation Model – Current Plant Treatment

A predictive model was developed for THM formation as a function of several variables. The model was adapted from Crittenden et al. (2012). The variables incorporated in the model are TOC, temperature, pH, chlorine consumed, and chlorine residual (i.e., all of the relevant variables for which SCADA data were available). The model was calibrated using two data sets: (1) last year's data alone (i.e., January 2016 – September 2017) and (2) the last two year's data (i.e., January 2016 – September 2018). The resulting models are described by the following equations:

1-Yr THM model (January 2016 – September 2017 data) **(Eqn. 1)**:

$$\log(THM) = 0.36 + 0.31\log(TOC) + 0.02\,Temp + 0.12\,pH + 0.03\log(Cl_{res}) + 0.05\log(Cl_{cons})$$

2-Yr THM model (January 2016 – September 2018 data) **(Eqn. 2)**:

$$\log(THM) = 1.15 + 0.26\log(TOC) + 0.01\,Temp + 0.03\,pH + 0.14\log(Cl_{res}) + 0.08\log(Cl_{cons})$$

where THM is the sedimentation basin effluent THM concentration (µg/L), TOC is the raw blend TOC (mg/L), Temp is the raw blend temp (˚C), pH is the settled water pH, Cl_{res} is the sedimentation basin chlorine residual (mg/L), and Cl_{cons} is the amount of chlorine consumed, i.e., the difference between the chlorine dose and residual concentration (mg/L).

While the equations for the two models have slightly different coefficients, they predict similar concentrations of THM (Figure 40). In general, the models track the measured THMs closely, though they are less accurate at predicting the more extreme values such as those that occur during winter storms. This difference likely stems from the fact that there are fewer of the extreme values and therefore their weight in the calibration is less than that of the more moderate values. The 1-Yr model appears to be more accurate at predicting peak values but generally overestimates THM formation during non-extreme times. In contrast, the 2-Yr model does a better job at predicting the THM formation during non-extreme times, but, compared to the 1-Yr model, under-predicts the peak values. The difference between the two models may be due to the fact that the winter of the 2017 Water Year had many large storms, leading to more extreme values for TOC and THMs; these values likely carried more weight in the model calibration. In comparison, the winter of WY 2018 had comparatively few storms, thus "diluting" the extreme values such that their contribution to the calibration was less significant.

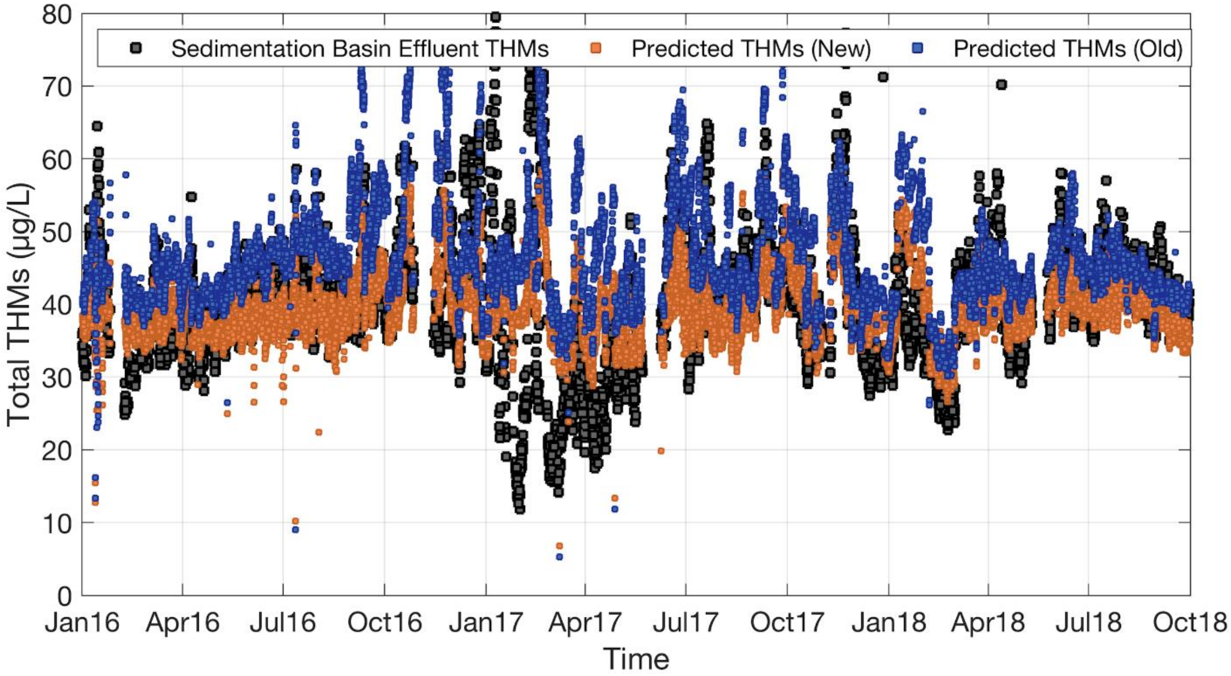

Figure 40. Comparison of THM predictions from the 1-Yr and 2-Yr model to measured THM concentration in the sedimentation basin effluent.

Both the 1-Yr and 2-Yr models for predicting sedimentation basin effluent DBPs show that TOC is a dominant parameter. Therefore, controlling the raw blend TOC should be an effective strategy to control DBP formation in the current operational configuration. This model may be used (1) to determine how low the TOC must be in order to meet a THM goal under different conditions, or, (2) to predict the impact of lowering the TOC via modification of a treatment process.

Application of DBP Model

For a particular DBP goal, a raw blend TOC limit can be developed using the model. One common goal is to maintain THM levels at 80% or less of the MCL (64 µg/L) at terminal locations in the distribution system. Given that DBPs will continue to form after leaving the treatment plant, a relationship between the treatment plant values and the distribution system values must be developed. Figure 41 shows the relationship between sedimentation basin effluent THMs and an experimental estimate of THM formation potential at a location in the distribution system with a 6-day water age[3]. The data indicate that THMs approximately double in the distribution system at terminal locations. Consequently, the sedimentation basin effluent THMs would need to be maintained at or below about 30 µg/L in order to meet the distribution system goal.

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[3] THM formation potential estimates are made with an Aquametrology THM meter by heating a water sample and holding it for a specified amount of time to simulate THM concentrations at a terminal location in the distribution system.

To determine the raw blend TOC that would be needed to maintain THMs below the threshold of 30 µg/L leaving the plant, we back calculated a TOC value from the model, using average values for the other model inputs. Because of the importance of temperature on THM formation, we estimated a TOC limit for both winter (15°C) and summer (19°C) based on the average temperatures of those seasons.

To maintain sedimentation basin effluent THMs at or below 30 µg/L, the raw blend TOC would need to remain below about 0.6 mg/L in the summer (April – September) and 0.8 mg/L in the winter (October – March). As shown in Figure 42, these values will be hard to maintain consistently; this limit was met in the summer about 0.1% of the time, and in the winter about 0.5% of the time. Thus, under the current operational strategy of chlorinating in the sedimentation basins, it will be difficult to consistently meet the goal of 80% of the THM MCL at terminal locations in the distribution system. This issue will likely be made worse if additional winter San Lorenzo River flows are treated because of the higher TOC of this source.

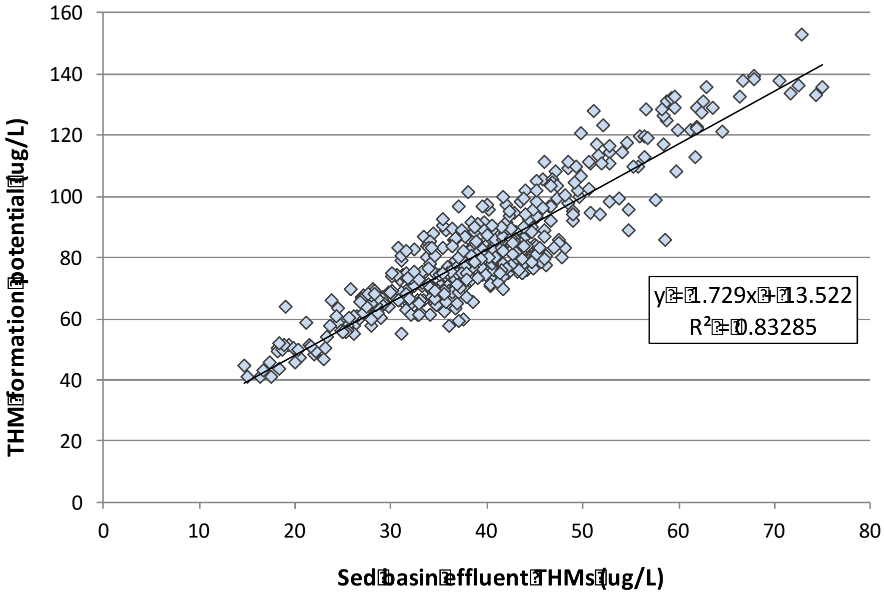

Figure 41. Correlation between sedimentation basin effluent THMs and estimated THM formation potential in the distribution system. THM formation potential is estimated with an Aquametrology THM meter using an experimental method to simulate THM formation in the distribution system.

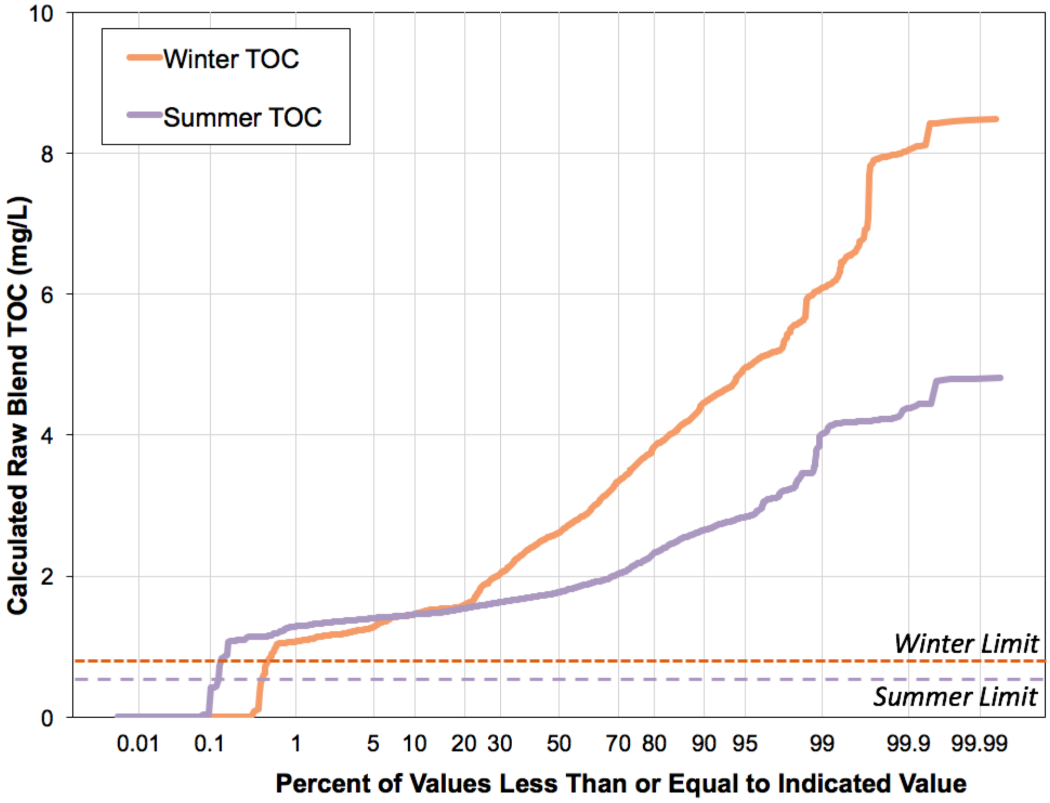

Figure 42. Probability plot of raw blend TOC in the winter (October – March) and summer (April – September) months. Dashed lines show estimated limits for maintaining sedimentation basin effluent THMs at or below 30 µg/L. TOC data are a combination of routine grab samples and operator grab samples collected between September 2011 and September 2017.

4.3 DBP Mitigation Strategies

DBP formation is an issue that GHWTP will continue to face, whether or not additional winter water flows are treated. Under the current operational strategy (pre-chlorination), the chlorine is exposed to DBP precursors for the duration of the residence time in the sedimentation basin, which promotes higher levels of DBP formation. One potential strategy for mitigating DBP formation is moving the chlorine dosing location downstream of the sedimentation basins to take advantage of the TOC removal that occurs through the flocculation and sedimentation processes. Chlorine dosed after the sedimentation basins would react with lower concentrations of TOC and potentially lead to lower DBP levels. An evaluation of post-chlorination strategies is discussed in the following section.

Post-Chlorination with Enhanced Coagulation

One strategy to optimize TOC removal prior to chlorine dosing is to use enhanced coagulation coupled with post-chlorination for CT compliance. Trussell Tech and West Yost conducted bench-scale jar testing at the GHWTP in March 2018 to evaluate the benefits of an enhanced coagulation strategy for THM formation (full technical memo found in Appendix B). Four source waters were evaluated: San Lorenzo River (high turbidity), San Lorenzo River (low turbidity), Raw Blend, and Loch Lomond. The goals of the tests were to (1) determine the optimal pH and coagulant dose for enhanced coagulation, and (2) evaluate the impact of the chlorination strategy on THM formation.

Enhanced coagulation involves lowering the pH to improve TOC removal during flocculation and sedimentation. Jar testing was used to determine the optimal pH and coagulant dose for enhanced coagulation. These tests involved a single pre-chlorinated jar as a control for THM formation comparison, with the remaining jars all receiving post-chlorination only. In addition, simulated distribution system (SDS) testing was performed for all jars, where samples were held for 7 days to simulate DBP formation in the distribution system.

To determine the optimal pH for enhanced coagulation, the coagulant dose was held constant while the pH was varied. To determine the optimal coagulant dose for enhanced coagulation, the pH was held constant at the optimal value while the coagulant dose was varied. The optimal pH was chosen to maximize DOC removal while minimizing chemical usage. The optimal coagulant dose was chosen based on achieving effluent turbidity <2 NTU, DOC reduction > 40%, and 7-day TTHM formation below 80% of the MCL, while minimizing chemical usage. The optimal conditions identified from the jar testing are summarized in Table 10.

The jar testing showed that compliance with the TTHM MCL could be achieved by lowering the pH below 6.7 and using an alum dose of 53-73 mg/L, along with a post-chlorination strategy. Post-chlorination resulted in up to 14% less TTHM formation than pre-chlorination after 7-days of hold time (all other conditions held constant). Newell Creek was the most challenging of the source waters, with TTHM concentrations between 80 and 100% of the MCL after 7 days.

Table 10. Summary of jar test results for enhanced coagulation optimization.

Source Water	Optimal Conditions		Raw Water Quality		Jar Test Performance				
	pH	Alum dose (mg/L)	Raw Turbidity (NTU)	Raw DOC (mg/L)	DOC Removal (%)	Settled Turbidity (NTU)	Settled DOC (mg/L)	TTHM (ug/L) (0 day)	TTHM (ug/L) (7 day)
San Lorenzo River	6.7	53	37.1	4.4	53	1.13	2.07	18	38
San Lorenzo River	6.6	53	5.27	2.94	43	1.67	1.68	16	55
Plant Blend	6.4	53	1.88	2.83	36	0.59	1.92	17	41
Newell Creek	6.5	73	2.66	4.37	44	1.09	2.44	21	74

*Results summarized in this table assume a post-chlorination strategy

Alternative Treatment Technologies

Additional jar testing was conducted with HDR at Trussell Tech's lab facility in Pasadena, CA to evaluate the benefits of alternative treatment technologies for DBP formation, and to generate data for a DBP formation model based on post-chlorination (full technical memo found in

Appendix C). The treatment technologies were selected to evaluate a range of potential mitigation strategies.

- Enhanced coagulation: used to optimize TOC removal upstream of a post-chlorination approach
- ACTIFLO Carb: an alternative technology that may provide higher TOC removal through the use of both (a) enhanced coagulation and (b) activated carbon upstream of a post-chlorination strategy
- Pre- and intermediate ozonation: an oxidation and disinfection process used to transform the quality of the TOC making it either (a) easier to remove through downstream processes or (b) less likely to form DBPs upon subsequent chlorine addition
- UV disinfection: an alternative disinfection process that does not lead to THM formation

Data from both the HDR and West Yost jar tests were collected and used to develop a DBP formation model to evaluate the benefits of the mitigation strategies.

4.4 DBP Formation Model – Post-Chlorination

Results from the enhanced coagulation jar testing conducted at GHWTP, and the additional testing conducted at Trussell Tech's lab facility, were used to develop site-specific DBP formation models. The DBP formation model and subsequent analysis lead to preliminary recommendations for future treatment train modifications. Models were developed for THM and HAA formation at both (a) the plant and (b) in the distribution system. To predict the total DBP formation at the end of the distribution system, the plant and distribution system models were calculated and added together.

The plant THM model predicts formation based on water temperature, alkalinity, DOC concentration, UV absorbance, chlorine consumed, pH, chlorine residual. Formation in the distribution system includes a time component. The plant HAA model predicts the formation of HAA based on DOC concentration and chlorine consumed; the distribution system model also includes a time component. The resulting models are described by the following equations:

THM formation at the plant **(Eqn. 1)**:

$$log(THM) = 0.429 + 0.011\,Temp\ - 0.0005\,Alk + 0.676\,log(DOC) + 0.168\,log(UVA) + 0.231\,log\left(Cl_{2,cons}\right) + 0.092pH + 0.324\,log(Cl_{2,res})$$

THM formation in the DWDS **(Eqn. 2)**:

$$log(THM) = -1.73 + 0.015\,Temp\ + 0.00307\,Alk + 0.126\,log(DOC) + 0.00557\,log(UVA) + 0.554\,log\left(Cl_{2,cons}\right) + 0.313pH + 0.0739\ log\left(Cl_{2,res}\right) + 0.747\,log(time)$$

HAA formation at the plant **(Eqn. 3)**:

$$\log(HAA) = -0.75 + 2.33log(DOC) + 2.12log(Cl_{2,cons})$$

HAA formation in the DWDS **(Eqn. 4)**:

$$log(HAA) = 0.517 + 0.02\log(\mathrm{time}) + 1.5\,log(DOC)\ + 0.98\ log\left(Cl_{2,cons}\right)$$

Where, THM is THM concentration in units of µg/L,
HAA is HAA concentration in units of µg/L,
Temp is temperature in units of degree Celsius,
Alk is alkalinity in units of mg/L as $CaCO_3$,
DOC is dissolved organic carbon concentration in units of mg/L,
UVA is UV absorbance in units of 1/cm,
$Cl_{2,\,cons}$ is the chlorine consumed (dose minus residual) in units of mg/L,
$Cl_{2,\,res}$ is the concentration of chlorine residual in units of mg/L, and
time is the contact time with chlorine in units of days.

To verify the predictive capability of the model, we tested the model against two independent sets of historical jar testing data, one provided by HDR and the other by GHWTP plant staff (Figure 43). Both sets of jar testing simulated a post-chlorination strategy with enhanced coagulation at the plant. To verify the models, the plant and DWDS equations were used to calculate the predicted THMs at 7 days under the same conditions as the HDR jar testing, and the predicted values were compared to the observed values. The city staff data set included THM values at 0-days (i.e., the concentration of THMs in the simulated plant effluent); therefore, only the plant model was used to calculate the predicted THMs under the city staff's jar testing conditions. In general, the concentration of THMs predicted by the models is similar to what was experimentally measured by both HDR and the city staff, indicating the models have good predictive capabilities when a post-chlorination strategy is used.

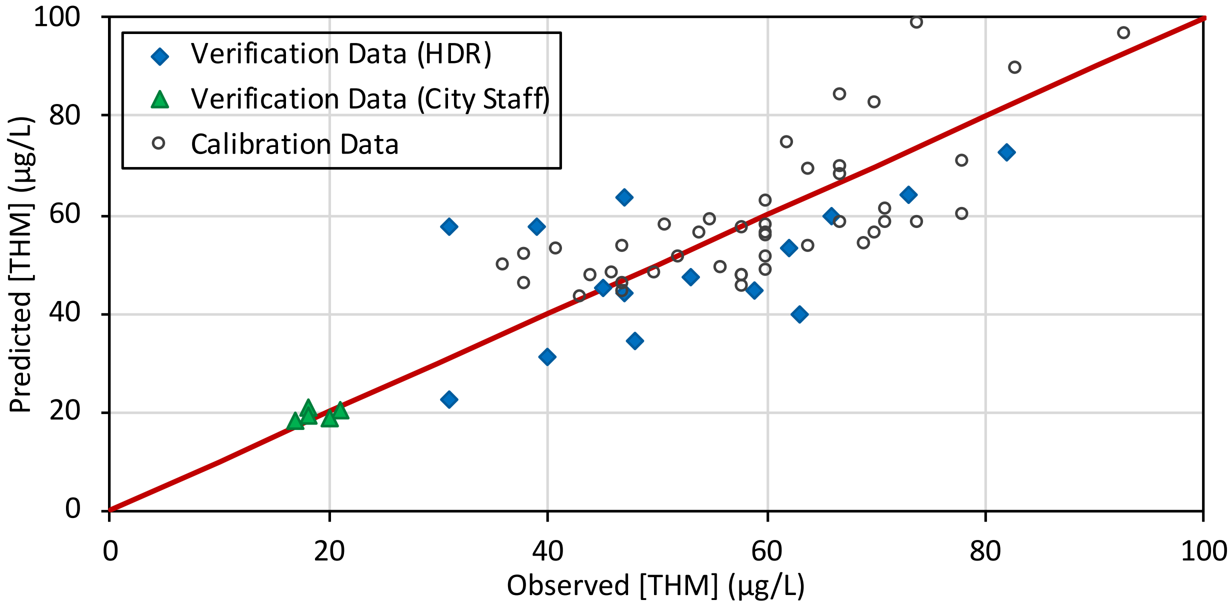

Figure 43. Comparison of model performance for data used to calibrate the model and data used to verify the model.

The THM formation model was used to determine the maximum settled water DOC concentration that would be allowable under different scenarios (Figure 44, Table 11). Scenario 1, the 'Base Case', is based on typical water quality (temperature, UVA), optimal enhanced coagulation conditions identified during jar testing, and using post-chlorination for CT credit.

Under these conditions, a settled water DOC of 2 mg/L is needed to achieve the THM goal of 80% of the MCL after 7 days in the distribution system.

To understand what the settled water DOC requirements would need to be in a realistic worst-case scenario, we also evaluated a scenario in which the highest water temperature was assumed but the variables that can be controlled at the plant (pH, chlorine residual) were given their best-case values. This is meant to simulate a realistic worst-case scenario. The modeling demonstrated that under these conditions, a settled water DOC of 2.5 mg/L would be low enough to achieve the THM formation goal. In the enhanced coagulation jar testing, all of the optimized conditions resulted in settled water DOC values less than or equal to 2.44 mg/L, even with the most challenging source waters.

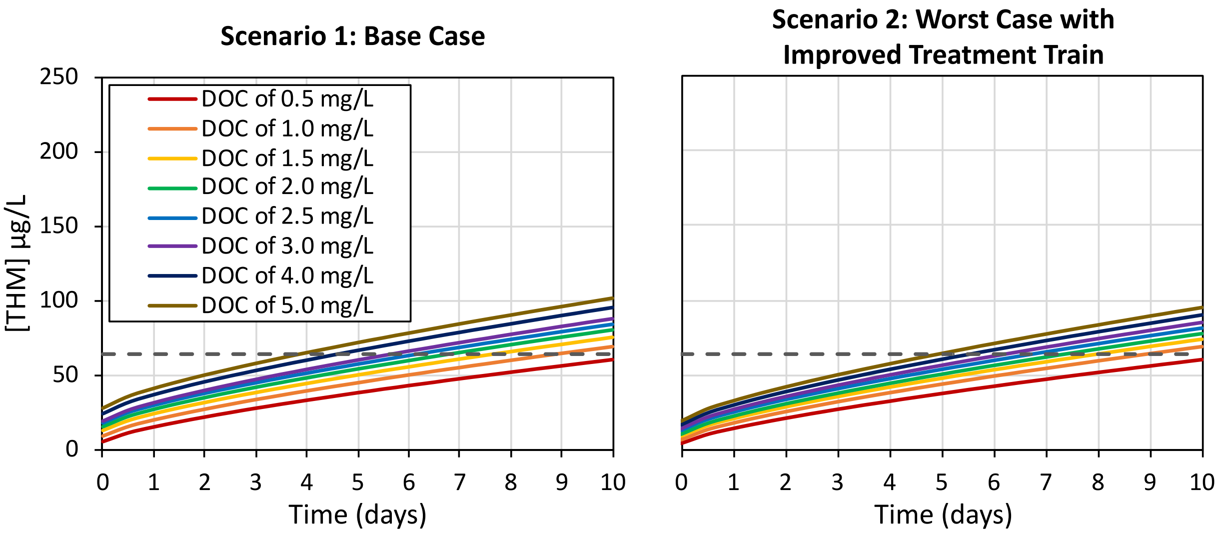

Figure 44. Modeled THM formation as a function of different settled water DOC concentrations for two scenarios. The dashed line represents a target value for THM of 64 µg/L (80% of the MCL).

Table 11. Conditions used in model scenarios shown in Figure 44

Variables	Scenario 1: Base Case		Scenario 2: Worst Case with Improved Treatment Train	
	Plant	DWDS	Plant	DWDS
Alkalinity (mg/L)	100	100	100	100
Temperature (°C)	15	15	20	23
UVA (1/cm)	0.02	0.02	0.01	0.01
pH	6.7	7.3	6.5	7.0
Cl_2 residual (mg/L)	1.2	0.2	1.2	0.3
Cl_2 consumed (mg/L)	2.0	1.0	0.5	0.9

4.5 Summary of DBP Formation Models and Applications

Multiple models have been developed for DBP formation both at the plant and in the distribution system, as summarized in Figure 45 and Table 12. The correct model to use will depend on the questions being asked. In all cases, estimating the DBP formation at a certain point in the distribution system requires summing the formation at the plant with formation in the distribution system. For example, calculating the 7-day THM formation with a post-chlorination strategy would require calculating both Model 3 and Model 4 and summing the results.

Current Plant Configuration – Plant Formation

There are two models that predict THM formation at the plant using the current treatment configuration, i.e. chlorination through the sedimentation basins. The first, Model 1a, is more effective at predicting THM formation in the 30 – 50 µg/L range, while Model 1b is more effective at predicting THMs in the higher ranges, i.e. >50 µg/L. Thus, Model 1b may be more appropriate to predict the impacts of high-TOC winter water on THM formation, whereas Model 1a may be more appropriate to predict the impacts of treatment modifications designed to reduce THM formation.

Current Plant Configuration – Distribution System Formation

The model for THM formation in the distribution system is based on the THM analyzer used at the plant that uses an experimental procedure to predict THM formation in the distribution system. This model is a simple linear regression, as shown above in Figure 41. This model may need to be modified if changes are made in the distribution system to reduce water age and/or volatilize THMs using a tray aerator. Model 2 should be used with either Model 1a or Model 1b.

Post-Chlorination Configuration – Plant Formation

There are two models for plant DBP formation using post-chlorination—one for THM formation (Model 3), and one for HAA formation (Model 5). This model would be appropriate to evaluate treatment modifications that involve moving chlorination after the sedimentation basins (or after a different clarification process, e.g. ACTIFLO Carb). For example, to use this model to evaluate an ACTIFLO Carb system, an estimate of the TOC, UVA, alkalinity, and temperature leaving the ACTIFLO Carb process could be plugged in as the starting point for the model.

Post-Chlorination Configuration – Distribution System Formation

There are two models for distribution system DBP formation using post-chlorination—one for THM formation (Model 4), and one for HAA formation (Model 6). This model should be paired with the corresponding plant formation DBP model based on post-chlorination to estimate the final distribution system concentration. Because time is an input to this model, it can be used to model different water ages in the distribution system.

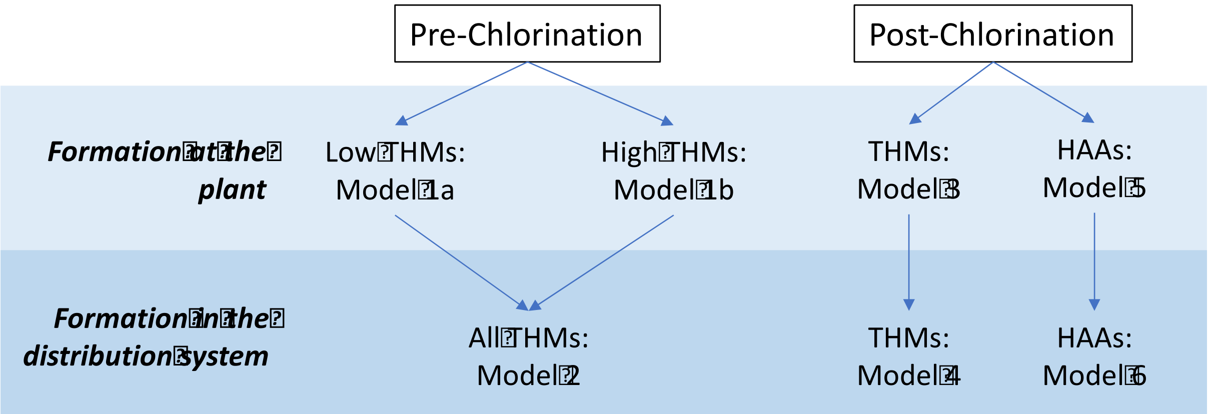

Figure 45. Decision chart for DBP formation models.

Table 12. Summary of disinfection byproduct formation models for GHWTP.

Model Number	*Description*	*Model*
Pre-Chlorination		
Model 1a	Low THMs at plant	$\log(THM) = 1.15 + 0.26\log(TOC) + 0.01\,Temp + 0.03\,pH + 0.14\log(Cl_{res}) + 0.08\log(Cl_{cons})$
Model 1b	High THMs at plant	$\log(THM) = 0.36 + 0.31\log(TOC) + 0.02\,Temp + 0.12\,pH + 0.03\log(Cl_{res}) + 0.05\log(Cl_{cons})$
Model 2	All THMs in distribution	$7\ day\ THM\ formation = 1.729\ (Plant\ effluent\ THMs) + 13.522$
Post-Chlorination		
Model 3	THMs at plant	$log(THM) = 0.429 + 0.011\,Temp - 0.0005\,Alk + 0.676\,log(DOC) + 0.168\,log(UVA) + 0.231\,log(Cl_{2,cons}) + 0.092pH + 0.324\,log(Cl_{2,res})$
Model 4	THMs in distribution	$log(THM) = -1.73 + 0.015\,Temp + 0.00307\,Alk + 0.126\,log(DOC) + 0.00557\,log(UVA) + 0.554\,log(Cl_{2,cons}) + 0.313pH + 0.0739\,log(Cl_{2,res}) + 0.747\,log(time)$
Model 5	HAAs at plant	$log(HAA) = -0.75 + 2.33\,log(DOC) + 2.12\,log(Cl_{2,cons})$
Model 6	HAAs in distribution	$log(HAA) = 0.517 + 0.02\log(\mathrm{time}) + 1.5\,log(DOC) + 0.98\,log(Cl_{2,cons})$

4.6 Summary of DBP Analysis

THM formation mitigation strategies need to be considered regardless of whether or not additional winter SLR flows are treated at the plant. The modeling presented here demonstrates that there may be multiple strategies to meet the THM limits in the distribution system. Strategies that reduce TOC prior to chlorination are likely to be effective—these strategies include enhanced coagulation, ACTIFLO Carb, and potentially other technologies that can reduce TOC levels below the 2.0-2.5 mg/L threshold. Additional testing of alternative approaches is recommended to evaluate the effectiveness of other control strategies, such as TOC transformation by ozone or UV disinfection.

5 Additional Regulatory Considerations

5.1 MCLs

Several of the constituents measured during routine monitoring have either primary or secondary MCLs, as summarized in Table 2 (see Section 2). For all constituents in this table, the MCLs or sMCLs are either met in the raw source waters or are not expected to be an issue for compliance after treatment (e.g., iron and manganese).

Additional sampling was performed for some volatile organic chemicals (VOCs), synthetic organic chemicals (SOCs), and radionuclides with MCLs – see Appendix B for a complete list. During Water Year 2017, only one of these constituents was detected in any source water. The one compound detected, uranium, was well below its MCL (20 pCi/L) in the San Lorenzo River (0.067 pCi/L) and Liddell Spring (0.94 pCi/L). None of the source waters are expected to pose a challenge to compliance with primary and secondary MCLs.

One group of contaminants of emerging concern are the perfluorinated compounds, or more specifically the perfluoroalkyl substances (PFASs). California has recently implemented Notification Levels (NL) for two of the PFAS chemicals: perfluorooctanoic acid (PFOA) and perfluorooctanesulfonic acid (PFOS) of 14 ng/L and 13 ng/L, respectively. These NLs were established in July 2018, in addition to the May 2016 EPA Health Advisory of 70 ng/L for the combined concentration of both PFOA and PFOS. The City has not previously conducted any sampling for PFOA and PFOS, and there are currently no requirements to monitor for these constituents.

5.2 Bin Classification for *Cryptosporidium* Regulations

As discussed above, the 2018 Sanitary Survey has resulted in a re-classification of GHWTP into Bin 2 for *Cryptosporidium* treatment. Bin 2 requires 3-log *Cryptosporidium* removal/inactivation through the treatment plant, which can be achieved through a combination of filtration and disinfection. Filtration credit can be achieved through compliance with more stringent combined filter effluent (CFE) and individual filter effluent (IFE) turbidity limits. 0.5-log *Crypto* removal credit can be obtained if the CFE turbidity measurements are $\leq$ 0.15 NTU in 95% of monthly samples. An additional 0.5-log credit is available for systems with individual filter effluent (IFE) turbidities meeting the following criteria: (1) IFE < 0.15 NTU in at least 95% of monthly samples, and (2) no individual filter has two consecutive measurements > 0.3 NTU (EPA 2010). No disinfection credit can be obtained for *Cryptosporidium* with chlorine disinfection; however, using UV as a primary disinfectant would be an effective strategy for achieving *Cryptosporidium* inactivation credit. UV would also provide additional protection against the other regulated pathogens and indicator organisms.

5.3 *Giardia* Removal Regulations

Based on the monthly median total coliform concentrations of the Raw Blend, the required removal/inactivation for *Giardia* is 4-logs (refer to Table 3 and Figure 20). Compliance is currently achieved through a combination of disinfection and filtration credit. Measured concentrations of total coliform in the San Lorenzo River did exceed 10,000 MPN/100 mL; if the raw blend monthly median were to exceed this threshold, it might result in changes to the *Giardia* treatment requirement. However, given that the treatment requirement is based on a monthly median and the SLR concentrations are not consistently above 10,000 MPN/100 mL, this is not likely to be an issue.

6 Conclusions and Recommendations

6.1 Source Water Quality

The GHWTP surface source waters exhibit varying degrees of seasonality, with the largest impacts to water quality coming from winter storms. We recommend continued routine water quality monitoring for all source waters, and event-based sampling for the San Lorenzo River. Additional routine and event-based sampling would continue to provide useful insight into source water quality. Event-based sampling for microbial constituents coupled with a targeted analysis with microbial source tracking could help elucidate the sources of increased microbial contamination during winter storms. Continued frequent sampling of TOC and/or DOC during storm events could expand the high TOC data available for the THM formation model, and could improve the model calibration at more extreme TOC and THM values. Adding online UVA monitoring for the raw blend would be an additional way to obtain more frequent data for the model calibration, as the UVA correlates well with the TOC.

Additional monitoring data could be used to further confirm seasonal trends in water quality and provide a larger database with which to evaluate potential implications for future treatment. The microbiological data, particularly the new event-based sampling in the SLR from WY18, further demonstrate the need to provide consistent control against pathogens. As treatment modifications are evaluated, disinfection performance should be prioritized in order to ensure the microbiological safety of the water.

6.2 Coagulation Performance

Variations in constituents such as alkalinity, pH, and hardness have implications for treatment, but the observed water quality ranges should not result in a raw blend that cannot be treated to all regulatory standards. Further bench-scale testing is recommended to optimize the coagulation strategy for raw blends consisting of higher winter water San Lorenzo River flows. Additionally, the plant could consider decoupling pH adjustment and coagulant dosing to better optimize control of both parameters. If there are scenarios where coagulant is being over-dosed to achieve pH reduction, there is potential for savings in terms of solids production.

6.3 Disinfection Byproduct Formation

Disinfection byproduct formation, particularly of THMs, is an issue for GHWTP. Under the current operational strategy of chlorinating in the sedimentation basins, it will be difficult to consistently meet the plant’s goal of 80% of the THM MCL at terminal locations in the distribution system. Based on the analysis provided in Section 4, the most effective way to control THM formation in the plant’s current configuration will be to limit the raw blend TOC concentration, which is not a feasible option. Alternative solutions include moving the primary chlorination location after the sedimentation basins and using enhanced coagulation to reduce the settled water DOC, or using an alternative primary disinfectant such as UV or ozone.

6.4 Solids Production

Treating additional winter San Lorenzo River flows will likely be problematic from both a DBP and solids production standpoint. If the DBP issue is mitigated through the use of an alternative disinfection strategy, it is likely that solids production will be the limiting factor in treatment of winter SLR water. Modifications to the solids handling infrastructure and/or practices at GHWTP may allow for additional treatment of winter flows. We recommend additional bench-scale testing on winter San Lorenzo River water to better understand the THM formation potential of higher turbidity waters, as well as the coagulant dose needed to meet effluent turbidity standards.

7 References

CDM (2007) Summary Report: Water Quality and System Improvements Study. Prepared for the City of Santa Cruz Water Department.

CDPH (2013) Regulations Related to Drinking Water. California Code of Regulations.

Crittenden, J.C., Trussell., R.R., Hand, D.W., Howe, K.J., Tchobanoglous, G., Borchardt, J.H. (2012) MWH’s Water Treatment Principles and Design, Third Edition. John Wiley & Sons, Inc: Hoboken, New Jersey.

DDW. Appendix B: Guidelines for determining when surface waters will require more than the minimum levels of treatment defined in the surface water treatment regulations.

EPA (1989) Surface Water Treatment Rule. 40 CFR 141.70-141.75.

EPA (1998) National Primary Drinking Water Regulations: Interim Enhanced Surface Water Treatment Rule. 40 CFR Parts 9, 141, and 142.

EPA (1999) Enhanced Coagulation and Enhanced Precipitative Softening Guidance Manual. Office of Water, Washington, D.C.

EPA (2006) National Primary Drinking Water Regulations: Long Term 2 Enhanced Surface Water Treatment Rule. 40 CFR Parts 9, 141, and 142.

EPA (2010) Long Term 2 Enhanced Surface Water Treatment Rule Toolbox Guidance Manual.

Kennedy/Jenks Consultants (2013) San Lorenzo Valley and North Coast Watersheds Sanitary Survey. Prepared for City of Santa Cruz.

Kennedy/Jenks Consultants (2018) San Lorenzo River and North Coast Watersheds Sanitary Survey Update. Prepared for City of Santa Cruz.

NRC (2004) Indicators for waterborne pathogens, National Research Council, National Academies Press, Washington, DC.

Ricker, J, and Peters, S. (2006) Assessment of Sources of Bacterial Contamination at Santa Cruz County Beaches.

U.S. Department of Health and Human Services Federal Panel on Community Water Fluoridation (2015) U.S. Public Health Service Recommendation for Fluoride Concentration in Drinking Water for the Prevention of Dental Caries. *Public Health Reports* (150), 318-331.

Water Supply Advisory Committee (2015) Final Report on Agreements and Recommendations. City of Santa Cruz. Available at (http://www.santacruzwatersupply.com/meeting/wsac-final-reportrecommendation-appendices).

Appendix A: Additional Sampling for Constituents with MCLs

Table B-1. Summary of monitoring results for additional constituents with MCLs. Units are mg/L unless otherwise noted.

Constituent	MCL	San Lorenzo River	Newell Creek	Majors Creek	Laguna Creek	Liddell Spring
1,1-Dichloroethane	0.005	ND	ND	ND	ND	ND
1,1-Dichloroethylene	0.006	ND	ND	ND	ND	ND
1,1,1-Trichloroethane	0.2	ND	ND	ND	ND	ND
1,1,2-Trichloro-1,2,2-Trifluoroethane (Freon 113)	1.2	ND	ND	ND	ND	ND
1,1,2-Trichloroethane	0.005	ND	ND	ND	ND	ND
1,1,2,2-Tetrachloroethane	0.001	ND	ND	ND	ND	ND
1,2-Dichlorobenzene	0.6	ND	ND	ND	ND	ND
1,2-Dichloroethane	0.005	ND	ND	ND	ND	ND
1,2-Dichloropropane	0.005	ND	ND	ND	ND	ND
1,2,4-Trichlorobenzene	0.005	ND	ND	ND	ND	ND
1,3-Dichloropropene	0.0005	ND	ND	ND	ND	ND
1,4-Dichlorobenzene	0.005	ND	ND	ND	ND	ND
Benzene	0.001	ND	ND	ND	ND	ND
Carbon Tetrachloride	0.0005	ND	ND	ND	ND	ND
cis-1,2-Dichloroethylene	0.006	ND	ND	ND	ND	ND
Dichloromethane	0.005	ND	ND	ND	ND	ND
Ethylbenzene	0.3	ND	ND	ND	ND	ND
Methyl-tert-butyl ether (MTBE)	0.013	ND	ND	ND	ND	ND
Monochlorobenzene	0.07	ND	ND	ND	ND	ND
Styrene	0.1	ND	ND	ND	ND	ND
Tetrachloroethylene	0.005	ND	ND	ND	ND	ND
Toluene	0.15	ND	ND	ND	ND	ND
trans-1,2-Dichloroethylene	0.01	ND	ND	ND	ND	ND
Trichloroethylene	0.005	ND	ND	ND	ND	ND
Trichlorofluoromethane	0.15	ND	ND	ND	ND	ND
Vinyl Chloride	0.0005	ND	ND	ND	ND	ND
Xylenes	1.75	ND	ND	ND	ND	ND
2,4-D	0.07	ND	ND	ND	ND	ND
Alachlor	0.002	ND	ND	ND	ND	ND
Atrazine	0.001	ND	ND	ND	ND	ND
Bentazon	0.018	ND	ND	ND	ND	ND
Carbofuran	0.018	ND	ND	ND	ND	ND
Diquat	0.02	ND	ND	ND	ND	ND
Endothall	0.1	ND	ND	ND	ND	ND
Ethylene Dibromide	0.00005	ND	ND	ND	ND	ND

Constituent	MCL	San Lorenzo River	Newell Creek	Majors Creek	Laguna Creek	Liddell Spring
Lindane	0.0002	ND	ND	ND	ND	ND
Oxamyl	0.05	ND	ND	ND	ND	ND
Simazine	0.004	ND	ND	ND	ND	ND
Gross alpha particle	15 pCi/L	ND	ND	ND	ND	ND
Radium-226	5 pCI/L (combined)	ND	ND	ND	ND	ND
Radium-228		ND	ND	ND	ND	ND
Uranium	20 pCi/L	0.067	ND	ND	ND	0.94

Appendix B: West Yost Jar Testing Technical Memo

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GHWTP Jar Testing and Sludge Settling Testing Results

Date: October 1, 2019 (Final)
March 8, 2019 (Revised)
June 14, 2018 (Draft)

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Subject: Final GHWTP Jar Testing and Sludge Settling Testing Results

1. Introduction

Bench-scale testing was conducted by Trussell Technologies (Trussell Tech) and West Yost at the City of Santa Cruz (City) Graham Hill Water Treatment Plant (GHWTP) to support solids handling system design work and treatment alternatives evaluation. The test goals were as follows:

Solids handling evaluation:
- Determine coagulant dose to treat high-turbidity water from both the San Lorenzo River and Loch Lomond to optimize removal of both turbidity and total organic carbon (TOC).
- Characterize settleability of semi-clarified spent filter backwash water and sedimentation basin sludge.

Enhanced coagulation optimization:
- Evaluate enhanced coagulation treatment to determine optimal pH and coagulant dose to minimize DBP formation.

Multiple source waters were tested over the course of two weeks of jar testing conducted onsite at the GHWTP lab facility, including both a high and low turbidity San Lorenzo River water, a source water blend, and a low turbidity Newell Creek water. This technical memorandum (TM) describes the testing methodology, summarizes the results, and provides conclusions from the bench-scale testing.

2. Enhanced Coagulation Testing Methods

The bench testing focused on the assessment of enhanced coagulation of the currently available GHWTP water sources, as well as the formation of disinfection byproducts (DBPs) – namely total trihalomethanes (TTHMs) – using simulated distribution system (SDS) tests. Free chlorine is used as a disinfectant at GHWTP and can react with natural organic matter (NOM) in the water to form TTHMs. In water treatment, total organic carbon (TOC) is typically used as a surrogate for NOM, and the removal of TOC–a surrogate for removal of precursor material–is regulated by the United States Environmental Protection Agency (USEPA) under the Stage 1 and 2 Disinfectants and Disinfection Byproducts Rules (D/DBP Rules) (USEPA, 1998, 2001, and 2006). The D/DBP Rules establish maximum contaminant levels (MCLs), routine monitoring requirements, as well as operational evaluation levels to mitigate exposure to high DBP levels. Compliance with the MCL value of 0.080 mg/L for TTHM is based on a locational running annual average (LRAA) for individual monitoring locations within the distribution system (USEPA, 2006). For some source waters, greater TOC removal can be achieved at a lower pH with the same coagulant dose. This process, enhanced coagulation, was tested as part of this project to determine the optimal pH and aluminum sulfate (alum) dose to achieve turbidity and TOC reduction in each source water. Additionally, the regulation requires different amounts of TOC removal for enhanced coagulation depending on the source water TOC and alkalinity (refer to Table 1).

Table 1. Stage 1 D/DBP Rules TOC Removal Requirement (USEPA, 1998).

Source Water TOC (mg/L)	Source Water Alkalinity (mg/L as $CaCO_3$) TOC removal (%)		
	0 – 60	>60 – 120	>120
>2.0 – 4.0	35	25	15
>4.0 – 8.0	45	35	25
>8.0	50	40	30

This section describes the preparation of stock chemical solutions for use in the testing, the procedure for titrations that was used to determine acid doses needed to achieve specific coagulated water pH targets, the jar testing procedure, and the procedure for simulated distribution system disinfection byproduct (SDS DBP) testing.

2.1. Stock Solution Preparation

Enhanced coagulation jar tests involved the use of several chemicals that are used for full-scale treatment at GHWTP. The preparation of stock solutions used during testing followed the *Standard Operating Procedure for Jar Tests* developed by Trussell Tech in 2017. Stock solutions were made weekly for aluminum chlorohydrate (ACH), Magnafloc LT7992 (cationic polymer used as a coagulant aid), and non-acidified liquid alum ($Al_2(SO_4)_3 \bullet 14H_2O$). For the purposes of this document, future uses of the word “alum” refer to the non-acidified liquid alum. Each stock solution was prepared to a target concentration as neat chemical used by the GHWTP staff for internal jar testing. An example calculation for the dilution of ACH and the stock solution concentrations used are provided in Appendix A.

Potassium permanganate ($KMnO_4$), sulfuric acid, and sodium hypochlorite were provided by City staff at strength. Potassium permanganate was 0.1% of neat strength, and sulfuric acid was 5%. The sodium hypochlorite stock was quantified by City staff daily using a Hach digital titrator kit.

2.2. Titration

In order to optimize pH and coagulant dose for enhanced coagulation, each of the waters tested was titrated with sulfuric acid to target specific pH values. Titrations were performed prior to the start of each jar test – using an aliquot of source water dosed with the associated chemicals for an individual test condition – to ascertain the combined effect of the chemical addition on pH and determine the necessary sulfuric acid dose at Flash Mix 2 to achieve the target pH for the given test condition.

Two steps were needed for titrating each source water. First, the non-alum chemicals (e.g. $KMnO_4$, Magnafloc, and ACH) were added in a stepwise manner to determine the impact on pH, followed by titration with increasing doses of alum to achieve the targeted range of alum concentrations. This process provides data on the resulting pH from each of the chemical additions in Flash Mix 1, as well as from the alum addition at Flash Mix 2 for a range of alum doses. An example of this process is provided in Table 2. Note that ACH was not included in the specific example shown in Table 2, though ACH is often used for Newell Creek water. The second titration was used to determine the dose of sulfuric acid needed to bring the water down to the target pH value for enhanced coagulation. This titration was performed for each alum dose condition, as the addition of alum resulted in a change in pH. An example is given below, in Table 3. The titration provides data on the sulfuric acid addition and resulting pH for each step of the titration. The information allows for the calculation of sulfuric acid needed to depress the pH from the value identified in Table 2, Step 3 (specific to the target alum dose) to the specific target pH value for each test condition. The full list of steps for this titration can be found in Appendix B, including example calculations for a hypothetical test condition consisting of pre-chlorination, an alum dose of 73 mg/L (as neat), and a target pH of 6.5.

Table 2. Example of chemical addition steps and resulting pH values for Newell Creek water.

Newell Creek		**Step 3 - Alum:**		
Volume (mL):	2000	Dose (mg/L)	Stock (mL)	pH
pH:	7.55	53	10.6	7.15
		63	12.6	7.09
Step 1 - KMnO4:		73	14.6	7.01
Dose (mg/L):	1	83	16.6	6.94
mL Stock (mL):	2	93	18.6	6.90
pH:	7.57	103	20.6	6.85
		113	22.6	6.80
Step 2 - Magnafloc:		123	24.6	6.74
Dose (mg/L):	2	133	26.6	6.71
mL Stock (mL):	4	143	28.6	6.67
pH:	7.58			

Table 3. Example of sulfuric acid titration for Newell Creek water.

Newell Creek		**Step 3 - H2SO4[a]:**		
Volume (mL):	2000	Initial Burette (mL)	Final Burette (mL)	pH
pH:	7.53	0	3.1	7.59
		3.1	18.5	7.16
Step 1 - KMnO4:		18.5	23	7.09
Dose (mg/L):	1	0	4.5	7.01
mL Stock (mL):	2	4.5	9	6.94
pH:	7.56	9	13.5	6.90
		13.5	17	6.85
Step 2 - Magnafloc:		17	20	6.79
Dose (mg/L):	2	20	22.7	6.75
mL Stock (mL):	4	0	5	6.72
pH:	7.58	5	9	6.68
		9	14	6.62
Step 3 - NaOCl:		14	23	6.53
Dose (mg/L):	3.9	0	12	6.42
mL Stock (mL):	0.639	12	23	6.32
pH:	7.73	19.4	25	6.24

[a] 5% H_2SO_4 diluted 100 times

2.3. Jar Testing

The *Standard Operating Procedure for Jar Tests* was used for the jar tests. During the testing, an assumed plant flow of 8 MGD was used for establishing the jar testing parameters related to plant flow (i.e., contact basin hydraulic retention time). The sedimentation time used for the jar test was based on settling velocity curves developed by City staff. Twenty minutes was selected as the preferred settling time to simulate the plant process based on the turbidity data of the settling velocity curve. The overall jar testing procedure is summarized as follows.

1. Measure and collect samples for water quality
2. Aliquot the chemicals required in plastic weigh boats prior to the start of each test
3. Perform jar tests with the following setpoints:
    a. Flash Mix 1: 10 sec @ 236 RPM
        i. Add ACH (if used) and KMnO4
    b. Carbon Contactor: 114 min @ 55 rpm
    c. Flash Mix 2: 30 sec @ 120 RPM
        i. Add sodium hypochlorite (if pre-chlorination condition), polymer, alum, and acid
    d. Flocculation Step 1: 42 min @ 28 RPM
    e. Flocculation Step 2: 42 min @ 19 RPM
    f. Sedimentation: 20 min

g. Post Sedimentation:
    i. Slowly decant water from the jars using the sampling ports
    ii. Measure and collect samples for water quality
    iii. Add sodium hypochlorite (for post-chlorination conditions) and hold for 60 minutes

2.4. SDS DBP Testing

The SDS DBP testing was performed in conjunction with jar testing to evaluate the formation of TTHMs for each test condition. The same chlorine dose was tested for each of the post-chlorination test conditions.

After the settled water quality was collected, 1L of each of the post-chlorination test condition waters were aliquoted into separate zero head-space containers. Sodium hypochlorite was then dosed for each of the test conditions while staggering the time to ensure adequate time for sampling of each test condition after 1 hour. Sodium hypochlorite was not dosed after the jar test for the pre-chlorination test conditions. The sample representing the 1-hour contact time for the pre-chlorination sample was collected at the same time as the 1-hour contact samples for the post-chlorination samples. Temperature, pH, chlorine residual, and TTHM were each measured from the 1-hour contact time sample. Simultaneously, approximately 250 mL was aliquoted into an amber bottle for each of the test conditions and held for 7 days to replicate a representative residence time in the City's distribution system. The pH of these samples was not adjusted prior to SDS testing and may impact DBP formation in the distribution system. The pH range present in the SDS tests after chlorination was 6.5 – 7.5, whereas the current plant operations target a pH leaving the plant of 7.1 – 7.3. A DBP formation model that has been developed as part of a parallel effort could be used to estimate the impact of increasing the pH in the distribution system for the conditions where it was below 7 in the SDS tests. The SDS samples were stored in the filter gallery to maintain a temperature similar to the treated water in the City's distribution system. The 7-day hold time samples were processed after 7 days, and the same water quality parameters were measured (temperature, pH, chlorine residual, and TTHM). The difference in the TTHM values from these two sample times provides a measure of the formation potential.

The formation potential samples (7-day hold time 250 mL sample in amber bottles) and the individual TTHM samples were stored in amber glass bottles with no headspace. In addition, the formation potential samples were covered in aluminum foil to ensure the samples were not affected by ambient light.

3. Enhanced Coagulation Results

Enhanced coagulation testing was carried out to determine the optimal pH and coagulant dose for removal of dissolved organic carbon (DOC) and turbidity in multiple source waters. Specific targets were not set for DOC and turbidity in the settled water; the selection of 'optimal' pH and coagulant dose was based on the relative performance between jars. Two sets of jar tests were conducted for each source water; in the first, the coagulant dose was kept constant while the pH was varied to select an optimal pH, and in the second the coagulant dose was varied while all the jars were kept at the optimal pH. Table 4 summarizes the source water quality and the test conditions performed during this study.

Table 4. Summary of jar test conditions.

Source water	Collection date	Test #	Turbidity (NTU)	DOC (mg/L)	pH tested	Alum dose tested (mg/L)
San Lorenzo River	3/1/18	1	37.1	4.4	6.3 – 7.04	73
		2	37.1	4.12	~6.7	73 - 113
		3	37.1	4.04	~6.7	73 - 113
San Lorenzo River	3/12/18	4	6.35	2.94	6.36 – 7.38	73
		5	5.27	2.93	~6.6	53 - 133
Plant Blend (56% Newell Creek / 44%San Lorenzo River)	3/6/18	6	2.00	2.99	6.2 – 7.27	73
		7	1.88	2.84	~6.4	33 - 83
Newell Creek	3/12/18	8	2.66	4.37	6.53 – 7.12	73
		9	2.66	4.34	~6.1	43 - 93

3.1. pH Optimization

The results of enhanced coagulation pH optimization for the different source waters are presented in Figure 1. The data demonstrate a correlation between pH and DOC removal in all source waters tested, where DOC removal increases with decreasing pH. The ‘optimal’ pH selected for each source water was based on a combination of optimizing DOC removal, while minimizing the need for chemical addition for pH adjustment at full scale. In some test conditions, decreasing the pH resulted in diminishing returns in terms of DOC removal; this was the case for the high turbidity San Lorenzo River water. In that case, the marginal gain from 54% to 57% DOC removal was not deemed sufficiently beneficial to justify the increased pH adjustment and corresponding cost associated with the both additional acid dose and post-treatment caustic soda. The pH selected for each source water to carry forward for the coagulant dose optimization is shown in Table 5.

The pH selected for further testing was based on DOC removal because the DBP formation data were not yet available at the time of selection. However, the same improved performance trend was observed at lower pH values in the SDS DBP testing results; both DOC and TTHM concentrations were typically lower for the lower pH coagulated water conditions (see Figure 2). Reducing pH during enhanced coagulation resulted in lower TTHM formation over the 7-day hold period and test conditions with pH values lower than 6.7 in the settled water typically achieved the compliance goal of 80% of the TTHM MCL after 7 days. It should be noted that some conditions had a pH lower than 7 in the SDS test after chlorination, which would result in lower THM formation than would occur at a higher pH.

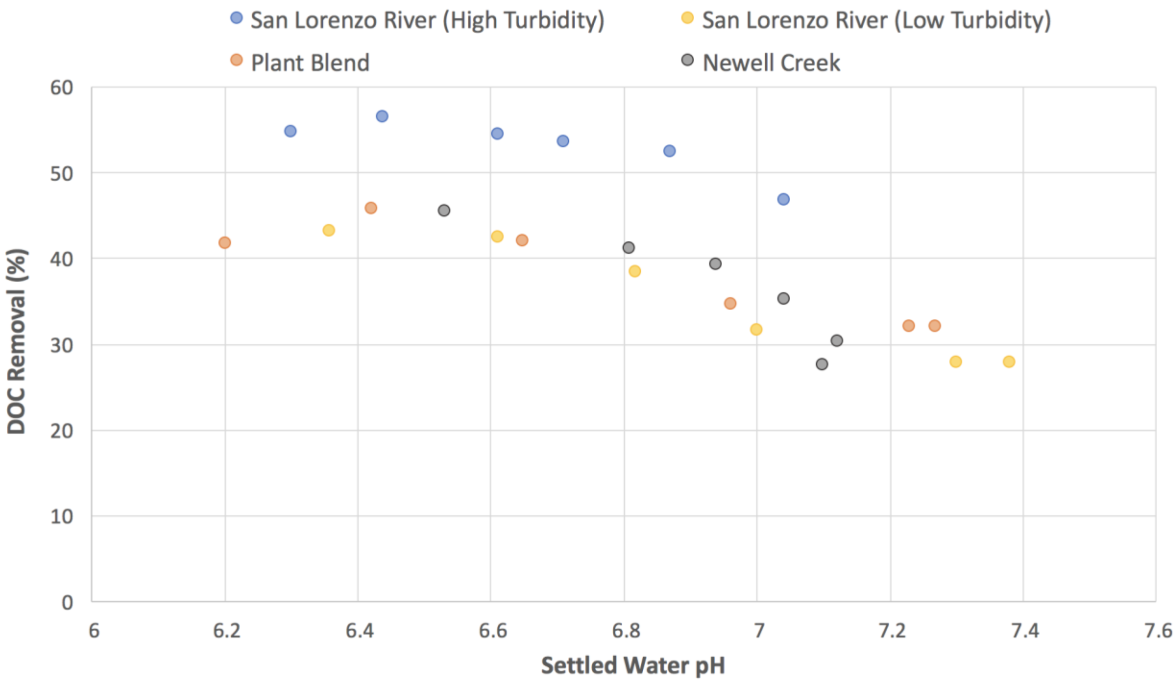

Figure 1. Impact of pH on DOC removal for different source waters

Table 5. Summary of jar test performance for each source water with optimal pH and alum dose identified.

Source Water	Optimal Conditions Identified		Jar Test Conditions and Performance						
	pH	Alum dose (mg/L)	Raw Turbidity (NTU)	Raw DOC (mg/L)	DOC Removal (%)	Settled Turbidity (NTU)	Settled DOC (mg/L)	TTHM (ug/L) (0 day)	TTHM (ug/L) (7 day)
San Lorenzo River	6.7	53	37.1	4.4	53	1.13	2.07	18	38
San Lorenzo River	6.6	53	5.27	2.94	43	1.67	1.68	16	55
Plant Blend	6.4	53	1.88	2.83	36	0.59	1.92	17	41
Newell Creek	6.5	73	2.66	4.37	44	1.09	2.44	21	74

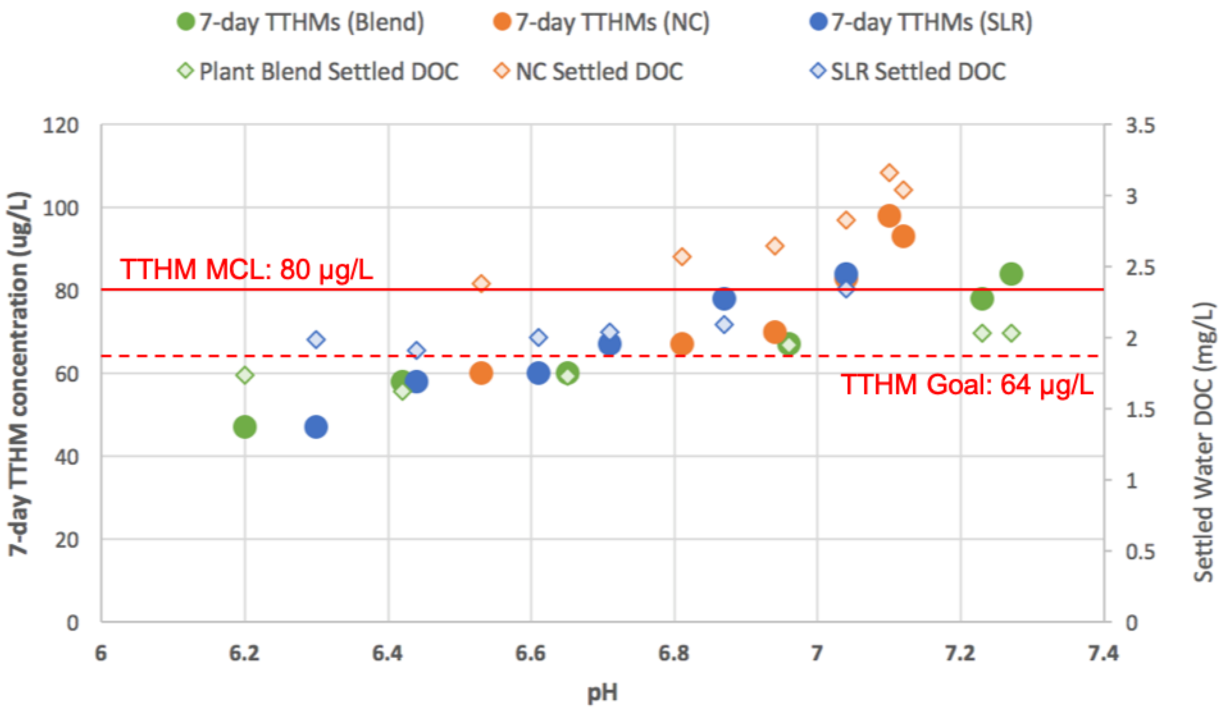

Figure 2. Impact of coagulation pH on concentration of TTHM and DOC for different GHWTP water sources. Alum dose for all tests shown was 73 mg/L.

3.2. Coagulant dose optimization

Multiple alum doses were tested for each source water with the pH held constant at the value shown in Table 5. The goal of the alum dose optimization was to identify the dose needed to achieve the target turbidity and DOC removals. Figure 3 shows the impact of alum dose on settled water turbidity for each source water. Higher alum doses did not result in significant turbidity reduction, even for the high-turbidity San Lorenzo River water. However, the use of higher alum doses did increase DOC removal to some degree (see Figure 4), but higher alum doses did not result in an observable decrease in 7-day TTHM concentration (see Figure 5). For the San Lorenzo River and plant blend test conditions, an alum dose of 53 mg/L was sufficient to produce a low settled water turbidity (<2 NTU) and a high degree of DOC reduction (>40%), while maintaining TTHM concentrations after 7-days of hold time below 80% of the MCL. For the Newell Creek source water, maintaining the TTHM below 80% of the MCL was only achieved for one of the replicates tested with an alum dose of 73 mg/L (Figure 5).

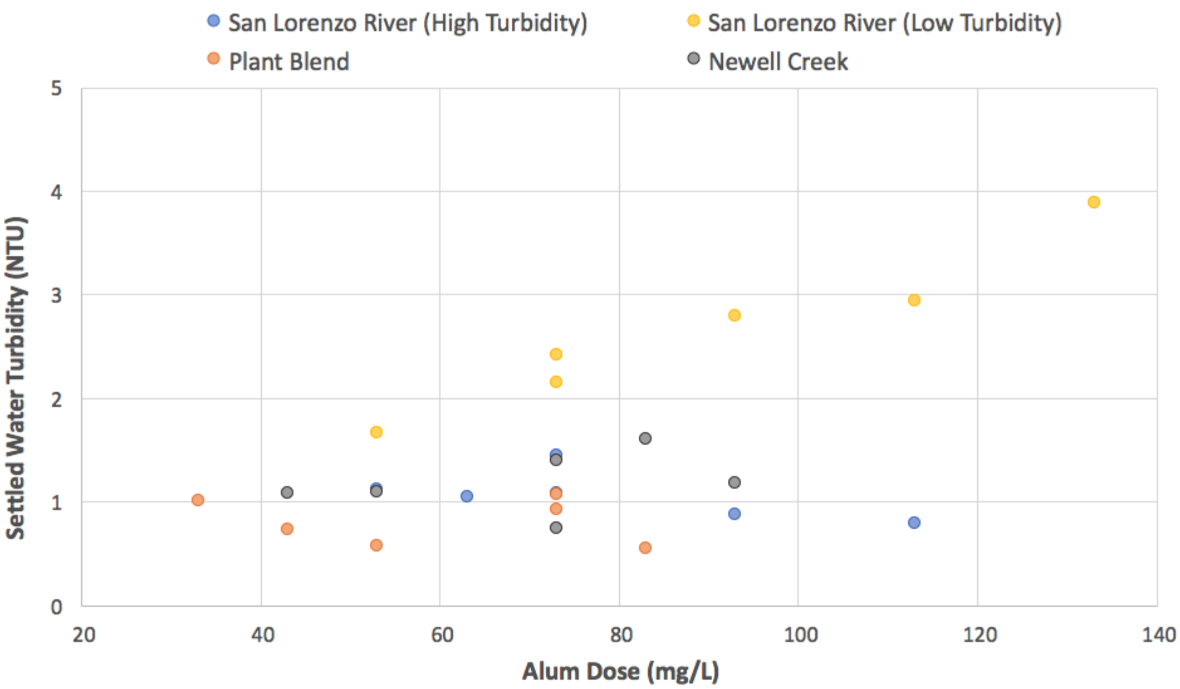

Figure 3. Impact of alum dose on settled water turbidity for different GHWTP water sources

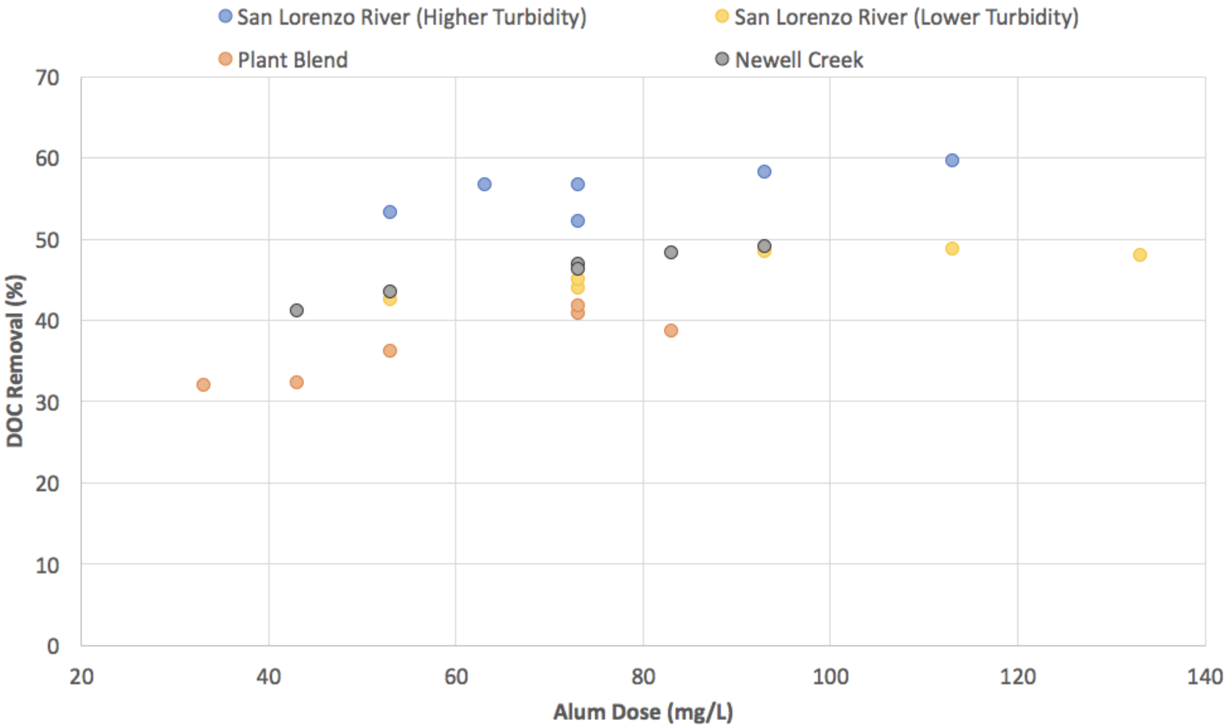

Figure 4. Impact of alum dose on DOC removal for different GHWTP source waters

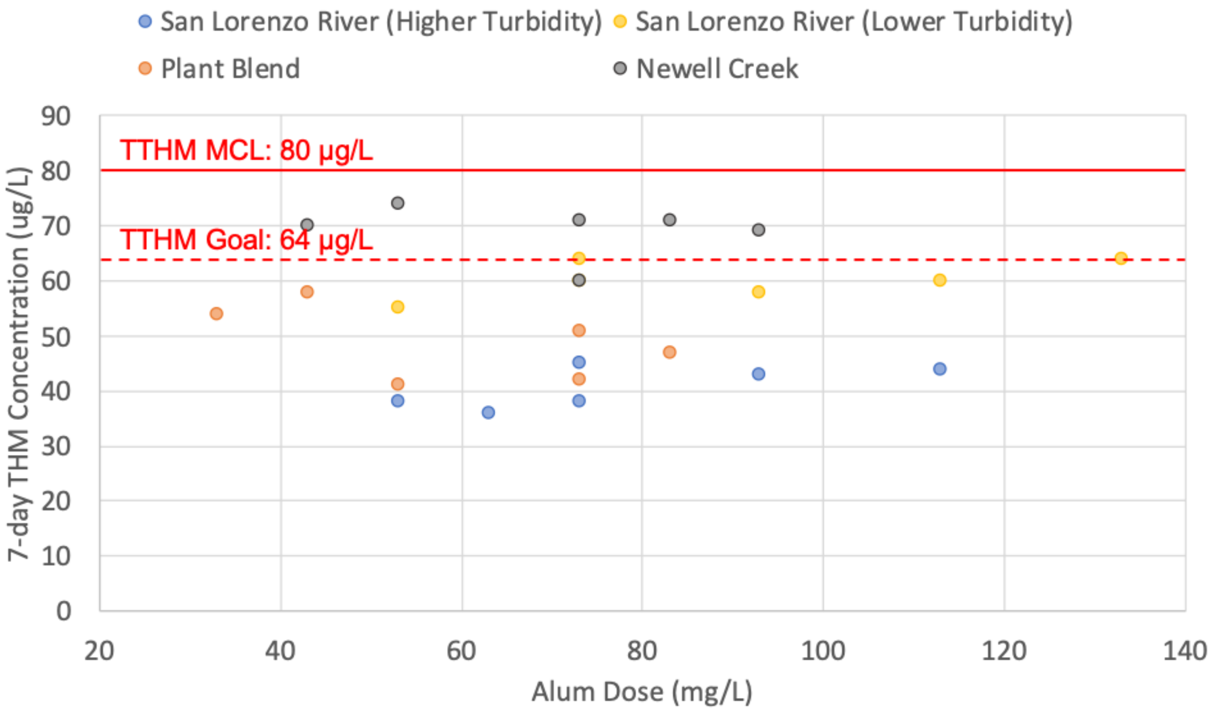

Figure 5. Impact of alum dose on 7-day TTHM concentration for different GHWTP water sources

3.3. DBP Formation

The DBP formation testing compared two scenarios – pre-chlorination (chlorine addition prior to enhanced coagulation) and post-chlorination (chlorine addition after enhanced coagulation). Pre- and post-chlorination scenarios were compared in order to evaluate the potential benefit of moving the chlorine injection downstream of the clarifiers to reduce DBP formation. In general, a reduction in TTHM formation was observed for the test conditions simulating chlorine addition following enhanced coagulation. Given the same simulated treatment conditions (i.e. same water, pH, and alum dose), the post-chlorination scenario resulted in up to a 14% reduction in TTHM formation compared to pre-chlorination. As expected during pre-chlorination, chlorine is in direct contact with DBP precursors (NOM) present in the raw water source, typically resulting in higher THM formation and formation potential. Enhanced coagulation improves removal of the precursor material (NOM), and when coupled with post-chlorination, this treatment strategy can result in lower TTHM concentrations both at the plant and in the distribution system.

Although TOC is a direct measure of water's bulk organic content, it does not directly measure the concentration of DBP precursors present in the water matrix. Specific ultraviolet absorbance (SUVA) has been shown to be a better indicator of the presence of humic compounds (hydrophobic aromatic compounds) in water matrices, which are often the main precursors for DBP formation. Studies have reported that enhanced coagulation is most effective for reducing DOC (>50%) in waters with SUVA above 4 L/mg-m (Sillanpää and Matilainen, 2015). The water sources tested in this study had SUVA values between 2 and 3 L/mg.m, indicating a mixture of humic and non-humic material (Table 6). For this SUVA range, DOC removal was within the expected range of 35-50%. Because the range of SUVA observed in the source waters is fairly narrow, it is difficult to draw conclusions about using SUVA to predict THM formation.

As an alternative, DOC and pH have been shown to be the most important factors for optimizing enhanced coagulation; during this testing, it was observed that in general, settled water DOC below 2.5 mg/L and pH below 6.7 can yield compliance with the TTHM regulations.

Table 6. SUVA measurements of different source waters

Source Water	Average Raw Water SUVA (L/mg.m)
San Lorenzo High Turbidity	3.0
San Lorenzo Lower Turbidity	2.93
Plant Blend	2.7
Newell Creek	2.42

4. Sludge Settleability Testing

The goal of the sludge settling testing was to characterize the settleability of sludge solids sent to from the reclaim (recycle) tank, which receives both spent filter backwash water and sedimentation basin sludge. Water from the recycle tank currently undergoes clarification and further settling, and the clarified recycle water is returned to the head of the water treatment plant. In order to return this clarified recycle water, it must comply with the turbidity limits of the Cryptosporidium Action Plan (CAP), i.e. < 2 NTU. Characterizing the settleability of recycle tank sludge will inform the design of the two new recycle tanks that can meet the CAP turbidity limits through discrete withdrawal of supernatant from different depths over time. Additional clarification using the two existing lamella plate settlers can also be used if the CAP turbidity limits cannot be met through clarification in the recycle tank alone.

Reclaim Water Settleability Testing in Settling Column

Initial characterization of sludge settleability was conducted in a 10-ft settling column, which was used to simulate settling in the reclaim tank. The column consists of 3-in diameter PVC pipe with multiple sample ports spaced at 1-foot intervals such that turbidity can be sampled along the full depth of the column over time and the sludge solids settling rate can be characterized (see Figure 6). Spent filter backwash water and water from the reclaim tank was collected prior to polymer addition and added to the top of the column. Testing involved sampling turbidity at Ports 1, 2, 3, and 6, which are 1, 2, 3, and 6 feet from the top of the column, respectively. Turbidity was sampled at 30-minute intervals for approximately 2 hours, then subsequently every one to two hours for about 4 hours.

Figure 7 illustrates an example of the changes in turbidity at different sample ports over multiple hours of settling time in the column. The sludge demonstrated rapid settling; after 1.5 hours, the turbidity in all of the sample ports tested was below 5 NTU. After 6 hours, the turbidity in all sample ports (ports 1,2,3 and 6) was below 2 NTU, which is the turbidity target for compliance with the CAP.

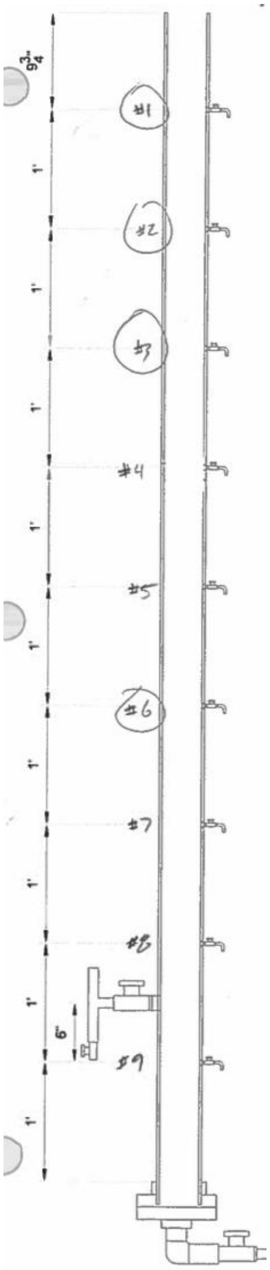

Figure 6. Schematic of sludge settling column.

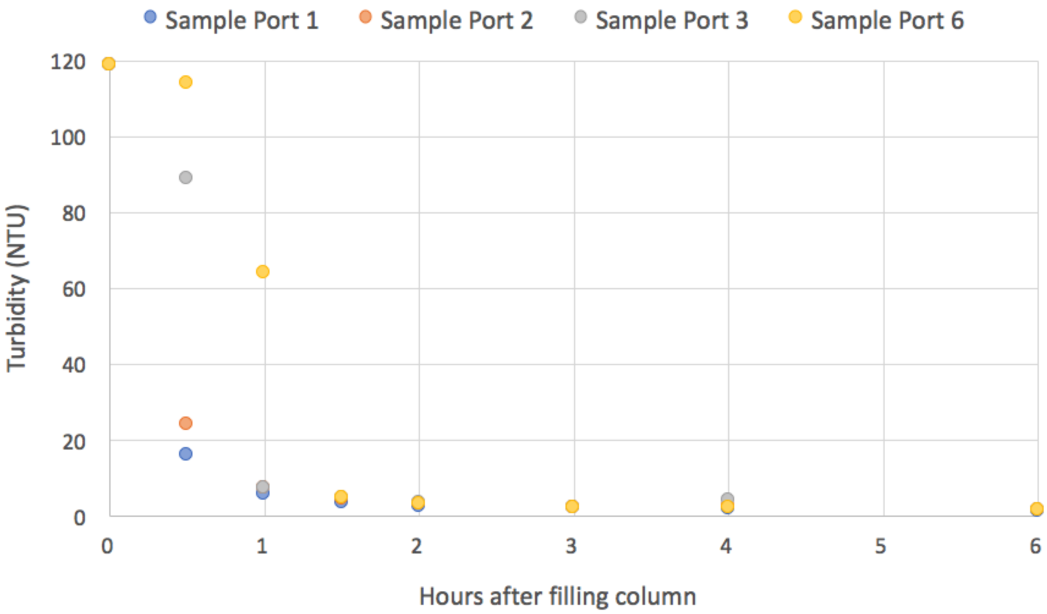

Figure 7. Changes in turbidity over time in sludge settling column. Testing was done with reclaim water with starting turbidity of 119 NTU.

High Turbidity Reclaim Water Settleability Jar Testing
If the CAP turbidity limit is not able to be met through settling in the two new recycle tanks, the two existing lamella plate clarifiers can continue to be used to facilitate further turbidity reduction. Jar testing was used to assess the settleability of higher-turbidity reclaimed water, simulating conditions where the CAP turbidity limits could not be met in the recycle tank. The turbidity range of interest for the testing was 25-50 NTU; because of the rapid settling observed in the column, the water for the jar tests was collected from the settling columns and spiked with additional filter backwash water to achieve a turbidity in the range of interest.

Figure 8 shows the results for two sets of jar tests that were conducted with different mixing speeds and times; both tests included polymer addition at the same concentration being used at full-scale, i.e. 3.5 mg/L of polymer 7736. Both conditions lead to a similar degree of settling after 90 minutes, with turbidities between 5-6 NTU. It was determined that the polymer dose used was not sufficiently high to achieve turbidity reduction below 2 NTU. Further optimization of the coagulant and polymer dosage could provide additional insight into whether the CAP turbidity targets can be met for sludge with turbidity in the range of 25 to 50 NTU.

An additional jar test was conducted with a lower starting turbidity, i.e. 8.34 NTU. In this test, 8181 nonionic polymer was used at a dose of 3 mg/L. The results of the jar testing are summarized in Table 7. Jars 1-4 reached a turbidity below 2 NTU within 30 minutes; Jar 6, which did not receive any mixing, reached a turbidity below 2 NTU after 2.5 hours.

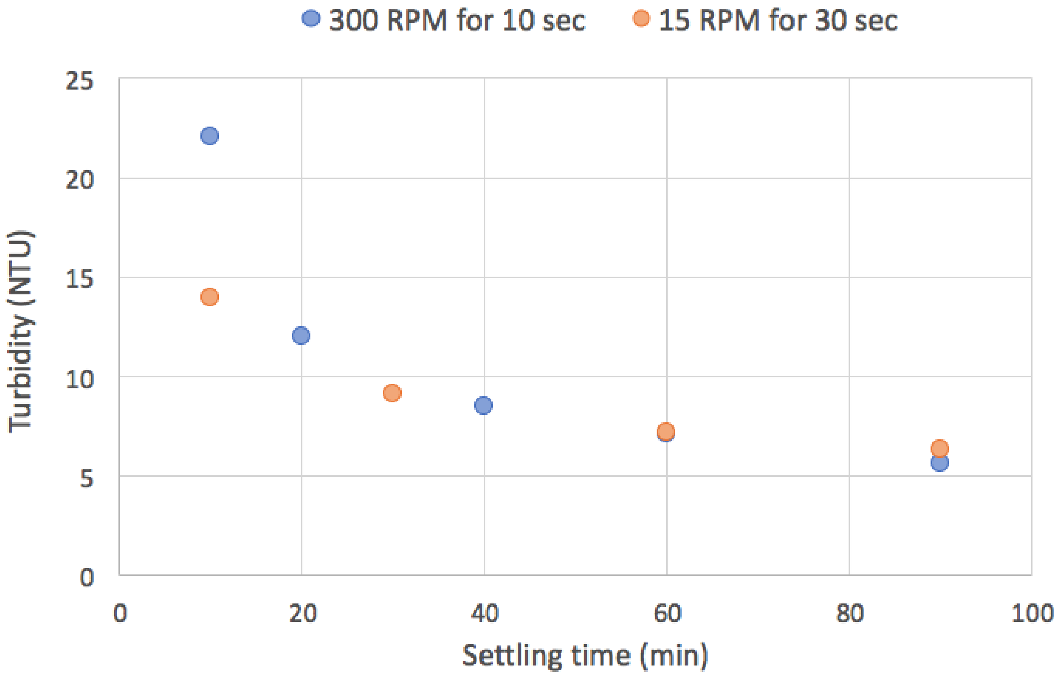

Figure 8. Sludge settling in two jar tests with different mixing speeds and times. Values shown are averages of six jars. Polymer dose for both sets of tests was 3.5 mg/L of polymer 7763.

Table 7. Summary of recycle tank water jar testing. All jars received a polymer dose (8181 nonionic polymer) of 3 mg/L. Turbidity measurements were stopped when each jar reached a turbidity below 2 NTU.

Jar	Mixing Speed (RPM)	Sample Turbidity (NTU)			
		30 min	1 hr	2 hr	2.5 hr
1	110	1.4	-	-	-
2	90	1.74	-	-	-
3	70	1.6	-	-	-
4	50	1.82	-	-	-
5	28	2.02	1.87	-	-
6	0	3.56	3.77	2.17	1.92

5. Conclusions

High-turbidity San Lorenzo River water testing:

- An enhanced coagulation pH of 6.7 and an alum dose of 53 mg/L coupled with a post-chlorination strategy were sufficient to achieve turbidity reduction and meet the TTHM goal after 7 days for the high turbidity San Lorenzo River water.

Enhanced coagulation testing & DBP formation:

- Compliance with the TTHM MCL was achieved for all source waters by lowering the pH below 6.7 and using an alum dose of 53 mg/L, along with a post-chlorination strategy.
- Under many of the enhanced coagulation conditions, the TTHM concentration in the Newell Creek water after 7 days was between 80 and 100% of the MCL. Further

optimization of DOC reduction would likely be needed in order to achieve a 7-day TTHM concentration below 80% of the MCL if this source water is treated without blending.

- Post-chlorination is preferred relative to pre-chlorination for meeting the THM MCL in drinking water. This study demonstrated that moving chlorine injection downstream of enhanced coagulation yields lower THM formation.
- For the water sources tested, DOC and coagulated water pH appear to be the most important factors in optimizing enhanced coagulation with respect to minimizing DBP formation. Although SUVA is a good indicator of DBP precursors (aromatic humic compounds), it does not seem to be a good surrogate for predicting THM formation in the source waters tested.

Sludge settling testing:

- Partially clarified blend of spent filter backwash water and settling basin sludge collected from the reclaim tank exhibited rapid settling in the settling column; sludge with turbidity as high as 119 NTU dropped below 2 NTU after 6 hours at all elevations tested (up to 6 feet in depth). This corresponds to a settling and clarification rate of no less than 1 ft/hr.
- Sludge with turbidity between 25 and 50 NTU may be more challenging to settle; additional optimization of chemical dosing for this strategy is recommended to determine whether the CAP goal of 2 NTU can be achieved in a single stage provided by the two new recycle tanks.

6. References

Sillanpää, M. and Matilainen, A. (2015) Chapter 3. NOM Removal by Coagulation.

USEPA (1998). Federal Register, Volume 63, No. 241, December 16, 1998, “National Primary Drinking Water Regulations: Disinfectants and Disinfection Byproducts; Final Rule”

USEPA (2001). Stage 1 Disinfectants and Disinfection By-Products Rule: EPA 816-F-01-014. Office of Water (4607).

USEPA (2006). Federal Register, Volume 71, No. 2, January 4, 2006, “National Primary Drinking Water Regulations: Stage 2 Disinfectants and Disinfection Byproducts Rule”

Appendix A: Stock Solution Concentrations and Example Calculation

Example Calculation – ACH (SWT2000) stock solution

Bulk chemical:
Specific gravity = 1.338
Concentration (lbs/gal) = (specific gravity) x (% solution) x (8.34 lbs/gal)
Concentration (lbs/gal) = (1.338) x (1.00) x (8.34 lbs/gal)
Concentration (lbs/gal) = 11.16
Concentration (mg/L ACH) = 1,338,000

Stock solution:
Target strength: 1% by weight = 100 mg/mL ACH = 10,000 mg/L ACH
Target volume (mL): 1000 (volumetric flask)

$C_1V_1 = C_2V_2$, where:

C = concentration (mg/L)
V = volume (mL)
1 = bulk chemical
2 = stock solution

$V_1 = (C_2V_2) / C_1$
$V_1 = (10{,}000 \text{ mg/L} \times 1000 \text{ mL}) / 1{,}338{,}000 \text{ mg/L}$
$V_1 = 7.5 \text{ mL}$

Table A1. Summary of stock solution preparations.

Chemical	Stock Solution		
	Target Strength (% by weight)	Bulk Chemical - Wet (mL)	DI Water (mL)
ACH (SWT2000)	1%	7.5	992.5
Magnafloc LT7992 (cationic coagulant aid)	0.1%	1.93	998.07
Liquid Alum (Al2(SO4)3•14H2O)	1%	7.5	992.5
NaOCl	0.8%	-	-
KMnO4	0.1%	-	-
H2SO4	5%	-	-

Appendix B: Titration Procedures

Procedure to determine the impact of different alum doses on pH:

1. Aliquot a known volume of sample water
2. Measure and record the initial pH
3. Add the target dose of ACH and measure the resulting pH
4. Add the target dose of potassium permanganate and measure the resulting pH
5. Add the target dose of polymer (Magnafloc LT7992) and measure the resulting pH
6. Add the alum dose and measure the resulting pH in a step-wise fashion
    a. Start by adding the lowest alum dose (e.g., 40 mg/L)
    b. Evaluate the resulting pH
    c. Add additional alum to meet the subsequent dose target (e.g., 10 mg/L to achieve 50 mg/L alum)
    d. Measure pH and continue process for remaining target doses of alum

Titration procedure to determine the required acid addition to achieve target pHs:

1. Dilute 5% sulfuric acid such that a reasonable amount of acid is used for the titration
    a. As more titrations are performed, previous data can be used for an informed selection of dilution factor
2. Aliquot a known volume of sample water
3. Measure and record the initial pH
4. Add the target dose of ACH and measure the resulting pH
5. Add the target dose of potassium permanganate and measure the resulting pH
6. Add the target dose of polymer (Magnafloc LT7992) and measure the resulting pH
7. Add sodium hypochlorite and measure the resulting pH
8. Titrate with sulfuric acid to achieve the pH prior to sodium hypochlorite addition
    a. Sodium hypochlorite does not add significant alkalinity; therefore, a new sample is not needed for succeeding titration with sulfuric acid
9. In a step-wise fashion, titrate water sample with sulfuric acid to target the corresponding pH values for the corresponding alum dose targets; at each pH target, recording the volume of sulfuric acid added since the previous pH target. Continue the titration and record the acid addition at the pH setpoints selected for testing

<u>Example Calculation –Pre-chlorination, pH target 6.5, and 113 mg/L alum neat</u>

Amount of 5% H_2SO_4 needed to achieve pH target 6.5 w/o pre-chlorination:

pH after alum addition = 6.8

Titrant (100x diluted 5% H_2SO_4) needed to reach pH 6.5 =

(22.7 mL – 4.5 mL) + (23 mL – 0 mL) = 41.2 mL

$$\frac{Titrant\ Amount}{Titrant\ Dilution} \times \frac{Jar\ Volume}{Sample\ Volume} = \frac{41.2\ mL\ H_2SO_4}{100} \times \frac{2000\ mL}{2000\ mL} = 0.412\ mL\ 5\%\ H_2SO_4$$

Amount of 5% H_2SO_4 needed to adjust for pre-chlorination sodium hypochlorite:

$$\frac{Titrant\ Amount}{Titrant\ Dilution} \times \frac{Jar\ Volume}{Sample\ Volume} = \frac{3.1\ mL\ H_2SO_4}{100} \times \frac{2000\ mL}{2000\ mL} = 0.031\ mL\ 5\%\ H_2SO_4$$

Total amount of 5% H_2SO_4 needed to adjust for pre-chlorination sodium hypochlorite:
0.412 mL H_2SO_4 + 0.031 mL H_2SO_4 = 0.443 mL H_2SO_4

Appendix C: HDR Jar Testing and Modeling Technical Memo

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Graham Hill Water Treatment Plant Disinfection Byproduct Formation Modeling

Final Report
July 15, 2019

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Trussell
TECHNOLOGIES INC

Table of Contents

1 Introduction

1.1 Background

Disinfection byproduct (DBP) formation is a key driver for treatment modifications at the Graham Hill Water Treatment Plant (GHWTP). The current disinfection strategy—chlorination through the sedimentation basins and the use of a free chlorine residual in the distribution system—leads to high DBP formation, particularly trihalomethanes (THMs). HDR is investigating multiple potential alternative treatment trains and evaluating their ability to achieve compliance with DBP maximum contaminant levels (MCLs). In support of this effort, Trussell Tech conducted bench-scale testing of these treatment alternatives to calibrate and use a DBP formation model to assess the impact of different treatment strategies on DBP control.

1.2 Source Water Quality

Jar testing was conducted on water from both the San Lorenzo River and Newell Creek. The water quality of the source waters is summarized in Table 1. The source waters were sampled in an effort to get ‘worst-case’ water quality to understand the treatment performance and DBP formation under challenging conditions. We were able to obtain a sample of high turbidity (25 NTU) winter water from the San Lorenzo River, which also had a relatively high total organic carbon (TOC) of 5 mg/L. At the time of sampling, the water from Newell Creek did not have high turbidity, but did have a characteristic TOC of 4.4 mg/L. This source is generally less impacted by storms and has a more consistent turbidity and TOC.

Table 1. Water quality of source waters for jar testing.

	San Lorenzo River	Newell Creek
Date Collected	3/2/18	3/2/18
Turbidity (NTU)	25	1.5
TOC (mg/L)	5	4.4
DOC (mg/L)	4.8	4.3
pH	7.8	7.7
UV 254 (cm^{-1})	0.17	0.12
Alkalinity (mg/L)	74	93
Bromide (ug/L)	27	40

1.3 Treatment Alternatives

The treatment configurations tested as part of this work were developed in conjunction with HDR and are presented in Figure 1. A complete description of jar testing methods can be found in Appendix A.

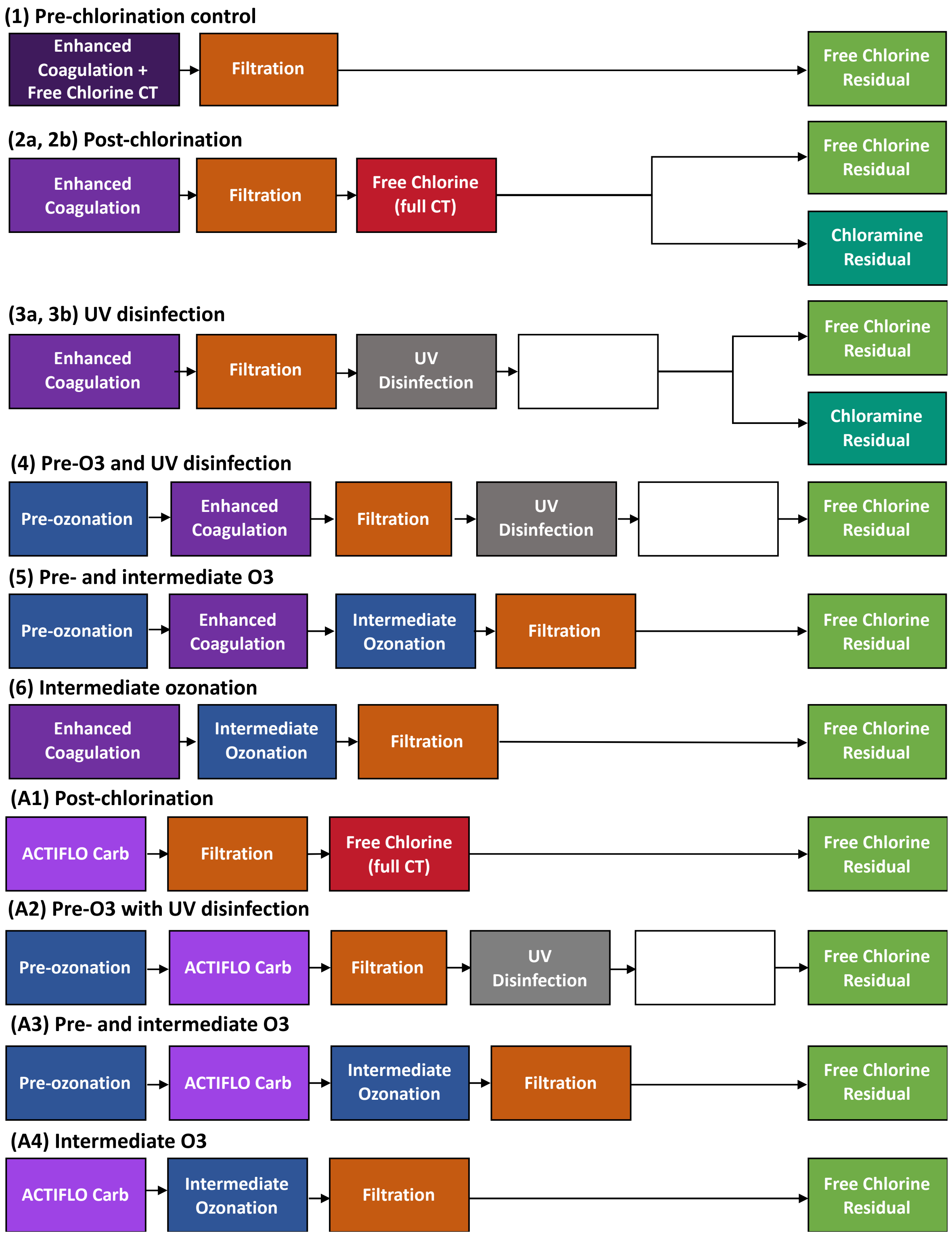

Figure 1. Summary of treatment trains tested using bench-scale jar testing to evaluate TOC removal and DBP

formation, both during treatment and in the distribution system.

In order to complete the jar testing, assumptions were made about the treatment conditions for each process; these are summarized in The data generated from the bench-scale testing conducted at Trussell Tech's Pasadena lab were supplemented with additional bench-scale data from enhanced coagulation testing conducted by Trussell Tech at the GHWTP lab. This combined data set was used to calibrate predictive models for DBP formation and is summarized in Appendix B. The goal of the models is to evaluate THM and HAA formation under a wide range of water quality conditions, thus helping in the selection of treatment alternatives.

Table 2. In general, all testing assumed a Bin 1 classification for *Cryptosporidium* (requiring 2-log reduction[1]), as well as an additional 1-log treatment requirement beyond the minimum requirements of the Surface Water Treatment Rule for virus and *Giardia*, i.e., 5- and 4- log reduction requirements, respectively. These pathogen reduction requirements are equivalent to the current requirements for the GHWTP.

Treatment conditions for both enhanced coagulation and ACTIFLO Carb were based on previous jar testing. The target pH for enhanced coagulation (6.5) was based on a previous enhanced coagulation optimization study conducted by Trussell Tech. During those studies, multiple pH values were tested and the optimal pH was selected based on maximizing DOC removal. The ACTIFLO Carb chemical doses were based on previous testing done by HDR during which multiple chemical doses were tested; optimal doses were also based on maximizing DOC removal for this process.

All treatment trains involving ozonation—both pre- and intermediate—required ozone demand testing to select the appropriate ozone dose; doses used are summarized in Table 3. For pre-ozonation, a single dose was identified for each source water such that an ozone residual was present for 2-3 minutes. For intermediate ozonation, two different doses were selected based on whether or not pre-ozonation was part of the treatment train (e.g., Train 5 vs. Train 6). The ozone doses for intermediate ozonation *without* pre-ozonation are higher because there is no reduction in ozone demand from the pre-oxidation step.

All jar tests looked at treatment performance as well as DBP formation in the distribution system using simulated distribution system (SDS) testing. Finished water from each jar test was adjusted to a pH target of 7 and held for approximately 7 days in an amber bottle. Where necessary, chlorine was added prior to the SDS testing to a target residual concentration of 1.2 mg/L. The impact of using chloramine rather than free chlorine was also tested in two scenarios (Trains 2b and 3b).

The data generated from the bench-scale testing conducted at Trussell Tech's Pasadena lab were supplemented with additional bench-scale data from enhanced coagulation testing conducted by Trussell Tech at the GHWTP lab. This combined data set was used to calibrate predictive models for DBP formation and is summarized in Appendix B. The goal of the models is to evaluate THM and HAA formation under a wide range of water quality conditions, thus helping in the selection of treatment alternatives.

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[1] As a result of water quality sampling undertaken in 2016 and 2017, the facility subsequently received a Bin 2 classification. The additional 1-log treatment requirement for *Cryptosporidium* under Bin 2 was assumed to be met through filter performance and not through disinfection.

Table 2. Summary of treatment conditions used in jar testing for different treatment processes.

Treatment Process (Relevant Treatment Train)	Assumptions/Treatment Conditions
Enhanced Coagulation (1, 2, 3, 4, 5, 6)	• pH target 6.5: optimal observed pH for DOC reduction from previous jar testing • Alum dose: 113 mg/L
Post-chlorination for full CT (2, A1)	• Chlorine dose: 3 mg/L, based on achieving CT for *Giardia* inactivation • Contact time: 60 minutes, based on the design contact time of the modified filtered water tank
UV disinfection with short CT free chlorine (3, 4, A2)	• Pathogen target: assuming UV disinfection will be designed for protozoa control, but not for full virus disinfection credit • Free chlorine CT of 3 mg-min/L to achieve 2-log virus reduction
Pre-ozonation (4, 5, A2, A3)	• Ozone dose: based on ozone decay curves for each water; target was to maintain ozone residual for 2-3 minutes
Intermediate ozonation (5, 6, A3, A4)	• Ozone dose: based on ozone decay curves for each source water and treatment train
ACTIFLO Carb (A1, A2, A3, A4)	• Anionic polymer dose: 0.8 mg/L[1] • Powdered activated carbon dose: 20 mg/L[1] • Microsand dose: 8 mg/L[1] • Alum dose: 113 mg/L • Ambient pH
Secondary disinfection with free chlorine (1, 2a, 3a, 4, 5, 6, A1, A2, A3, A4)	• Free chlorine residual at start of SDS testing: 1.2 mg/L, based on current practice at the plant
Secondary disinfection with chloramine (2b, 3b)	• Chloramine target: 4 mg/L selected as a conservative dose

[1] Chemical doses based on previous jar testing conducted by HDR

Table 3. Summary of ozone doses selected for different jar testing conditions.

Source Water	Pre-Ozonation		Intermediate Ozone with Pre-Ozone		Intermediate Ozone without Pre-Ozone	
	Ozone dose (mg/L)	O3:TOC Ratio	Intermediate ozone dose (mg/L)	O3:TOC Ratio	Intermediate ozone dose (mg/L)	O3:TOC Ratio
San Lorenzo River	3.25	0.6	1.1	0.2	1.4	0.6
Newell Creek	1.7	0.4	0.78	0.18	1	0.45

Technical challenges were encountered during the bench-scale testing that rendered some of the data not usable for the model calibration. For example, in several of the jars where ozone was used for disinfection, changes in water quality made it difficult to maintain and reliably measure a chlorine residual (e.g., trains 4, 5, 6, A2, A3, and A4). These data were not used for the distribution system model (and in some cases the plant model) because it was not known how long the chlorine residual was present or the precise concentration of the chlorine residual. In addition, some of the data were not relevant for the model calibration because the jar test involved a chloramine residual rather than a free chlorine (e.g., train 2b and 3b) or the jar test modeled a pre-chlorination dosing strategy rather than a post-chlorination dosing strategy (e.g., train 1). The results from the jar tests that were not included in the model calibration are provided in Appendix C.

2 Model Development

Formation models were developed for both THMs and HAAs. The models were developed in two parts: the first part describes DBP formation at the treatment plant, and the second part describes formation in the distribution system. All models assume the use of free chlorine in the distribution system. While the use of chloramine was evaluated during the jar testing, there was not enough data to develop a model for DBP formation with a chloramine residual in the distribution system. The two parts of the model are added together to calculate the total DBP formation at different time points in the distribution system.

2.1 DBP Formation Model Selection

The THM and HAA models were developed based on existing empirical models in the literature (see Table 4, and Equations 1 and 2 below). Each model predicts DBP formation as a function of multiple independent variables, which are summarized for THMs and HAAs in Table 4.

Table 4. Summary of disinfection byproduct formation model sources and independent variables.

Disinfection Byproduct	Model Source	Independent Variables
Trihalomethanes	Obolensky and Singer (2008)	Temperature, alkalinity, dissolved organic carbon, UV absorbance at 254 nm, pH, chlorine consumed, chlorine residual, and contact time
Haloacetic Acids	EPA (1998)	DOC, chlorine consumed, bromide concentration[1], and contact time

[1] Bromide concentration was not included in the HAA model for this study. The concentrations observed were sufficiently low, i.e. < 40 µg/L, that it is not expected to be a significant predictive factor in HAA formation.

THM Formation Model (Obolensky and Singer 2008) **(Eqn. 1)**:

$$log(THM) = -1.371 + 0.015\,Temp\ - 0.0005\,Alk + 0.188\,log(DOC) + 0.326\,log(UVA) + 0.291\,log(Cl_2\,consumed) + 0.087\,pH + 0.291\ log(Cl_2\,residual) + \ 0.119\,log(time)$$

HAA Formation Model (EPA 1998) **(Eqn 2)**:

$$log(HAA) = 0.85 + 0.159\,\log(\mathrm{time}) + 0.581\,log(DOC) + 0.529\ \ log(Cl_2\,consumed)$$

2.2 Model Calibration

The models were calibrated using the water quality data from jar tests (Appendix B). For each jar test, the relevant water quality data were measured (all independent variables shown in Table 4) as well as the concentration of THMs and HAAs at the end of the jar tests and after a 7-day hold time. THMs were also measured at an intermediate time between 1-3 days. A multilinear regression was performed to determine the 8 coefficients in the THM model and the 3 coefficients in the HAA model. Given the limited range of conditions evaluated during the jar testing, adjustments were needed for certain model coefficients. Specifics of these adjustments are discussed in Section 3.1.

The amount of DBPs formed at the plant (concentration at day 0 in the distribution system) was subtracted from the concentration at each time point in the distribution system before performing the multilinear regression. When using the model to predict future treatment scenarios (as done in Section 4), the total DBP formation is the sum of the predicted DBP formation at the plant (using the plant model) and the predicted DBP formed in the distribution system at a given time (using the DWDS model).

2.3 Limitations of the Model

- The model is only as good as the calibration data. While a large variety of treatment trains were examined, each unique test was only performed during a single round in duplicate. Greater certainty in the model could be achieved with additional data points.

- Due to schedule and experimental constraints, the range of values tested for each variable was narrow. The model is anticipated to be most reliable in the range at which each variable was tested. For example, the temperature in the SDS DBP tests only varied between 14 and 18°C. Users of the model should be cautious when extrapolating to temperature values outside of this range. The ranges of water quality for each parameter are summarized in Table 5.

Table 5. Range of values observed in jar tests

Parameter	Plant	DWDS
Temperature (°C)	16 - 21	14 - 18
pH	6.1 - 7.9	6.5 - 7.8
DOC (mg/L)	1.5 - 3.0	1.5 - 3.0
Br (μg/L)	27 - 40	
UVA (cm^{-1})	0.023 - 0.28	0.023 - 0.28
Alkalinity (mg/L)	61 - 108	61 - 108
Cl_2 residual (mg/L)	0.01 - 2.9	0.06 - 1.9
Cl_2 consumed (mg/L)	0.08 - 1.9	0.17 - 1.9
Chlorine Contact Time	2.5 - 60 min	1.2 - 9.1 days
THM (μg/L)	5 - 36	7 - 123
HAA (μg/L)	0 - 20	18 - 62

3 Model Results

3.1 Summary of Coefficients for the THM and HAA models

THM and HAA formation at the treatment plant and drinking water distribution system (DWDS) were modeled using data collected during jar testing and fit to Equations 3-6. The DBP modeling coefficients for both the plant and DWDS model are summarized in Table 6 and Table 7.

THM formation at the plant **(Eqn. 3)**:

$$log(THM) = 0.429 + 0.011\,Temp\ - 0.0005\,Alk + 0.676\,log(DOC) + 0.168\,log(UVA) + 0.231\,log(Cl_2\ consumed) + 0.092pH + 0.324\,log(Cl_2\ residual)$$

THM formation in the DWDS **(Eqn. 4)**:

$$log(THM) = -1.73 + 0.015\,Temp\ + 0.00307\,Alk + 0.126\,log(DOC) + 0.00557\,log(UVA) + 0.554\,log(Cl_2\ consumed) + 0.313pH + 0.0739\ log(Cl_2\ residual) +\ 0.747\,log(time)$$

HAA formation at the plant **(Eqn. 5)**:

$$log(HAA) = -0.75 + 2.33\,log(DOC) + 2.12\,log(Cl_2\ consumed)$$

HAA formation at the DWDS **(Eqn. 6)**:

$$log(HAA) = 0.517 + 0.02\,\log(\text{time}) + 1.5\,log(DOC) + 0.98\ log(Cl_2\ consumed)$$

Where, THM is THM concentration in units of µg/L,
HAA is HAA concentration in units of µg/L,
Temp is temperature in units of degree Celsius,
Alk is alkalinity in units of mg/L as $CaCO_3$,
DOC is dissolved organic carbon concentration in units of mg/L,
UVA is UV absorbance in units of 1/cm,
Cl_2 consumed is the chlorine consumed (dose minus residual) in units of mg/L,
Cl_2 residual is the concentration of chlorine residual in units of mg/L, and
time is the contact time with chlorine in units of days.

Table 6. Summary of Coefficients for the THM Model

	Literature[1]	Plant	DWDS
Intercept	-1.371	0.429	-1.73
Alkalinity	-0.0005	-0.0005	0.00307
Temperature	0.015	0.011	0.015[3]
Log(UVA)	0.326	0.168	0.00557
Log(DOC)	0.188	0.676	0.126
pH	0.087	0.092	0.313
Log(Cl_2 residual)	0.167	0.324	0.0739
Log(Cl_2 consumed)	0.291	0.231	0.554
Log(time)	0.119	0[2]	0.747

[1] Obolensky and Singer (2008)

[2] Data did not include sufficient time variability to model this coefficient, thus was not included in the model for the plant

[3] Coefficient set to the literature value

Table 7. Summary of coefficients for HAA formation models.

	Literature[1]	Plant	DWDS
Intercept	0.848	-0.75	0.517
Log(DOC)	0.581	2.33	1.5
Log(Cl_2 consumed)	0.529	2.12	0.98
Log(time)	0.159	0[2]	0.02
Log([Br^-])	0.080		

[1] EPA (1998)

[2] Data did not include sufficient time variability to model this coefficient, thus was not included in the model for the plant

The variable for time was excluded from the DBP models at the plant because the database did not include enough different contact times to gain meaningful results. In addition, the contact time in the plant is substantially less than the contact time in the distribution system and experience suggests that some precursors react more quickly than others. Hence it should not be assumed that the rate of change with time in the plant is the same as the rate of change with time in the distribution system. If there is value in understanding this relationship more deeply, a bench study could be conducted to show the shape of the formation curve from 0 to 120 minutes or so, representative of time in the treatment plant.

Generally, the data available for modeling at GHWTP was somewhat limited, so the coefficients for some variables were modified to avoid unreasonable outcomes. For example, the coefficient for temperature for the DWDS data set was fixed at the literature value of 0.015 and the multivariable regression was repeated with the remaining variables. Additionally, bromide was excluded from the HAA model because at the low concentrations studied, bromide did not have a major impact on HAA formation. The issues with the coefficients likely stem from the narrow range of values tested in the jar tests. While the data used in this model are limited, we chose to use the plant-specific data in cases where it was consistent with principles; where it was inconsistent, we adopted the literature coefficients (temperature) or removed the parameter from the model (chlorine contact time at the treatment plant and bromide concentration).

3.2 Comparison of Predicted and Observed DBP Formation

The model predictions for THM and HAA concentrations are compared to the observed THM and HAA concentrations in Figure 2 to Figure 5. To evaluate the goodness of fit between the model and actual values, the one-to-one line is plotted along with the data points. Data points below this line indicate the model underestimates DBP formation and data points above this line indicate the model overestimates DBP formation.

Data are limited, particularly for HAAs, but, based on these figures the THM plant model overestimates THM formation at low levels of THMs and underestimates at high levels of THMs (Figure 2). The other models provide reasonable fit.

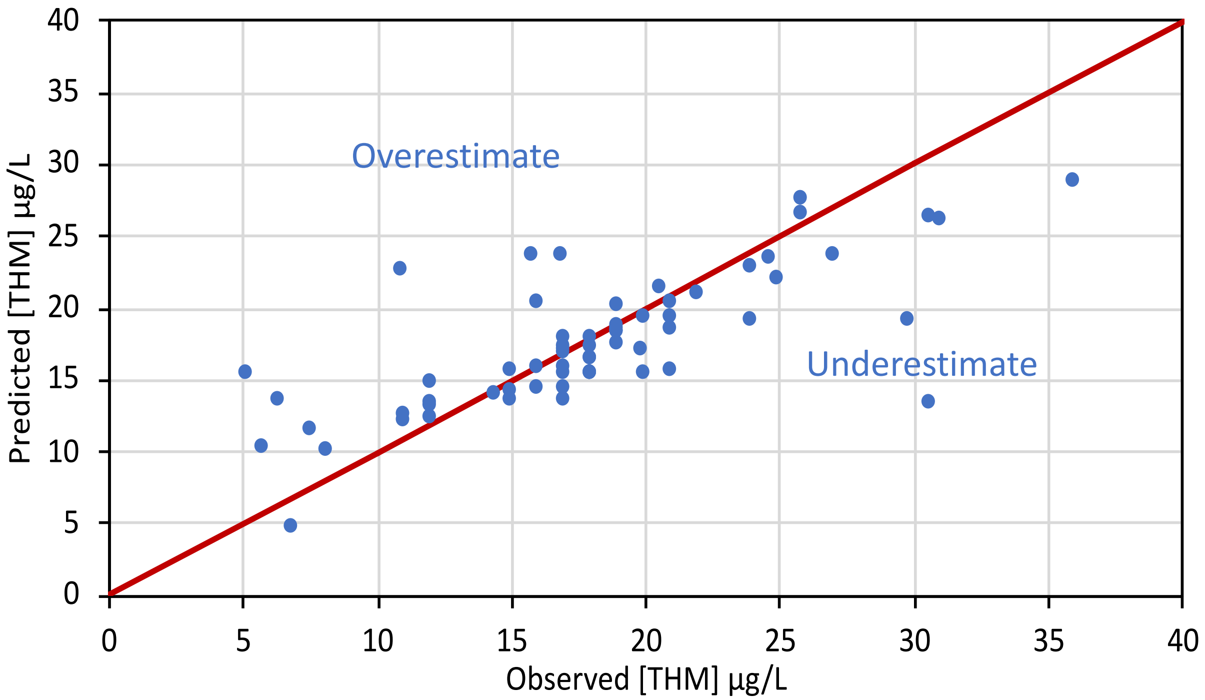

Figure 2. Relationship between the observed concentration of THMs at the end of the drinking water treatment plant and the predicted formation (calculated using the proposed model). The red line represents the one-to-one line.

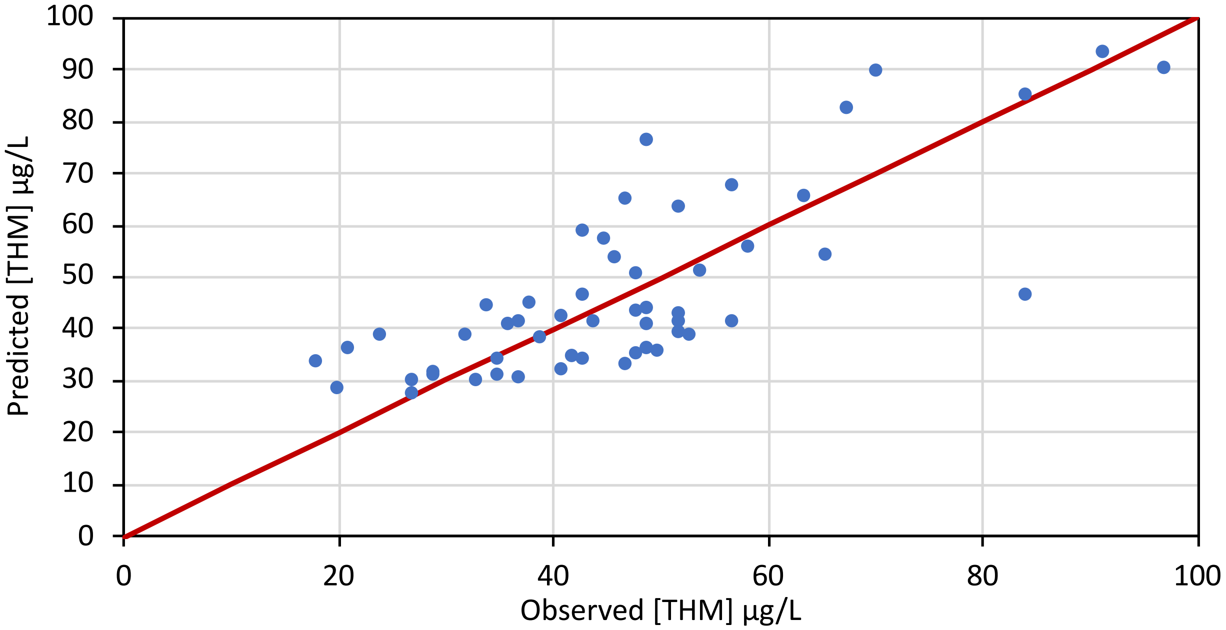

Figure 3. Relationship between the observed concentration of THMs in the distribution system (at a given time) and the predicted formation (calculated using the proposed model). The red line represents the one-to-one line.

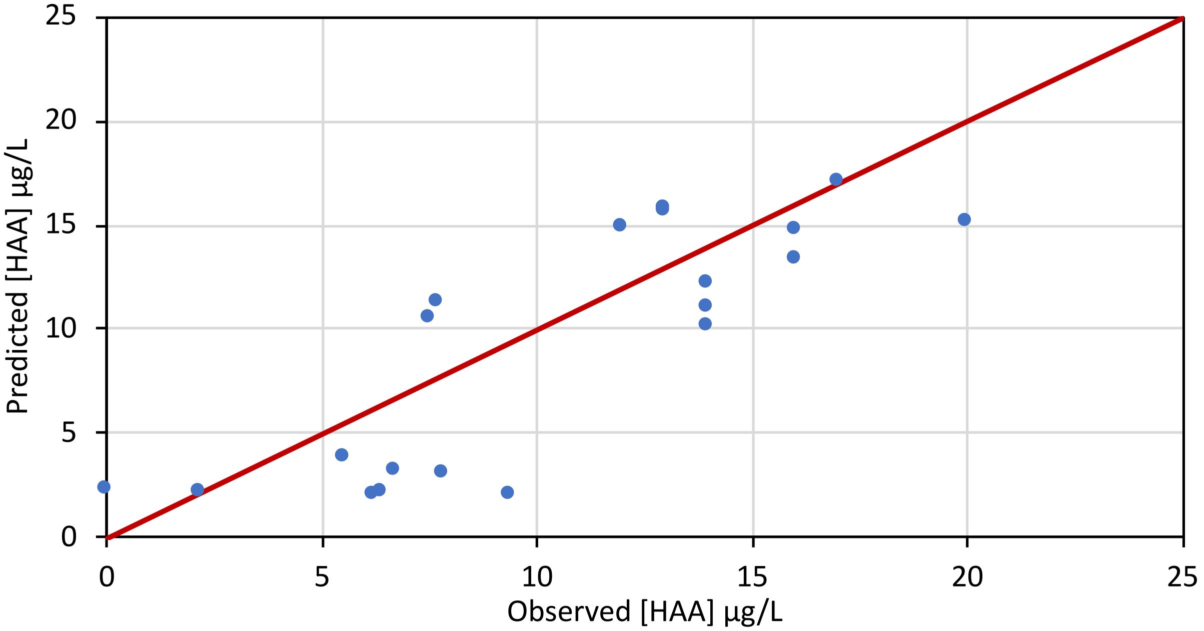

Figure 4. Relationship between the observed concentration of HAAs at the end of the drinking water treatment plant and the predicted formation (calculated using the proposed model). The red line represents the one-to-one line.

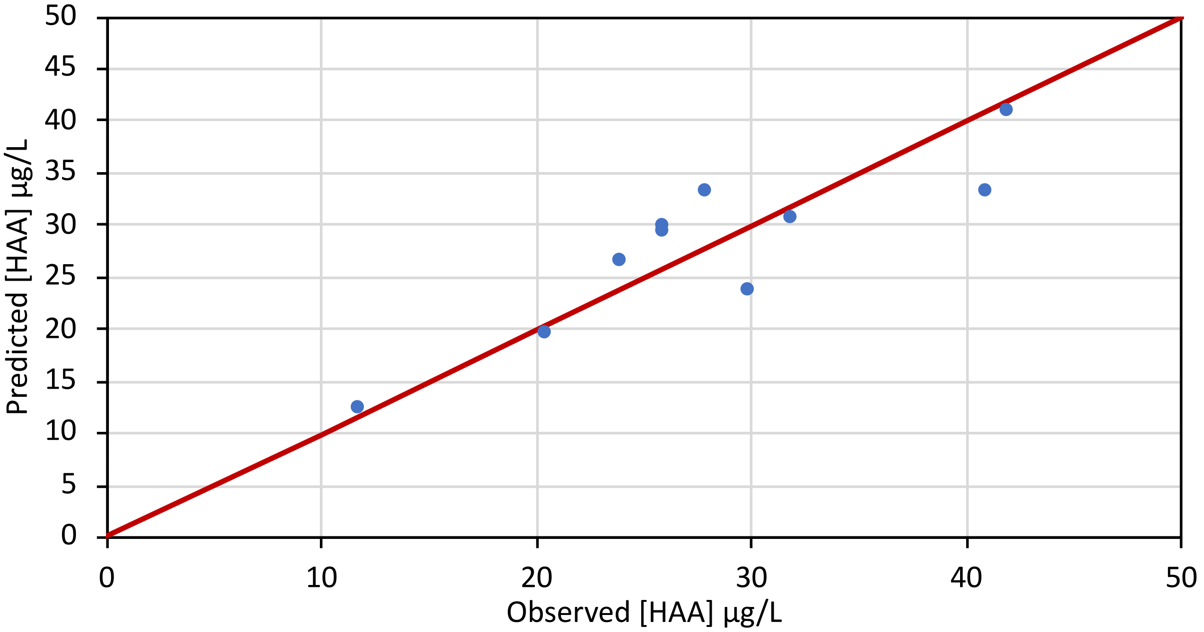

Figure 5. Relationship between the observed concentration of HAAs in the distribution system (at a given time) and the predicted formation (calculated using the proposed model). The red line represents the one-to-one line.

3.3 Model Verification

As an additional verification step, the THM model was tested using a separate set of jar testing data that was previously collected by HDR in 2010 (see Appendix B for summary of data used). The comparison between the measured and predicted 7-day THM formation from that jar testing work is shown in Figure 6. One limitation of the HDR data set is that there was no DOC concentration data. As a workaround, a correlation was developed between UVA and DOC in the GHWTP finished water and this correlation was used to estimate the settled water DOC from the jar test UVA data[2].

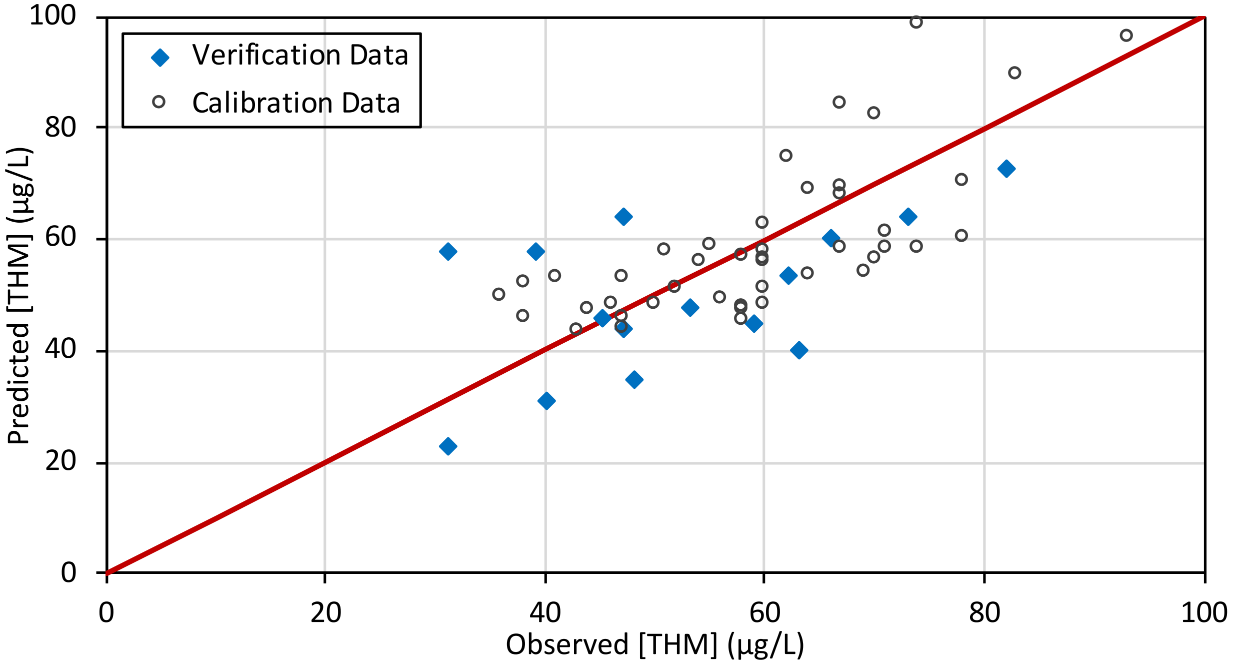

Figure 6. Comparison of model performance for data used to calibrate the model and historical HDR jar testing data used to verify the model.

4 Model Interpretation

4.1 THM Sensitivity Analysis

In this section, the sensitivity of THM formation to key variables is examined. Table 8 and Table 9 show the input values that were assumed in simulations of the worst-, base-, and best-case conditions in both the plant and the distribution system. These values were selected based on existing conditions at GHWTP and data from the jar tests; the basis for each is explained in the tables.

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[2] DOC = 65.8 * UVA – 0.0263

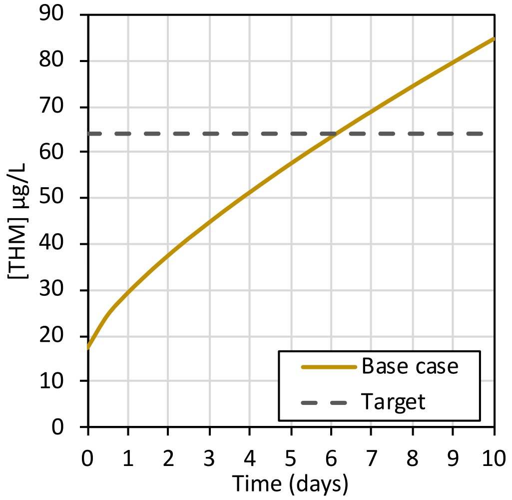
Table 8. Values used for independent variables in the sensitivity analysis of the plant model.

Variable	Worst-case	Base-case	Best-case	Basis
Alkalinity (mg/L)	100			Not a significant factor in DBP formation; did not vary to minimize number of test conditions
Temperature (°C)	20	15	10	Based on typical ranges observed at GHWTP
UVA (1/cm)	0.04	0.02	0.01	Based on typical ranges observed at GHWTP, with worst-case being higher than generally observes
pH	7.5	6.7	6.5	Based on assumption of enhanced coagulation using pH adjustment for base-case; improved performance with lower pH for best-case; no pH adjustment for worst-case
Cl_2 residual (mg/L)	2	1.2	1.2	Base- and best-case conditions based on current plant practice; worst-case based on residuals observed during jar testing
Cl_2 consumed (mg/L)	3	2.0	0.5	Base-case based on assumption of post-chlorination for full CT; best-case assuming short free chlorine CT
DOC (mg/L)		2.5		Base-case based on typical value observed during jar testing

Table 9. Values used for independent variables in the sensitivity analysis of the distribution system model.

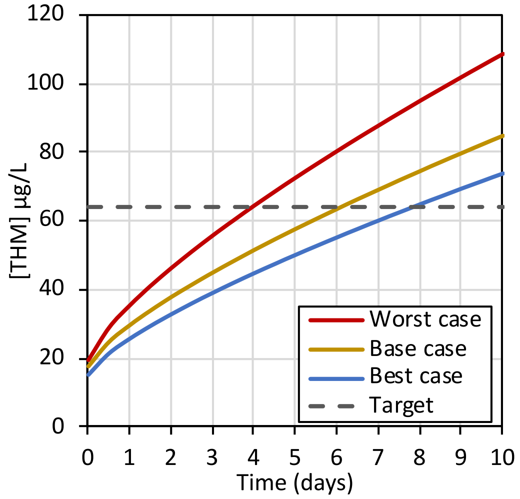
Variable	Worst-case	Base-case	Best-case	Basis
Alkalinity mg/L	100			Not a significant factor in DBP formation; did not vary to minimize number of test conditions
Temperature (°C)	23	15	11	Based on typical values observed in Santa Cruz distribution system
UVA (1/cm)	0.04	0.02	0.01	Based on typical ranges observed at GHWTP, with worst-case being higher than generally observed
pH	7.6	7.3	7.0	Based on typical values leaving GHWTP
Cl2 consumed / residual (mg/L)	1.8 / 0.2	1.0 / 0.2	0.9 / 0.3	Chlorine consumed is based on typical conditions in the distribution system; worst-case value was observed during the jar testing. Chlorine residuals are calculated from the residual leaving the plant and the chlorine consumed
DOC (mg/L)		2.5		Base-case based on typical value observed during jar testing

The impact of varying each variable individually to model worst and best-case scenarios while holding all other variables at their base-case values is shown in Figure 7 to Figure 11. THM formation over time is compared to the maximum target value of 64 μg/L, (80% of the regulation value of 80 μg/L). Variance in alkalinity was not examined because the coefficient for alkalinity is small and therefore changes in alkalinity (within a normal range) are expected to have only a minor impact. The importance of DOC levels will be examined in the next section. Note that in all graphs, the y-intercept at time 0 represents the DBP formation at the plant; time 0 represents the beginning of the distribution system.



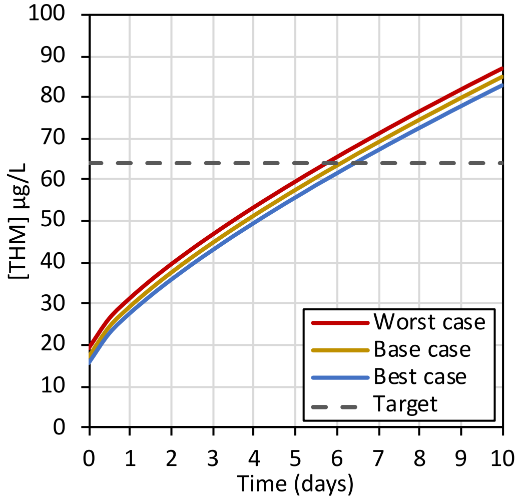
Variable	Plant	Distribution System
Alkalinity (mg/L)	100	100
Temperature (°C)	15	15
UVA (1/cm)	0.02	0.02
pH	6.7	7.3
Cl_2 residual (mg/L)	1.2	0.2
Cl_2 consumed (mg/L)	2.0	1.0
DOC (mg/L)	2.5	2.5

Figure 7. Modeled THM formation with base-case conditions (values shown in table). The dashed line represents the maximum target value for THM, 64 μg/L, assuming the goal is to not exceed 80% of the MCL (80 μg/L).



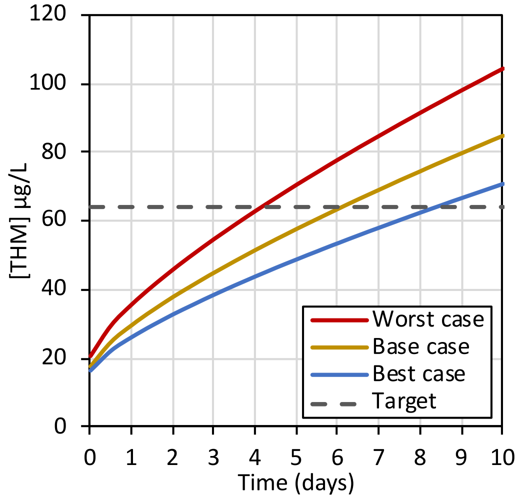
Temperature (°C)	Plant	Distribution System
Worst Case	20	23
Base Case	15	15
Best Case	10	11

Figure 8. Modeled THM formation with various temperatures (worst, base, and best-case values). All other variables are held at their base-case values. The dashed line represents the maximum target value for THM, 64 μg/L, assuming the goal is to not exceed 80% of the MCL (80 μg/L).



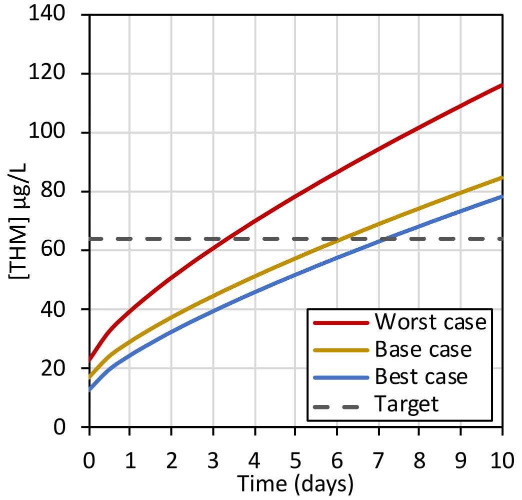
UV Absorbance	Plant	Distribution System
Worst Case	0.04	0.04
Base Case	0.02	0.02
Best Case	0.01	0.01

Figure 9. Modeled THM formation with various UV absorbance values (worst, base, and best-case values). All other variables are held at their base-case values. The dashed line represents the maximum target value for THM, 64 μg/L, assuming the goal is to not exceed 80% of the MCL (80 μg/L).



pH	Plant	Distribution System
Worst Case	7.5	7.6
Base Case	6.7	7.3
Best Case	6.5	7.0

Figure 10. Modeled THM formation with various pH values (worst, base, and best-case values). All other variables are held at their base-case values. The dashed line represents the maximum target value for THM, 64 μg/L, assuming the goal is to not exceed 80% of the MCL (80 μg/L).



Cl_2 Consumed (mg/L)	Plant	Distribution System
Worst Case	3	1.8
Base Case	2	1.0
Best Case	0.5	0.9

Cl_2 Residual (mg/L)	Plant	Distribution System
Worst Case	2	0.2
Base Case	1.2	0.2
Best Case	1.2	0.3

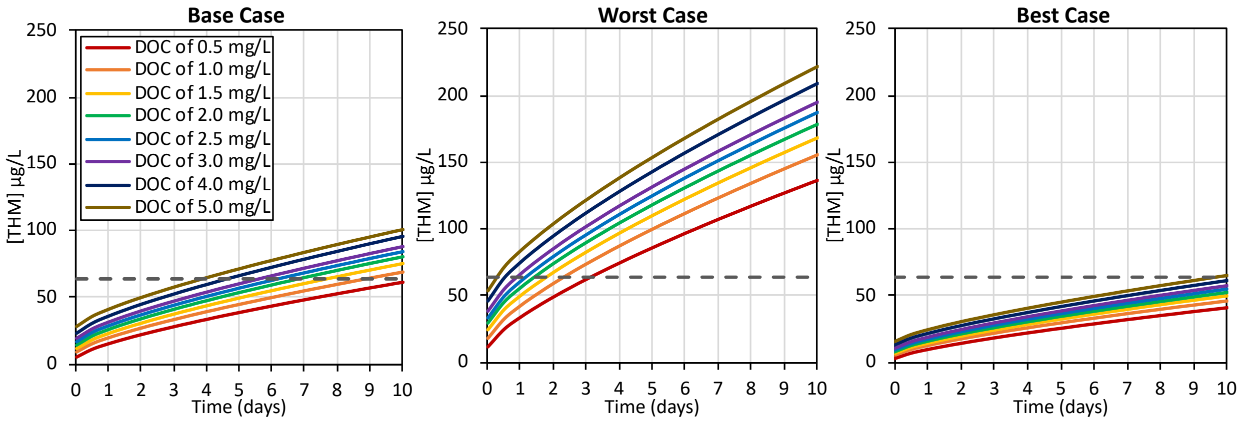
Figure 11. Modeled THM formation with various values of Cl_2 consumed and Cl_2 residual (worst, base, and best-case values). All other variables are held at their base-case values. The dashed line represents the maximum target value for THM, 64 µg/L, assuming the goal is to not exceed 80% of the MCL (80 µg/L).

Based on the modeling, the variables that have the greatest impact on THM formation are temperature, pH, and chlorine consumed/residual. While some of these parameters are relatively insensitive to design considerations (e.g., temperature), others can be more easily modified by treatment (e.g., pH and chlorine consumed). The modeling can therefore be used to identify and prioritize relevant parameters during the design phase.

UV absorbance does not appear to have a large impact on THM formation. This is due to the fact that our model has a much smaller coefficient for UV absorbance in the distribution system (0.0056) than the literature value of 0.326. Were these coefficients to be more similar, we would expect UV absorbance to have a larger impact on THM formation. A larger UV absorbance is indicative of the presence of higher levels of more aromatic carbon molecules, which are more reactive with chlorine than aliphatic carbon. One advantage of adding ozone to the treatment train prior to chlorine addition is that the ozone is highly reactive with aromatic carbon, which can reduce both UV absorbance and consequently the formation of THMs.

4.2 Impact of DOC on THM Compliance

The concentration of DOC is another key factor in DBP formation. To aid in determining the impact of DOC on DBP compliance, THM formation was modeled over a range of DOC values. This DOC analysis was done while setting the remaining variables to either their base-case, worst-case, or best-case values (Figure 12). The results demonstrate that the assumptions about the conditions for the other variables have significant impacts on the settled water DOC that would be needed to achieve the goal of 80% of the THM MCL after 7 days in the distribution system. The settled water DOC that would be required ranges between 0.5 and 5 mg/L, depending on the assumptions for the other variables (see Table 10). It is worth noting that the range of settled water DOC values used to calibrate the model ranged between 1.5 and 3 mg/L (see Table 5). Because the modeled scenarios extend beyond this calibration range (i.e., 0.5 to 5 mg/L), greater uncertainty is associated with the extrapolated values.



	Base Case		**Worst Case**		**Best Case**	
Variable	*Plant*	*DWDS*	*Plant*	*DWDS*	*Plant*	*DWDS*
Alkalinity (mg/L)	100	100	100	100	100	100
Temperature (°C)	15	15	20	23	10	11
UVA (1/cm)	0.02	0.02	0.04	0.04	0.01	0.01
pH	6.7	7.3	7.5	7.6	6.5	7.0
Cl_2 residual (mg/L)	1.2	0.2	2	0.2	1.2	0.3
Cl_2 consumed (mg/L)	2.0	1.0	3	1.8	0.5	0.9

Figure 12. Modeled THM formation with various DOC levels assuming base-, worst-, and best-case conditions. The dashed line represents the maximum target value for THM, 64 μg/L, assuming the goal is to not exceed 80% of the MCL (80 μg/L).

Table 10. Maximum allowable settled water DOC to limit 7-day THM formation for different assumed conditions in the plant and distribution system.

Scenario	**DOC Concentration (mg/L)**
Worst Case	< 0.5 mg/L
Base Case	1.5-2.0 mg/L
Best Case	> 5.0 mg/L

The proposed model suggests that THMs will be a problem under worst-case conditions, resulting in unrealistic requirements for the settled water DOC. As has been discussed, however, treatment design can be used to modify some of these parameters towards more favorable values. For example, (a) chemical addition can be used to control the pH through both the plant and the distribution system, (b) ozone can be used to decrease UVA prior to chlorine addition (see Section 4.1), and (c) chlorine demand can be reduced by maximizing organics removal and through the use of alternative disinfectants such as ozone. The worst-case scenario for a treatment train with these improvements is modeled in Figure 13. The variables that can be controlled by treatment train modifications (UVA, pH[3], chlorine residual, and chlorine consumed) were given their best-case values. The variable that cannot be controlled, temperature, was assigned the worst-case value (summer temperatures). In this modified worst-

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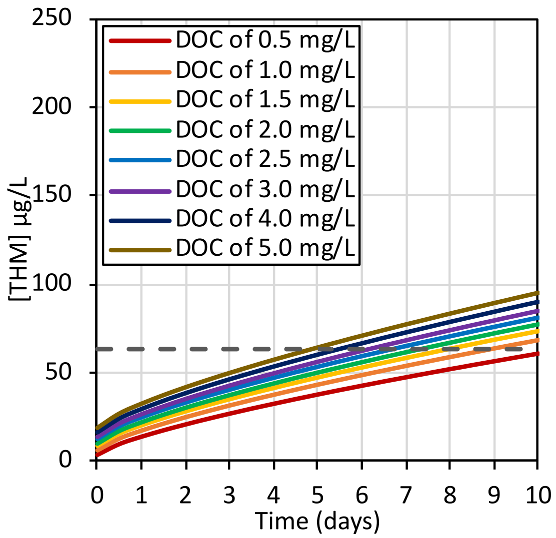
[3] Although pH is a key parameter in the plant THM formation model, it should be noted that the distribution system pH is the dominant pH factor in terms of 7-day THM formation.

case scenario, a settled water DOC of 2.5 mg/L would be low enough to achieve the THM goal of 80% of the MCL at 7 days in the distribution system.



Variable	Plant	DWDS
Temperature (°C)	20	23
UVA (1/cm)	0.01	0.01
pH	6.5	7.0
Cl_2 residual (mg/L)	1.2	0.3
Cl_2 consumed (mg/L)	0.5	0.9

Figure 13. Modeled THM formation with various DOC levels assuming the worst-case value for temperature and the best-case values for all other variables (values given in the table). The dashed line represents the maximum target value for THM, 64 µg/L, assuming the goal is to not exceed 80% of the MCL (80 µg/L).

4.3 Sensitivity to Source Water: Newell Creek versus San Lorenzo River

Model parameters were also determined for the Newell Creek (NC) and San Lorenzo River (SLR) source waters individually (Table 11). Only the plant data was fit individually for the two sources because the data set for the DWDS was too small to get reliable model results.

Table 11. Summary of model coefficients for THM formation at the plant for the combined source waters and for Newell Creek and San Lorenzo River individually.

Parameter	Combined	Newell Creek	San Lorenzo River
Intercept	0.429	0.254	0.777
Alkalinity	-0.0005	0.006	-0.002
Temperature	0.011	0.007	-0.004
Log(UVA)	0.168	0.407	0.102
Log(DOC)	0.676	1.228	0.246
pH	0.092	0.047	0.109
Log(Cl_2 residual)	0.324	0.828	0.217
Log(Cl_2 consumed)	0.231	0.405	-0.023

The effect of DOC levels on the formation of THM over time (with base-case conditions) was modeled for the two source waters (Figure 14). Both source waters result in similar DOC requirements (maximum DOC of 2.0 mg/L) if the residence time in the DWDS is 7 days. There does appear to be significant differences in the THM formation from Newell Creek water and San Lorenzo River water at higher DOC values (i.e. above 2.5 mg/L). However, if the DOC concentration can be reduced to levels similar to what is currently achieved at the plant (i.e. 2.5 mg/L and below), the predicted THM formation for the source waters is quite similar. This is illustrated in Figure 15, in which the predicted THM formation is shown for the individual source water models, as well as the combined model (i.e. the model calibrated with all source water data) for a DOC concentration of 2.5 mg/L. In this case, the predicted THM formation is similar for all three models.

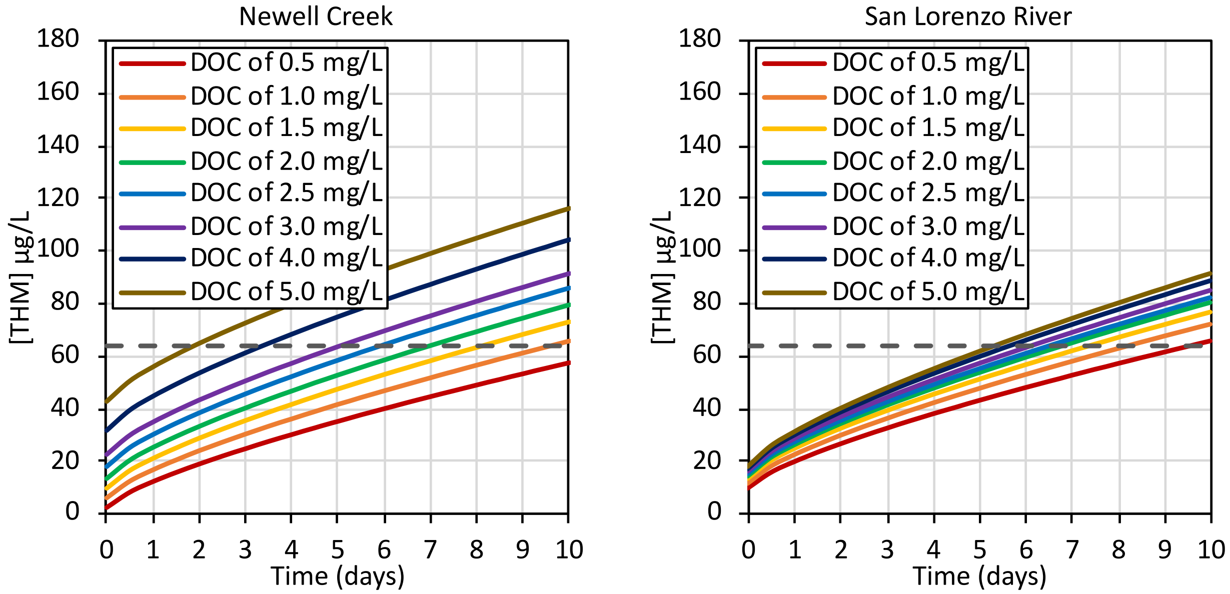


Figure 14. Modeled THM formation with various DOC levels (assuming all other variables are at their base-case values) for Newell Creek and San Lorenzo River source waters. The dashed line represents the maximum target value for THM, 64 µg/L, assuming the goal is to not exceed 80% of the MCL (80 µg/L).

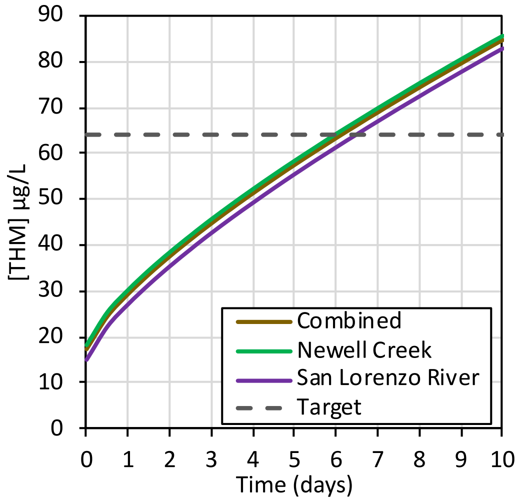


Figure 15. Modeled THM formation for Newell Creek (NC) and San Lorenzo River (SLR) source waters individually and combined (with base-conditions for all variables, including 2.5 mg/L DOC concentration).

4.4 HAA Model

HAA formation was also modeled and compared to the drinking water MCL of 60 µg/L. Using the selected model, HAA formation is dependent on three parameters: time, DOC, and chlorine consumed. The predicted formation of HAAs over time under base-case, worst-case, and best-case conditions is summarized in Table 12. Under base- and best-case conditions, a DOC of 3.0 mg/L does not result in HAA formation above 80% of the MCL. Under worst-case conditions, a DOC of 3 mg/L does result in HAAs above 80% of the MCL, but they are still below the MCL. Given that a DOC of 3 mg/L is likely to be too high to maintain compliance with the THM goals, HAAs are unlikely to be an issue for the GHWTP in the operating range of DOC that we would expect in the future.

Table 12. Summary of HAA formation under base, worst, and best-case conditions at the plant and after 7 days.

DOC (mg/L)	**Base Case (µg/L)**		**Worst Case (µg/L)**		**Best Case (µg/L)**	
	Plant	*7 days*	*Plant*	*7 days*	*Plant*	*7 days*
1	0.8	4.2	1.8	7.9	0.04	3.1
2	3.9	13.6	9.2	26.4	0.2	8.9
3	10.0	27.8	23.6	55.2	0.5	16.6

5 Conclusions

The results of this DBP modeling effort reveal key conclusions that are relevant moving forward with the treatment alternatives evaluation:

- Controlling the pH in the distribution system can play an important role in minimizing THM formation. Because GHWTP adds phosphate to the water leaving the plant, there is potentially an opportunity for further optimization of distribution system pH. As currently operated, the pH ranges between 7 and 7.6; achieving a pH of 7 more consistently in the distribution system may be an effective strategy to help limit THM formation.
- Newell Creek and San Lorenzo River do exhibit differences in THM formation, although it is primarily at high concentrations of DOC (>3 mg/L). If the DOC is controlled below 2.5 mg/L in the settled water, the differences are minimal and the combined model can be used for THM formation predictions.
- With sufficient operational control of key variables such as pH (6.5 or less in the plant and 7.0 in the DWDS), UVA (0.01 or less), and DOC, it does appear that the 7-day THM goals can be met with a DOC of 2.5 mg/L or less, even under worst-case temperature conditions. Given the importance of achieving DOC reduction, there may be a benefit to further investigating treatment options such as ACTIFLO Carb optimization and ozone coupled with biologically active filtration.
- HAAs are not expected to be a problem from a regulatory standpoint. Under worst-case modeled conditions, with a DOC of 3 mg/L, they did exceed 80% of the MCL after 7 days, but did not exceed the MCL under any conditions. At the expected future settled water DOC target of 2.5 mg/L or lower, compliance with the HAA goal should be achievable.

6 References

EPA (1998) “Empirically Based Models for Predicting Chlorination and Ozonation By-Products: Trihalomethanes, Haloacetic Acids, Chloral Hydrate, and Bromate.” Office of Water, EPA 815-R-98-005.

Obolensky A., and Singer P. C., (2008) ‘‘Development and Interpretation of Disinfection Byproduct formation Models using the Information Collection Rule Database,’’ *Environ. Sci. Technol.*, **42**, 15, 5654–5660.

Appendix A: Materials and Methods

A.1 Chemicals

Several chemicals were used for enhanced coagulation and ACTIFLO Carb jar testing. Table A1 summarized the characterization of these chemicals and their stock concentration used for this testing. The liquid alum was diluted to a stock solution with concentration of 5000 mg/L as alum for the jar testing. A stock solution contains 0.1% (wt) anionic polymer was used for the jar testing.

Table A1. Characterization of Coaugulants and Aids during Jar Tests

Property	Alum	Microsand	Anionic Polymer	PAC
Vendor	CHEMTRADE	Veolia	NALCO	Cal-Pacific Carbon, LLC.
Name	Liquid Alum	Silica Sand	NALCLEAR 7763	PACarb Plus
Chemical Formula	$Al_2(SO_4)_3 * 4H_2O$	SiO2		C
Specific Gravity	1.332	2.65		1.043
Apparent Density, g/cm3	--	--	1.03-1.08	0.42-0.48
Strength	8.18% as Al2O3	100%	100%	0.65% by weight

A.2 Jar Test Procedures

Prior to starting the jar tests, the water was taken out of the refrigerator and allowed to warm to approximately the same temperature as when collected. The raw water was analyzed for pH, temperature, turbidity, alkalinity, dissolved UV-254, TOC, dissolved organic carbon (DOC), and bromide.

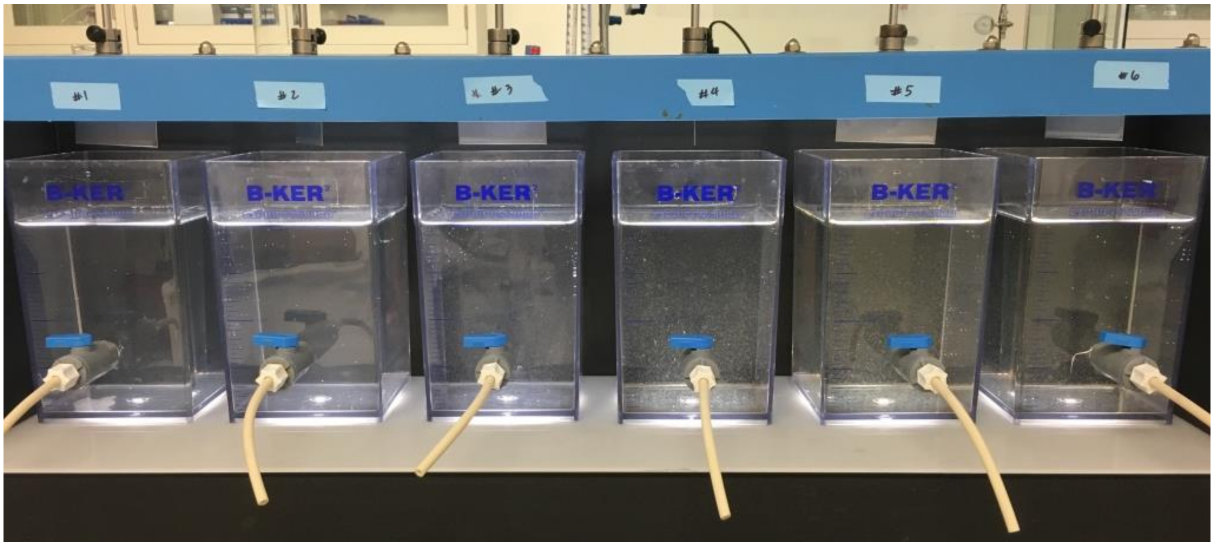


Figure A1. Jar Testing Apparatus

The procedures used for the jar tests followed generally accepted procedures as described in (Kawamura, 2000) and (AWWA, 2011), using square 'gator' jars as shown in Figure A1. The mixing regime for enhanced coagulation tests included rapid mix followed by four steps of tapered flocculation. The maximum speed of the mixer was used for rapid mix, for 1 minute. The velocity gradients used during tapered flocculation were 55, 40, 25, and 15 sec^{-1}, which correspond to mixing speeds of 55, 45, 33, and 22 rpm, respectively. Each step of tapered flocculation had a duration of 7.5 minutes, for a total flocculation time of 30 minutes. Following coagulation, the water was allowed to settle for 30 minutes prior to sample collection. The sampling point is the same in all jars, and is located 10 centimeters below the water surface.

For ACTIFLO Carb jar testing, both PAC and polymer were used during the procedure. The jar testing for ACTIFLO were carried out by following the test procedure shared by the technology vendor VEOLIA. Table A2 summarized the chemical addition, mixing speed, and mixing time for both enhanced coagulation and ACTIFLO Carb jar testing.

Table A2. Chemical Addition, Mixing Speed, and Mixing Duration for Jar Testing in This Study

Enhanced Coagulation			ACTIFLO Carb		
Chemical Addition (in order)	Mixing Speed (rpm)	Mixing Duration (minutes)	Chemical Addition (in order)	Mixing Speed (rpm)	Mixing Duration (minutes)
Liquid Alum	300	1	PAC	200	8-10
--	55	7.5	Microsand	200	--
--	45	7.5	Liquid Alum	300	2
--	33	7.5	Anionic Polymer	300	0.25
--	22	7.5	--	220	0.75
	0	30	--	0	2

A.3 Solution Ozone Test

A modification of the solution ozone test (SOT) described by Rakness (2005) was used for bench scale ozone dosing and assessment of ozone demand. In brief, a stock ozone solution was prepared by bubbling ozone through deionized (DI) water. Oxygen was used as the feed gas for the ozone generator. Dry ice was packed around a custom-made ozone stock solution vessel to maintain a water temperature just below zero degrees Celsius for better gas transfer as ozone was bubbled in through a ceramic diffuser. The equipment setup for the SOT is shown in Figure A2. The resulting ozone stock solutions used for the SOT tests had very high ozone concentrations, up to 90 mg/L of dissolved ozone. The SOT procedure delivers ozone in the form of an aqueous solution, rather than as a gas stream bubbled through the test sample. As a result, the applied ozone dose is the same as the transferred ozone dose, and ozone gas transfer efficiency is not a factor in determining ozone dose.

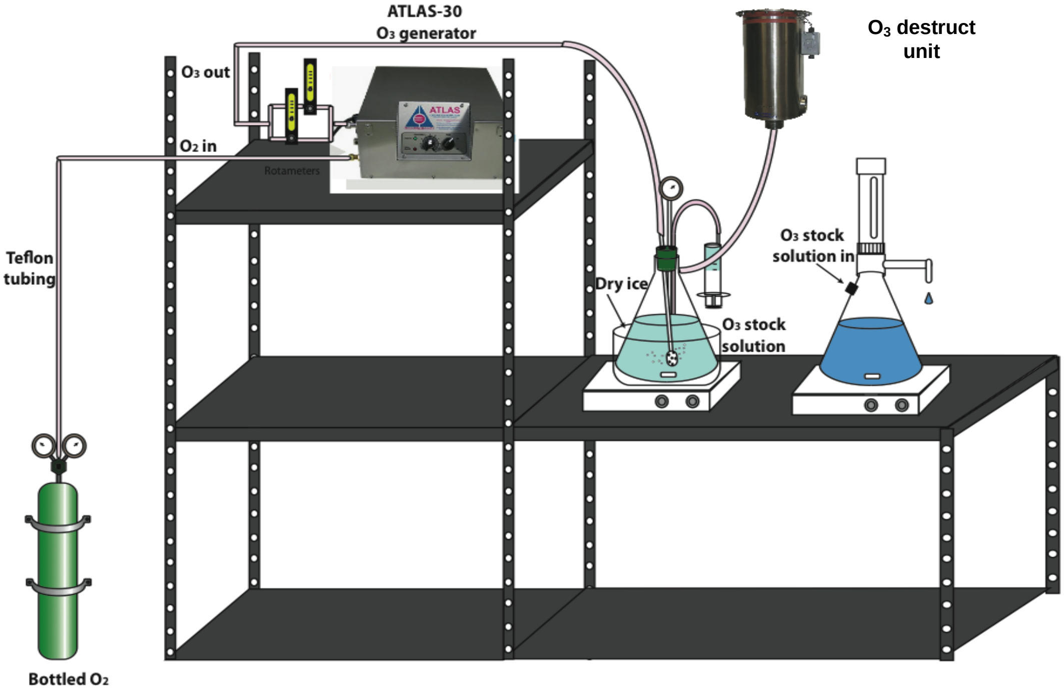


Figure A1. SOT Apparatus

After bubbling ozone into the chilled DI water for at least 10 minutes to make the concentrated ozone solution, an aliquot of the ozone stock solution was removed, using a glass syringe, for measuring the ozone concentration. The gravimetric indigo standard method (Rakness, 2005) was used for ozone analysis. The ozone stock solution was gradually added to a known volume (50 mL) of indigo solution using the syringe, and the blue color of the indigo solution faded as it reacted with the ozone. Once the blue color was almost gone, the mass of the sample was measured for gravimetric determination of the ozonated solution volume dispensed into the indigo solution, and the residual indigo concentration was measured at 600 nanometers (nm) using the spectrophotometer. The ozone concentration of the stock solution was calculated from these measurements.

Ozone demand, and consequently the required ozone dose, can vary with changes in source water quality (e.g., TOC, turbidity, reduced iron and manganese, nitrite, etc.). Particulate matter (measured as turbidity) can be particularly variable when the source water is a river.
Ozone can be applied on both raw source water (preozonation) and the coagulated/settled water (intermediate ozonation). In this study, solution ozone test was utilized to simulate preozonantion and intermediate ozonation.

A.4 Ozone Decay

The ozone demand and decay coefficients of the water were determined using ozone decay curves. Once the concentration of the ozone stock solution was measured (Section 0), a known volume of the ozone stock solution was added to a known volume of sample, typically dosed as a ratio of ozone-to-TOC (e.g., 0.6 mg/L O_3 for each 1.0 mg/L TOC in the sample), and stirred for about 10 seconds. The ozone residual was measured at the following time intervals (each in terms of time elapsed after the ozone dose was added): 20 seconds, 40 seconds, then 1, 2, 3, and 4 minutes. The residual ozone was measured at each time interval by withdrawing an aliquot of the ozonated sample (using a dispenser system) into a small vial containing indigo solution and using the gravimetric indigo standard method.

A.5 Disinfection

Various disinfection scenarios were compared with the testing. The control used to simulate the current treatment condition included pre-chlorination ahead of coagulant addition. For all other test conditions, disinfection was added after filtration. The filtration step was approximated by filtering the sample through a 0.45-µm membrane filter. Primary disinfection conditions targeted 3 mg/L as Cl_2 for 60 minutes contact times. Final secondary disinfection targeted 1.2 mg/L as Cl_2 after 2.5 minutes contact time in the finished water. A few test conditions compared the use of free and combined chlorine – in these cases, the combined chlorine had a target of 4 mg/L as Cl2 in the finished water. The stock chlorine solution used in this step had concentrations around 10,000 mg/L as Cl_2. The stock ammonia solution concentration was 1,000 mg/L as N.

A.6 SDS DBP Test Procedures

To simulate DBP formation of various finished water conditions, following the disinfection step(s), the pH was adjusted to 7.5 with the addition of sodium hydroxide (NaOH, 0.3% by wt) – the ACTIFLO jar tested waters were not performed with enhanced coagulation, thus the finished water pH was not adjusted. The sample was bottled with no head space, then held at 15°C for the timeframe of interest.

Both free chlorine and chloramine disinfection were used with the SDS test conditions. Three hold times were tested for each finished water condition to establish a DBP formation profile over time: initial (time zero), a hold time between 1 and 3 days, and a hold time of 7 days (one set of samples was held for 9 days, instead of 7). Residual chlorine and pH were measured and recorded at each holding time. The SDS testing was completed in glass bottles without headspace and light exposure. Once the desired hold time was achieved, the sample was transferred into amber glass vials for the analysis of HAA_5 and THMs. Each of the sample collection bottles contained sufficient quenching agent for up to 4 mg/L as Cl_2.

A.7 Analytical Methods

Eurofins Eaton Analytical Lab analyzed samples for HAA_5, bromide, and bromate. All other analyses were performed on-site at the TT Lab in Pasadena. The analytical methods implemented are shown in Table A3. Samples for DOC and UV-254 analysis were filtered through a 0.45-µm membrane filter prior to analysis.

Table A3. Analytical Methods Used during the Jar Tests

Parameter	Method	Lab	Comments
pH	SM 4500-H+	TT	Hach IntelliCAL Standard pH probe
Temperature		TT	Digital thermometer
Turbidity	SM 2130	TT	HACH 2100AN
UV-254	SM 5910 B	TT	HACH DR5000. Samples were filtered using 0.45-µm filter before analysis.
Alkalinity	SM 2320	TT	Titration with 0.002N HCl
TOC	SM 5310C	TT	GE Sievers 5310C
DOC	SM 5310C	TT	GE Sievers 5310C. Sample filtered through a 0.45-µm filter.
SDS DBP	SM 5710C	TT	Sample filtered through a 0.45-µm Gelman Supor 450 (PES) membrane filter prior to setup
TTHM	Purge-and-Trap Gas Chromatography	TT	Parker THM-1000 Analyzer
Free Chlorine	HACH 8021 (DPD Method)	TT	HACH DR900
Total Chlorine	HACH 8061 (DPD Method)	TT	HACH DR900
Bromide	EPA 300	Eurofins	
Bromate	LC-MS-MS	Eurofins	
HAA_5	SM 6251B	Eurofins	

Appendix B: Jar Test Data Used for Model Development

Table B1. Data used for calibration of plant model include both jar test results conducted as part of the current study (highlighted rows) as well as data from jar tests conducted at GHWTP.

Treatment Train	Source Water	Temp (C)	pH	DOC (mg/L)	Br (µg/L)	UVA	Alk (mg/L)	Cl2 dose (mg/L)	Cl2 residual (mg/L)	Cl2 consumed (mg/L)	Time (min)	THM (µg/L)	HAA (µg/L)
2a	NC	18.5	7.12	3.04		0.059	85	3	1.78	1.22	60	36	
2a	NC	18.5	7.04	2.83		0.051	85	3	1.92	1.08	60	31	
2a	NC	18.6	6.94	2.65		0.045	85	3	2.12	0.88	60	27	
2a	NC	18.7	6.81	2.57		0.042	85	3	1.86	1.14	60	24	
2a	NC	19.1	6.53	2.38		0.039	85	3	2	1	60	19	
2a	NC	18.1	6.26	2.55		0.05	86	3	1.9	1.1	60	21	
2a	NC	18.0	6.26	2.45		0.045	86	3	1.98	1.02	60	21	
2a	NC	18.2	6.32	2.33		0.041	86	3	1.98	1.02	60	19	
2a	NC	18.3	6.37	2.24		0.037	86	3	1.88	1.12	60	19	
2a	NC	18.8	6.38	2.21		0.038	86	3	1.52	1.48	60	19	
2a	NC	18	6.14	2.1	40	0.04	92.5	3	2.44	0.56	60	18	14
2b	NC	17.8	6.14	2.28	40	0.04	92.5	3	2.22	0.78	60	19.9	14
3a	NC	17.9	6.18	2.16	40	0.041	92.5	1.4	1.17	0.23	2.5	5.8	6.2
3b	NC	17.9	6.23	2.15	40	0.04	92.5	1.4	1.19	0.21	2.5	8.2	9.4
4	NC	18	6.5	2.19	40	0.031	92.5	1.4	0.95	0.45	60	7.6	2.2
4	NC	18	7.23	2.26	40	0.03	92.5	1.4	0.97	0.43	60	6.4	0
A1	NC	19.2	7.32	2.63	40	0.05	92.5	3	1.92	1.08	60	25.8	17
A1	NC	18.8	7.28	2.47	40	0.05	92.5	3	2.4	0.6	60	24.7	16
A2	NC	18	7.44	2.53	40	0.033	92.5	2.5	1.9	0.6	60	22	7.5
A2	NC	18	7.37	2.6	40	0.033	92.5	2.5	1.79	0.71	60	20.6	7.7
2a	SLR	20	7.3	2.12		0.036	92	3	1.78	1.22	60	25	
2a	SLR	19	7	2.01		0.033	92	3	1.78	1.22	60	20	
2a	SLR	19	6.82	1.81		0.032	92	3	1.88	1.12	60	17	
2a	SLR	19	6.61	1.69		0.028	92	3	1.86	1.14	60	18	
2a	SLR	19.6	6.36	1.67		0.025	92	3	1.78	1.22	60	16	
2a	SLR	16.2	6.49	1.68		0.28	94	3	1.94	1.06	60	16	
2a	SLR	16.2	6.49	1.61		0.026	94	3	1.72	1.28	60	12	
2a	SLR	15.6	6.49	1.51		0.025	94	3	1.86	1.14	60	11	
2a	SLR	15.9	6.49	1.5		0.024	94	3	2.22	0.78	60	11	
2a	SLR	16.4	6.52	1.52		0.023	94	3	2.26	0.74	60	12	
2a	SLR	18.9	6.72	1.92		0.03	61.2	3	1.15	1.85	60	18	
2a	SLR	18.6	6.66	1.78		0.028	61.2	3	1.97	1.03	60	18	
2a	SLR	18.6	6.55	1.78		0.028	61.2	3	2.16	0.84	60	17	
2a	SLR	18.8	6.6	1.72		0.027	61.2	3	1.94	1.06	60	16	
2a	SLR	19.0	6.63	1.66		0.026	61.2	3	1.98	1.02	60	15	
2a	SLR	16	6.87	2.09		0.033	61	3	1.5	1.5	60	21	
2a	SLR	16	6.71	2.04		0.036	61	3	1.65	1.35	60	18	
2a	SLR	16	6.61	2		0.028	61	3	1.75	1.25	60	18	
2a	SLR	16	6.44	1.91		0.028	61	3	1.8	1.2	60	17	
2a	SLR	15.8	6.3	1.99		0.028	61	3	1.72	1.28	60	20	
2a	SLR	18.0	6.54	2.09		0.037	61	3	1.5	1.5	60	19	
2a	SLR	17.8	6.57	2.01		0.034	61	3	1.68	1.32	60	17	
2a	SLR	17.6	6.57	1.93		0.032	61	3	1.74	1.26	60	17	
2a	SLR	17.6	6.61	1.91		0.03	61	3	1.54	1.46	60	17	
2a	SLR	20.7	7.11	2.48	27	0.043	74	3	2.4	0.6	60	15.8	12
2b	SLR	20.7	7.01	2.54	27	0.043	74	3	2.4	0.6	60	16.9	13
3a	SLR	20.3	6.98	2.61	27	0.042	74	1.5	1.39	0.11	2.5	30.6	5.5
3b	SLR	20.4	7.14	2.53	27	0.043	74	1.2	0.01	1.19	2.5	6.9	6.4
4	SLR	20	6.73	2.38	27	0.033	74	1.5	0.61	0.89	60	14.4	7.8
4	SLR	20	6.98	2.4	27	0.032	74	1.5	0.8	0.7	60	5.2	6.7
A1	SLR	20.5	7.81	2.37	27	0.05	74	3	2.36	0.64	60	25.8	16
A1	SLR	20.2	7.87	2.5	27	0.053	74	3	2.92	0.08	60	29.8	20
A2	SLR	20	7.34	2.99	27	0.04	74	2.5	1.61	0.89	60	30.6	13
A2	SLR	20	7.17	2.57	27	0.036	74	2.5	1.61	0.89	60	10.9	14
2a	Blend	16.9	7.23	2.03		0.036	105.8	3	1.57	1.43	60	24	
2a	Blend	17.4	6.96	1.95		0.03	105.8	3	1.75	1.25	60	19	
2a	Blend	17.4	6.65	1.73		0.029	105.8	3	2.06	0.94	60	12	
2a	Blend	17.9	6.42	1.62		0.025	105.8	3	1.81	1.19	60	15	
2a	Blend	18.2	6.2	1.74		0.023	105.8	3	2.06	0.94	60	12	
2a	Blend	18.9	6.32	1.93		0.033	108	3	2.24	0.76	60	18	
2a	Blend	18.9	6.46	1.92		0.034	108	3	2.32	0.68	60	21	
2a	Blend	18.9	6.41	1.81		0.03	108	3	2.34	0.66	60	17	
2a	Blend	19.2	6.41	1.65		0.027	108	3	2.32	0.68	60	17	
2a	Blend	19.6	6.45	1.74		0.026	108	3	2.32	0.68	60	15	

Table B2. Data used for calibration of distribution system model include both jar test results conducted as part of the current study (highlighted rows) as well as data from jar tests conducted at GHWTP.

Treatment Train	Source Water	Temp (ºC)	pH	DOC (mg/L)	UVA	Alk (mg/L)	Cl2 entering DWDS (mg/L))	Cl2 residual (mg/L)	Cl2 consumed (mg/L)	Time (days)	THM (µg/L)	HAA (µg/L)
2a	NC	15.7	7.42	3.04	0.059	85	1.78	0.39	1.39	7	93	
2a	NC	14.8	7.34	2.83	0.051	85	1.92	0.49	1.43	7	83	
2a	NC	14.9	7.16	2.65	0.045	85	2.12	0.56	1.56	7	70	
2a	NC	15	6.93	2.57	0.042	85	1.86	0.41	1.45	7	67	
2a	NC	15.3	6.83	2.38	0.039	85	2	0.71	1.29	7	60	
2a	NC	16.4	6.52	2.55	0.05	86	1.9	0.55	1.35	7	70	
2a	NC	16.6	6.62	2.45	0.045	86	1.98	0.66	1.32	7	74	
2a	NC	16.6	6.6	2.33	0.041	86	1.98	0.52	1.46	7	71	
2a	NC	16.4	6.73	2.24	0.037	86	1.88	0.34	1.54	7	71	
2a	NC	16.6	6.71	2.21	0.038	86	1.52	0.41	1.11	7	69	
2a	NC	15	7.51	2.1	0.04	92.5	2.44	1.31	1.13	3	60.5	
2a	NC	15	7.51	2.1	0.04	92.5	2.44	0.85	1.59	7.19	85.6	38
3a	NC	15	7.5	2.16	0.041	92.5	1.17	0.07	1.1	7.17	64.2	18
A1	NC	15	7.68	2.63	0.05	92.5	1.92	0.46	1.46	6.94	110	43
A1	NC	15	7.65	2.47	0.05	92.5	2.4	0.68	1.72	6.94	116	44
3a	NC	15	7.5	2.16	0.041	92.5	1.17	0.24	0.93	3	50.8	
A1	NC	15	7.68	2.63	0.05	92.5	1.92	1.75	0.17	1.2	32.3	
A1	NC	15	7.65	2.47	0.05	92.5	2.4	1.51	0.89	1.2	46.9	
2a	SLR	16.7	7.57	2.12	0.036	92	1.78	0.06	1.72	7	74	
2a	SLR	16.4	7.37	2.01	0.033	92	1.78	0.49	1.29	7	67	
2a	SLR	16.6	7.2	1.81	0.032	92	1.88	0.62	1.26	7	62	
2a	SLR	16.7	7.07	1.69	0.028	92	1.86	0.47	1.39	7	64	
2a	SLR	16.7	6.83	1.67	0.025	92	1.78	0.64	1.14	7	60	
2a	SLR	14.3	6.89	1.68	0.28	94	1.94	0.97	0.97	7	55	
2a	SLR	14.6	6.87	1.61	0.026	94	1.72	0.85	0.87	7	60	
2a	SLR	14.6	6.74	1.51	0.025	94	1.86	0.94	0.92	7	58	
2a	SLR	14.7	6.73	1.5	0.024	94	2.22	0.51	1.71	7	60	
2a	SLR	14.6	6.74	1.52	0.023	94	2.26	0.87	1.39	7	64	
2a	SLR	17.3	6.89	1.92	0.03	61.2	1.15	0.34	0.81	7	38	
2a	SLR	16.8	6.83	1.78	0.028	61.2	1.97	0.89	1.08	7	36	
2a	SLR	16.9	6.8	1.78	0.028	61.2	2.16	0.86	1.3	7	38	
2a	SLR	16.4	6.68	1.72	0.027	61.2	1.94	1	0.94	7	43	
2a	SLR	16	6.81	1.66	0.026	61.2	1.98	0.9	1.08	7	44	
2a	SLR	16.1	7.27	2.09	0.033	61	1.5	0.52	0.98	7	78	
2a	SLR	15.9	7.11	2.04	0.036	61	1.65	0.45	1.2	7	67	
2a	SLR	16.2	6.85	2	0.028	61	1.75	0.52	1.23	7	60	
2a	SLR	15.5	6.69	1.91	0.028	61	1.8	0.35	1.45	7	58	
2a	SLR	16.2	6.6	1.99	0.028	61	1.72	0.28	1.44	7	47	
2a	SLR	15.7	6.8	2.09	0.037	61	1.5	0.41	1.09	7	56	
2a	SLR	15.7	6.73	2.01	0.034	61	1.68	0.54	1.14	7	50	
2a	SLR	15.6	6.85	1.93	0.032	61	1.74	0.49	1.25	7	52	
2a	SLR	15.6	6.95	1.91	0.03	61	1.54	0.63	0.91	7	46	
2a	SLR	15	7.11	2.48	0.043	74	2.4	1.6	0.8	2	36.6	
2a	SLR	15	7.11	2.48	0.043	74	2.4	0.81	1.59	9.1	79.3	53
3a	SLR	15	6.98	2.61	0.042	74	1.39	0.4	0.99	2	32.8	
3a	SLR	15	6.98	2.61	0.042	74	1.39	0.06	1.33	9.1	68.6	26
A1	SLR	15	7.76	2.37	0.05	74	2.36	0.45	1.91	6.9	123	48
A1	SLR	15	7.67	2.5	0.053	74	2.92	1.03	1.89	6.9	100	62
A1	SLR	15	7.76	2.37	0.05	74	2.36	0.99	1.37	1.9	61.2	
A1	SLR	15	7.67	2.5	0.053	74	2.92	1.89	1.03	1.9	65.6	
A2	SLR	15	7.34	2.99	0.04	74	1.61	0.12	1.49	6.9	96.1	39
A2	SLR	15	7.17	2.57	0.036	74	1.61	0.09	1.52	6.9	95	44
A2	SLR	15	7.34	2.99	0.04	74	1.61	0.64	0.97	2.9	56.2	
A2	SLR	15	7.17	2.57	0.036	74	1.61	0.57	1.04	2.9	59.8	
2a	Blend	16.6	7.21	2.03	0.036	105.8	1.57	0.76	0.81	7	78	
2a	Blend	16.5	7.11	1.95	0.03	105.8	1.75	0.85	0.9	7	67	
2a	Blend	16.1	6.85	1.73	0.029	105.8	2.06	1.09	0.97	7	60	
2a	Blend	16.1	6.7	1.62	0.025	105.8	1.81	1.02	0.79	7	58	
2a	Blend	16.2	6.46	1.74	0.023	105.8	2.06	1.2	0.86	7	47	
2a	Blend	16.5	6.56	1.93	0.033	108	2.24	1.06	1.18	7	54	
2a	Blend	16.7	6.6	1.92	0.034	108	2.32	1.2	1.12	7	58	
2a	Blend	16.6	6.59	1.81	0.03	108	2.34	1.33	1.01	7	41	
2a	Blend	17	6.68	1.65	0.027	108	2.32	1.15	1.17	7	51	
2a	Blend	17.5	6.62	1.74	0.026	108	2.32	1.38	0.94	7	47	

Table B3. Data used for verification of THM model (sum of plant and distribution system formation) from a 2010 jar test campaign undertaken by HDR.

HDR Test Label	Temp (C)	pH	DOC (mg/L)	UVA	Alk (mg/L)	Plant Cl2 residual (mg/L)	Plant Cl2 consumed (mg/L)	DWDS Cl2 residual (mg/L)	DWDS Cl2 consumed (mg/L)	Time (days)	Measured THM (µg/L)	Predicted THM (µg/L)
#4 with 10 mg/L Alum	13.5	6.99	3.8	0.0624	100	1.46	0.74	0.16	1.3	7	82	72
#4 with 20 mg/L Alum	13.5	6.87	3.1	0.0502	100	1.54	0.66	0.15	1.39	7	73	64
#4 with 30 mg/L Alum	13.5	6.78	2.9	0.0462	100	1.67	0.53	0.38	1.29	7	66	60
#5 with 20 mg/L Alum	12.8	6.65	2.8	0.045	100	1.59	0.61	0.31	1.28	7	62	53
#5 with 30 mg/L Alum	12.8	6.54	2.5	0.0404	100	1.55	0.65	0.61	0.94	7	59	45
#8 with 10 mg/L Alum	13.1	7.03	2.8	0.0448	100	1.47	0.73	0.05	1.42	7	47	64
#8 with 20 mg/L Alum	13.1	6.9	2.5	0.0398	100	1.42	0.78	0.15	1.27	7	39	58
#8 with 30 mg/L Alum	13.1	6.83	2.3	0.0362	100	1.74	0.46	0.36	1.38	7	31	58
#9 with 10 mg/L Alum	13.8	6.56	2.6	0.0414	100	1.3	0.9	0.11	1.19	7	53	48
#9 with 20 mg/L Alum	13.8	6.43	2.5	0.0394	100	1.46	0.74	0.24	1.22	7	45	45
#9 with 30 mg/L Alum	13.8	6.38	2.2	0.035	100	2.09	0.11	0.66	1.43	7	47	44
#11 with 10 mg/L Alum	11.8	6.19	2.8	0.0452	100	1.38	1.32	0.05	1.33	7	63	40
#11 with 20 mg/L Alum	11.8	6	2.2	0.0356	100	1.57	1.13	0.06	1.51	7	48	34
#11 with 30 mg/L Alum	11.8	5.98	1.9	0.0304	100	1.38	1.32	0.1	1.28	7	40	31
#12 with 30 mg/L Alum	11.6	5.5	1.9	0.03	100	1.2	1.5	0.09	1.11	7	31	23

Appendix C: Jar Test Data Not Included in Model Calibration

Some of the jar testing data were not included in the model calibration for various reasons:

- Jar tests modeling a pre-chlorination dosing strategy (e.g., train 1) were excluded from both the plant and DWDS models because these models were exclusively for post-chlorination dosing strategies.
- Jar tests with a chloramine residual (e.g., trains 2b and 3b) were excluded from the DWDS model because this model was only for free chlorine systems.
- It was difficult to maintain and reliably measure a chlorine residual in the trains with ozonation; therefore, data from these trains were not included in the models (e.g., train 5, 6, A3, and A4 for the plant model and train 4, 5, 6, A2, A3, and A4 for the DWDS model).

Table C1. Additional jar tests simulating the plant (not included in the model calibration)

Treatment Train	Source Water	Temp (°C)	pH	DOC (mg/L)	Br (µg/L)	UVA	Alk (mg/L)	Cl_2 dose (mg/L)	Cl_2 residual (mg/L)	Cl_2 consumed (mg/L)	Time (min)	THM (µg/L)	HAA (µg/L)
1	NC	17.8	6.29	2.21	40	0.035	92.5	4	2.94	1.06	--	21.1	30
1	NC	18.1	6.26	2.27	40	0.035	92.5	4	2.98	1.02	--	17.6	32
5	NC	18	6.04	2.31	40	0.028	92.5	1.4	1.21	0.19	--	1.2	ND
5	NC	18	5.98	2.32	40	0.029	92.5	1.4	1.31	0.09	--	0.7	ND
6	NC	18.2	6.45	2.79	40	0.046	92.5	1.2	1.55	0.00	--	1.1	ND
6	NC	18.1	6.45	2.49	40	0.042	92.5	1.2	1.66	0.00	--	0.4	ND
A3	NC	18	7.73	2.8	40	0.036	92.5	2.3	2.16	0.14	--	8.5	2.8
A3	NC	18	7.74	2.77	40	0.035	92.5	2.3	1.92	0.38	--	3.4	2.5
A4	NC	18	7.38	2.7	40	0.05	92.5	1.6	1.55	0.05	--	5.0	3.5
A4	NC	18	7.65	2.83	40	0.051	92.5	1.6	1.59	0.01	--	4.2	2.3
1	SLR	20.4	6.67	2.59	27	0.041	74	4	1.94	2.06	--	26.2	46
1	SLR	20.5	6.65	2.44	27	0.041	74	4	1.83	2.17	--	24.7	45
5	SLR	18	6.49	2.47	27	0.03	74	1.4	1.15	0.25	--	0.8	ND
5	SLR	18	6.48	2.46	27	0.03	74	1.4	1.18	0.22	--	1.4	ND
6	SLR	17.9	6.71	2.3	27	0.044	74	1.2	2.00	0.00	--	0.3	5.6
6	SLR	17.9	6.79	2.42	27	0.047	74	1.2	1.86	0.00	--	0.3	5.6
A3	SLR	18	7.72	2.62	27	0.037	74	2.3	2.00	0.3	--	2.0	3.9
A3	SLR	18	7.81	2.6	27	0.037	74	2.3	2.07	0.23	--	1.7	3.9
A4	SLR	18	7.48	2.3	27	0.051	74	1.6	1.42	0.18	--	4.7	2.8
A4	SLR	18	7.79	2.53	27	0.051	74	1.6	1.49	0.11	--	5.9	3.7

Table C2. Additional jar tests simulating the distribution system (not included in the model calibration)

Treatment Train	Source Water	Temp (°C)	pH	DOC (mg/L)	Br (µg/L)	UVA	Alk (mg/L)	Cl_2 entering DWDS (mg/L)	Cl_2 residual (mg/L)	Cl_2 consumed (mg/L)	Time (days)	THM (µg/L)	HAA (µg/L)
1	NC	15	7.51	2.21	40	0.035	92.5	2.94	0.98	1.96	3.2	78.7	--
1	NC	15	7.51	2.21	40	0.035	92.5	2.94	0.63	2.31	7.39	110	61
1	NC	15	7.5	2.27	40	0.035	92.5	2.98	1.59	1.39	3.2	78.2	--
1	NC	15	7.5	2.27	40	0.035	92.5	2.98	1.20	1.78	7.39	91.5	68
2b	NC	15	7.5	2.28	40	0.04	92.5	4.00	0.04	3.96	3	38	--
2b	NC	15	7.5	2.28	40	0.04	92.5	4.00	0.03	3.97	7.19	50.3	19
3b	NC	15	7.5	2.15	40	0.04	92.5	3.90	1.88	2.02	3	12.7	--
3b	NC	15	7.5	2.15	40	0.04	92.5	3.90	1.39	2.51	7.17	14.4	8.3
4	NC	15	7.5	2.19	40	0.031	92.5	0.95	0.07	0.88	2	30.4	--
4	NC	15	7.5	2.19	40	0.031	92.5	0.95	0.03	0.92	6.98	39.9	8.6
4	NC	15	7.5	2.26	40	0.03	92.5	0.97	0.06	0.91	2	20.5	--
4	NC	15	7.5	2.26	40	0.03	92.5	0.97	0.01	0.96	6.99	32	6.7
5	NC	15	7.52	2.31	40	0.028	92.5	1.21	0.07	1.14	2	25.1	--
5	NC	15	7.52	2.31	40	0.028	92.5	1.21	0.03	1.18	6.98	43.7	9.1
5	NC	15	7.5	2.32	40	0.029	92.5	1.31	0.01	1.30	2	3.7	--
5	NC	15	7.5	2.32	40	0.029	92.5	1.31	0.00	1.31	6.98	5.1	ND
6	NC	15	7.5	2.79	40	0.046	92.5	1.55	0.26	1.29	0.93	22.8	--
6	NC	15	7.5	2.79	40	0.046	92.5	1.55	0.04	1.51	6.96	224.4	10
6	NC	15	7.5	2.49	40	0.042	92.5	1.66	0.35	1.31	0.85	21.3	--
6	NC	15	7.5	2.49	40	0.042	92.5	1.66	0.04	1.62	6.96	25.2	8
A2	NC	15	7.78	2.53	40	0.033	92.5	1.90	0.92	0.98	2.9	62.2	--
A2	NC	15	7.78	2.53	40	0.033	92.5	1.90	0.16	1.74	6.89	84.4	32
A2	NC	15	7.72	2.60	40	0.033	92.5	1.79	0.77	1.02	2.8	61.2	--
A2	NC	15	7.72	2.60	40	0.033	92.5	1.79	0.30	1.49	6.88	85.1	35
A3	NC	15	7.72	2.80	40	0.036	92.5	2.16	0.89	1.27	2.8	49	--
A3	NC	15	7.72	2.80	40	0.036	92.5	2.16	0.28	1.88	6.85	84.2	25
A3	NC	15	7.79	2.77	40	0.035	92.5	1.92	0.94	0.98	2.8	51.2	--
A3	NC	15	7.79	2.77	40	0.035	92.5	1.92	0.38	1.54	6.85	92.5	27
A4	NC	15	7.71	2.70	40	0.05	92.5	1.55	0.44	1.11	2.9	51.5	--
A4	NC	15	7.71	2.70	40	0.05	92.5	1.55	0.06	1.49	6.87	66.3	19
A4	NC	15	7.79	2.83	40	0.051	92.5	1.59	0.33	1.26	2.9	56.9	--
A4	NC	15	7.79	2.83	40	0.051	92.5	1.59	0.04	1.55	6.87	57.6	17

Treatment Train	Source Water	Temp (°C)	pH	DOC (mg/L)	Br (µg/L)	UVA	Alk (mg/L)	Cl_2 entering DWDS (mg/L)	Cl_2 residual (mg/L)	Cl_2 consumed (mg/L)	Time (days)	THM (µg/L)	HAA (µg/L)
1	SLR	15	7.5	2.59	27	0.041	74	1.94	0.99	0.95	2	46.3	--
1	SLR	15	7.5	2.59	27	0.041	74	1.94	0.16	1.78	9.1	84.5	92
1	SLR	15	7.5	2.44	27	0.041	74	1.83	0.93	0.90	1.9	45.9	--
1	SLR	15	7.5	2.44	27	0.041	74	1.83	0.26	1.57	9.1	91.7	100
2b	SLR	15	7.5	2.54	27	0.043	74	4.00	3.22	0.78	2	19.5	--
2b	SLR	15	7.5	2.54	27	0.043	74	4.00	2.42	1.58	9.1	59.4	20
3b	SLR	15	7.5	2.53	27	0.043	74	4.00	3.46	0.54	1.9	8.6	--
3b	SLR	15	7.5	2.53	27	0.043	74	4.00	2.36	1.64	9.1	16.6	11
4	SLR	15	7.5	2.38	27	0.033	74	0.61	0.03	0.58	2	31.2	--
4	SLR	15	7.5	2.38	27	0.033	74	0.61	0.01	0.60	7	31.4	9.5
4	SLR	15	7.52	2.40	27	0.032	74	0.8	0.06	0.74	2	35.8	--
4	SLR	15	7.52	2.40	27	0.032	74	0.8	0.02	0.78	6.9	42	6.7
5	SLR	15	7.5	2.47	27	0.03	74	1.15	0.03	1.12	2	23.8	--
5	SLR	15	7.5	2.47	27	0.03	74	1.15	0.01	1.14	6.9	25	4.5
5	SLR	15	7.5	2.46	27	0.03	74	1.18	0.04	1.14	2	19.4	--
5	SLR	15	7.5	2.46	27	0.03	74	1.18	0.00	1.18	6.9	22.8	5.7
6	SLR	15	7.5	2.30	27	0.044	74	2.00	0.52	1.48	1	19.1	--
6	SLR	15	7.5	2.30	27	0.044	74	2.00	0.09	1.91	7	29.7	20
6	SLR	15	7.5	2.42	27	0.047	74	1.86	0.53	1.33	1	28.3	--
6	SLR	15	7.5	2.42	27	0.047	74	1.86	0.10	1.76	7	40.2	18
A3	SLR	15	7.72	2.62	27	0.037	74	2.00	0.48	1.52	2.9	41	--
A3	SLR	15	7.72	2.62	27	0.037	74	2.00	0.12	1.88	6.8	78.3	28
A3	SLR	15	7.81	2.60	27	0.037	74	2.07	0.47	1.60	2.9	44.7	--
A3	SLR	15	7.81	2.60	27	0.037	74	2.07	0.05	2.02	6.8	76	27
A4	SLR	15	7.5	2.30	27	0.051	74	1.42	0.21	1.21	2.9	44.5	--
A4	SLR	15	7.5	2.30	27	0.051	74	1.42	0.03	1.39	6.8	58	20
A4	SLR	15	7.79	2.53	27	0.051	74	1.49	0.03	1.46	2.9	25	--
A4	SLR	15	7.79	2.53	27	0.051	74	1.49	0.01	1.48	6.8	36.6	11